PHYSICAL AND GEOCHEMICAL BEHAVIOR OF MINE ROCK STOCKPILES IN HIGH RAINFALL ENVIRONMENTS

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Abstract

A full-scale trial dump was constructed at Grasberg Mine, Indonesia, to investigate the geochemical processes and flow behavior within a waste rock pile in a high rainfall environment. The effectiveness of using different limestone blending methods with acid rock and impervious covers to minimize acid rock drainage (ARD) was assessed. The chemistry, temperature, and oxygen data from the trial dump showed that oxidation occurred throughout the full profile of the dump; however, the oxidation rates at the trial dump were approximately 11% to 58% lower than those calculated for smaller scale experiments. The low-permeability cover that was constructed using the materials available at the Grasberg Mine was not effective in reducing infiltration into the trial dump and had little benefit in reducing the acid generation. Conversely, it was found that limestone blending with a crusher/stacker system was an effective method to mitigate ARD when the geochemical criteria are adequately fulfilled.

Excavations were conducted in the test panels at the end of experiment revealing that inclined coarse and fine layers formed the internal structure of the waste rock pile. The presence of this coarse and fine layered structure within the profile tended to increase the potential for preferential flow and thus was found to adversely affect the performance of limestone blending and the limestone cover. In order to develop a fundamental understanding of water flow and geochemical mixing within the layered mine rock, a meso-scale laboratory experiment that contained the inclined
coarse and fine layers of acid mine rock and limestone was constructed. The results of the meso-scale experiment demonstrated that the distribution of water flow in the coarse and fine layered pile was dependent on rainfall intensity. Furthermore, the flow path and residence time were found to control effluent quality from the layers. When rainfall intensity was increased, the flow of water was distributed more evenly and vertical flow became the dominant flow mechanism, promoting chemical mixing between the leachate from the different layers. In summary, this study indicates that limestone blending and limestone covers are most effective in mitigating ARD in waste rock in high rainfall environments.
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Co-Authorship Statement

The main body of this manuscript-based thesis consists of five draft manuscripts (Chapters 2, 3, 4, 5 and 6) that were prepared for publication. Although these manuscripts were co-authored by my research supervisors, Dr. Ward Wilson and Dr. Stuart Miller, I conducted the research and manuscript preparation myself. My role in this research was to design, construct, and monitor the field and laboratory experiments, collect and compile the data, perform analyses of the data, and to prepare the drafts of the individual manuscripts. My research supervisors only provided guidance for the research and manuscript preparation.

Chapters 2, 3 and 4 present the analyses of the data collected from the field experiment located at the PT Freeport Indonesia mine site. PT Freeport Indonesia provided access to the collected data and gave permission for its use in this thesis.
Chapter 1 Introduction

1.1 Background

Acid rock drainage (ARD) is low pH water (i.e. pH less than 4) that is produced by the oxidation of sulfide minerals contained in rock. The resulting acidic water causes other metals associated with the sulfide minerals to also be dissolved and leached out with the pore water that drains from the rock mass. When ARD enters an aqueous system, it often deteriorates water quality and damages the ecosystem due to its acidic nature (Blowes et al. 2004; Lottermoser, 2003). Furthermore, the metals that are dissolved by the acidic water are frequently highly toxic to life. While ARD can occur as a natural phenomenon, the process is relatively slow. In contrast, mining activities associated with the construction of tailings and mine rock deposits dramatically accelerate ARD generation due to the exposure of previously isolated sulfide minerals to the atmosphere.

Although the mining industry is not the sole source of ARD, the vast scale and quantity of mine waste (i.e. waste rock and tailings) produced worldwide makes the industry a primary source of ARD. ARD has become one of the major problems facing the mining industry. For example, potential liability costs related to ARD from mines in Canada is estimated to be between 1.3 and 5 billion $CDN (INAP, 2003). The increasing demand for metals in recent years has led either to the opening of new mines or enhancing the capacity of existing ones. It is therefore logical to speculate that a
greater potential for ARD will be generated in the future since metal deposits are frequently associated with sulfide minerals.

It is not economical to process waste rock because it contains little valuable minerals. Nonetheless, waste rock must be removed to provide access to ores. There is always a predetermined cut-off grade that establishes whether the rock is classified as ore or waste. The amount of waste rock at a mine that must be excavated can reach hundreds of thousands of tonnes per day. Every mine produces different types and quantities of waste rock, depending on the mineralogy of deposits and the processing techniques applied in the mine (Blowes et al. 2004). When the waste rock that contains sulfide minerals is exposed to oxygen and water, ARD is generated and oxidation may continue for centuries when the neutralization potential is exhausted.

While many studies have been established to understand the long-term leaching behavior of waste rock, most of them were carried out using small-scale kinetic cells, such as humidity cells (for example Sapsford et al. 2008; Hakkou et al. 2008; Hollings et al. 2001; Botz and Mason, 1990; Fernandes et al. 1998) and free drainage leach columns (Miller et al. 2003; Stromberg and Banwart, 1999; Eriksson et al. 1997). While the results of the small-scale kinetic tests are useful for providing information as to whether or not the leachate from the waste rock will become acidic, the results do not always represent the leachate quality from a large-scale waste rock pile. These studies are not able to take into account, for example, the hydrological and physical processes within the waste rock piles that interact with the geochemical process that ultimately generate
the acid drainage and the environmental conditions of the test such as temperatures, climates are also different.

A review of the literature indicated that experiments blending acidic waste rock with materials that have a high neutralization capacity had been conducted in the laboratory (for example Heikkinen, et al. 2009; Mylona et al. 2000; Lapakko et al. 1997; Day, 1994), meso-scale experiments (Ziemkiewicz and Meek, 1994) and field applications (Brady et al. 1990). Although some blending techniques are found to be successful at a small scale, it has yet to be proven whether or not a similar performance can be achieved on a full scale waste rock system. The application of limestone at a field scale as reported by Brady et al. (1990) was mostly related to limestone used for backfilling or surface amendment to abate acid drainage in coal mines. The field application of limestone to mitigate acid drainage was also applied in hard rock mines, however, the blending performance from these sites was unknown due to a lack of documentation on the blending details and the seepage quality monitoring data (Mehling et al. 1997).

The internal structure and bulk permeability of a waste rock dump determines the water flow and oxygen transfer within the dump, which in turn, influences the oxidation of sulfide minerals and migration of oxidation products. When a waste rock dump is constructed with the end dumping method, the inclined layers comprised of both coarse and fine grain materials form the internal structure of the waste rock pile (Azam et al. 2007; Smith and Beckie, 2003; Herasymuik, 1996). Since one type of layer is
typically more conductive than other layer types, this particular internal structure can create preferential flow paths. A review of the literature reveals that only a few previous studies on the influence of stratification and inclined coarse and fine layers of waste rock have been completed using a numerical model (Fala et al. 2005; Fines et al. 2003; Wilson et al. 2000), and laboratory experiments have only been carried out for vertical coarse and fine grain layers (Newman, 1999). Furthermore, both these laboratory and numerical studies focused solely on water flow with no assessment of the chemical composition of the leachate. A geochemical study of the coarse and fine layers of a deconstructed waste rock pile has only been conducted by Tran et al. (2003) who investigated waste rock samples collected from the pile and also by Poisson et al. (2009) who used the geophysical techniques to assess the internal structure of an existing waste rock pile at Laronde Mine in Quebec.

The present thesis describes a research program that has been undertaken at PT Freeport Grasberg Mine in Indonesia. The primary objective of the research program is to understand the interaction among the hydrological, physical, and geochemical processes in a waste rock pile with respect to acid generation. The study was conducted using a field-scale waste rock pile termed ‘trial dump’, that was constructed in a series of panels. The primary benefit of the field site selected is that it allowed for the evaluation of the influence of high rainfall environments on the hydrological and geochemical behavior of the waste rock dump. In addition, it allowed for different ARD mitigation options such as a low-permeability cover, a limestone cover, and the
application of limestone blending on the individual panel sections of the trial dump in order to assess their effectiveness in minimizing acid generation at a large-scale waste rock pile. Since the influence of the inclined coarse and fine layers that form the internal structure of the waste rock pile on water flow and geochemical behavior were difficult to observe in the trial dump, a meso-scale laboratory experiment was also constructed at the University of British Columbia to further address this issue.

1.2 Objectives of Research

The primary purpose of this research was to investigate the physical and geochemical characteristics of acid-generating waste rock dumps in high rainfall environments, and to assess the efficacy of an impermeable cover, a limestone cover, and limestone blending for long-term ARD mitigation, as well as to understand the influence of the existence of coarse and fine layers in the internal structure of the waste rock pile on flow and geochemistry behaviors.

The specific objectives of the thesis are outlined below:

- To provide a description of the physical and geochemical behaviors within the dump that affect the ARD evolution kinetics and leaching characteristics in a high rainfall environment, based on data collected from the trial dump.
- To assess the temperature and oxygen concentrations in the interior of the waste rock pile and to evaluate correlations between the temperature and oxygen concentration and the quality of the drainage from the waste rock pile.
• To assess the performance of the large-scale application of a low-permeability cover, a limestone cover and limestone blending for mitigating ARD.

• To understand the appropriate geochemical criteria necessary to increase the effectiveness of the limestone blending method.

• To develop a fundamental understanding of the flow paths within the inclined coarse and fine layered structure of a waste rock pile.

• To investigate the influence of the rainfall intensity on the flow path and the quality of the leachate from the inclined coarse and fine layers.

1.3 Scope of Research

The research program focused on two phases of research. The first phase involved the in situ investigation of a large scale trial dump, while the second study focused on the design, construction and observation of a meso-scale laboratory experiment conducted at the University of British Columbia.

1.3.1 Field Experiment

A large scale waste rock dump was constructed for a trial at Grasberg Mine site in Indonesia in 1999. Different ARD mitigation scenarios were applied in this dump trial including an low-permeability cover, a limestone cover and a technique of blending limestone with the acid waste rock. Monitoring for oxygen concentrations, internal dump temperatures, leachate quality, and flow rates in the trial dump was carried out.
for approximately four years. Although record-keeping was not continuous for all of the parameters at all locations, the extent and coverage of the program was significant since the systems were installed during construction of the dump, and provided a unique data set for dump hydrology, dump oxidation, and ARD evolution at field-scale at a waste rock dump. The effectiveness of the ARD mitigation methods applied were also able to be evaluated from the data gathered.

In March 2004, an excavation underneath the cover treatment was completed to observe the occurrence and nature of oxidation. Further excavations were completed in December 2004 to focus on the panels constructed with limestone blending and limestone covers. The results and observations obtained during the excavation are included with the evaluation of the ARD mitigation methods described in this thesis.

1.3.2 Laboratory Experiment

Using material shipped from the Grasberg Mine, a meso-scale laboratory experiment was conducted at the University of British Columbia, to address concerns related to the limitation of the field experiment in assessing the influence of the inclined coarse and fine layers on the flow behavior and the effluent quality from the waste rock pile. The design of the laboratory experiment was based on observations made during the post-mortem study completed on the field trial, which revealed inclined coarse and fine layers within the internal structure of the waste rock dump. Three meso-scale panels were specifically built to emulate the internal structure of the waste dump. The panels consisted of acid rock layers only, acid rock layers covered with
limestone, and inter-bedded acid rock and limestone layers. The last two panels were used to independently investigate the effectiveness of the limestone cover on inclined coarse and fine layers of acid rock, and the influence of the limestone layers in reducing acid production. Several infiltration tests were also conducted between September 2006 and April 2008.

1.4 Thesis Organization

This thesis consists of an introductory chapter, five research papers in the manuscript format prepared for submission to journals, presented in Chapters 2 to 6, and a concluding chapter. The trial dump results are presented in Chapters 2, 3 and 4, while Chapters 5 and 6 present the results of the meso-scale laboratory experiment.

Chapter 2 describes the design and construction of the trial dump. It also includes an analysis of the responses of the measured leachate outflow from lysimeters to the various rainfall events. The role of a low-permeability cover in reducing the amount of infiltrated water into the trial dump is also discussed in this chapter.

Chapter 3 presents the analysis of the evolution of the leachate chemistry, dump temperature, and oxygen profiles in the trial dump and the correlations between these measured parameters. The release rates of constituents for ARD and metal leaching such as acidity, sulfate, and dissolved metals were evaluated and compared with the results from a previous 500-tonne test pad experiment using a similar type of waste rock at the same site. Chapter 3 also includes a discussion of the implications of the
placement of the low-permeability cover on the quality of leachate collected from the lysimeter beneath the cover treatment.

An evaluation of the ARD mitigation methods deployed in the trial dump, including limestone blending and the limestone cover, is discussed in Chapter 4. This chapter also presents information regarding the observed leachate quality and measured temperatures that were obtained within the trial dump to assess the effectiveness of limestone blending and the limestone cover treatment. Observations made during the excavation of the trial dump are also included in this chapter.

Chapter 5 presents results from the meso-scale laboratory experiment which investigated the effect of different rainfall rates on the distribution of water flow within the rock layers. The influence of bottom boundary conditions on the water flow distribution is also explained.

Chapter 6 provides observations and discussions on the quality of the leachate collected from the meso-scale laboratory. The parameters that influence the quality of leachate collected from each layer are also analyzed. The effect of the inclined layers on the performance of the limestone blending and limestone cover is also discussed in this chapter. Finally, a summary of the findings, the results and the conclusions of the thesis are given in Chapter 7.

An appendix is attached to this thesis that contains all the data obtained from the trial dump experiment. This data is mainly presented in graphic form and includes the volume of leachate and chemistry of leachate measured by the lysimeters at each
panel, temperatures recorded from the thermistors located at the base and the face of the trial dump, and finally oxygen concentrations measured from oxygen probes located at the base and in the sloping profiles at the trial dump. Data collected prior to the placement of the cover treatment on the trial dump is also included in the Appendix. It is used as a comparison to evaluate the performance of the cover materials.

The relationship among the chapters is illustrated in Figure 1.1 below.
Figure 1.1 Relationship between the chapters of this thesis
References


Chapter 2 Design, Construction and Flow Behavior of a Waste Rock Dump Trial in a High Rainfall Environment

2.1 Introduction

Weathering of waste rock may produce acid rock drainage (ARD) at mine sites. When waste rock piles are exposed to the atmosphere and water, the sulfide minerals contained in the waste rock are oxidized to form acid. In the presence of infiltrating water, the acid generated is transported and drained to the environment. Some of the hazardous metals contained within the waste rock are also dissolved due to the acidic conditions and leached out by the drainage. The flow of water in a waste rock dump has important implications with respect to the quality of the effluent from the waste rock pile. Both the quantity and pathway of the infiltrating water influences the weathering rate of waste rock (Molson et al. 2005; Frostad et al. 2005; Malmström et al. 2000).

Many researchers have investigated water flow in waste rock dumps during the last decade by studying the physical characteristics of existing waste rock dumps (Poisson et al. 2009; Azam et al. 2007; Stockwell et al. 2006; Matin et al. 2005; Fines et al. 2003; Smith et al. 2001; Herasymuik, 1996) by conducting experiments in large columns (Newman 1999; Li, 2000; Eriksson et al. 1997; Murr, 1979), and by constructing large-scale field experiments (Williams and Rohde, 2008; Corazaou Galegas, 2007; Blowes et al. 2006; Nichol et al. 2005). In addition to these types of physical

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1 A version of this chapter will be submitted for publication. Andrina, J., Wilson, G.W. and Miller, S., Design, Construction and Flow Behavior of a Waste Rock Dump Trial in a High Rainfall Environment
investigations, scientists have used results from laboratory and field studies to develop
numerical models that simulate water flow in a waste rock pile (Fala et al. 2005), and
also to study the coupled processes of water flow and chemical reactions (Linklater et
al. 2005; Molson et al. 2005; Sracek et al. 2004; Lefebvre et al. 2001; Eriksson and
Destouni, 1997). Although the results of these previous studies provide a good
understanding of water transport and geochemical processing in the waste rock pile,
there remains a need to improve our understanding of waste rock hydrology and its
effects on the weathering process in full scale waste rock piles, including the role of
climatic regimes on flow behavior and the leaching process.

The materials that comprise waste rock piles have a wide range of particle sizes
as a result of the blasting of the mine deposits. Fine particles (i.e. silt size and smaller)
have diameters less than 0.02 mm, while boulders in the piles can be larger than 1
meter. When waste rock is dumped from a truck, using either end-dumping or push-
dumping methods, the material segregates due to the various sizes. Larger waste rock
tends to ravel out to the bottom of the pile, while finer materials remain on the slope.
As the waste rock pile is further developed, its structure becomes even more complex
with variations in rock type, as well as the formation of inclined coarse and fine layers
caused by segregation. An internal, horizontally-compacted layer is also formed due to
traffic during construction (Poisson et al. 2009; Smith and Beckie, 2003). Among other
variations are changes in particle size due to weathering and particle breakdown over
time. In summary, the hydraulic properties of the waste rock pile are controlled by particle size distribution, stratification, and the texture of the materials in the pile.

When precipitation falls on the surface of an uncovered waste rock dump, the water infiltrates into the dump, evaporates back to the atmosphere, or flows off the dump surface as runoff. The amount of water that infiltrates into the dump depends on the hydraulic properties of the waste rock surface. The infiltrated water may form a channel in the dump that flows through the coarse media into the dump, or it may make its way through the fine matrix by capillary flow (Smith and Beckie, 2003; Nichol et al. 2005). The dominant flow mechanism in a waste rock pile is determined by the physical properties of each particular waste rock pile and the rainfall rate. When water infiltrates into the waste rock pile, it can either be stored by the capillary properties of the waste rock material or it can flow directly through the pile as outflow. The energy balance at the surface of the waste profile is determined by both climate type and dump surface conditions, such as the presence of vegetation (if a cover system exists) and its effect on the actual evapotranspiration rate from the waste rock dump (Wilson et al. 1997). The amount of runoff water from the waste rock dump depends on the infiltration capacity of the surface (Smith and Beckie, 2003; Miyazaki, 2005).

Preferential flow, which refers to an uneven and rapid movement of water, also occurs in porous media, such as waste rock. There are several forms of preferential flow including finger flow (Fetter, 1993; Paralnge and Hill, 1974), funnel flow (Kung, 1990), film flow (Tokunaga and Wan, 2001), and macro-pore flow (Flury et al. 1994;
The presence of large constituents, such as boulders or cobbles, or the existence of free-matrix zones (i.e., those with large pores and no fine matrix) in the waste rock pile can create preferential flow through macro pores. Preferential flows, which are characterized by quick responses to rainfall events and rapid changes in water flow rates at the toe of the dump, are dominant in many waste rock piles (Eriksson et al. 1997; Li, 2000; Nichol et al. 2005). Preferential flow has implications for the quality of effluent due to the limited contact time between the aqueous and solid phase of the waste rock material. Hence, preferential flow can prolong the release of the contaminants from waste rock piles (Murr, 1979; Younger et al. 2002; Eriksson et al. 1997).

The amount of water that infiltrates into a waste rock dump can be limited by the installation of an engineered cover system. In addition, the quantity of acid that is generated by the waste rock dump can be reduced if oxygen diffusion is limited by maintaining high saturation in the cover profile. This type of cover system is suitable for humid climates where precipitation exceeds potential evaporation such that the cover remains close to saturation (Wilson, 2003; Nicholson et al. 1989). Engineered cover systems are usually made of low permeability and high water retention capacity materials such as composite soil (Adu-Wusu and Yanful, 2006; Woyshner and Yanful, 1995; Yanful et al. 1993), run-of-mine waste (O’Kane et al. 2000), low sulphide tailings (Demers et al. 2008; Bussière et al. 2006; Sjoberg et al. 2003), or a mixture of tailings and waste rock (Wilson et al. 2006). A chemical cover is another alternative that may
be used to minimize the generation of acid. This approach utilizes specific chemical properties in the cover material to produce alkalinity in the infiltrating water that subsequently neutralizes the acid in the underlying waste rock (Taylor et al., 2006; Miller et al. 2003b). A chemical cover can also be used to consume the available oxygen (Peppas et al. 2005; Pierce et al. 1994) or to passivate sulphide-containing minerals (Evangelou, 1995).

A waste rock trial dump was constructed in 1999 at the Grasberg Mine in Indonesia. The purpose of the construction was to investigate the leaching behavior of waste rock within a large scale dump and to evaluate the appropriate treatments needed to reduce the long-term generation of acid. The design of the field experiment was intended to allow for an evaluation of the effect of a specific climate type on the flow behavior in the waste rock pile and to assess the weathering process. The field experiment for the trial dump was part of a scale-up program for an ARD mitigation study. Prior to designing the trial dump, a smaller-scale experiment that involved leach columns and test pads containing 500 tonnes of material had already been built and operated for several years at the Grasberg Mine site (Miller et al. 2003a). The trial dump was designed to mimic the behavior of a typical bench from a larger waste rock pile. The design of the trial dump included the use of 24 large lysimeters (10 m X 10 m) for monitoring and collecting the drainage from the base of the pile, as well as the use of instrumentation to monitor temperature and oxygen levels inside the profile of the dump. Equipping the field experiment with instrumentation to monitor physical
characteristics of the waste rock pile allowed for the evaluation of the interactions of the coupled physical and geochemical processes. Relatively low-permeability cover systems were also placed on the surface of three panels of the trial dump in order to assess the effectiveness of the cover materials in reducing the release of acid. Another ARD mitigation method involved the use of a limestone cover. It was also applied to two of the selected panels over the flat surface of the trial dump.

The primary objective of this chapter is to describe infiltration in the waste rock dump. This chapter describes the design and construction of the trial dump and presents the data for the leachate outflow obtained from the lysimeters over a four year period. The influence of precipitation events on the flow behavior was also determined. Finally, the influences of the various cover treatments on the infiltration and leachate production rate were also assessed.

2.2 Site Description

The Grasberg Mine is located on the southern flanks of the Jayawijaya Mountains in the Papua Province of Indonesia. The mine site has a tropical mountain climate which lacks any significant seasonal changes in temperature and rainfall quantities. The typical tropical alpine/sub-alpine temperature that was recorded at the mine site consists of a daily temperature range from 2°C to 14°C, with an average annual temperature of approximately 7°C. The annual precipitation ranges between 3,000 mm and 5,000 mm, and the average daily precipitation in the mining area is typically 10 mm throughout the year. The computed potential evaporation from the
mine is 1 mm/day. This is based on net radiation, humidity and air temperature data that was collected from a meteorological station at the mine site (Freeport, 1999).

2.3 Materials and Methodologies

The methods used in this study include field programs and laboratory tests. The field programs consisted of constructing the trial dump and measuring the volume of leachate collected from the basal lysimeters that were installed underneath the trial dump. The laboratory program consisted primarily of the grain size analysis of the waste rock samples collected during the construction of the trial dump.

2.3.1 Design, Construction and Monitoring of the Trial Dump

In 1999, construction of the trial dump began in the West Grasberg dumping area of the Grasberg Mine. The location was selected because it was available for at least a five-year operational period and was situated where there was not much interference with mining activities. A continuous waste rock dump was constructed that was 480 meters long, 80 meters wide and 20 meters high. These dimensions created a suitable model of a full-scale waste rock pile. Internal sections within the trial were subdivided into eight 60 m by 80 m panels that were named panels 1 through 8. Each panel was divided into three sections to accommodate selected types of waste rock. Figures 2.1, 2.2 and 2.3 show the configuration of the trial dump, a simplified plan view of the site and a typical cross section of the trial dump, respectively.
Figure 2.1 Sketch of trial dump
Figure 2.2 Simplified plan view of trial dump

- **RW** = red waste
- **BW** = blue waste
- **BW + LS** = blue waste and limestone

Legend:
- □ = lysimeter, 10 m x 10 m
- HDPE cover
- Road mud cover
- Weathered waste, Limestone cover
- L3, L2, L1: Lysimeters
- 80 m, 60 m, 480 m, 1%: Distances and slope

Directions:
- North
- Drainage from lysimeter to storage tank
- Pre-mixing
- Alternate dumping
- Layered dumping
- Crusher and stacker
Figure 2.3 Cross section of trial dump
The north side of the selected area was prepared at the start of construction by dumping Run-of-Mine (ROM) waste rock to provide an access road and an initial tipping spot. The foundation was then constructed with ROM waste rock, which was later compacted and graded to form a 1% slope to the north and east such that the seepage could be directed for collection.

Upon completing of the foundation, three lysimeters measuring 10 m x 10 m were installed at the base of each panel to collect the seepage from the pile. The centers of these lysimeters were located approximately 15 m (L1), 35 m (L2) and 65 m (L3) from the toe of each panel, as shown on Figure 2.3. Catchment areas were created by placing a 750-mm high berm on the perimeters of the lysimeters. A 1.5-mm High Density Poly Ethylene (HDPE) liner was placed on the 10 m x 10 m base area of the lysimeter and extended up to the berm, as shown in Figure 2.4. Since the base and dumping material might have contained sharp objects that could puncture the HDPE liner, a protective layer, made from 25 mm of rubber conveyor belt, was placed over the HDPE liner. The lysimeters were subsequently loaded with ROM waste rock that was geochemically and hydrologically similar to the waste rock that would be placed within the panels. This material was installed over the lysimeter to avoid impact damage due to the tipping and rolling of large or sharp rocks onto the face of the lysimeter during dumping of the waste rock. A total of 24 lysimeters, three for each panel, were constructed and installed. It should be noted that the trial dump did not have complete
Figure 2.4 Leachate collection system

closure on the water balance since there was no continuous collection system beneath the base.

A 50-mm drain hole was installed at the lowest (northeast) corner of the lysimeter. The drain hole was capped with a durable plastic screen and covered with waste rock before the lysimeter was loaded with material to avoid clogging and damage. Prior to the placement of the HDPE liner, a trench with a 1% gradient towards the toe of the panel was excavated to install a 50-mm-internal diameter (ID) pipe. The
pipe connected the drain of each lysimeter to a one-cubic-meter storage tank that was installed beyond the toe of the dump for leachate collections.

Overflow from the storage tank was directed to the tipping bucket through a 50-mm ID HDPE pipe. The tipping buckets were made of PVC and had two symmetrical chambers separated by a central wall that was placed on an axle, as shown on Figure 2.4. Each chamber had a 3-liter capacity. When the first chamber was full of water, it was tipped instantaneously, and the second chamber would start to collect water. Once the second chamber was full, the same process occurred, and the pattern was repeated continuously. The tipping bucket was equipped with a trip counter to record the number of tips over a given time period. The leachate discharged from the tipping bucket was directed to the southwest through a culvert that was installed along the south end of the trial area.

The instrumentation included thermistors and gas sampling ports that were installed at select locations to measure the temperature and oxygen concentration in the interior of the waste rock dump. Approximately three to five sets of thermistor strings and pore gas samplers were installed alongside the base and on the face of each panel. Further details regarding the design and installation of these instruments can be found in Chapter 3.

Two different types of acid forming waste rock were placed in the panels of the trial dump. The first type was a lower capacity potentially-acid forming (PAF) rock identified as blue waste, while the other type was a higher capacity PAF material.
termed red waste. In addition, the limestone that was to be mixed with the blue waste was placed in the outer or front sections of the panels. A haul truck with an approximate capacity of 250 tonnes was used to place the waste rock in progression from north to south on all the panels, except on the front face of panels 7 and 8, where a stacker was used. The end dumping method was deployed for truck placement on all sections of panels 1 to 8. The construction of the rear and middle sections was completed by February 2001. A lack of specified rock types in the mining sequence that was required for the construction of the front sections of the panels resulted in significantly slowing the overall construction of the trial dump. The trial dump was finally completed in April 2002. Figure 2.5 shows the progress of construction for the trial dump. Details regarding the types of waste rock placed on each section for each panel are described in Chapter 3, while detailed information for the placement of blended limestone and low PAF materials is presented in Chapter 4.

2.3.2 Cover Placement

Different types of covers were placed on the surfaces of selected panels to assess their effect on the quality and quantity of leachate. Panel 2 remained uncovered (control) for comparison. Relatively low-permeability cover systems were constructed on the flat surface of panel 1 in April 2002 and on the top of panels 3 and 4 in July 2002. In addition, three meters of ROM limestone was placed on the top of panels 5 and 6 in May 2002. Figure 2.6 shows the cover treatment at the different panels. The covers
Figure 2.5 Construction progress at the trial dump
Figure 2.6 Cover treatments
were placed only on the upper flat surface of the dump, and the panel slope remained uncovered, as shown on Figure 2.3.

Panel 1 was covered with 1.5-mm thick HDPE liner to test the effects of an impervious oxygen and water barrier. It was also to observe the formation of the oxidation products within the dump under the conditions where rain water infiltration was excluded. The edges of the HDPE liner were tucked into the shallow trench that was already dug around the panel and then the trench was backfilled using the material previously excavated from the same trench. Since the width of the HDPE sheet was shorter than the width of the panel, HDPE liner sheets were therefore overlapped and welded using a hot wedge welding machine. Seams between HDPE liners were checked by ‘air pressure testing’ to ensure there were no leaks at the seams.

Finding the appropriate materials at the mine site was the greatest challenge for the application of a barrier cover system since no suitable, naturally-occurring soils were available onsite. Therefore two separate test pads, measuring 12 m x 12 m, with a depth of 2 m, as shown on Figure 2.7, were constructed in March 2002 prior to the placement of the covers for the trial dump. Two materials were evaluated for the construction of covers that, based on previous laboratory tests, appeared to have appropriate characteristics to serve as oxygen and infiltration barriers.

The first cover material consisted of a mixture of road mud and limestone. The road mud was a fine grained material that was produced from the continuous
Figure 2.7 Sketch of Barrier Cover trial

breakdown of the road base material under the continuous heavy truck traffic. It was necessary to excavate this material from the road surfaces to maintain good trafficability but the material was usually discarded, however, because of the road mud properties it was considered a potential source for the cover material. Blending of the road mud, which had a high in-situ water content, with limestone that had low water content, was found to create a low-hydraulic conductivity material that achieved adequate compaction and which was suitable for the cover material.

The second material selected as a cover was weathered red waste rock, which was obtained from a historic waste rock dump that was being removed for the purpose of open pit operation. Because this weathered waste rock had been placed more than five years earlier, it was found to be highly altered and consisted mainly of fine grained
material, due to its high reactivity. The laboratory test results showed hydraulic conductivities for both of these materials to be approximately $10^{-8}$ m/s, as presented in Table 2.1.

**Table 2.1 Hydraulic conductivity of cover materials**

<table>
<thead>
<tr>
<th>Sample</th>
<th>Description</th>
<th>Hydraulic Conductivity, m/s</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weathered waste rock</td>
<td>Clayey silty gravel, wet soft</td>
<td>$1 \times 10^{-8}$ - $2 \times 10^{-8}$</td>
</tr>
<tr>
<td>Mixture of road mud and limestone</td>
<td>Well graded gravel with silt and clay, wet, soft</td>
<td>$2 \times 10^{-8}$ – $9 \times 10^{-8}$</td>
</tr>
</tbody>
</table>

Based on the results of the small scale trial, a 750 mm thick cover of blended road mud and limestone was placed on top of panel 3 and a 1,000 mm thick cover of compacted weathered waste rock was placed on top of panel 4. Both covers were compacted in 250-mm thick layers using a rolling compacter and a wheel loader. Since the experimental site had been exposed to a wet climate, the compaction was performed with the cover materials near fully saturated conditions or at wet of Proctor optimum.

**2.3.3 Sampling and Monitoring of the Trial Dump**

Waste-rock samples were collected during the construction of the trial dump at locations directly above the lysimeters for each panel, but not including any material larger than boulder-sized. Approximately 20 kg samples of waste-rock material were
collected from the tip face of the dump as it advanced over each lysimeter. Samples were obtained at three different times during the dump construction as follows: 1) when the lysimeter was initially being covered with rocks; 2) when the lysimeter was partially covered with waste rock; and 3) when the lysimeter was completely covered with rock. Samples were collected at the crest, at the mid-slope, and at the toe above each lysimeter resulting in a total of nine waste-rock samples from each lysimeter location. Photographs of the sampling points were taken during the sampling to provide further information for the typical grain sizes of the material at each location. Waste rock samples were analyzed for geochemical characteristics and grain size. All grain size analyses were conducted according to ASTM test method, D422-63 (ASTM, 1994).

The volume of leachate collected from each lysimeter was recorded daily (except Sundays and holidays) commencing August 2000. Monitoring continued until May 2004 for most panels, although several panels (i.e. blended blue waste and limestone panels) continued until the decommissioning of the trial dump in December 2004.

During the four-year operational period, several problems were encountered related to the drainage outlet that temporarily prevented the leachate from draining. While attempts were made to repair the outlet drainage, the outflow stopped completely from L3 lysimeter for panel 1 in November 2000, L3 lysimeter for panel 2 and L2 lysimeter for panel 7 in February 2001, and L2 lysimeter for panel 2 in May 2001. These four lysimeters operated only for a short period of time, ranging from one month
to eight months. In June 2003, the drainage for L3 lysimeter in panels 3 and 4 and L2 lysimeter in panel 1 stopped. The drainage of L3 lysimeter for panel 7 also stopped in December 2003. Although it could not be proven, it was believed that the termination of drainage was related to problems in the piping outlet system rather than changes in the permeability of the waste rock pile. Near the end of 2002, the collection tank and tipping bucket for panel 8 were destroyed by a mud slide that came from an adjacent area, and no further leachate data was collected from panel 8.

2.4 Results and Discussion

This section describes the results obtained from the laboratory testing and the field monitoring. Laboratory testing involved the grain size analyses of the waste rock samples collected during the construction of the trial dump. The field monitoring program recorded the volume of leachate draining from the basal lysimeters for the period August 2000 to May 2004 with some panels continuing until decommissioning of the trial dump in December 2004. An analysis of the outflow data along with the factors that influence the infiltration process in the trial dump is discussed.

2.4.1 Characteristics of the Material

In general, the waste rock samples were dominated by cobble and gravel-size rock with less than 3\% clay and silt. However, the results of the grain size analysis from individual samples confirmed heterogeneity among the samples. For example, some
samples consisted of more than 90% cobble, while the other samples contained more than 90% gravel and sand.

Results of the grain-size analyses indicated differences between the blue waste and the red waste. The red waste samples yielded averages of 42% cobble, 46% gravel, 10% sand, and 2% fine grained material based on seventy-two (72) samples. In contrast, the blue waste samples were comprised of 10% cobble, 73% gravel, 15% sand and 2% fine grained materials according to analysis of ninety-nine (99) samples. The blue waste tended to have finer particles than the red waste since most of the blue waste materials originated from poker-chip rock, which is a naturally-friable type of rock. Figure 2.8 shows typical grain size curves for both the red waste and the blue waste along with the upper and lower bound curves.

The grain size of samples collected from the front sections of panels that were constructed with blended blue waste and limestone (panels 4 to 8) also showed variation in the results. These samples consisted on average of 15% cobble, 60% gravel, 22% sand and 3% fine-grain material based on forty-five (45) samples collected from the front sections of the panels. The upper and lower bounds, as well as the typical grain size curves for the blended blue waste and limestone are shown in Figure 2.8. Samples that were collected from the panels constructed by truck dumping, particularly those samples obtained from the layer blended panel (panel 6), mainly contained cobble material. The presence of limestone in the panel contributed to there being a larger fraction of cobble-sized rocks due to it being naturally harder than the acid rock.
Figure 2.8 Grain size distribution of waste rock samples

On the other hand, a higher portion of sand and gravel was identified in samples collected from panels 7 and 8 that were constructed using the crusher-stacker method. Prior to placement, the material was passed through a crusher in order to ensure that the maximum particle size of the waste rock would fit into the chute opening for the stacker. Thus, the maximum grain size of the material placed on this section (i.e. 300 mm) was smaller than on the other panels.
Waste rock piles can be distinguished as a rock-like pile or a soil-like pile according to its hydrological properties. In a rock-like pile, water flows mainly through the pores between the coarse particles, while in a soil-like pile, the properties of the fine material control the flow behavior of the water in the pile as matrix flow. Dawson et al. (1998) indicated that at least 20% sand (2 mm) content is adequate to create a pile with soil-like behavior, while Smith et al. (2001) indicated 10% to 30% of material finer than sand (<4.75 mm) will behave as soil-like. According to these classifications, the results for the particle size analyses described above suggest that the trial dump should behave with the properties of a rock-like pile.

Because a large portion of the trial dump was constructed using the tipping and end-dumping method, segregation occurred during the dump construction. In the case of most panels, coarser materials were observed in the samples collected from the bottom of the dump, while finer fractions were generally found in samples collected from higher sampling points. However, due to the limited quantity of material that could be placed into the sample bags for testing, the results for the particle size analyses did not consistently show this segregation. Photographs taken during sampling provided evidence of segregation and Figure 2.9 shows an example of material segregation that occurred on the slope of the dump at panels 2 and 3. The length of the pen which was used as comparison for waste rock particle size scale in these photographs was 120 mm.
Figure 2.9 Waste rock segregation at the dump slope
2.4.2 Rainfall Data

The mine site had several automatic and manual rainfall gages at different locations. Due to the variable topography and altitude, the rainfall data was different among the stations. Rainfall data used for this experiment was collected from a rainfall gage located about 500 m southwest of the trial dump area. The altitude for the rainfall gage was approximately the same as the altitude of the trial dump. Annual rainfall data collected during the four-year monitoring of the outflow from the trial dump was consistent with an average annual rainfall of 3,600 mm and a coefficient of variation equal to 3%. The average daily precipitation was 9.8 mm during the monitoring period. Rainfall data indicated 46% of the time rainfall exceeded 10 mm/day. The greatest 24-hour rainfall measured at the station was 64 mm. About 134 days, or 8.6% of total monitoring days, were recorded as zero-rainfall days. The longest consecutive zero-rainfall period was 12 days, which occurred in August 2002.

2.4.3 Outflow of Individual Lysimeters

The outflow data from the lysimeters was recorded since the end of August 2000. At that time the lysimeters located at the rear and middle sections of the panels had been covered by the waste rock material for four months. All the lysimeters located in the front sections of the panels were not yet covered by waste rock at that time. Data collection from the front lysimeters commenced once waste rock placement was completed. Monitoring for the uncovered lysimeters indicated that, although the lysimeters had not yet been completely covered with the waste rock, outflows had
already started to drain from the lysimeters due to the frequent rainfall at the experimental site. Since it took several days to complete the placement of waste rock over the lysimeters, the leachate that was collected initially was considered to be mainly rain water that had previously infiltrated into the pile and was stored in the pile during the construction period.

The outflows from the lysimeters were measured using tipping buckets as described above. The number of tips for each tipping bucket was calculated based on a daily record of the tip counter that was attached to the instrument. The tipping bucket was periodically calibrated to ensure the precise volume from each tip. The amount of outflow was calculated according to the number of tips and the volume of each tip. The daily outflow data for most of the lysimeters were compiled from the end of August 2000 to May 2004 (or November 2004 for the front sections of panels 4 to 8), except for lysimeters located at the front of the panels, which commenced in May 2001, and the lysimeters that had drainage problems as explained in Section 2.3.3. The daily outflow data in mm/day was compiled and is plotted as box and whisker plots in Figures 2.10 and 2.11. The dot in the box indicates the mean value of the outflow flux from the lysimeter and 75 and 25 percentiles are shown as the upper and lower limits of each box, respectively, while the horizontal line in the middle of the box shows the median value. The ends of the vertical lines, or whiskers, indicate the minimum and maximum values of outlier data (i.e. 1.5 inter-quartile range of the lower quartile, and 1.5 inter-quartile range of the upper quartile). Figures 2.10 and 2.11 illustrate the spatial and
Figure 2.10 Spatial and temporal variation in flux outflow measured for (a) L1 lysimeters and (b) L2 lysimeters from August 2000 to November 2004
Figure 2.11 Spatial and temporal variation in flux outflow measured for (a) L3 lysimeters from August 2000 to November 2004 and (b) Panel 8 from August 2000 to December 2002.
temporal variability of outflow from the lysimeters located at the front (L1), middle (L2) and rear (L3) sections of the trial dump, respectively. For improved clarity of presentation, the plot for panel 8 is presented separately in Figure 2.11. The complete data sets for the monthly outflow measured from each lysimeter for individual panels are presented in the Appendix.

The median values for daily outflow do not always lie in the center of the box and were frequently skewed either towards the minimum or the maximum values. Since the data population was skewed, outliers appeared in the plots. Data were considered to be outliers when the values were less than 1.5 times the first quartile or when the values were greater than 1.5 times the third quartile. All of the outlier data in the daily outflow occurred in the range above the third quartile. Outliers also occurred in the rainfall data from approximately 2.0% of the data. The outlier data among the lysimeters varied from 0.02% to 14.3% and the lysimeters varied from 0.02% to 14.3% and the highest percentage occurred for the front lysimeter (L1) of panel 3.

The temporal variation of the daily rainfall data was measured, and the coefficient of variation (CV) was found to be 82%. The variability in the rainfall data caused temporal variability in the daily outflow from the lysimeters, and also resulted in outlier data among the daily outflow data. The average CV for the daily flux outflow data from the lysimeters located at the front section of the panels (L1) was 67%, which was less than the CV of the daily rainfall. The variability of the daily flux outflow for the lysimeters located at the middle section of the panels (L2) was more pronounced than
the lysimeters located at front section of the panels (L1). The average CV was 83%, which was comparable to the CV of daily rainfall. The temporal variability of daily flux outflow was most significant at the lysimeters located at the rear section (L3) of the panels with an average CV of 121% due to the significant decrease in the daily outflow from several lysimeters as will be explained in Section 2.4.6.

Spatial variability was also recorded in the daily outflow for the lysimeters. The mean daily flux outflow for individual lysimeters varied from 1.1 to 76.3 mm/day. The two lysimeters that were located at the middle (L2) and the rear section (L3) of panel 8 had high outflows almost continuously during the experiment due to the runoff that came from the adjacent dump (as explained in the next section). The mean daily flux outflows for L2 lysimeter and L3 lysimeter of panel 8 were 20.2 mm/day and 76.3 mm/day, respectively. The mean daily flux outflow for the lysimeters located at the front section (L1) of the panels varied from 6.6 mm/day to 16.3 mm/day. A lower mean daily flux outflow, which ranged from 2.1 mm/day to 9.6 mm/day, was recorded at the lysimeters located in the middle section (L2) of the panels. Except for panel 3, the mean daily flux outflow of lysimeters located at the rear (L3) was slightly lower than lysimeter L2 and it ranged between 1.1 mm/day and 12.3 mm/day.

2.4.4 Net Infiltration of Individual Lysimeters

Net infiltration is defined as the amount of water that infiltrates into the surface of the waste rock and is transported to the base to be recorded as outflow from the lysimeters. For analytical purposes, the net infiltration is calculated based on the
measured outflow from each lysimeter and is presented as a percentage or portion of the rainfall that fell on the surface of the waste rock pile. The amount of surface evaporation and surface run off were not considered in the net infiltration calculation, since no measurements were available for these parameters during the experiment. The net infiltration values presented here are for monthly and annual periods for each lysimeter.

Similar to daily flux outflow, the monthly net infiltration values showed temporal and spatial variability, as can be seen in Figures 2.12 and 2.13. The trend in the monthly net infiltration for each group of lysimeters was consistent with the trend shown in the daily flux outflow. Although the data of net infiltration from lysimeters in panel 8 are presented in the Figures 2.12, these data are not considered in the following discussion since the data may not be representative due to the extremely high value as a result of the runoff from the adjacent area and the fact that the tipping bucket and collection tank were destroyed by a mudslide at the end of 2002. The average monthly net infiltration of L1 lysimeters varied from 21% to 91%, and the CV ranged from 28% to 69%. The lowest CV value was in the L1 lysimeter of panel 7. A wider range of monthly net infiltration occurred in the L2 lysimeters. The average monthly net infiltration for L2 lysimeters was between 18% and 130%, and the CV varied from 24% to 82%. The highest CV values belonged to the L2 lysimeter in panel 1 where the outflow increased significantly in the third year. The variability in monthly net infiltration was the most pronounced in the L3 lysimeters. The average monthly net infiltration ranged from 10%
Figure 2.12 Monthly net infiltration from panels 1 to 4
Figure 2.13 Monthly net infiltration from panels 5 to 8
to 454% and the CV for each panel between ranged from 69% to 141%. The highest CV value was recorded in the L3 lysimeter for panel 7 where the outflow fluctuated significantly.

A relatively high coefficient of variation reflects the variability in the volume of outflow. The relatively high coefficient of variation in the outflow net infiltration should not be interpreted as an error or that a problem occurred during data acquisition. It simply reflects the heterogeneity of waste rock particles in the internal structure of the waste rock dumps. This resulted in the variability of the outflow volume. The variability of the outflow from a waste rock pile at Cluff Lake in Northern Saskatchewan was also reported by Nichol et al. (2005), who monitored the net infiltration from 16 lysimeters that were placed beneath the waste rock pile that was 8 m x 8 m in area and 5 m in height. The net infiltration measured by Nichol et al. (2005) for the individual lysimeters varied between 30% and 121%. In general, the variability of the outflow from the lysimeters was more pronounced for the trial dump as compared to the waste rock pile reported by Nichol et al. (2005). It is suggested that the primary reason for the difference between the two experimental sites can be attributed to climate. The precipitation at the site for the trial dump is much wetter and therefore more variable. Total mean annual precipitation at the Grasberg mine is 3,000 mm/year to 5,000 mm/year, compared to 455 mm/year for the Cluff Lake site reported by Nichol et al. (2005). Furthermore, the two waste rock piles were constructed using different methods, which may have created different internal structures, that could affect the
infiltration process. The waste rock pile described by Nichol et al. (2005) was constructed using random placement of the different particle sizes, a process in which segregation of the particles according to size is not expected to occur during dumping. In contrast, the trial dump was constructed using the end-dumping method based on truck placement. Although the height of the test dump was only 20 m, segregation by particle size was observed along the face of the dump during construction.

Figure 2.14 presents the cumulative net infiltration data for each lysimeter from the beginning of the experiment until the end life of the experiment. The trend in the cumulative net infiltration for each panel is also shown in the Appendix. The data are presented based on L1, L2 and L3 lysimeter grouping. The cumulative net infiltration for panel 8 is presented in a separate graph in Figure 2.14(d).

The heterogeneity within the waste rock pile resulted in variability of the cumulative net infiltration, which can be seen to vary between 21% and 183%. The outflows from six lysimeters exceeded the cumulative value of the rainfall, as shown in Figure 2.14. These six lysimeters were found in the front section lysimeters (L1) for panels 5, 6, and 7, and the middle section lysimeters (L2) for panel 1. In addition, extreme values were recorded at L2 and L3 lysimeters for panel 8 where the cumulative net infiltrations were approximately two and six times the cumulative rainfall, respectively. The cumulative net infiltration outflow for the three lysimeters that were located at the rear section (L3) of panels 1 and 2, and the lysimeter located in the middle (L2) of panel 7 was much lower than the cumulative net infiltration of the other
Figure 2.14 Cumulative Net Infiltration of L1 Lysimeters, L2 Lysimeters, L3 Lysimeters and panel 8 lysimeters from August 2000 to November 2004

lysimeters. These were some of the lysimeters that had problems with outlet drainage, where the outflow flux gradually decreased before it stopped completely after operating in these cases for less than six months.

Runoff that came from an area outside of the trial dump area could not be completely avoided or diverted, especially during rain storm events which occurred periodically. The runoff was observed to come from the adjacent area located on the
southwest side of the trial area, which was relatively higher in elevation than the trial
dump area. The buffer area that was built to separate the trial area from the actual
waste rock dump was not able to completely prevent surface runoff from reaching the
surface of the trial dump. However, this runoff was directed only toward a particular
area of the test dump located at the rear section of panel 8. The mud slide event that
subsequently destroyed the tipping buckets and the storage tanks at panel 8 was an
example of the problems caused by surface runoff from the adjacent area. High
volumes of leachate were recorded continuously throughout the operation of panel 8,
particularly for the lysimeters located at the middle and rear of panel. These
observations were also likely due to surface runoff from the adjacent area. The
cumulative net infiltration collected on these two lysimeters (L2 and L3) showed that
the amount of infiltrated water represented an area that ranged from two to six times
larger than the footprint of the lysimeter. In summary, the amount of leachate from just
these two lysimeters counted for between 30% and 50% of the total outflow flux from
the 24 lysimeters.

2.4.5 Response of Individual Lysimeters to Rainfall Events

The daily outflow of individual lysimeters along with the daily rainfall was
examined to investigate the behavior of each lysimeter in response to rainfall events.
The data that was collected in August 2001 was selected for the analysis presented
here. The climate data that was recorded during this month was adequate to represent
the heavy rainfall events, recorded during the first week, as well as a dry period that
began in the third week of August where there were 10 consecutive days with no rain. Analyses of the outflow data from the lysimeters for the August 2001 period were expected to identify the responses of each section of the waste rock pile to the different rainfall events.

The responses of the lysimeters to rainfall events can be categorized into three types of responses, as shown in Figure 2.15. The first type of response, quick response, (Figure 2.15(a)) as shown in L1 lysimeter for panel 3, was sensitive to and emulated rainfall events. Thus, the outflow patterns were similar to rainfall patterns, although the magnitude of the outflow could be lower or higher than the intensity of rainfall. During the period in which the rainfall fluctuated from day to day, the lysimeter response showed the appearance of distinct peaks, and the lysimeter showed a decreasing trend in outflows during the days when there was no rainfall. Although the lysimeters were sensitive to changes in rainfall events, the responses of the lysimeters to these events lagged the actual events by two to five days.

Figure 2.15(b) for L2 lysimeter of panel 5 illustrates the second type of response, medium response, shown in the lysimeters. These lysimeters were sensitive to the absence of rainfall but not to the daily fluctuations of rainfall events. The decreasing trend in their outflows was the result of the absence of rainfall. Similar to the first type of response, there were also three to five day lags in their responses to rainfall events. The response to the daily fluctuations of rainfall was not obvious, since there was a gradual increase in the outflows. The peak, as shown Figure 2.15(b), was only observed
Figure 2.15 Lysimeter response to rainfall events as (a) a quick response, (b) medium response, and (c) slow response
to occur after a consecutive series of large rainfall events occurred, and the magnitude was typically lower than the rainfall intensity. In summary, a pattern between the rainfall events and the outflow is not apparent for this type of lysimeter response.

The third type of response, slow response, is represented by L3 lysimeter in panel 6 in Figure 2.15(c). The response seen in this type of lysimeter was that they were not sensitive to changes in daily rainfall events. The daily outflow did not change significantly during the large rainfall events. Unless consecutive storm events occurred, a peak in the lysimeter outflow was not obvious, and the magnitude of the peak was much less than the peak of the storm event. The zero precipitation period was followed by gradually decreasing outflows. As a result, the patterns of the daily outflows were not similar to the patterns of rainfall events.

The type of response was strongly related to the net infiltration. The lysimeters with the first type of response, sensitive to changes in the rainfall events, for the most part had relatively high net infiltration rates and, therefore, were classified as fast-flowing lysimeters. The outflow fluxes for the fast-flowing lysimeters were greater than the average outflow for all the lysimeters. The group of fast-flowing lysimeters consisted mainly of the L1 and panel 8 lysimeters as previously discussed, however, the panel 8 lysimeters were extremely affected by runoff from the adjacent waste rock dump. In contrast, the group of lysimeters that had slow response to the variation of rainfall events had a low net infiltration. The majority of L3 lysimeters were classified as slow-flowing lysimeters. The group for the second type of lysimeters had a medium
value of net infiltration, and the group consisted mainly of L2 lysimeters and some L1 lysimeters.

Statistical analyses were used to determine the correlation between daily outflow flux and daily rainfall amounts. The results of statistical analysis showed a weak correlation between these two parameters. The coefficient of correlation ranged from 4% to 70%, and only two lysimeters had coefficient correlations greater than 50%. Faster-flowing lysimeters usually had better coefficients of correlation. Further statistical analyses showed that the coefficient of correlation could be improved when a moving average approach was used to assess the relationship between the outflow of the lysimeters and rainfall events. When using one-week and two-week moving averages, the coefficient of correlation for the front lysimeters (L1) improved to near or more than 50% (except for panels 1 and 2). However, when the same approach was used for lysimeters located in the middle (L2) and in the rear sections (L3) of the panels, a different result was obtained since only one lysimeter, L2 lysimeter for panel 5, had a coefficient of correlation near 50%. In summary, the L1 lysimeters had the strongest correlation with the rainfall events. The statistical analysis results, as well as the response of individual lysimeters, indicated that the daily outflow was not determined by a single daily rainfall event, but it was influenced by the cumulative effects of the previous rainfall events.

In a particular case, such as a period that had a fluctuation of daily rainfall with a rain storm event, the magnitude of the peak outflow flux of the fast lysimeters could be
higher than the magnitude of the rainfall peak itself. Outflow for a basal lysimeter that was higher than the magnitude of rainfall after the rain storm event was also observed by Nichol et al. (2005). The tracer test performed by Nichol et al. (2005) confirmed the high outflow response during the rain storm event and did not solely imply that the current storm drained through the preferential flow path, but it could also be attributed to the release of old water that had been stored in the pile. Pressure gradients that form in the profile under high infiltration rates expelled water that had been retained in the pile during previous rainfall event (Rasmussen et al. 2000; Williams et al. 2002; Marcoline, 2008) resulting in outflow peaks that are greater than the peak associated with the current rainfall.

2.4.6 Water Balance of the Trial Dump

Figure 2.16(a) shows the trend in monthly net infiltration from the lysimeters in the trial dump during the period August 2000 to November 2004. The net infiltration was estimated by comparing the total amount of net infiltration from the lysimeters to the total amount of rainfall presented on the area of the trial dump that was monitored by the lysimeters. The estimation of net infiltration was estimated using only the lysimeters that were observed to function properly during the experiment period. Therefore, lysimeters where the drainage stopped completely after several months such as the L3 lysimeters in panels 1 and 2, and the L2 lysimeters in panels 2 and 7 were not included in the calculation, as well as the anomalous lysimeters of panel 8 which tended to have an extremely high volume of outflow during the experiment due to
Figure 2.16 (a) Trend of monthly net infiltration and (b) trend of cumulative net infiltration in the trial dump from August 2000 to November 2004

runoff effects from the adjacent waste rock dump. In total, 17 of 24 lysimeters were included in this analysis.

The monthly net infiltration, as represented by the 17 lysimeters, varied with respect to time, as shown on Figure 2.16(a). The changes in total monthly net infiltration did not always reflect the net increase or decrease in infiltration shown in the individual lysimeters. For example, the influence of the cover treatment applied to panels 3 and 4 that reduced the outflow from lysimeters 2 and 3 in panel 4 did not
appear to reduce the net infiltration from the combined lysimeters in the trial dump. The decreasing outflow from these covered panels seemed balanced by the temporal increase of net infiltration from the other lysimeters.

Table 2.2 shows the annual net infiltration calculated for groups of the L1 lysimeters, the L2 lysimeters, and the L3 lysimeters. The data presented here is only for the period of August 2000 to December 2003 because the tipping buckets could not be accessed in 2004 as well as the modifications that were required to extend the collection pipe system in early 2004.

### Table 2.2 Yearly net infiltration of entire trial dump

<table>
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<tbody>
<tr>
<td>Rainfall, mm</td>
<td>3,670</td>
<td>3,680</td>
<td>3,420</td>
<td></td>
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<tr>
<td>Infiltration, %</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>L1 Lysimeters uncovered</td>
<td>91</td>
<td>90</td>
<td>106</td>
<td>108</td>
</tr>
<tr>
<td>L2 Lysimeters uncovered</td>
<td>89</td>
<td>66</td>
<td>62</td>
<td>63</td>
</tr>
<tr>
<td>L2 Lysimeters Covered</td>
<td>N/A</td>
<td>123</td>
<td>106</td>
<td></td>
</tr>
<tr>
<td>L3 Lysimeters uncovered</td>
<td>87</td>
<td>50</td>
<td>27</td>
<td>70</td>
</tr>
<tr>
<td>L3 Lysimeters- covered</td>
<td>N/A</td>
<td>N/A</td>
<td>43</td>
<td></td>
</tr>
</tbody>
</table>

In general, higher annual net infiltration was recorded during the first year period for most groups of lysimeters. Furthermore, the net infiltration recorded during the first year for all groups of lysimeters are similar. The group of L1 lysimeters consistently shows high net infiltration values for all periods, while the net infiltration
values from the deeper L2 and L3 lysimeter groups show a decreasing trend in the second year. The net infiltration from the uncovered L2 lysimeters was relatively stable after July 2001. The highest net infiltration value, which was higher than 100%, was recorded from the covered L2 lysimeters. This high value was primarily due to the outflow volume from the L2 lysimeter in the panel 1 that increased significantly after July 2002.

It can also be seen in Table 2.2., the annual net infiltration for the uncovered L3 lysimeters fluctuated during the time period considered. A significant decrease in outflow from several lysimeters such as the L3 lysimeter in panel 4 during the second year, and the L3 lysimeter in panel 6 during the second and third year contributed to the low net infiltration values from the L3 lysimeters. Alternately the outflow volume from the L3 lysimeter in panel 6 increased significantly during August to December 2003, causing the net infiltration value from the combined group of L3 lysimeters to increase to 70%. The net infiltration for the covered L3 lysimeter can only be evaluated for the period of August 2002 to July 2003 due to a drainage problem in the L3 lysimeter for panel 1. The outflow volume from the L3 lysimeters tended to decrease after November 2001 prior to placement of the low-permeability cover on panel 4 and this trend continued until the end of experiment.

This significant decrease in net infiltration was mostly observed for the L3 lysimeter in most of the panels, the L2 lysimeter in panels 4, 5 and 6, and L1 lysimeter for panel 1. The placement of cover treatments contributed to the decreasing of the net
infiltration in the L2 lysimeters for panel 4 and the L3 lysimeter for panels 3 and 4, although the decreasing trend from the L3 lysimeter in panel 4 was observed prior to the cover placement. For most of the other L3 lysimeters, the actual reason for the decrease in the net infiltration was not fully known and was probably related to changes in the permeability of the waste rock or the changes in the flowpath. The increase in the annual net infiltration occurred during the third year for the L1 lysimeters in the most panels and for the lysimeter 3 in panel 6 during August to December 2003. The reason for this increase was probably related to the changes in the flow path and/or redistribution of the rain water.

The contribution of monthly net infiltration to the cumulative net infiltration from the 17 lysimeters is presented in Figure 2.16(b). The cumulative net infiltration from these lysimeters tended to decrease and was assumed to represent the estimated net infiltration from the trial dump. The cumulative net infiltration in the trial dump accounted for approximately 82% of the total rainfall. These values, when expressed as a percent of total precipitation, are higher than the cumulative net infiltration for waste rock dumps in low rainfall semi-arid/arid climate as reported by Williams and Rohde (2008) where the cumulative net infiltration varied from 50% to 60%. According to water balance calculations, approximately 18% of the rainfall water can be accounted for as evaporation, surface runoff, and storage. Since no measurements for either evaporation, runoff, or changes in water storage were obtained during the experiment, the actual percentage for each parameter remains unknown. The calculated potential
evaporation at the site was known to be 1 mm/day or about 10% of the average daily rainfall. The potential evaporation is so low because the humidity is high and the temperature is low on the site of the trial dump. In addition, this location is cloudy during most of the daytime, which leads to a low net radiation. The actual evaporation was expected to be the same as potential evaporation, which is 10%. Therefore, assuming zero change in storage, the estimated runoff is approximately 8%. This computed value is similar to the values reported by Harries and Ritchie (1983) who recorded approximately 15% run-off during high intensity rainfall events for the waste rock dump at Rum Jungle in Northern Territory, Australia that had a relatively smooth surface with no vegetation. Since the surface of the trial dump was quite irregular, the run-off water from the trial dump could be less than 15%. The visual observation made during the experiment period shows that the surface run off from the flat surface as well as from the slope of the trial dump was not significant.

The water balance on the trial dump estimated a cumulative net infiltration which was 82% of the total rainfall based on reliable lysimeters. The water balance for the 500-tonnes test pad indicated that the cumulative net infiltration was approximately 90% of the total rainfall. Since the test pad had not been exposed to surface traffic during the construction, and did not have a cover treatment, the surface of the test pads can be expected to be more permeable than the trial dump. Furthermore, the construction methods used for the trial dump and the test pad were different. The trial dump was constructed by end-dumping, while the test pad was built
with random placement using an excavator. Therefore, the waste rock segregation was more pronounced for the trial dump as compared to the test pad and resulted in an internal structure that was more complex. The difference in these physical characteristics between the trial dump and 500 tonne test pad contributed to the difference in the net infiltration. Due to both of these reasons, the lower cumulative net infiltration measured for the trial dump compared to the test pad is considered to be reasonable.

In summary, the outflow data from the individual lysimeters within the trial dump can not be used to accurately predict the water balance for the entire trial dump. Ideally, the entire base of the trial dump should be lined in order to provide the best estimation of the water balance for the trial dump. However, the estimation of the water balance in the trial dump can be improved by using the summation of outflow data from all lysimeters within the trial dump. The number and combined area of the lysimeters, which represented about 6% of the total trial dump area, is considered adequate to represent the water balance in the trial dump.

2.4.7 Infiltration in the Trial Dump

The different amount of leachate collected from the lysimeters over the same period of time prior to the cover placement reflects the spatial variability within the dump. The characteristics of the waste rock in the dump, i.e., the heterogeneity of the grain sizes of the waste rock, determine the amount of rain water that infiltrates into the waste rock dump. As a result of the heterogeneity, particular areas in the waste
rock pile were more permeable than other areas. The particle size analyses for the waste rock samples collected above the lysimeters showed differences among the samples that were collected. However, when assessing the range of particle size for the waste rock samples from a particular lysimeter with the amount of leachate collected, the correlation between those two parameters was neither obvious nor consistent. For example, samples from the L2 lysimeter in panel 5 contained more cobble and gravel than the samples for the L2 lysimeter in panel 6, but the net infiltration was lower in the L2 lysimeter for panel 6 compared to L2 lysimeter in panel 5. Since there was a maximum range of grain sizes that could be collected from the waste rock, the samples might not be representative of the bulk waste rock located above the lysimeters. Therefore, the results of the grain size analyses only are not considered useful for assessing the hydraulic properties of the waste rock material above the lysimeter, since the flow may occur in ways other than just through the fine matrix. In addition, the grain size is not the only factor that imply to the flow in the waste rock pile.

Marcoline (2008) suggested that an irregular surface configuration can affect the distribution of rainfall water due to surface runoff. Grading the surface of the trial dump after construction was completed, seemed to improve the direction of runoff at the trial dump. Although the actual amount of runoff from the trial dump was not measured, surface runoff between the panels (i.e. form one panel to the next) was not visible prior to cover treatment. However, the installation of the cover treatments later in April, May and July 2002 modified the contour and the texture of the trial dump surface, which
likely resulted in redistribution of the rainwater. Since the top surfaces of the covered panels were higher than the surface of the uncovered panels, runoff from the covered areas was directed to the lower region at adjacent panels. For example, the placement of the 1 m weathered waste rock cover on the top of panel 4 reduced the surface permeability of the panel. Thus, the rainfall rates exceeded the surface permeability of panel 4, and the excess rainwater became surface runoff which was subsequently redirected to another area near panel 4 (i.e. panels 3 and 5). Although the influence of the cover placement resulted in little variation in the overall water balance for the trial dump, changes in the net infiltration for the individual lysimeters were distinguishable. The significant increase of the net infiltration from the L2 lysimeter in panels 3 and 5, as shown on Figures 2.12 and 2.13, after placement of the surface cover on panel 4 may be attributed to the redirection of the surface runoff. The influence of the limestone cover (panel 5) in redirecting surface runoff appears not to be as significant as the low-permeability cover. Although the height of the limestone cover was approximately three meters, the placement of the cover using the plug-dump or paddock dump method without further compaction (as shown on Figure 2.6) decreases runoff from the limestone cover panels.

Variability in the outflow from the lysimeters suggests that the geometry of the test dump had influenced the infiltration process. As described in section 2.4.3, the outflow fluxes for the L1 lysimeters were relatively higher than the outflow fluxes for the L2 and L3 lysimeters. The L1 lysimeters were located underneath the slope of the
dump, while the flat dump surface covered the L2 and L3 lysimeters. Corazaou Galegas (2007) and Williams and Rohde (2008) also showed similar results in that more leachate was collected from the lysimeters than were located beneath the sloped portions of the dump.

The higher amounts of leachate collected from the L1 lysimeters implied that the materials above these lysimeters were more permeable than the materials that overlaid the L2 and the L3 lysimeters. When the test dump was constructed, the rear sections of the panels, which were above the L3 lysimeters, were built first, and then these sections were used as an access road for trucks and as a tipping face while dump construction was progressing. The significant amount of truck traffic passing over the rear and the middle sections of the panels made these areas become relatively more compacted compared to the front and sloping sections of the panel. The presence of puddles or ponded water, which usually appeared at the rear section of the trial dump after rain storm events, suggested that the material in those areas had a relatively low surface permeability. Smith and Beckie (2003) stated that puddles of water occur when the intensity of rainfall events exceeds the infiltration capacity of the dump. Thus, the surface infiltration rate was slower for the L3 and L2 lysimeters compared to the L1 lysimeters.

In addition to the surface variability described above, the vertical distance between the lysimeters and the dump surface was 23% to 67% less for the sloped section compared to the flat surface section. Therefore, when rainfall occurred,
infiltration was evident earlier at the L1 lysimeters compared to the L2 and L3 lysimeters. Williams and Rohde (2008) also suggested that greater leachate volumes are collected in the lysimeters (located directly) underneath the slope because the storage capacity is reduced due to the shorter flow path.

Several lysimeters such as L1 lysimeters in panels 1 and 2, and L3 lysimeters in panels 3, 4 and 7 showed a decreasing trend in the cumulative net infiltration, as shown on Figures 2.12 and 2.13. Net infiltration values could have also been decreased as a result of the changes in the flow path within the waste rock pile due to decreases in the permeability of the waste rock. The acidic waste rock that was used for the construction of the trial dump was a highly reactive material that reached the maximum sulfate release rate between one and two years after the waste rock was exposed to oxidant (Miller et al. 2003a). Hence, the waste rock material became finer due to particle break down and loss of mass due to sulfate release, thereby reducing the hydraulic conductivity of the waste rock pile. Giere et al. (2003) suggested that the precipitation of secondary minerals as a result of the oxidation of sulfide-containing minerals could also alter the permeability of the waste rock pile.

The change in the net infiltration among lysimeters in the trial dump, as shown on Figures 2.12 and 2.13, may have also resulted from changes in water storage and run off from the waste rock pile. Changes in water storage capacity depended on the rainfall pattern and the properties of the waste rock. When the waste rock material weathers, the particle sizes may become finer, and this could change the capillary pressures or
matric suction in the waste rock material. Since no measurements of pore water or the
water content of the waste rock material were made during the experiment, changes in
the water storage capacity of the test dump could not be calculated. Marcoline (2008)
reported no change in the water storage capacity of a reactive waste rock pile that had
been exposed to a climate that was relatively drier than this trial dump area during four
years of monitoring. However, it is important to note that the climate in Northern
Saskatchewan, the experiment location of Marcoline (2008), is much colder than the
one at the Grasberg Mine.

As presented in Section 2.4.4, some variation occurred in the cumulative net
infiltration for each lysimeter. Some lysimeters such as the L1 lysimeters for panels 5, 6
and 7, the L2 lysimeters for panels 1 and 8, and the L3 lysimeter in panel 8 had
cumulative net infiltrations higher than 100% (as shown on Figure 2.14). A value over
100% indicates that the amount of leachate collected from these lysimeters was greater
than the amount of rainfall that fell on an equivalent area of the footprint of these
lysimeters. This result suggested that rainwater from outside the footprint of the
lysimeters had to be directed to the lysimeters. The rainwater could be in the form of
surface runoff, as explained in the previous section, or it could be sub-surface lateral
flow. On the other hand, the sub-lateral flow could also reduce the amount of net
infiltration in a lysimeter because it can also direct the infiltrated water towards regions
outside of the footprint of the lysimeter. As a result, the low cumulative net infiltration
recorded at some particular lysimeters, such as L3 lysimeters in panels 4, 6 and 7, may under report actual net infiltration.

Sub-surface lateral flow may occur in the waste rock pile due to the variability in grain size among the waste rock particles, as well as, the segregation that occurred during dumping of the waste rock that led to the formation of layers with different hydraulic properties. Furthermore, the capillary barrier effect may also exist due to variability in the hydraulic conductivity of the different layers in the pile, and thus may change flow paths and promote lateral flow (Fala et al. 2005; Stormont, 1995). Thus, the existence of sub-surface lateral flow likely alters the distribution of infiltrated water within the waste rock pile and the amount of water collected in the basal lysimeters may be higher or lower than the amount of rainfall received on an equivalent area of the footprint above the lysimeters.

Analysis of the outflow data for the lysimeters, excluding lysimeters with drainage problems (i.e. L3 lysimeters for panels 1 and 2, and L2 lysimeters for panels 2 and 7) and the lysimeters for panel 8, indicated that the overall outflow from the trial dump was transported mainly through preferential flow paths in the macro pores or fast-flowing lysimeters, which are described in Section 2.4.5. These fast-flowing lysimeters are the L1 lysimeters in panels 5, 6 and 7, and the L2 lysimeter in panels 1. Only 4 of the 17 lysimeters were identified as fast-flowing lysimeters. Analysis of the annual infiltration for the second year (September 2001 to August 2002) and the third year (September 2002 to August 2003) indicates the fast flowing-lysimeters contributed
approximately 44% and 59% of the total outflow, respectively. Each individual fast-flowing lysimeter contributed between 6% and 14% of the total outflow compared to only 1% to 4% for each of the slow-flowing lysimeters, as shown in Figure 2.17. In addition, the wet climate for which rainfall occurs throughout the year, plays an important role in the flow path of the infiltrated water in the trial dump. As reported by Newman (1999) and Nichol et al. (2003), water tends to flow through the coarser material under higher infiltration rates, while the majority of water flows through the fine layers when the infiltration rate is low. In summary, considering the quick response of the outflow from the lysimeters described above, together with the high rainfall rate recorded at the location of the trial dump, it is concluded that preferential flow through the macro-pore or non-matrix zone is the dominant flow mechanism in the trial dump rather than flow through the fine matrix zone.

2.4.8 Performance of Cover Treatments

The performances of different cover treatments in panels 1, 3 and 4 are presented in Figure 2.18, which shows the trends for cumulative net infiltration from the lysimeters under the covered panels. Figure 2.18 does not include the performance of the L3 lysimeter in panel 1 as it could not be assessed since the outflow for this lysimeter stopped before the placement of the HDPE cover.

Figure 2.18 shows placement of the HDPE cover did not affect to the amount of the leachate collected in the L2 lysimeter from panel 1. The cumulative net infiltration for the L2 lysimeter increased significantly from 63% in April 2002 when the cover was
Figure 2.17 Contributions of individual lysimeters to total outflow for the periods from (a) September 2001 to August 2002, and (b) from September 2002 to August 2003.
Figure 2.18 Cumulative infiltration of panels with HDPE cover (panel 1) and mixture of road mud and limestone cover (panel 3) and weathered waste rock cover (panel 4)

placed to 129% in May 2003. The cumulative net infiltration value over 100% suggested that the water from the outside footprint of this lysimeter entered as a run off or non-vertical flow (as previously described in Section 2.4.7) and then collected in the L2 lysimeter.

The cumulative net infiltration in panel 3, where a mixture of road mud and limestone was used to construct the cover, decreased by approximately 5% for the L2 lysimeter for the period of July 2002, when the cover was placed to February 2004, and 11% for the L3 lysimeter from July 2002 to May 2003.
The placement of weathered waste rock on panel 4 reduced the cumulative net infiltration for the L2 lysimeter from 54% to 39% for the period from July 2002 to February 2004, while the cumulative net infiltration at the L3 lysimeter decreased from 37% to 30% from July 2002 to May 2003.

The net infiltration for the panels that had the mixture of road mud and limestone (panel 3), or the weathered waste rock (panel 4), as the cover materials did not decrease infiltration as much in the small-scale experiment of the cover treatment, as shown in Figure 2.7 and summarized in the Appendix. The long-term performance of the small-scale experiment showed that net infiltration could be reduced up to 80%, as shown in the Appendix. The different results were most likely affected by the different designs used for the two experiments. The small-scale experiment was constructed as a single-test pad. The cover was placed on the entire test pad area including the top and the slopes of the test pad or entire footprint of the lysimeter as shown on Figure 2.7. On the other hand, the covers on panels 3 and 4 were placed only on the flat portion of the test dump as shown in Figure 2.3. Furthermore, the lysimeters covered only a selectively small region of the total base area, and therefore, sub-lateral flow and runoff between the uncovered and covered areas could not be prevented nor completely collected. In addition, the compaction level on the covers placed at the trial dump might not have been as high as that for the cover placed at the small-scale experiment due to difficulty in achieving the maximum standard proctor during construction.
The performance of the cover treatments suggested that the available cover materials at Grasberg Mine, such as the weathered waste rock and the mixture of road mud and limestone, have a negligible impact in reducing infiltration in high rainfall environments. However, their performances may not be effective when the slope remains uncovered and when the required compaction specifications are not achieved. Considering that rainfall occurs continuously throughout the year, and that the height of the slopes on the full-scale waste rock dump can reach hundreds of meters, construction of these types of covers over the full-scale dump at Grasberg Mine would be quite challenging.

2.5 Conclusions

This chapter presented the information regarding the design and construction of the trial dump at Grasberg Mine, Indonesia along with the results of monitoring from the leachate outflow from the lysimeters installed underneath the trial dump. The data obtained from the trial dump provided information regarding water-flow behavior in a large-scale waste rock pile within a high rainfall environment. The followings key points and conclusions are drawn from this experiment:

- Monitoring of the outflows from the lysimeters demonstrated that the net infiltration in the waste rock pile was temporally and spatially variable due to the heterogeneity of the physical characteristics from the trial dump. The cumulative net infiltration that was recorded from the lysimeters ranged from 20% to 163%.
• The water balance for the trial dump was calculated based on data collected from the lysimeters that were considered to be reliable over the four year monitoring period and shows that about 82% of the rainfall was collected as a net infiltration. Although the cumulative net infiltration data from each individual lysimeter cannot be used to estimate the water balance from the trial dump due to variations among the lysimeters, the estimation made by using the collective net infiltration data from the 17 working lysimeters is considered adequate to represent the water balance for the trial dump.

• Each lysimeter showed a different response to rainfall events as well as to the absence of rainfall events. Several lysimeters were found to be sensitive to rainfall events and the outflow response resembled the rainfall pattern. These lysimeters, such as L1 lysimeters in panels 5, 6, 7 and L2 lysimeter in panel 1 tended to have high outflow and thus were classified as fast-flowing lysimeters. Other medium flow lysimeters were found to be sensitive to the absence of rainfall but not to the daily fluctuations of rainfall events. The outflow patterns from these lysimeters were quite different when compared to the rainfall pattern and consisted mainly of L2 Lysimeters and some L1 Lysimeters. The outflow from the non-sensitive lysimeters were also found to be relatively low. These lysimeters were classified as slow-flowing lysimeters, were not sensitive to changes in rainfall and were mainly the L3 lysimeters.
• Preferential flow paths through macro pores were observed to occur in the trial dump through the fast-flowing lysimeters. Although these lysimeters had only 30% of the total footprint area of the lysimeters, the contribution of this preferential flow could reach as high as 59% of the total outflow. Each individual fast-flowing lysimeter contributed between 6% and 14% of the total outflow.

• The geometry of the dump affected the amount of the infiltration. In general, the net infiltration was higher for lysimeters located underneath the front slope (L1 lysimeters) of the trial dump compared to the lysimeters located at the middle (L2 lysimeter) and rear section (L3 lysimeter) of the panels. The daily outflow for the lysimeters located under the slope varied from 6.6 to 16.3 mm/day, which was higher than the daily outflow produced from lysimeters located under the flat surface, which ranged from 1.1 to 12.3 mm/day. The outflow from the lysimeters located underneath the front slope of the waste rock pile also had a stronger correlation coefficient with rainfall, a quicker response to the rainfall events, and a pattern that more closely resembled the rainfall pattern when compared to the outflows for the lysimeters placed under the flat surface of the trial dump. These observations may be attributed to the fact that the ponding condition never developed on the sloping regions (i.e. free run off), the vertical distance to these lysimeters was less and that no cover treatments were applied at the front slope.
• Several lysimeters, including L1 lysimeters in panels 5, 6 and 7, and L2 lysimeter in panel 2, recorded cumulative net infiltration values that were higher than 100% of total precipitation. This result suggests that the water originated from an area outside of the projected footprint area for the capture zone of the lysimeter, either as surface-lateral flow (runoff) or sub-surface lateral flow. Sub-surface lateral flow could also be occurring in the opposite direction, reducing the amount of net infiltration in those lysimeters with low flow. The surface conditions on the dump, such as varying compactive effort and texture control the distribution of runoff, while the heterogeneity of the grain size, segregation of the layers, capillary barrier effects and subsequent change in hydraulic conductivity within the waste rock pile promotes sub-lateral flow within the pile.

• In general the application of the low-permeability cover treatment from locally available materials was found to not be effective in reducing infiltration into the trial dump due to difficulty in achieving the maximum compaction and the inability to cover the sloped surface of the pile, although sub-lateral flow may have influenced the measured performance for the lysimeters within the cover treatments. The road mud and limestone cover, and the weathered waste rock cover were able to reduce the cumulative net infiltration, but only by 11% and 14%, respectively.
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Chapter 3 Physical and Geochemical Behavior of a Large Mine Rock Pile in a High Rainfall Environment

3.1 Introduction

Waste rock can be a major source of acid rock drainage (ARD) in mining. The amount of waste rock that must be excavated at a large open pit mine can reach hundreds of thousands of tons per day, and it is usually placed in engineered stockpiles near the pit. Waste rock piles that contain sulfide minerals produce acidic water when they are exposed to oxygen and water. The generation of ARD is of particular concern for the decommissioning of mines, since ARD is associated with acid and metal contamination of the environment.

Acid generation in a waste rock pile is the result of complex interactions between physical, geochemical, and biological processes. Physical processes include the flow of water and oxygen within the waste rock profile, while the geochemical processes consist of the oxidation of sulfide minerals, the solution/dissolution process, and the metal leaching and buffering reactions that occur when the waste rock contains carbonate and silicate minerals. Heat that is generated as a result of sulfide mineral oxidation often strongly affects the flow of oxygen in the waste rock dump (Ritchie,

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2 A version of this chapter will be submitted for publication. Andrina, J., Wilson, G.W. and Miller, S., Physical and Geochemical Behavior of a Large Mine Rock Pile in a High Rainfall Environment
2003; Lefebvre et al. 2001b). In addition, the specific microorganisms present in the dump have critical roles in accelerating the oxidation rate of the sulfide minerals.

The most common sulfide mineral found in metal deposits is pyrite. Pyrite oxidation occurs in both oxic and anoxic environments, as shown in the following reaction equations (after Singer and Stumm, 1970).

\[
\begin{align*}
\text{FeS}_2 + \frac{7}{2} \text{O}_2 + \text{H}_2\text{O} & \rightarrow \text{Fe}^{+2} + 2 \text{SO}_4^{2-} + 2 \text{H}^+ ; \text{ heat } = 1440 \text{ kJ/mol} \quad (3.1) \\
\text{Fe}^{+2} + \frac{1}{4} \text{O}_2 + \text{H}^+ & \rightarrow \text{Fe}^{+3} + \frac{1}{2} \text{H}_2\text{O} ; \text{ heat } = 102 \text{ kJ/mol} \quad (3.2) \\
\text{FeS}_2 + 14 \text{Fe}^{+3} + 8 \text{H}_2\text{O} & \rightarrow 15 \text{Fe}^{+2} + 2 \text{SO}_4^{2-} + 16 \text{H}^+ ; \text{ heat } = 11 \text{ kJ/mol} \quad (3.3) \\
\text{Fe}^{+3} + 3 \text{H}_2\text{O} & \rightarrow \text{Fe(OH)}_3 + 3 \text{H}^+ \quad (3.4)
\end{align*}
\]

Pyrite initially reacts with oxygen to produce dissolved ferrous, sulfate, and hydrogen ions. Unless neutralizing materials are available in the waste rock host, the pore waters becomes acidic. Ferrous ions generated in Equation 3.1 are oxidized to ferric ions in Equation 3.2 and, at low pH, microorganisms such as *T. ferrooxidans* control and accelerate this oxidation. The resulting ferric ions subsequently act as an oxidizing agent and react with pyrite to produce more ferrous ions as shown in Equation 3.3. As long as ferric ions are produced, the oxidation of pyrite will continue to take place.

Small-scale kinetic tests, such as a humidity cell or a free drainage column, are commonly used to investigate the rate of release of acid and metals from the waste rock. Kinetic testing usually involves a 1 kg to 2 kg waste rock sample that is exposed to dry and wet cycles that accelerate the weathering process. Various testing methods for a humidity cell (ASTM D5744-96, 2000) and also for the free drainage leach column
(IWRI and EGI, 2002) have been developed and documented over the past few decades. Despite the useful information obtained from the small kinetic test, the data produced from these methods do not directly represent the weathering rate that occurs at the field scale. For example, in many cases, the weathering rate at the field scale can be several orders of magnitude less than the weathering rate obtained from the small-scale kinetic test, as reported by Wagner et al. (2006), Frostad et al. (2005), Ritchie (2003), Malström et al. (2000), and Stromberg and Banwart (1999).

Frostad et al. (2005) evaluated several different testing protocols for the small kinetic cell, and the results indicated that the discrepancy between the effluent quality from a small kinetic test and full scale waste rock dumps was mostly due to the hydrological assumptions that were made. The physical properties of the waste rock pile, such as grain size distribution and the internal structure of the waste rock pile, control the distribution of water flow and air flow within the waste rock dump. These properties often cannot be duplicated in the small kinetic test. Furthermore, external factors, such as climatic conditions at the site that have specific temperatures, precipitation rates, and precipitation patterns, are not taken into account in small scale test protocols, and this results in different outcomes between the small kinetic test and the full scale dump (Sapsford et al. 2008; Younger et al. 2002).

Malström et al. (2000) proposed an empirical model to extrapolate weathering rates obtained from the small scale kinetic test to a large-scale waste rock dump, based on the physical and chemical characteristics of the mine, such as pH, temperature,
mineral content, particle size, and hydrological factors. The complexity and physical scale of a mine site must be well understood before this empirical model can be applied. The proposed model was verified with the well-characterized waste rock from the Aitik mine in Sweden (Stromberg and Banwart, 1999) and showed some agreement between the model and the field results. However, the applicability of this empirical model to the waste rock from other mines is still unknown.

The importance of hydrological, physical, and geochemical factors in predicting effluent quality for waste rock piles has often led to field investigations for a number of full-scale waste rock dumps in the last few decades. The initial studies for large-scale dumps were focused on existing, large waste rock dumps such as the studies at Doyon Mine in Quebec, Canada (Sracek et al. 2004; Choquette et al. 2001; Lefebvre et al. 2001a), Whistle Mine in Ontario, Canada (De Vos et al. 2001), Aitik Mine in Sweden (Eriksson et al. 1997; Stromberg and Banwart, 1994), Rum Jungle Mine in Australia (Ritchie and Bennett, 2003), Goldstrike Mine in US (Martin et al. 2005), and Laronde Mine in Quebec, Canada (Poisson et al. 2009). Instrumentation including piezometers, lysimeters, thermistors, and oxygen probes were installed to collect hydrological information, effluent data, temperatures, and oxygen concentrations within the waste rock pile. In addition, core drilling was performed and the core samples were used to obtain the geochemical characteristics of the waste rock.

Large-scale field trials have been a useful approach to investigate the physical and chemical characteristics of waste rock dumps and to improve our understanding of
the kinetic evolution of waste rock. The field studies mentioned above include investigations for leaching characteristics along with the interactions between the physical and geochemical processes that occur in waste rock piles. However, there are difficulties associated with the installation of instrumentation into existing dumps. Many of these problems can be eliminated through the installation of the sensors into fresh waste rock pile. This approach makes it possible to collect the information from the beginning of the weathering process in order to study the evolution of the leachate chemistry from the waste rock pile.

The following points provide recent examples and dimensions for large-scale field experiments along with the specific objectives for each experiment:

- A single waste rock pile 5 m high, 8 m long, and 8 m wide at Cluff Lake Mine in Saskatchewan, Canada to study the hydrological and geochemical behaviors of the pile (Nichol et al. 2005).

- Two piles 15 m high, 60 m long, and 50 m wide at Diavik Diamond Mine in the Northwest Territories, Canada to investigate the behavior of the waste rock pile in a permafrost region (Blowes et al. 2006), and a third pile 15 m high, 125 m long, and 80 m wide to investigate the performance of a till cover beneath a 3 m cap of non-reactive waste.

- Five waste rock piles each of which were 10 m high, 36 m long, and 36 m wide at the Antamina Mine in Peru, constructed to study the behavior of neutral drainage (Corazaou Galegas, 2007).
The large-scale field experiment described here was conducted at the Grasberg Mine and began in 1999. The aim of the experiment was to understand the kinetic behavior of the waste rock in the full scale dump and to observe the physical and geochemical characteristics of the acidic waste rock dump in the high rainfall environment at Grasberg Mine. This field experiment was part of the scale-up program for previous, smaller-scale ARD tests that involved 2-kg and 35-kg free drainage leach columns together with the 500-tonne test pads (Miller et al. 2003a). The large scale field experiment termed ‘trial dump’ was equipped with lysimeters, thermistors, and oxygen probes so that the quantity and the quality of the leachate, the internal temperatures, and the oxygen levels within the dump could be monitored.

This paper summarizes the data collected from the trial dump over a four-year period specifically related to leachate chemistry, internal dump temperatures, and oxygen concentrations. The sulfide weathering rate and associated release rate of the dissolved metals of the trial dump are compared to the rates measured in the 500-tonne test pads as well as laboratory scale leach columns. The relationships between leachate chemistry, temperatures, and oxygen levels are further assessed to help provide insight into the physical and chemical characteristics in the full-scale waste rock dump at Grasberg Mine. Finally, the influence of the low-permeability cover to the weathering process is discussed.
3.2 Site Background

The trial dump was constructed at the Grasberg Mine located in the Papua Province in the eastern part of Indonesia. The mine is in an equatorial mountainous area, which is approximately 3,500 m to 4,300 m above sea level. The Grasberg Mine has a tropical sub-alpine/alpine climate with no significant seasonal changes. The mean annual temperature is 7°C. The annual rainfall generally ranges between 3,000 mm and 5,000 mm, and precipitation occurs throughout the year.

The Grasberg Mine is a porphyry Cu/Au deposit hosted within limestone. Copper is primarily in the form of chalcopryte (CuFeS₂) with some bornite (Cu₃FeS₄), chalcocite (Cu₂S), and covellite (CuS) (Miller et al. 2003a). The Grasberg Mine will produce approximately 2,750 million tons of waste rock by the end of the year 2015 (Rusdinar, 2006). The waste rock has been placed in stockpiles adjacent to the mine pit, using truck dumping and crusher/stacker methods. About 63% of the waste rock is potentially acid forming (PAF) material and the remaining portion is comprised of limestone. The PAF material is further classified according to its Net Acid Generation (NAG) value. The lower capacity-potentially-acid forming (PAF) waste rock, which is coded as blue waste, has a NAG value less than 35 kg H₂SO₄/ton waste, and contains less than 5% pyrite. The higher capacity potentially-acid forming (PAF) material, or red waste, has a NAG value greater than 35 kg H₂SO₄/ton waste, and it contains approximately 5% to 10% pyrite.
3.3 Materials and Methodology

The methods used in this paper cover field programs and laboratory tests. The field programs consisted of the construction of the trial dump, the installing of instrumentation and the collecting of leachate samples from the basal lysimeters that were installed underneath the trial dump. The monitoring of temperature and oxygen was also included in the field programs. The laboratory program covered the geochemical tests for waste rock samples that were collected during the construction of the trial dump and also the chemical analysis of leachate samples for constituents related to ARD such as acidity, sulfate and dissolved metals.

3.3.1. Design and Construction of the Trial Dump

Construction of a waste rock trial dump within the West Grasberg dumping area began in 1999. A single 480 m long, 60 m wide, and 20 m deep waste rock dump was constructed. The dump was divided into eight 60 m by 80 m panels. Each panel was subsequently divided into three sections, each to accommodate a selected type of waste rock. Twenty four (24) lysimeters were installed at the base of each section (i.e. 8 panels X 3 lysimeters) to collect the seepage from the waste rock in the pile. The detailed design and construction of the lysimeters and the collection system were previously described in Chapter 2 of this thesis.

Approximately 1.2 million tonnes of waste rock was used to construct the trial dump. A truck dispatch system was adopted to obtain the appropriate waste rock type directly from the open pit. The waste rock was placed using a haul truck for the
installation of each panel, except for the front sloping face of panels 7 and 8, where a stacker was used. Two different types of acid waste rock were placed on the rear and middle sections of the panels. Red waste was placed on the rear sections of panels 1 to 6, while the rear section of panels 7 and 8 had blue waste. Alternatively, the middle of panels 1 through 6 and the middle panels of 7 and 8 had blue waste and red waste, respectively. Figure 3.1 illustrates the composition of each panel. The different arrangement of waste rock between those panels allowed for the examination of acid and contaminant leaching when the oxygen might be restricted. The front sloping face of panels 1 to 3 had blue waste, while the front sloping face of panels 4 to 8 had combinations of acid rock and limestone to assess the effectiveness of limestone blending for neutralization of acid drainage. The method used to blend the limestone and the acid rock for each panel is described in Chapter 4 of this thesis.

The placement of waste rock progressed from north to south, and truck dumping was used to place the waste rock for all panels, except for the front of panels 7 and 8, where a stacker was used. The end-dumping method was deployed with truck placement. The construction of the rear and middle sections was completed by February 2001, while the construction of the front sections of panels 1 to 3 was finished in May 2001.

3.3.2 Instrumentation

Thermistors and gas sampling ports were installed to measure both temperature and oxygen concentrations within the interior of the trial dump. The instrumentation
Figure 3.1 Arrangement of waste rock types at each panel

- RW = red waste
- BW = blue waste
- BW + LS = blue waste and limestone
was installed in the advancing tip face of selected panels as construction proceeded. A 500 mm deep trench was excavated down the freshly deposited tip face for instrumentation placement. After the instrumentation had been put in place, the trench was backfilled, using the excavated material. Figure 3.2 shows a typical arrangement of the instrumentation of each panel. Thermistor strings with sensors at 10 m intervals were placed along the base of all the panels, which were located approximately 70 m, 60 m, 40 m, 30 m, 15 m, and 5 m from the final location of the toe for each panel. High temperatures were recorded at the base of the panels for the trial dump early in the experiment, thus indicating the oxygen advection might be present within the pile. Therefore, thermistor strings were also installed down the slope of the dump at the middle and the front sections of the selected panels. Thermistor strings were installed at 4-m intervals (from 8 m to 37 m of the dump toe) down the slope. The thermistor had a range temperature measurement from -80 °C to 75 °C with the accuracy 0.01 °C.

At the beginning of construction, the air sampling ports (10-mm ID HDPE pipe) were placed at the base of the dump, next to the basal thermistors. A 38-mm ID HDPE pipe was also installed near the center of the lysimeter and used as a port for measurements of both oxygen and barometric pressure. A newly designed pore-air sampling system, as shown in Figure 3.3, was installed down the face of the waste rock dump in which the sampling size 3.2-mm ID nylon tubes were placed inside 38-mm ID HDPE pipes to avoid being crushed by large rocks. Once the tubes were installed, the
Figure 3.2 Location of instrumentation at the trial dump
pipe was filled with a mixture of cement and bentonite to further increase the strength of the sampling tube. The oxygen sampling ports were positioned on 3-m or 4-m centers, allowing each one to be placed at the same elevation in the dump as the thermistors. Photographs of thermistor and oxygen probes are shown in Figure 3.3.

![Photographs of thermistor and oxygen probes](image)

**Figure 3.3** Instrumentation at the trial dump including thermistors (top left) and (top right), oxygen tubes (bottom left), and oxygen placement at the face of the trial dump

3.3.3 Sampling and Monitoring

During construction of the trial dump, waste-rock samples were collected from various locations within the test panels as the over dumping on the lysimeters was progressing. The waste-rock samples were analyzed for geochemical characteristics
following the Acid-Base Accounting (ABA) method, for determining sulfur and Acid Neutralizing Capacity (ANC), and also the Net Acid Generation (NAG) test. Both ANC and NAG tests followed the methods described by IWRI and EGI (2002). In addition, copper content was analyzed.

Leachate sampling was commenced in October 2000, while construction was in progress. The leachate samples were obtained every two weeks. The sampling was changed to a monthly basis starting June 2001. Leachate temperatures, pH values, and electrical conductivities (EC) were measured in the field. Analyses for total suspended solids (TSS), alkalinity, acidity, total dissolved solids (TDS), major cations and anions, and dissolved metals were conducted in the laboratory. Regular monitoring for oxygen concentrations and temperature was carried out for the duration of the trials. Temperatures were measured using an Ohmmeter (Fluke multimeter) with accuracy 0.025%, and pore gases were measured using a portable O₂/CO₂ meter (Nova 309 WP) with accuracy 2%.

3.3.4 Low-permeability Covers and Limestone Cover Placement

The placement of the different types of covers as a treatment to reduce ARD began in April 2002. A relatively low-permeability cover system, including an HDPE liner, a mixture of road mud and limestone, and a weathering acid waste rock were placed on the top horizontal surfaces of panels 1, 3, and 4, respectively. The details for placement of the covers were described in Section 2.3.2 in Chapter 2 of this thesis. A 3-m limestone cover was placed on the flat surface of panels 5 and 6 to evaluate potential
ARD generation shutdown through armoring mechanisms previously observed on one of the limestone-capped, 500-tonne test pads (Miller et al. 2003b). The ROM limestone was placed as a free-dump cover, and no compaction or other surface treatment was applied. Panels 2, 7, and 8 had no cover treatment.

Excavation was conducted on the surface of panels 1, 3, and 4 in March 2004 to observe oxidation products beneath the low-permeability covers. A 1.5 m-deep trench was excavated using a bull dozer. The selected zones were visually inspected, and the paste pH of the waste rock within the excavation was measured directly in the field by selecting relatively fine grained materials that were mixed with distilled water at a ratio solid to water of 1:2 (w/w) and then stirred thoroughly. Samples were also collected for geochemical analysis.

3.4. Results and Discussion

This section provides the results obtained from the laboratory test and the field monitoring. The laboratory tests involved a geochemical analysis to characterize the waste rock samples that were collected during the construction of the trial dump. Results of the laboratory tests and the release rates for key constituents for ARD and metal leaching such as acidity, sulfate and dissolved copper in the leachate are presented in the following sections. The field monitoring on temperature and oxygen concentrations in the interior of the trial dump are also covered. This section includes the data from selected thermistors, oxygen ports, and key constituents of leachate. The complete temperature and oxygen data for all thermistors and oxygen ports, and
chemistry of leachates are presented in the Appendix. The results of leachate quality and field measurement on temperature and oxygen from the blended limestone and blue waste panels (front section of panels 4 to 8) are not included in this chapter. They are presented separately in Chapter 4.

The quantity and quality of seepage captured from the lysimeters is examined in this section, along with temperature and oxygen measurements. This is to investigate potential correlations and improve the understanding of ARD kinetics and oxidation mechanisms at the full-scale waste rock dump. The quality of leachate obtained from the trial dump is compared to the quality of leachate collected from the leach columns and 500-tonne test pad in order to assess the release rates of oxidation products from the different scales of the waste rock piles.

3.4.1 Material Characterization

The waste rock samples that were collected during the construction were characterized for geochemical properties, including Net Acid Generation (NAG), Acid Neutralization Capacity (ANC), total sulfur, and copper content. Table 3.1 presents the average values for the geochemistry of the blue and the red waste rock in the trial dump and the test pad. For the trial dump, the average value of sulfur content for the blue waste samples was 2% with an average NAG value of 35 kg H₂SO₄/ton. The average copper content was 0.162%. The average sulfur content of the red waste was around 4.8%. The average NAG value of the red waste samples was 107 kg H₂SO₄/ton while the copper content in the red waste samples was approximately 0.025%. Average ANC
values of both waste rock types were low, and indicated that the blue and red waste had low carbonate minerals. The waste rock samples collected during the construction of the trial dump were obtained from different locations of the trial dump and showed geochemical heterogeneity. For example, the coefficients of variation for the sulfur content and NAG were 62% and 80% for the blue waste samples, respectively, and 32% and 35% for the red waste samples, respectively.

Table 3.1 Average Geochemistry of waste rock in the trial dump and test pad

<table>
<thead>
<tr>
<th>Waste type</th>
<th>Parameter</th>
<th>Trial Dump</th>
<th>Test Pad</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blue waste</td>
<td>% Sulfur</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>NAG, kg H₂SO₄/ton</td>
<td>35</td>
<td>27</td>
</tr>
<tr>
<td></td>
<td>ANC, kg H₂SO₄/ton</td>
<td>20</td>
<td>no data</td>
</tr>
<tr>
<td></td>
<td>% Copper</td>
<td>0.162</td>
<td>0.45</td>
</tr>
<tr>
<td>Red waste</td>
<td>% Sulfur</td>
<td>4.8</td>
<td>4.7</td>
</tr>
<tr>
<td></td>
<td>NAG, kg H₂SO₄/ton</td>
<td>107</td>
<td>95</td>
</tr>
<tr>
<td></td>
<td>ANC, kg H₂SO₄/ton</td>
<td>13</td>
<td>no data</td>
</tr>
<tr>
<td></td>
<td>% Copper</td>
<td>0.025</td>
<td>0.07</td>
</tr>
</tbody>
</table>

AMIRA (2000) reported mineralogy test results from typical red waste and blue waste samples indicated that pyrite was a sulfide mineral in the red waste, while the blue waste contained chalcopyrite as well as pyrite. Gangue minerals in the red and blue waste samples were quartz and various micas such as muscovite, biotite, sericite and phlogopite.
Table 3.1 also shows the characteristics of the blue and red waste samples collected from the test pad constructed in 1996. These results were published by Miller et al. (2003a). The average sulfur content for both waste materials were similar to the average sulfur content of the samples of the wastes obtained during construction of the test pads. The copper content in the samples that were collected from the trial dump was less when compared to those at the test pad. This could be attributed to the change in the cut-off grade, which was lower during the time that the trial dump was being constructed. The NAG values for the samples from the trial dump were also similar to the NAG values for the test pad samples. In summary, except for copper content, the geochemical characteristics of the samples of the blue waste and red waste obtained during construction of the trial dump were similar to the characteristics for the samples obtained from the 500 tonne test pads.

Although quality control was implemented during dump construction, the geochemical results for the individual waste rock samples indicated a discrepancy between the geochemistry of waste rock that was actually dumped and the geochemistry of typical waste rock that was intended to be dumped. For example, two of the nine samples collected from the front section of panel 1 and three of the nine samples collected from the middle section of panel 2, which were supposed to be the blue waste, had different geochemical characteristics from those typical of the blue waste. These samples had a NAG value approaching zero with a sulfur content that was less than 1% and an average ANC value of approximately 79 kg H₂SO₄/ton. These
particular samples did not agree with the classification of the blue waste, which has positive NAG value less than 35 kg H\textsubscript{2}SO\textsubscript{4}/ton and low ANC value of 20. Although these particular samples showed different characteristics from the typical blue waste, overall the total sample set collected from the front section of panel 1 and the middle section of panel 2 still had characteristics typical of blue waste, with average NAG values of around 35 kg H\textsubscript{2}SO\textsubscript{4}/ton and 11 kg H\textsubscript{2}SO\textsubscript{4}/ton, respectively.

Further geochemical classifications of the samples are presented in Figure 3.4, which shows the plot of NAGpH versus Net Acid Producing Potential (NAPP) based on the work of Miller et al. (1997). The plot is divided into quadrants representing samples that were Non-Acid Forming (NAF) materials, Potentially Acid Forming (PAF) materials, Uncertain (UC) materials with low NAPP values, and Uncertain (UC) materials with high NAPP values. A sample is classified as NAF material when the NAPP of the sample is negative and its NAGpH is greater than or equal to 4.5. On the other hand, a sample is classified as PAF material when the NAPP of the sample is positive and its NAGpH is less than 4.5. Samples are classified as uncertain when there is a disagreement between the value of NAPP and NAGpH such as when the NAPP is negative and NAGpH is less than 4.5 or the NAPP is positive and NAGpH is equal to or greater than 4.5.

Three samples of blue waste were classified as NAF material, and five samples of blue waste were classified as UC materials with low NAPP value. The UC material samples were collected from the front section of panel 1 and the middle section of panel 2, which as previously explained, had low, near-zero NAG values and higher ANC.
Figure 3.4 Geochemical plots of waste rock samples for (a) blue waste and (b) red waste
values. The plots also show that not all blue waste samples had low PAF capacities and that not all red waste samples had high PAF capacities. These results support geochemical heterogeneity in the samples and were expected since the trial dump was built using run of mine (ROM) rock.

3.4.2 Leachate Quality

In this section, results for leachate chemistry are grouped according to type of waste (blue or red) and location of the lysimeter in each panel (L1 for front section, L2 for middle section, and L3 for rear section). The centers of L1, L2 and L3 lysimeters were located approximately 15 m, 35 m, and 65 m from the toe of the dump, respectively. Although many major cations, anions, and dissolved metals were analyzed from the collected leachate samples, this section presents only the key parameters that were considered to represent the oxidation process in the waste rock. These include pH, acidity, sulfate, and copper. These parameters are presented as a concentration and also as a release rate. The results from the smaller scale experiment, 500-tonne test pads and leach columns, are also presented in the graphs for comparison. This section does not include the quality of leachate collected from the front sections of panels that had a blending of the blue waste with limestone (panels 4 to 8) since these results are presented and discussed separately in Chapter 4.

The results presented here exclude the quality of leachate from several lysimeters that encountered a drainage outlet problem and prevented drainage of the leachate. These lysimeters include the L3 lysimeter for panel 1, which stopped flowing
in November 2000, the L3 lysimeter for panel 2 and L2 lysimeter for panel 7 which stopped flowing in February 2001, and the L2 lysimeter for panel 2 which stopped flowing in May 2001. These four lysimeters had operated only for a short time, ranging from one month to eight months. Also the quality of leachate from panel 8 is not presented since the leachate of this panel was likely mixed with water that came from another area, as explained in Chapter 2. Leachate quality for the red waste is presented solely from the red waste located at the rear of the panels (i.e. L3 lysimeters).

3.4.2.1 pH

Figure 3.5 shows the pH trend in leachate from the blue waste and red waste in the trial dump. The first sampling of the leachate from lysimeters located at the middle (L2) and the rear (L3) sections of panels 1 to 8 was conducted in October 2000. At that time, the waste rock had already been in place for approximately 7 months. Meanwhile, the first sampling of the leachate collected from the lysimeters located at the front section (L1) of the panels 1 to 3 was carried out in May 2001 or one month after the waste rock had been placed in these sections.

The leachate from all of the lysimeters had already turned acidic except for the leachate from the L1 lysimeter in panel 1. The first measurement of the red waste pH ranged from 1.9 to 2.6. The initial pH measurement of blue waste leachates was comparable with the red waste, and varied between from 1.9 and 2.9, except for the leachate obtained from the L1 lysimeter in panel 1 which was measured to be at 6.2. The pH from the L1 lysimeter in panel 1 decreased to 2.3 after three months. Samples
Figure 3.5 pH trends in leachate from (a) blue waste and (b) red waste

for the waste rock obtained above the L1 lysimeter in panel 1 had relatively low NAG values as well as significant ANC values, as described in Section 3.4.1, and thus likely contributed to the lag observed in the first pH measurements of the leachate. The acid initially produced from this waste rock was neutralized by the ANC contained in the same materials. Once the ANC was depleted, low pH values started to appear in the leachate as acid continued to be generated from the waste rock material.

As the monitoring continued, the pH decreased gradually, and it reached the minimum value between six and twelve months. The pH continued at approximately the
minimum value for several months, and then tended to increase gradually, as shown in Figure 3.5. At the end of the experiment, when the waste rock had been exposed for three to four years, the leachate pH of the red waste varied between 2.9 and 3.8, and the pH for leachate of blue waste ranged from 2.2 to 3.3. A relatively low pH for the blue waste was measured in the leachates from the front sections of panels 1 and 2. The exposure time of blue waste located at the front panels was relatively shorter as compared to the blue waste and the red waste located in the middle and rear sections of the panels. Thus, as a part of pH evolution, the increasing pH in the leachate for blue waste located in the front sections of panels 1 and 2 was not as high as the increasing pH of leachates from the blue and red waste located in the middle and rear sections of other panels.

3.4.2.2. Concentration of Acidity, Sulfate and Copper

Figures 3.6, 3.7 and 3.8 show the trends in acidity, sulfate and copper concentration for the blue waste located in the front, middle, and rear sections of the trial dump panels.

The maximum acidity and sulfate concentration in the leachate from blue waste located in the front sections (i.e. L1 lysimeters) of panels 1 to 3 ranged between 48,000 mg CaCO₃/l and 80,000 mg CaCO₃/l and between 48,000 mg CaCO₃/l and 98,800 mg CaCO₃/l, as shown in Figures 3.6(a) and 3.7(a), respectively. The maximum copper concentration for the blue waste from the same location varied between 4,010 mg/l to 6,540 mg/l, as shown in Figure 3.8(a). The maximum acidity, sulfate and copper
Figure 3.6 Acidity trends in leachates from blue waste located at (a) front sections (i.e. L1 lysimeters) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeters) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeters) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
Figure 3.7 Sulfate concentration trends in leachates from blue waste located at (a) front sections (i.e. L1 lysimeters) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeters) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeters) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
Figure 3.8 Copper concentration trends in leachates from blue waste located at (a) front sections (i.e. L1 lysimeters) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeters) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeters) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
concentration occurred approximately between seven months to twenty months after placement of the waste rock.

The acidity, sulfate and copper concentration in the leachate from the blue waste located at middle sections (i.e. L2 lysimeters) of panel 1 and panels 3 to 6 are shown in Figures 3.6(b), 3.7(b) and 3.8(b), respectively. The maximum acidity varied between 46,000 mg CaCO$_3$/l to 59,300 mg CaCO$_3$/l, and the maximum sulfate concentration ranged between 26,537 mg/l to 96,200 mg/l. The maximum copper in the leachate from the blue waste at the same location ranged between 1,450 mg/l to 7,260 mg/l. The maximum acidity, sulfate and copper concentrations were observed between ten months and twenty-six months after placement of the waste rock. As can be seen in Figures 3.6(b), 3.7(b), and 3.8(b), the concentrations of these three constituents in the leachate of the L2 lysimeter of panel 1 were much higher as compared to the concentrations in the leachate from the other panels. The concentrations of these constituents in the leachate of the L2 lysimeter of panel 1 started to increase steeply after one year of waste rock placement. At the same time, the volume of leachate outflow from this lysimeter also started to increase due to the sub-lateral flow from the area outside of the lysimeter as presented and explained in Chapter 2 of this thesis. Thus, the steep increase in acidity, sulfate and copper concentration might be related to the increase in the volume of leachate outflow. In addition, the high NAPP value (approximately 229 kg H$_2$SO$_4$/ton) that was detected in one sample collected from the L2 lysimeter of panel 1, as shown on Figures 3.4(a), may
have also contributed to the high acidity, sulfate and copper concentrations in the leachate.

The trend in the acidity, sulfate and copper concentrations from the blue waste located at the rear section (i.e. L3 lysimeter) of panel 7 are shown on Figures 3.6(c), 3.7(c) and 3.8(c), respectively. The maximum acidity and sulfate concentration were 37,180 mg CaCO₃/l and 41,330 mg/l, respectively. The maximum copper concentration was 568 mg/l. The maximum release rates for these constituents occurred approximately fourteen months after placement of the waste rock.

In general, acidity, sulfate and copper concentrations in the leachate for the blue waste from the front sections of the panel were higher than the concentration of these constituents in leachates of blue waste located at the middle and rear sections of the panels.

A similar concentration evolution was observed for acidity, sulfate and copper concentrations in the leachate collected from the red waste in the trial dump. The maximum concentration for acidity and sulfate occurred within one or two years after the waste rock placement, as shown in Figures 3.9, 3.10 and 3.11, respectively. Since the data from L2 lysimeter in panels 7 and 8 were not available, the concentration in the leachate from the different sections of these panels could not be compared. Although the sulfur and NAG were higher in the red waste than in the blue waste, there was no significant difference in the maximum acidity and sulfate concentration that were detected in the leachate. The maximum acidity and sulfate concentration in the
Figure 3.9 Acidity trends in leachates from red waste located at (a) rear sections (i.e. L3 lysimeter) of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns

Figure 3.10 Sulfate concentration trends in leachates from red waste located (a) at rear sections (i.e. L3 lysimeter) of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns
Figure 3.11 Copper concentration trends in leachates from red waste (a) at rear sections (i.e. L3 lysimeter) of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns

Leachate for red waste located at the rear zones (i.e. lysimeter 3) of panels 3 to 6 varied from 24,000 mg CaCO₃/l to 90,830 mg CaCO₃/l and from 19,450 mg/l to 101,600 mg/l, respectively, which occurred between eight months to twenty-one months after placement of the waste rock. The peak copper concentration for the red waste was much less than the peak concentration of the blue waste because the copper content was lower in the red waste than in the blue waste (see Table 3.1).
Although fluctuations in the concentration of leachate was observed for a short time period due to variations in rainfall intensity, the evolution of key parameters such as acidity, sulfate, and dissolved metals in the leachates from the trial dump had similar characteristics. The leachates were characterized by fast initial kinetics, attainment of maximum value within a relatively short period of time, and then rapid decrease followed, in most cases, by a slow trending downwards.

The concentration trends in the leachate from the trial dump were consistent with the results observed for the smaller scale test program using the same waste rock. These tests involved 2 kg and 35 kg leach column tests and a 500-tonne test pad reported by Miller et al. (2003a). The results from the test pads for both the blue and red waste can be seen in Figures 3.6 to 3.8(d) and 3.9 to 3.11(b). In general, the observed concentrations tended to be lower for the leachate from the test pads and leach columns when compared to the concentration of constituents in the leachate from the trial dump. This can be considered reasonable since the mass of waste rock involved in the trial dump was greater compared to the small scale experiments. Furthermore, the flow path was generally longer in the trial dump (i.e. 20 m) compared to the test pad (i.e. 3 m), thus infiltrating water has more time to dissolve more constituents resulting in a higher concentration of leachate since there was no (or low) solubility constraints. The low pH in the leachate along with the continuous exposure of the trial dump to rainfall prevented the precipitation of the oxidation products. Thus, the oxidation products that were flushed continuously in the drainage system led to
almost no solubility constraints in the leachate for the key constituents such as copper and sulfate.

3.4.3 Estimation of Average Release Rates of Acidity, Sulfate and Copper

The release rate of constituents in the leachate outflow was estimated according to the concentration and the volume of leachate, and then divided by the waste rock mass. The release rates for the leachate from the trial dump were estimated by assuming that the leachate was generated only by the waste rock mass that was directly above the area of the lysimeters. These values are presented in gram/ton per day.

Figures 3.12, 3.13 and 3.14 show acidity, sulfate and copper release rate trends from the blue waste located at the front, middle and rear sections of the panels at the trial dump. The maximum release rates of acidity and sulfate for the blue waste located at the front sections (i.e. L1 lysimeter) of panels 1 to 3 ranged from 20.3 g CaCO₃/t per day to 26.7 g CaCO₃/t per day and from 22.2 g/t per day to 32.3 g/t per day, respectively, as shown in Figures 3.12(a) and 3.13(a). The maximum copper release rate for the front sections of the same panel varied between 1.69 g/t per day and 2.95 g/t per day, as shown in Figures 3.14(a). The maximum release rate for all three constituents occurred approximately nine months after placement of the waste rock material.
Figure 3.12 Acidity release rate trends from blue waste located at (a) front sections (i.e. L1 lysimeter) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeter) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeter) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
Figure 3.13 Sulfate release rate trends from blue waste located at (a) front sections (i.e. L1 lysimeter) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeter) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeter) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
Figure 3.14 Copper release rate trends from blue waste located at (a) front sections (i.e. L1 lysimeter) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeter) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeter) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
The release rates of acidity, sulfate and copper from the blue waste located in the middle sections (i.e. L2 lysimeter) of panels 1, and panels 3 to 6 are presented in Figures 3.12 (b), 3.13 (b) and 3.14(b), respectively. As can be seen from these figures, the release rate for each constituent from the L2 lysimeters of panel 1 was much higher than the release rates from the L2 lysimeters of other panels. The acidity and sulfate release rates from panels 3 to 6 varied between 5.3 g CaCO$_3$/t per day to 19.8 g CaCO$_3$/t per day and between 6.4 g/t per day and 13.7 g/t per day, respectively. On the other hand, the acidity and sulfate release rates from the L2 lysimeter of panel 1 were 45.6 g CaCO$_3$/t per day and 68.5 g/t per day, respectively. A similar trend was also shown by the copper release rate, where the highest release rate recorded from the L2 lysimeter of panel 1 was 5.92 g/t per day as compared to 0.3 g/t per day to 0.8 g/t per day for lysimeter L2 of panels 3 to 6. The maximum release rate from panels 3 to 6 occurred between eleven months and thirteen months after the waste rock placement, while the maximum release rate from the L2 lysimeters in panel 1 occurred much later, approximately 2 years after the waste rock placement.

The trends observed in the leachate from the L2 lysimeter in panel 1 that were relatively different from the trends observed in the leachates collected from the L2 lysimeter of the other panels (i.e. panels 3 to 6) did not only occur in the release rates, but also in the concentrations as explained in the previous section and in the increasing volume of leachate outflow as explained in Chapter 2. The leachate that came from outside of the lysimeter footprint area entered the capture zone of the L2 lysimeter of
panel 1 and may have affected the quality of leachate, as well as the quantity of leachate.

The release rates trends for acidity, sulfate and copper for the blue waste located at the rear zone (L3 lysimeter) of panel 7 are shown on Figures 3.12(c), 3.13(c) and 3.14(c). The maximum release rates of acidity, sulfate and copper were 14.9 g CaCO$_3$/t-day, 17.5 g/t per day, and 0.32 g/t per day, respectively. The maximum release rates for these constituents occurred approximately 8 months after the placement of the waste rock.

In general, the release rate from the blue waste located within the front zones (i.e. L1 lysimeter) had the highest release rate, while the release rates from the blue waste located at the middle (L2 lysimeter) sections of the panels were comparable with the blue waste located at the rear sections (L3 lysimeters) of the panel. These trends are consistent with the concentration trend in the leachate as shown in Figures 3.6 to 3.8. In addition, a similar trend was also observed in the volume of leachate outflow in which the greatest volume of leachate outflow was generally measured from lysimeters located at the front sections (i.e. L1 lysimeters) of the panels while the lysimeters located at the rear sections (L3 lysimeters) of the panels had the least volume of leachate outflow, as presented in Chapter 2 of this thesis.

As shown in Figures 3.15(a) and 3.16(a), the maximum release rates of acidity and sulfate for the red waste ranged between 26.3 g CaCO$_3$/t per day and 42.8 g
Figure 3.15 Acidity release rate trends from red waste located (a) at rear sections (i.e. L3 lysimeter) of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns

Figure 3.16 Sulfate release rate trends from red waste located (a) at rear sections (i.e. L3 lysimeter) of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns
Figure 3.17 Copper release rate trends from red waste located (a) at rear sections (i.e. L3 lysimeter of panels 3 to 6, and (b) in the 500 tonne test pad and leach columns

CaCO$_3$/t per day, and between 24.6 g/t per day and 42.8 g/t per day, respectively. The maximum copper release rate for the red waste in the same panels varied between 0.15 g/t per day and 0.67 g/t per day, as shown in Figure 3.17(a). All of these maximum release rates for these constituents occurred between 6 months to 8 months after the placement of the waste rock.

Release rates of acidity, sulfate and copper from the blue and red waste rock were compared between the trial dump and the smaller scale experiments (i.e. leach columns and test pad). The data for the leach columns and the 500-tonne test pad were obtained from Miller (2003c). Figures 3.12(d) to 3.14(d) show the acidity, sulfate and
copper release rates for the blue waste from the leach columns and from the test pad, respectively. The acidity, sulfate and copper release rates for the red waste from the test pad and leach columns are shown in Figure 3.15(b), 3.16(b) and 3.17(b). As explained previously, the release rates for the constituents from the waste rock in the trial dump were estimated assuming the leachate generated only from the waste rock mass directly above the area of the lysimeters. Although there is a degree of uncertainty with this assumption, the results, as can be seen in Figures 3.12 to 3.17, indicate that the acidity, sulfate and copper release from the waste rock in the panels of the trial dump had a similar trend as compared to the smaller scale test programs using the same waste rock. In general, release rates of acidity, sulfate and copper were high at the beginning of the experiment with the peak load reaching within one to two years after placement of the waste rock. It usually takes a few years for the construction of a waste rock dump to reach the final height since one waste rock dump is designed to accommodate waste rock material produced from mining in the long-term (i.e. several years to several decades). Therefore the implication of this observation with respect to the full-scale dump is that the significant release of key constituents may actually occur during the construction of the full-scale dump.

Contrary to the concentration trends, the release rates of key constituents such as acidity, sulfate and copper were higher in the smaller scale experiments (i.e. leach column and test pad) as compared to the trial dump. For example, the peak acidity and sulfate release rate from the blue waste in the test pad was higher by up to ten times
when compared to the peak release rate of the blue waste from the trial dump. Maximum copper release rates that were approximately forty times higher were also recorded from the blue waste located in the test pad and leach columns as compared to the blue waste located in the trial dump. Similar to the blue waste, the acidity, sulfate and copper release rates from the red waste were generally lower for the red waste located in the trial dump compared to the red waste from the test pad and leach columns. The peak acidity and sulfate release rates of the red waste in the smaller scale experiment was higher by up to five times as compared to the peak release rates from the trial dump. The difference in the peak copper release rate from the red waste in the trial dump and the smaller scale tests was much wider, and the peak copper release rate from the trial dump was lower by up to fifty times. This influence of scale on the testing of the oxidation of waste rock is further discussed in Section 3.4.4.

The decreasing pattern observed in the concentration and release rate suggests that the oxidation of sulfide minerals in the trial dump followed a shrinking core model that was introduced by Levenspiel (1972). The shrinking core model has been widely used in numerical models to describe the oxidation and dissolution reactions in waste rock and tailings (Ourangrawa et al. 2009; Hansen et al. 2008; Molson et al. 2005; Linklater et al. 2005; Mayer et al. 2002; Gerke et al. 1998; Wunderly et al. 1996; Davis and Ritchie, 1986). In the shrinking core model, the oxidation rate of the sulfide mineral is limited by the diffusion of oxygen to the sulfide mineral grain due to a protective surface coating. Miller (2007) reported that only very little pyrite remained in the
oxidized samples from the historic Manado dump in Grasberg Mine. The Manado dump had a similar type of waste rock as that which was used in the trial dump at the Grasberg mine. Results of mineralogical tests that were performed on samples collected from this historical dump indicated that these pyrite minerals were in a less reactive form and locked within gangue minerals that were predominantly quartz and micaceous particles as described at IWRI (2002). Hence, a steady oxidation rate that was much lower than the initial oxidation rate continued to develop after several years of exposure.

3.4.4 Scale-Up

As mentioned at the beginning of this chapter, the construction of the trial dump was a scale up program for the ARD study from the smaller scale tests that involved a 500-tonne test pad, and 2 kg and 35 kg leach columns. In order to understand the kinetics of the waste rock for the different scale of experiments, the cumulative sulfate loadings produced from these different experiments are examined. Figures 3.18 and 3.19 show cumulative sulfate loads leached from the blue waste and red waste, respectively from the trial dump and the test pad and leach column. The vertical scale of each graph is different with the lowest scale presented for lysimeter 3 of the trial dump and the highest vertical scale used for the test pad and leach columns. The cumulative sulfate load from these experiments increased towards a maximum sulfate loading in the range between 3,500 g/t and 11,000 g/t for the blue waste at the
Figure 3.18 Cumulative sulfate loads from blue waste located at (a) front sections (i.e. L1 lysimeter) of panels 1 to 3, (b) middle sections (i.e. L2 lysimeter) of panel 1 and panels 3 to 6, (c) rear sections (i.e. L3 lysimeter) of panel 7 in the trial dump, and (d) in the 500 tonne test pad and leach columns
trial dump and from 18,000 g/t to 20,000 g/t for blue waste from the 500 tonne test pad and leach column during three to four years of exposure time. The cumulative sulfate loading from the red waste at the trial dump was also compared to the test pad and leach columns. The cumulative sulfate loading varied between 3,000 g/t and 8,000 g/t for the red waste from the trial dump during four years of exposure. On the other hand, between 50,000 g/t and 65,000 g/t of cumulative sulfate was released from the
red waste in the 500 tonne test pad and leach column during the four year period following placement of the waste rock.

Based on the sulfate loadings that are presented in Figures 3.18 and 3.19, oxidation rates for waste rock within the trial dump and the 500 tonne test pad and leach columns can be estimated. The intrinsic oxidation rate (IOR) of waste rock depends on the sulfide mineral concentration and morphology in the waste rock, oxygen concentration, temperature, microbiology and other parameters (Bennett et al. 2000). The IOR for small scale experiments such as humidity cells and leach columns can be referred to as the oxidation rate for the waste rock material itself, while the IOR at a larger scale experiment such as the trial dump can be considered as the oxidation rate of the entire waste rock dump. The difference between the IOR values for the small scale experiment and the oxidation rate at the large scale experiments is considered a scale up factor. The IOR is calculated based on the cumulative sulfate that was released from the waste rock over the duration of the experiment. Reaction (3.1) from Section 3.1 was used to calculate the IOR and oxygen consumption rates equal to 0.583 times the sulfate loadings based on the stoichiometry of reaction (Bennett et al., 2000).

Table 3.2 shows the computed oxidation rate for the waste rock in the trial dump, test pad and leach columns. The ratio of the oxidation rate for the waste rock in the trial dump compared to the IOR from the smaller scale experiments is presented in Table 3.3, and is expressed as a percentage. Since the sulfate release rates for blue
Table 3.2 Comparison of oxidation rates from the different scale experiments

<table>
<thead>
<tr>
<th>Waste Type</th>
<th>Experiment Type</th>
<th>Oxidation Rate, kg O₂/kg material-second</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Year 1</td>
</tr>
<tr>
<td>Blue Waste</td>
<td>Front section of panels (panels 1 to 3)</td>
<td>9.5 × 10⁻¹¹</td>
</tr>
<tr>
<td></td>
<td>Middle section of panels (panels 3 to 6)</td>
<td>3.5 × 10⁻¹¹</td>
</tr>
<tr>
<td></td>
<td>Rear section of panel (panel 7)</td>
<td>4.5×10⁻¹¹</td>
</tr>
<tr>
<td></td>
<td>Leach column</td>
<td>1.2×10⁻¹⁰</td>
</tr>
<tr>
<td></td>
<td>Test pad</td>
<td>1.2×10⁻¹⁰</td>
</tr>
<tr>
<td>Red Waste</td>
<td>Rear section of panels (panels 3 to 6)</td>
<td>7.5 × 10⁻¹¹</td>
</tr>
<tr>
<td></td>
<td>Leach column</td>
<td>3.4×10⁻¹⁰</td>
</tr>
<tr>
<td></td>
<td>Test pad</td>
<td>4.6×10⁻¹⁰</td>
</tr>
</tbody>
</table>

Table 3.3 Scale up Ratio

<table>
<thead>
<tr>
<th>Period</th>
<th>Front section of panels (panels 1 to 3)</th>
<th>Middle section of panels (panels 3 to 6)</th>
<th>Rear section of panel (panel 7)</th>
<th>Rear section of panels (panels 3 to 6)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Trial dump/Leach column</td>
<td>Trial dump/Leach column</td>
<td>Trial dump/Test pad</td>
<td>Trial dump/Leach column</td>
</tr>
<tr>
<td>Year 1</td>
<td>79</td>
<td>29</td>
<td>38</td>
<td>22</td>
</tr>
<tr>
<td>Year 2</td>
<td>58</td>
<td>28</td>
<td>14</td>
<td>5</td>
</tr>
<tr>
<td>Year 3</td>
<td>39</td>
<td>36</td>
<td>29</td>
<td>4</td>
</tr>
<tr>
<td>Year 4</td>
<td>N/A</td>
<td>26</td>
<td>34</td>
<td>18</td>
</tr>
<tr>
<td>Overall</td>
<td>58</td>
<td>47</td>
<td>29</td>
<td>17</td>
</tr>
</tbody>
</table>
waste located in the middle section (i.e. L2 lysimeter) of panel 1 were much higher as compared to the blue waste located in the L2 lysimeter for the other panels, the data for the L2 lysimeter in panel 1 was not included in the calculation.

The overall range in oxidation rate of the trial dump for the blue waste ranged from $2.2 \times 10^{-11}$ kg/kg-second to $7.5 \times 10^{-11}$ kg/kg-second with the lowest oxidation rate measured in the blue waste located at the rear section of panel 7. The overall oxidation rate of the blue waste for the different sections of the trial dump were up to one order of magnitude lower compared to the overall IOR of the blue waste in the leach column and the test pad. The overall oxidation rate for the blue waste in the leach column and 500 tonne test pad were $1.3 \times 10^{-10}$ kg/kg-second and $1.6 \times 10^{-10}$ kg/kg-second, respectively. The overall oxidation rate for the red waste located in the rear sections of the trial dump was $5.1 \times 10^{-11}$ kg/kg-second. This was also lower by approximately one order of magnitude when compared to the overall IORs for the red waste from the leach columns and the test pad which were $4.7 \times 10^{-10}$ kg/kg-second and $4.0 \times 10^{-10}$ kg/kg-second, respectively.

The scale up factor for the blue waste located in the front and middle sections of the trial dump were approximately 24% to 58% of the overall oxidation rate for the blue waste material in the test pad and leach columns. Furthermore, the oxidation rate for blue waste and red waste located in the rear sections of the trial dump was only approximately 11% to 17% of the overall IOR for the blue waste and red waste obtained from the smaller scale experiments. The different oxidation rates at the different scale
experiments were also reported by Stromberg and Banwart (1999), Malström et al. (2000), Frostad et al. (2005), and Wagner et al. (2006) who conducted the scale-up experiment from the laboratory scale to the meso-scale using large columns and test piles that were comparable to the size of the 500 tonne test pad. Results from these previous experiments indicated that higher oxidation rates were observed in the small scale kinetic tests such as the humidity cell. However, the results for these previous studies were different when compared to the results obtained from the scale up program reported here which showed that the oxidation rates for the waste rock were similar between the leach columns and the 500 tonne test pad. The test pad was exposed to continuously high rainfall rates which resulted in a continuous flushing of the oxidation products. Thus, the test pad acted like a large column. The results from the trial dump experiment demonstrated that the oxidation rate determined from the 500 tonne test pad still over-estimated the oxidation rates measured in the trial dump.

The cumulative amount of sulfate release in the leachate was used to estimate the total amount of sulfur that had leached from the trial dump during the experiment. The calculation was based on the cumulative sulfate released from the lysimeter for each panel that and is presented in Figures 3.18 and 3.19. These data were compared to the initial amounts of sulfur that were obtained from the ABA tests performed on the waste rock sample collected during the construction, the results of which are presented in Table 3.1. The average amount of sulfur leached from the blue waste in the L1 lysimeter for panels 1 to 3 ranged between 4% and 11% over the approximate three
year period. Due to a longer time of exposure, the average amount of sulfur leached from the blue waste located in the L2 lysimeters for panels 1, and 3 to 6 was higher compared to the blue waste located in the L1 lysimeters for panels 1 to 3, which varied from 11% to 21% of the total sulfur after four years. The amount of sulfur leached from the red waste was comparable with the blue waste located in the middle of the panels 1, and 3 to 6. Since the initial amount of total sulfur was higher in the red waste compared to the blue waste, the percentage of sulfur that was leached from the waste was lower compared to the blue waste. For example, approximately 2% to 7% of the total sulfur in the red waste was leached after four years of exposure. Miller (2007) reported that approximately 50% of the total sulfur initially contained in the waste rock from the test pad was leached after 10 years, and that most of that sulfur leached within the first 5 years. Since the oxidation rate at the trial dump was between 11% and 58% of the IOR from the waste rock at the test pad, it is estimated that to leach 50% of the total sulfur from the waste rock contained in the trial dump, it would take between twelve and fifty years.

The experimental leach columns used waste rock containing particles less than 39 mm, and the waste rock used in the test pad had a maximum particle size of up to 300 mm. Although the ranges of particle size between these two experiments were different, the oxidation rates from both these tests were comparable as presented in Table 3.2. The range of particle size distribution for the waste rock used in the trial dump was much wider compared to the one for the smaller scale tests. Waste rock as
large as 1 m in size was observed in the trial dump. Although the portion of waste rock with sizes larger than 300 mm at the trial dump was unknown, the portion of larger-size waste rock (i.e. larger than 300 mm) may have contributed to lower oxidation rates for sulfide minerals since the total exposed surface for the large particles would be greatly reduced compared to an equivalent mass of fine waste rock. In other words, the oxidation rate for sulfide bearing minerals is higher for smaller particles (Janzen et al. 2000).

The size of trial dump, which was much larger than the 500 tonne test pad, may have influenced the oxygen transport within the panels that could possibly have reduced oxidation rates. However, oxygen concentrations observed in the panels, as will be discussed in Section 3.4.6, suggest that the oxygen availability was not limited in the trial dump, even though low oxygen concentrations were also observed occasionally.

The hydrological processes within the trial dump are considered to be more complicated when compared to the hydrological process in the test pad pile due to the existence of a complex structure such as coarse and fine layers that influence the leaching process and discharge of oxidation products. As discussed in Chapter 2, the average net infiltration in the trial dump (82%) was lower when compared to the average net infiltration in the test pad (i.e. 90%). Furthermore, the net infiltration for particular sections of the panels, such as the rear sections of the panels, was lower than the average net infiltration from the entire trial dump. The highest net infiltration was
recorded at the front sections of the panels and the lowest net infiltration occurred from the rear section of the panels. The rate of net infiltration can be compared with the oxidation rate for each section in the panel. The greatest oxidation rate was recorded for the waste rock located within the front sections of the trial dump and the lowest oxidation rate occurred in the rear of the panels. Thus, the oxidation rate that was calculated according to the sulfate release rate was comparable with the net infiltration rate in the panels.

An oxidation rate that is calculated according to the quantity of sulfate released can be underestimated if some of the sulfate precipitates such as gypsum and jarosite. The sulfate leaching rate is then lower than the sulfate generation rate. The geochemical speciation code PHREEQC (Parkhurst and Appelo, 2000) was used to examine the presence of sulfate and copper minerals. The concentration of these constituents in the leachate produced from the panels was then input to calculate the saturation index (SI) of the minerals. A SI equal to zero indicates an equilibrium condition. A SI greater than zero indicates super-saturated where precipitation has likely occurred. On the other hand, an SI less than zero implies undersaturation and precipitation is unlikely. Table 3.4 shows an example of SI from PHREEQC where the SI of the various secondary minerals are computed on the basis of the leachate chemistry from lysimeter 1 in panel 3. The input data can be seen in the Appendix.

The results of the speciation model for the leachates collected from the lysimeters in all panels indicated occasional oversaturation with respect to gypsum
Table 3.4 Saturation index of minerals for the speciation model of leachate from the L1 lysimeter in panel 3

<table>
<thead>
<tr>
<th>Minerals</th>
<th>SI log</th>
</tr>
</thead>
<tbody>
<tr>
<td>Al(OH)$_3$</td>
<td>-5.84</td>
</tr>
<tr>
<td>AlumK, Al(SO$_4$)$_2$·12H$_2$O</td>
<td>-5.87</td>
</tr>
<tr>
<td>Alunite, KAl$_3$(SO$_4$)$_2$(OH)$_6$</td>
<td>0.30</td>
</tr>
<tr>
<td>Anhydrite, CaSO$_4$</td>
<td>-0.13</td>
</tr>
<tr>
<td>Antlerite, Cu$_2$ (OH)$_3$SO$_4$</td>
<td>-7.09</td>
</tr>
<tr>
<td>Basaluminite, Al$_4$ (OH)$_3$SO$_4$</td>
<td>-10.73</td>
</tr>
<tr>
<td>Bianchite, ZnSO$_4$·6H$_2$O</td>
<td>-3.77</td>
</tr>
<tr>
<td>Boehmite, AlOOH</td>
<td>3.62</td>
</tr>
<tr>
<td>Brochantite, Cu$_6$ (OH)$_6$SO$_4$</td>
<td>-11.11</td>
</tr>
<tr>
<td>Chalcanthite</td>
<td>-2.22</td>
</tr>
<tr>
<td>Cu(OH)$_2$</td>
<td>-5.61</td>
</tr>
<tr>
<td>CuSO$_4$</td>
<td>-8.24</td>
</tr>
<tr>
<td>CuMetal</td>
<td>0.71</td>
</tr>
<tr>
<td>CuOOCuSO$_4$</td>
<td>-13.36</td>
</tr>
<tr>
<td>CupricFerrite CuFe$_3$O$_4$</td>
<td>-9.87</td>
</tr>
<tr>
<td>Cuprite Cu$_4$O</td>
<td>-0.75</td>
</tr>
<tr>
<td>CuprousFerrite CuFeO$_2$</td>
<td>4.26</td>
</tr>
<tr>
<td>CuSO$_4$</td>
<td>-7.87</td>
</tr>
<tr>
<td>Diaspore AlOOH</td>
<td>-1.91</td>
</tr>
<tr>
<td>Epsomite MgSO$_4$·7H$_2$O</td>
<td>-2.76</td>
</tr>
<tr>
<td>Fe(OH)$_2$, 7Cl$_3$</td>
<td>-2.50</td>
</tr>
<tr>
<td>Fe(OH)$_2$(a)</td>
<td>-8.40</td>
</tr>
<tr>
<td>Fe$_3$[OH]$_8$</td>
<td>-24.63</td>
</tr>
<tr>
<td>Gibbsite Al(OH)$_3$</td>
<td>-3.15</td>
</tr>
<tr>
<td>Goethite FeOOH</td>
<td>-2.51</td>
</tr>
<tr>
<td>Goslarite ZnSO$_4$·7H$_2$O</td>
<td>-3.57</td>
</tr>
<tr>
<td>Gypsum CaSO$_4$·2H$_2$O</td>
<td>0.09</td>
</tr>
<tr>
<td>Hematite Fe$_3$O$_5$</td>
<td>3.01</td>
</tr>
<tr>
<td>Jarosite(ss) (K·Na0.3H$_2$O)Fe$_6$ (SO$_4$)$_2$(OH)$_8$</td>
<td>16.65</td>
</tr>
<tr>
<td>Jarosite-K Fe$_6$ (SO$_4$)$_2$(OH)$_8$</td>
<td>-17.32</td>
</tr>
<tr>
<td>Jarosite-Na NaFe$_6$ (SO$_4$)$_2$(OH)$_8$</td>
<td>-21.10</td>
</tr>
<tr>
<td>JarositeH[FeH$_2$O]Fe$_6$ (SO$_4$)$_2$(OH)$_8$</td>
<td>-20.92</td>
</tr>
<tr>
<td>Jurbanite AlOOHSO$_4$</td>
<td>0.30</td>
</tr>
<tr>
<td>Langite Cu$_4$ (OH)$_6$SO$_4$·H$_2$O</td>
<td>-12.56</td>
</tr>
<tr>
<td>Maghemite Fe$_3$O$_4$</td>
<td>-13.40</td>
</tr>
<tr>
<td>Magnetite Fe$_3$O$_4$</td>
<td>-8.14</td>
</tr>
<tr>
<td>Melanothallite CuCl$_6$</td>
<td>-14.21</td>
</tr>
<tr>
<td>Melanterite FeSO$_4$·7H$_2$O</td>
<td>-3.08</td>
</tr>
<tr>
<td>Mirabilite Na$_2$SO$_4$·10H$_2$O</td>
<td>-6.92</td>
</tr>
<tr>
<td>Mn$_3$ (SO$_4$)$_2$</td>
<td>-51.65</td>
</tr>
<tr>
<td>MnSO$_4$</td>
<td>-8.79</td>
</tr>
<tr>
<td>Morenosite NiSO$_4$·7H$_2$O</td>
<td>-5.10</td>
</tr>
<tr>
<td>Ni$_4$(OH)$_6$SO$_4$ Ni$_6$ (OH)$_6$SO$_4$</td>
<td>-38.15</td>
</tr>
<tr>
<td>Portlandite Ca(OH)$_2$</td>
<td>-19.40</td>
</tr>
<tr>
<td>Retgersite NiSO$_4$·6H$_2$O</td>
<td>-5.42</td>
</tr>
<tr>
<td>Tenorite CuO</td>
<td>-4.59</td>
</tr>
<tr>
<td>Thenardite Na$_2$SO$_4$</td>
<td>-7.85</td>
</tr>
<tr>
<td>Zincosite ZnSO$_4$</td>
<td>-8.54</td>
</tr>
<tr>
<td>Zn$_2$[OH]$_2$SO$_4$ Zn$_3$ (OH)$_2$SO$_4$</td>
<td>-10.67</td>
</tr>
<tr>
<td>Zn$_3$O(SO$_4$)$_2$ ZnO:2ZnSO$_4$</td>
<td>-27.72</td>
</tr>
<tr>
<td>Zn$_4$ (OH)$_6$SO$_4$ Zn4(OH)$_6$SO$_4$</td>
<td>-26.84</td>
</tr>
<tr>
<td>ZnSO$_4$·H$_2$O</td>
<td>-4.96</td>
</tr>
</tbody>
</table>
(CaSO₄), while the precipitation of jurbanite (AlOHSO₄) occurred in most of the leachates collected. On the other hand, results of the speciation model for leachates collected from the test pad and leach columns showed only over-saturation with respect to jurbanite (AlOHSO₄). The precipitation of gypsum depended on the concentration of calcium that was dissolved from the waste rock (Sracek et al. 2004), while the precipitation of jurbanite depends on the pH of the leachate (Nordstrom and Ball, 1986). In addition, some leachates collected from the trial dump and test pads were also oversaturated with respect to Cuprous Ferrite (CuFeO₂), which was a copper-iron secondary mineral, although the precipitation of the copper mineral only occurred occasionally. Shum and Lavkulich (1999) indicated that pH controlled the precipitation of the copper secondary mineral.

In summary, it is therefore hypothesized that when precipitation of these secondary minerals occurred in the pile, the actual amount of sulfate and copper released from the trial dump is probably higher than indicated by the measured concentrations. Examination of the results for secondary minerals from the historic Manado dump located near the trial dump, showed the existence of the secondary minerals hematite, goethite, ferric hydroxide, jarosite and oxidized copper sulfides, but the amount was low (Miller, 2007). Furthermore, the blue waste and the red waste used in this trial contained low carbonate minerals. Therefore, it is argued that the relatively steady rainfall over the year leads to a continuous flushing of oxidation
products from the pile. Thus, the precipitation of the secondary minerals is considered to be minimal in the trial dump panels as well as the historic dumps.

3.4.5 Temperature at Interior of Trial Dump

Temperature measurement commenced as soon as the thermistors were covered with waste rock materials during the construction of the panels. The start-up time for each thermistor was different since it depended on the progress of construction for the trial-dump. A multimeter was used to measure the resistance of the thermistors and the measurement was converted to the temperatures based on a calibration curve provided by the manufacturer for the thermistor. During the first year of the experiment, the temperatures were measured every two weeks and then changed to once a month after the first year.

Figures 3.20 and 3.21 show temperatures from thermistors located at the base of the trial dump beneath the blue waste and red waste, respectively. These thermistors were placed between 5 m and 70 m from the toe of dump, as illustrated on Figure 3.2. The trends for temperature were consistent among the thermistors. The temperatures started to increase and reached the maximum temperature followed by a gradual decline, but remained elevated at the base of the piles until the end of the experiment. The maximum and minimum temperatures recorded in the panels 1 to 5, and 7 at the end of the experiment were 58°C and 15°C, respectively after the waste rock had been in place for three to four years.
Figure 3.20 Temperature trends in the basal lysimeters beneath the blue waste located (a) 5 m and 15 m, (b) 30 m, (c) 40 m and (d) 60 m and 70 m from the toe of the dump.
Figure 3.21 Temperature trends in the basal lysimeters beneath the red waste located (a) 30 m, (b) 40 m and (c) 60 m and (d) 70 m from the toe of the dump
Although the temperature trends were similar, the amount of time to reach the peak temperature was different among thermistors located at different distance from the toe of the dump, as shown in Figure 3.20. In general, thermistors in panels 1 to 3, which were located approximately 5 m and 15 m from the toe of the dump, reached peak temperature between seventeen and twenty-seven months. On the other hand, thermistors in panels 1 to 5 which were located approximately 30 m and 40 m from the toe of the dump achieved peak temperature after twenty-six to thirty-eight months, and between thirty-seven and forty-two months, respectively. While the temperature for thermistors located approximately 60 m and 70 m from the toe of the dump in panel 7 reached the maximum thirty months after placement of the waste rock. The magnitude of temperatures was also different among the thermistors. The maximum temperature ranged from 31°C to 56°C. The highest temperature was 56°C recorded approximately twenty months after the placement of the waste rock for the thermistor located 15 m from the toe of the dump in panel 2. The lowest maximum recorded was at panel 1 where the thermistor was located 5 m from the toe of the dump and this temperature occurred approximately twenty-three months after placement of the waste rock.

Temperature trends measured from basal thermistors located underneath the red waste are shown in Figure 3.21. These thermistors were located approximately 30 m, 40 m, 60 m and 70 m from the toe of the dump. Although the pyrite content was higher in the red waste than in the blue waste, the magnitude of the maximum
temperature recorded at the base of the panels containing the red waste was generally similar to the maximum temperature recorded in the blue waste, except for the thermistor at panel 7 which was located approximately 30 m and 40 m from the toe. The maximum temperatures in most of these thermistors varied from 17°C to 50°C. The highest peak temperature was recorded in panel 1, 70 m from the toe of the dump, which occurred approximately forty-three months after placement of the waste rock. The lowest maximum temperature occurred in a thermistor located 30 m from the toe of the dump in the panel 7, which was recorded approximately thirty-four months after placement of the waste rock. The peak temperature for the thermistors in panels 1 to 5, located approximately 60 m from the toe of the dump and thermistors at panels 7 and 8 located 30 m and 40 m from the toe of the dump occurred between thirty and forty-three months after placement of the waste rock. On the other hand, the peak temperature for thermistors from the same panels located approximately 70 m from the toe of the dump was achieved after a longer period, between forty and fifty-one months after placement of the waste rock.

The variation in the temperature data indicated that the temperature was independent of the location of the thermistor such as distance from the dump toe. It is most likely that the heterogeneity of the waste rock geochemistry determines temperature. For example, relatively low temperature was recorded at the thermistors in panel 1 located 5 m and 15 m from the toe of the dump; where higher temperatures may have been expected as a result of the availability of oxygen. As previously
discussed at Section 3.4.1, the geochemistry results for the waste rock samples from this location showed relatively higher ANC values and an almost zero NAG value; and therefore can be expected to be less reactive compared to the blue waste collected from the other panels that had higher NAG value and a low ANC value.

The thermistors were placed in the face of the panels, as illustrated in Figure 3.2 and elevated temperatures were recorded in these thermistors. Figures 3.22, 3.23 and 3.24 show temperature from thermistors located at the face of the panels 2, 4, and 6, respectively. Thermistors were placed approximately 15 m, 20 m and 24m from the crest of panel 2, while thermistors were located at approximately 8 m, 16 m, 24 m and 31 m from the crest of panels 4 and 6. Locations of thermistors are shown as dots at the first contour in the graph. The temperatures from other panels are shown in the Appendix of this thesis. Temperatures recorded at the face of the panel were higher than the temperatures recorded at the base of the trial dump. For example, higher maximum temperatures were measured at the thermistors located at the face of panel 2, which reached 72°C as shown on Figure 3.22. The rate of temperature increase became slower with time and the temperature for several thermistors began to decline after 2 years, however temperatures as high as 56°C were still recorded at many sensors at the end of the experiment after four years.
Figure 3.22 Measured temperatures in thermistors located at the face of panel 2
Figure 3.23 Measured temperatures in thermistors located at the face of panel 4
Figure 3.24 Measured temperatures in thermistors located at the face of panel 6
The front of panel 2, or up to 20 m from the trial dump toe contained the blue waste, while a blend of blue waste and limestone was placed in the front of panels 4 and 6. The influence of limestone on the temperature of the panel was observed at the front section of panel 4 where the increase in temperature was slower compared to other sections within the panel. On the other hand, the presence of limestone had limited influence on the temperature in the front section of panel 6. The increase in the temperature at the front of panel 6 was similar to the increase in temperature at other sections in panel 6 that contained the blue waste only.

The temperatures recorded at the face of the dump showed that the increases in temperature for each sensor on the same thermistor string at a different elevation was not consistent, even though these sensors had the same waste rock type and installation time. The temperature contour at panel 2, as illustrated in Figure 3.22, demonstrated that temperatures close to the base of the panel increased faster than temperatures at other locations. Thus, a temperature difference, as high as 40°C was recorded between the sensors along a single thermistor string located 26 m from the crest of panel 2. A different pattern was observed at panels 4 and 6, which demonstrated that temperatures near the top of the dump increased faster than at other locations. These profiles were recorded for the thermistor located 31 m from the crest of panels 4 and 6.

Although the same waste rock was placed along a given single thermistor string, variation and heterogeneity was still observed. Waste rock containing higher NAG
values was expected to be more reactive, thus faster increases in temperature might be expected. However, the measurement of temperature at the face of a panel did not always agree with this hypothesis. For example, the sulfur and NAG of a waste rock sample from the lower section of panel 6 were 1.1% and 20 kg H$_2$SO$_4$/ton, respectively, while the waste rock sample from the upper part of panel 6 had 0.7% and a NAG value of 11 kg H$_2$SO$_4$/ton. With these geochemistry characteristics, it was expected that the increase in temperature in the lower section of panel 6 would be faster than the rise in temperature in the upper section of panel 6. In fact, the opposite trends were observed where the temperature increased faster in the upper section of panel 6 as shown in Figure 3.24. This temperature trend implies that the heterogeneity of waste rock geochemistry was not the sole factor that determined the evolution of temperature. The availability of oxygen in this location also contributed to the oxidation process, as will be discussed in Section 3.4.7.

The high temperatures recorded in the trial dump were found to be comparable with those recorded for other studies by Ritchie (2003), and Sracek et al. (2006). The steady incremental increase in temperature that was recorded at most thermistors support the conclusion that the high temperatures were produced by a continuous sulfide mineral oxidation instead of remaining as heat supplied by earlier oxidation that was still trapped in the dump as reported by Bennett and Timms (1999).
3.4.6 Oxygen Concentration at Interior of Trial Dump

Oxygen measurement at the base of the trial dump started approximately 8 to 13 months after the placement of waste rock. The oxygen concentration was monitored regularly once every month. The oxygen concentrations measured in the gas collection tubes located at the bottom of the trial dump are presented in Figure 3.25. No oxygen measurements were available for panels 5 and 8 due to water problem in these oxygen tubes.

In order to compare the oxygen concentrations at different locations, the oxygen data are plotted as box plots. Figure 3.26 summarizes the results of the statistical analyses of the oxygen concentration measured at the front (L1), middle (L2) and back (L3) sections of the base of the trial dump. L1, L2 and L3 were approximately 15 m, 35 m and 65 m, respectively, from the toe of the trial dump. The mean of oxygen concentration measured at L3 was around 15%, which was lower than those measured at L1 and L2 at approximately 18.5%. In addition, a t-test was used to compare the mean of the oxygen concentration from L1, L2 and L3. Results of the t-test with using a 95% confidence interval indicated that the oxygen concentrations measured in L3 were significantly different from the ones measured in L1 and L2. On the other hand, the oxygen concentrations measured in L1 and L2 were not significantly different from each other.
Figure 3.25 Measured oxygen concentrations at the base of the panels
Figure 3.26 Box chart for oxygen distribution at the base of panels 1 to 4, and panels 6 and 7 at Lysimeters 1, 2 and 3

A further look at the charts in Figures 3.25 and 3.26 indicate that a temporal and spatial variability in the oxygen concentration was recorded over most of the locations in the trial dump. Additionally, there were no apparent trends over time in any of these panels. Low oxygen concentrations approaching anoxic conditions occurred temporarily and were generally measured in the rear of the panels (i.e. L3). Based on the oxygen measurement, shown in Figure 3.25 and 3.26, it was reasonable to conclude that there was little or no reduction in oxygen concentration as far back as lysimeter 2 (35 m from the dump toe), and only minimal oxygen reduction as far back as lysimeter 3 (65 m from the dump toe).
Figures 3.27, 3.28 and 3.29 show the box plots for oxygen concentration measured from oxygen tubes located at the face of panels 2, 4 and 6. The measurement results from the other panels are shown in the Appendix of this thesis. The temporal and spatial variability of oxygen concentration were also recorded from these oxygen probes. The oxygen concentrations varied from 0.1% to 20.5% measured at different locations. Low oxygen concentrations were measured at particular locations in panels 2 and 4. The cover constructed from weathered red waste was placed on the top of panel 4. Since the cover did not effectively decrease water infiltration as discussed in Chapter 2, it is also believed that the cover did not work effectively in reducing oxygen concentration. Since high oxygen concentrations were detected at locations next to the low oxygen concentrations, it is not clear if the low oxygen concentrations were due to the restriction of oxygen transfer to that location or if there were other factors involved such as a problem with the oxygen tubes.

Similar to oxygen concentrations in the base of the trial dump, there were no obvious apparent trends over time for any of the oxygen tubes located at the face of the panels. In addition, inconsistent trends occurred for the oxygen concentrations measured in probes located at the same dump elevation. The data also indicated that there was a difference in oxygen concentration between the front sections (i.e. probes located 8 m to 16 m from the crest) and the middle sections of the panels (probes located 26 m and 31 m from the crest). Low oxygen concentrations (i.e. less than 2%) were measured at several locations in the slope of the dump. However, these low
Figure 3.27 Oxygen concentrations in the blue waste for probes in panel 2 located (a) 15 m and (b) 26 m from the dump crest

Figure 3.28 Oxygen concentrations in the blue waste for probes in panel 4 located (a) 8 m, (b) 16 m, and (c) 26 m from the dump crest
Figure 3.29 Oxygen concentrations in the blue waste for probes in panel 6 located (a) 8 m, and (b) 31 m from the dump crest

Oxygen concentrations only occurred locally since high oxygen concentrations were measured at the sampling points located next to these low oxygen concentration sampling points, either at higher or lower elevation.

Oxygen is transported through diffusion due to oxygen concentration gradients at the interface between the dump surface and atmosphere, as well as through advection mechanisms driven by temperature and air density gradients (Lefebvre et al., 2001b, Ritchie, 2003). Higher oxygen concentrations, which were observed at locations closest to the base of the dump profile, and also near the dump surface as shown in Figures 3.25 to 3.29 imply both diffusion and advection mechanisms were involved in the oxygen transport. However, the occasional low oxygen concentration measured at the
rear of the panels indicated that oxygen transfer through the advection mechanism might be temporally limited at the rear of the panel. This observation agreed with Ritchie and Miskelly (2000) who suggested that the geometry of the dump influenced the mechanism for oxygen transport in the pile and that diffusion was a dominant mechanism in the centre of the pile while advection was the dominant mechanism in the region of the slope of the dump. The gas tubes at the rear of the panel were located about 65 m from the toe of the dump and the rear of the panels was surrounded by the extension to the trial dump area that was used for the access road. Hence, the location at the rear of the panels in the trial dump is considered to be relatively isolated with respect to oxygen supply. Figure 3.30 illustrates a conceptual diagram of oxygen transfer in the trial dump.

Figure 3.30 Conceptual illustration of oxygen transfer in the trial dump
Studies by Wels et al. (2003), Smolensky et al. (2000) and Harries and Ritchie, (1985) also showed that this oxygen variation in concentration occurred due to changes in atmospheric pressure. The changes in atmospheric pressure can enhance or restrict oxygen entry by the advection mechanism through the face of the waste rock dump. An increase in atmospheric pressure can enhance oxygen entry. On the other hand, oxygen releases from the waste rock pile when the atmospheric pressure decreases. In order to understand the impact of barometric pressure on oxygen concentration in the trial dump, a series of barometric pressure measurements were obtained using oxygen ports located along the base of panel 4. Although a change in oxygen concentration was observed in some tubes, as shown in Figure 3.31, it can be seen that the correlation between changes in barometric pressure and oxygen concentration is weak. The results of this observation is similar to the results observed by Phillip and Hockley (2007) at Sullivan Mine who indicated that the barometric pressure is not a dominant influence in oxygen transfer. Phillip and Hockley (2007) did however show that changes in the atmospheric temperature can affect the oxygen concentration in the waste rock dump. When the atmospheric temperature is lower than the temperature inside of the waste rock dump, oxygen is transferred from the atmosphere into the waste rock dump. On the other hand, oxygen is transferred from the waste rock dump into the ambient air, when the temperature inside of the waste rock dump is lower than the ambient temperature. Since the temperature within the trial dump (i.e. greater than 25°C) is higher as compared to the atmospheric temperature (i.e. annual average 7°C),
Figure 3.31 Correlation between oxygen concentration and barometric pressure

Oxygen is transferred continuously into the waste rock pile. The change in the daily temperature that occurred during the oxygen monitoring period could however have also affected the oxygen transfer resulting in a variation in oxygen concentration.

The variability in oxygen concentration within the trial dump may have resulted from rainfall events. Kabwe et al. (2005) reported that the concentration of gas at the waste rock dump could be changed due to changes in the water content of the pile. Although instrumentation to measure water content changes in the pile was not installed in the trial dump, it can be expected that the degree of saturation in the panels also varied during the period of oxygen monitoring since a variability in the daily rainfall intensity and duration occurred during the oxygen measurement at the trial dump.
location. In order to understand how much influence rainfall events have on oxygen concentration within the trial dump, a numerical simulation that includes oxygen diffusion and advection must be conducted, which is out of the scope of this thesis.

3.4.7 Correlation between Physical and Chemical Processes

The influence of the fluctuation in the volume of leachate outflow from the lysimeters on the pollutant concentrations such as acidity and sulfate can be assessed in order to help understand and illustrate the correlation between these two parameters. In general, the volume of the leachate appears to have a negative correlation with the concentrations of constituents (i.e. Figures 3.6 to 3.11 and Figures 2.12 and 2.13 in Chapter 2), although the leachate from several lysimeters such as the L1 lysimeter in panel 2, the L2 lysimeter in panel 1, and the L3 lysimeter in panel 3 show a positive correlation. The negative correlation was mostly weak (less than 0.5) and the coefficient of correlation varied between 0.1 and 0.56. The kinetic evolution characteristics of the waste rock likely determined the amount of oxidation products that controlled the quality of the leachate. Leachate chemistry was also found to be somewhat susceptible to fluctuations in rainfall. During the period of July to September 2002, due to what appeared to be a weak El Nino effect, monthly rainfall was reduced to approximately 50% to 65% of normal values. This did not however appear to have been sufficient to significantly change the general patterns of concentrations observed over the entire duration of the experiment.
The amount of net infiltration from every panel had a strong correlation with the release rate of the constituents, as discussed previously in Section 3.4.3. The highest release rates occurred from the panels that had the greatest net infiltration such as the waste rock located at the front sections of the panels. The lowest release rates occurred from the panels that had the lowest net infiltration which included the waste rock located at the rear sections of the panels.

The temperature profile in the trial dump panels implied strong oxidation of sulfide minerals occurred. The evolution in temperatures observed at the base of the panel was found to be consistent with the evolution of constituents such as acidity and sulfate in leachate; therefore, it was determined that temperatures can be used as an indicator of the progress of oxidation within the trial dump. As shown in Figures 3.20 and 3.21. The rate of increase in temperature at the base of the panel was reduced the further the thermistors were located from the toe of the dump. Although the magnitude of temperatures recorded between the thermistor located close and far from the dump toe were not that different, the time to reach the maximum temperature was longer when the thermistors were located farther from the toe dump. These observations are in agreement with the concentration of constituents leached from the panel and the oxygen observed at the probes located at the base of the trial dump. As previously discussed in Sections 3.4.2 and 3.4.3, the concentration and release rates of acidity, sulfate and copper were lower for the waste rock located at the rear of panels as compared to the waste rock at the front sections of the panel. The
average oxygen concentration measured at the rear section of the base of the trial
dump was slightly lower than the average oxygen concentration at the front and middle
sections of the base of trial dump, as shown in Figure 3.25 and 3.26.

In contrast, the relationship between oxygen concentrations and temperature at
the face of panels 2 and 6 was not obvious. The oxygen availability within the panel
affected the oxidation of sulfide minerals which lead to the elevated temperature.
Assuming that oxygen is an independent variable, it suggested that at this location,
there was little or no relationship between the temperature of a particular location in
the trial dump and the corresponding measured oxygen level at the same location, with
respect to time. Figure 3.32 shows the oxygen data collected from oxygen probes in
panel 6 located 31 m from the crest of the dump and in panel 2 located 26 m from the
toe. Temperatures measured from the thermistor strings located near these oxygen
probes in panels 2 and 6 are previously shown in Figures 3.22 and 3.24, respectively.
Since the nearest oxygen probes were located approximately 11 m apart from the
thermistor string at panel 4 which is presented in Figure 3.23, the oxygen data from this
panel was not included in this analysis. Due to the temporal variability of the oxygen
concentration, the data are presented as box plots where the mean and the median of
the data are shown as dots and center lines, respectively in the boxes in Figure 3.32.
The average oxygen concentration near the base of panel 2 (i.e. 1 m) was higher than
the oxygen concentration near the surface of the panel (i.e. 19 m). In addition,
Figure 3.32 Oxygen concentration in the blue waste from oxygen probes at the face of (a) panel 6, 31 m from the crest of the dump, and (b) panel 2, 26 m from the crest of the dump
temperatures measured near the base of panel 2 tended to be higher as compared to
temperatures measured near the surface of panel 2, as shown in Figure 3.22. In
contrast, the average oxygen concentration in Figure 3.32 near the surface of panel 6
(i.e. 19 m) was comparable to the average oxygen concentration near the base of panel
6 (i.e. 1 m), but temperatures near the surface of panel 6 tended to be higher than
temperatures near the base of panel 6, as shown in Figure 3.24. This data suggests that
the temperature of the panel may have a weak correlation with oxygen concentrations
recorded at similar locations. The warm buoyant air from the base of the panel could
rise through the coarse/fine structure that formed the internal structure of the waste
rock pile leading to increases in temperature near the dump surface and subsequent
changes in oxygen concentration.

Another view of the same comparison is shown in Figure 3.33 where results
from thermistor and oxygen probes at panel 6 located approximately 31 m from the
crest are plotted together. This figure shows that for oxygen levels ranging from 5% to
19%, temperatures ranged between 42°C and 60°C in April 2002. Similarly, even within
a tighter oxygen range of, for example 13% to 20%, temperatures can still range from
60°C to 70°C. This weak correlation indicates that the temporal variation in oxygen
measurements do not strongly affect sulfide mineral oxidation as expressed in the
temperature profiles.

The temperature that depends not only on the oxygen availability but also on
the geochemistry of the waste rock resulted in an inconsistent correlation between the
Figure 3.33 Temperature and oxygen profiles in the blue waste at the face of panel 6, 31m from the toe in (a) April 2002 and (b) April 2003

trends in temperature and oxygen that were measured at the face of the dump. In the end, the combined effect of heterogeneity in the geochemistry of waste rock, the oxygen availability and the internal structure of the waste rock pile determined the temperature at this location, assuming it is reasonable that the thermal conductivity is uniform in the trial dump.

3.4.8 Performance of Cover Treatments

The acidity release rates for the panels that had low-permeability and limestone covers are presented in Figures 3.34 and 3.35 for the L2 and L3 lysimeters, respectively.
Figure 3.34 Acidity release rates from the L2 lysimeters of panels with (a) the limestone cover (panels 5 and 6), (b) the HDPE cover (panel 1), and (c) the mixture of road mud and limestone cover (panel 3) and the weathered waste rock cover (panel 4)
Figure 3.35 Acidity release rates from the L3 lysimeters for panels with (a) the low-permeability covers (panels 3 and 4), and (b) the limestone covers (panels 5 and 6)

Due to problems with the outflow from the L3 lysimeter of panel 1, leachate chemistry data was not available for this lysimeter.

The decreasing pattern in the acidity release rate for the low-permeability cover treatment at panels 3 and 4 was generally similar. The use of the road mud cover and the watered red waste reduced insignificantly the acidity release rate at the L2 and L3 lysimeters as shown on Figure 3.34 and 3.35, respectively.

The acid loading rate from the L2 lysimeter at panel 1 did not decrease when the HDPE was placed. As discussed in Section 2.4.8 of Chapter 2, the volume of leachate collected from the L2 lysimeter increased significantly during the period after the
placement of the HDPE cover due to run off and sub-lateral flow from the area outside of the lysimeter area. As a result, the acid loading from the L2 lysimeter of panel 1 also increased after the cover placement, as shown in Figure 3.34.

The 1.5-meter deep trenches excavated in the panels with the various cover treatments were intended to determine whether oxidation products would continue to be generated within the dump if the infiltration water was excluded, i.e., if there would be flushing of oxidation products from the dump. Visual inspection in the trenches showed oxidation products were found beneath the cover materials, as shown in Figure 3.36. In addition, visual observations indicated that the accumulation of oxidation products was more pronounced under the HDPE liner than under the other covers.

The low-permeability covers used in this experiment were designed to reduce the amount of infiltration of water and specifically not to decrease the oxygen entry. Since the HDPE liner used as cover material is impervious to water flow, it was expected that the oxygen entry from the flat surface of the trial dump might also be reduced. However, it was observed, as shown in Figure 3.26 that oxygen concentration did not change significantly, in sampling points located at the base of the waste rock pile in panel 1. A decrease in oxygen concentration was only detected one year after the HDPE liner was placed at the sensor at the base of the dump located 15 m from the dump toe. In general, oxygen continued to enter into panel 1 by diffusion and/or an advection mechanism from the exposed face of the dump. In addition, the observation of the accumulation of oxidation products, increased temperatures, and a lack of decline in
Figure 3.36 Oxidation products discovered beneath the HDPE cover (panel 1) during excavation of the trial dump containing blue waste

oxygen concentrations suggested that airflow through the coarse rock at the base of the dump was a primary mechanism for oxygen transfer in the HDPE cover panel. It appeared that the placing of the HDPE liner only on the flat surface of the panel rather than down the slope of the panel as well did not stop the oxidation of the sulfide minerals.

The limestone cover had no impact on the acidity release rate as shown in Figures 3.34 and 3.35. The limestone cover test that was previously conducted at the
500-tonne test pad demonstrated that it took approximately twenty-three months to increase the pH of leachate to circum neutral (Miller et al. 2003b). Since the size of the trial dump was much larger than the test pad, it was expected that it would take a longer time to see the impact of the limestone cover on the quality of leachate from the trial dump. A change in the quality of the leachate was not yet seen because the limestone cover was just in place for around two years.

3.5. Conclusions

The quality of leachate and the recorded temperature profiles in the trial dump demonstrate that the waste rock was a highly reactive material that generated acid within a short period of time (i.e. weeks). The characteristics of the waste rock kinetics in the large-scale experiment were found to be consistent, but not quantitatively equivalent with the performance observed in the smaller-scale experiments. The waste rock kinetics were characterized by fast initial kinetics, attaining some maximum value within one to two years, followed by a rapid decrease with a longer term decay and a slower downward trend. Since it may take several years to complete the construction of a large dump, the exposure of the waste rock during the completion time can lead to a fast oxidation rate at the beginning, particularly for the type of waste rock that was tested in the trial dump at the Grasberg mine.

Elevated temperatures were measured at the different locations at the trial dump. Temperatures as high as 56° C were recorded at the base of the trial dump, while higher temperatures (i.e., approximately 72° C) were measured at the slope of the
dump. High temperatures continued to be recorded until the end of the experiment when the waste rock had been in place for four years. High oxygen concentrations (above 18%) were measured in the trial dump, however low oxygen concentrations (i.e. less than 2%) were also recorded at several oxygen tubes located at the face of the dump. These low oxygen concentrations seem to occur locally since the high oxygen concentrations were also recorded at the locations next to the sampling points where the low oxygen concentrations were measured. These low oxygen concentrations were not indicative of the sulfide mineral oxidation since high temperatures and acid leachate were still present. The high oxygen concentration that was measured near the base of the dump and also near the surface of the dump demonstrated that oxygen transport into the pile through both advection and diffusion mechanisms was occurring.

The oxidation rate for the waste rock that had been exposed for three to four years appeared to be lower in the trial dump when compared to the 500-tonne test pad and 2 kg and 35 kg leach columns. The oxidation rates at the trial dump varied from 2.2 \( \times 10^{-11} \) kg/to 7.5 \( \times 10^{-11} \) kg/kg-second for the blue waste, and 5.1 \( \times 10^{-11} \) kg/kg-second for the red waste, compared to 1.3 \( \times 10^{-10} \) kg/kg-second and 1.6 \( \times 10^{-10} \) kg/kg-second, and 4.7 \( \times 10^{-10} \) kg/kg-second and 4.0 \( \times 10^{-10} \) kg/kg-second of oxidation rates in the leach column and the test pad for the blue waste and the red waste, respectively. The oxidation rate at the trial dump was determined to be up to one order of magnitude less than the test pad and leach column. As a result, the scale up factor overall from the trial dump ranged between 11% and 58% from the test pad and leach columns. The
larger scale up factor was at the front section of the trial dump, while the rear section of the trial dump had a lower scale up factor.

The differences in oxidation rates observed between the trial dump and the smaller scale experiment were attributed to the difference in dump hydrology and particle size distribution. The trial dump had a lower net infiltration and a wider particle size range when compared to the smaller scale experiment, therefore the oxidation rate and the release of oxidation products were lower in the trial dump than the test pad and leach columns. In the real mining operation, the waste rock pile could be hundreds of meters in height and thousands of hectares in area. In the long run, the release of acid from the larger waste rock piles would take longer due to the lower oxidation rate, as illustrated above. The low oxidation rate observed at the trial dump implies that acid will be generated for a much longer time in the full-scale waste rock pile.

A strong correlation between the temperature and the oxygen concentration at the base of the trial dump and the sulfide mineral oxidation expressed as leachate was identified during the four-year monitoring of the trial dump. The oxygen concentration that was relatively lower at the rear of the panels appeared to slow down the oxidation rate resulting in a slower temperature increase and lower concentrations of key constituents such as acidity, sulfate and dissolved copper in the leachate. In addition, the net infiltration that was relatively lower in the rear sections of the panels caused the release rates of key constituents from the waste rock located at the rear of the panels
to be lower when compared to the waste rock from the front and middle sections of the panels.

On the other hand, the correlation between oxygen concentration and the temperature at the face of the panel was relatively weak. The variability in the oxygen concentrations observed at the face of the panels may be attributed to the structure of coarse/fine layers that promote advection at the dump face.

The placement of low-permeability cover had few implications with respect to the oxidation of sulfide minerals since acid was generated continuously due to oxygen transferred through advection mechanisms from the toe of the dump face, as well as the basal rubble zone.
References


Chapter 4 Performance of Limestone Blends for Long-Term Control of Acid Rock Drainage (ARD) in High Rainfall Environments

4.1 Introduction

Acid rock drainage (ARD), characterized by low pH and elevated dissolved metals, can be problematic when it is released from waste rock dumps and tailings impoundments to the environment. ARD deteriorates the quality of surface water and groundwater and often poses hazards to aquatic life. The concern regarding the impact of long-term ARD to the environment has led to research over the past several decades to find mitigative measures. Established strategies for mitigating ARD generation have attempted to exclude one or more of the factors that cause sulfide oxidation, such as the sulfide mineral, oxidants (i.e., oxygen and ferric ion), water, and microorganisms (Lottermoser, 2003). Over the past decade, research on ARD mitigation strategies have been developed, and some of the results have been implemented to achieve long-term environmental objectives using cost-effective methods. Some mitigative strategies include:

- Implementing geomembrane and soil cover (Swanson et al. 2003; Lundgren, 2001; Yanful et al. 1999) and a capillary barrier (Bussière et al. 2002).

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3 A version of this chapter will be submitted for publication. Andrina, J., Wilson, G.W. and Miller, S., Performance of Limestone Blends for Long-Term Control of Acid Rock Drainage Control in High Rainfall Environments
2007; Bussière et al. 2003; Khire et al. 2000) to reduce infiltration and oxygen diffusion into waste rock and tailing impoundments,

- Placing a store and release cover to control the oxygen diffusion into the dump (Williams et al. 2006; Durham et al. 2000),
- Constructing chemical covers such as phosphate encapsulation (Evangelou, 1995; Fytas et al. 2000) and alkaline covers (Heikkinen et al. 2009; Taylor et al. 2006; Miller et al. 2003a),
- Using organic covers (Peppas et al. 2005; Tremblay, 1994; Pierce et al. 1994;),
- Mixing of tailings and waste rocks (Wickland et al. 2006; Wilson et al. 2006)
- Using bactericides (Sand et al. 2007),
- Mixing acid rock with neutralization material (Miller et al. 2006; Day, 1994).

Adding neutralization material such as limestone to the acid waste rock can potentially minimize ARD generation. Effective blending can improve the pH of drainage and reduce the mobility of dissolved metals (Mehling et al. 1997). The amount of limestone or lime added to the acid waste rock and the intimacy of the mixing determine the effectiveness of this method. Determining the appropriate amount of the limestone or lime that is required for day to day operation was found to be quite a challenge due to the heterogeneity of sulfide minerals in the waste rock, slow
dissolution of carbonate minerals in the absence of the acid, and the possibility of the coating of carbonate grains by the iron precipitation (Ziemkiewicz and Meek, 1994).

The amount of limestone that should be added to the acid waste rock is usually assessed by the maximum potential acid (MPA) of the acid waste rock and the acid neutralizing capacity (ANC) of the limestone (Lapakko et al. 1997; Morin and Hutt, 1994; Price, 2003). Different terms are sometimes used, such as acid potential (AP) instead of MPA and neutralization potential (NP) instead of ANC. The MPA and the AP are calculated according to the sulfur content in the acid waste rock, while the NAPP and the NP are based on the neutralization capacity of the limestone obtained from the static test.

Brady et al. (1994) stated that an ANC/MPA ratio of 2 had been used in past practice at the coal mines and that this ratio was established according to a stoichiometry of the following neutralization reaction, which involved 2 moles of calcite and 1 mole of pyrite.

\[
FeS_2 + \frac{7}{2} H_2O + \frac{15}{4} O_2 + 2 CaCO_3 \rightarrow Fe(OH)_3 + 2SO_4^{-2} + 2 H_2CO_3 + 2 Ca^{2+} \quad (4.1)
\]

The ANC/MPA ratio of 2 was considered to be effective in preventing acid generation at that time, although the results of long-term practice has shown that application of ANC/MPA ratio of 2 has not always been successful in preventing acid rock drainage. To improve the performance of limestone blending, Cravotta et al. (1994) suggested increasing the ANC/MPA ratio to 4 based on correction of the following chemical reactions involved in calcite and pyrite neutralization.
\[
\text{FeS}_2 + \frac{7}{2} \text{H}_2\text{O} + \frac{15}{4} \text{O}_2 + 2\text{CaCO}_3 \rightarrow \text{Fe(OH)}_3 + 2\text{SO}_4^{2-} + 2\text{HCO}_3^- + 2\text{Ca}^{2+} + 2\text{H}^+ \quad (4.2)
\]

Price (2003) reported that the majority of blending cases for coal and metal mines had an ANC/MPA ratio of less than 2; however, the required ratio or safe ratio of waste rock blends that would prevent ARD and metal leaching at those sites was not identified. A leach column experiment conducted by Chapman et al. (2000) showed that an ANC/MPA ratio of 2.7 would not likely produce acid. Morin and Hutt (1994) reviewed the existing work database and indicated that an ANC/MPA ratio of 1.3 to 4 was required to maintain the long-term neutral pH; however, the safe ratio should be determined on a case-by-case basis.

Since past practice has shown that a specified bulk ANC/MPA ratio does not always accurately predict drainage quality, Morin and Hutt (2000) introduced the discrete zone mixing concept to improve the intimacy of the limestone blending system. In this approach, one Potential Acid Forming (PAF) zone, which represents the acid waste rock, must be separated and managed individually from other PAF zones. Each individual NAG zone must be mixed with a sufficient amount of ANC zone to ensure that the acid released from each PAF zone is neutralized. The minimum distance between the discrete PAF zones is thus determined by the excess ANC and the hydrology factors of the waste rock dump such as channel flow and lateral flow. A similar approach was also shown by Kempton et al. (1997), who developed a conceptual model for placement of the acid generating and net neutralization material in the waste rock dump in arid
and semi-arid climates. The model results showed that thinner layers of the acid waste rock provided more efficient mixing and could reduce potential acid generation.

The number of published works on the application of blending acid waste rock with limestone is limited. The successful blending of limestone to reduce ARD was mostly reported for experiments at laboratory scale (Jahn et al. 1990; Lapakko et al. 1997; Mylona et al. 2000). Application at the field scale was mostly related to coal mines and not all field applications showed acid generation was abated successfully (Brady et al. 1990). Hence, the application of blending waste for the large scale mine waste dump remains in serious doubt, and further investigation into the application of this method is required.

Grasberg Mine will have produced significant quantities of limestone between 2003 and 2013 when operation of the open pit is complete. Thus, it is important to develop strategies for designing a waste rock dump that optimize the use of limestone to effectively minimize ARD generation over time. The initial study to assess the effectiveness of limestone blending for typical acid waste rock produced at the Grasberg Mine was carried out in 1996 (Miller et al. 2003a). The study assessed the blending of limestone with acid rock in small leach columns (i.e. 2 kg and 35 kg) and meso-scale 500-tonne test pads. Although, the results provided important information such as the minimum amount of limestone required to prevent acid generation and/or drainage, the suitability of the blending or the methods to be used for large scale waste rock dumps was still not known.
A large scale field experiment was constructed at Grasberg Mine in 1999. In addition to studying the kinetics of the acid generating waste rock in a large scale experiment, a part of the field experiment was also designed to assess the performance of the different methods for blending limestone and acid waste rock. The various dumping equipment such as haul trucks and a crusher-stacker system usually used for the construction of the full scale waste dumps at the Grasberg Mine were examined in the trial.

The present study is aimed at a thorough evaluation of blending methods for ARD mitigation in full scale waste rock dumps in a high rainfall environment. The site has an average daily precipitation of 10 mm, which persists throughout the year. As a result, the waste rock experiences continuous leaching throughout the year and the seepage is produced at a constant rate. The high rainfall environment at the site may affect the performance of the various blending treatment methods since the high infiltration rate may reduce lime dissolution due to the limit in residence time (Hossner and Doolittle, 2003).

This paper describes and evaluates the various blending methods used in the construction of the large field experiment. The results of the quality of the leachate collected from each panel where the various blending methods were used are discussed. Finally, temperature data measured during the experiment is also presented. The implication of the field experiment results with respect to full scale waste rock dumps is also addressed.
4.2 Site Description

Grasberg Mine, is a copper and gold mine located in the equatorial mountainous region in the Papua Province of Indonesia. The Grasberg Mine is a porphyry Cu/Au deposit hosted within limestone. Approximately 2,750 million tonnes of waste rock will be produced by the year 2015 and about 37% of the waste rock consists of limestone (Rusdinar, 2006). Calcite is the dominant carbonate mineral within the limestone, although magnesium carbonate and some pyrite occurs locally (Miller et al. 2003b). The limestone contains a carbonate equivalent of more than 85% CaCO₃. The acid waste rock that was assessed in the blending method was considered to have a lower capacity of potential acid forming (PAF), which was locally identified by the mine as blue waste. This material was found to contain up to 5% pyrite or as 2% sulfur and has a NAG value of less than 35 kg H₂SO₄/ton waste.

4.3 Methodology

The methods used in this paper include field programs and laboratory tests. The field programs consisted of constructing blended panels at the trial dump, installing instrumentation, and collecting leachate samples from the basal lysimeters L1 that were installed at the base of the trial dump. In addition, the monitoring of temperature and oxygen was also included in the field programs. The laboratory program included the geochemical tests for waste rock samples that were collected during the construction of the trial dump, and also the chemical analysis on the leachate samples.
4.3.1 Design and Construction of Trial Dump

In 1999, the mine began construction of the waste rock dump trial at a West Grasberg dumping area. A waste rock dump 480 m long, 80 m wide and 20 m thick consisting of 8 panels (or sections) was constructed. Each panel was then divided into three sections, each to accommodate selected types of waste rock. A 10 m x 10 m lysimeter was installed at the base of each section to collect the seepage giving a total of 24 (i.e. 8 panels X 3 lysimeters) leachate sampling points. Thermistors and gas sampling ports were installed at the base and the slope of the dump to measure both the temperature and oxygen concentration in the interior of the waste rock dump. A detail of the dump design, construction and instrumentation can be found in Chapters 2 and 3 of this thesis.

The front sections of five panels (panels 4 to 8) were used to assess the effectiveness of the various limestone blending methods for ARD control. As illustrated in Figure 4.1, panels 4, 5 and 6 were constructed using a conventional truck dumping method while the fronts of panels 7 and 8 were constructed using a crusher and stacker system. Different blending methods were deployed for each panel as shown in Figure 4.2. The front of panel 4 was constructed using three (3) truck loads of run-of-mine (ROM) blue waste to every one (1) load of ROM limestone. The loads were tipped at the face along the crest of the dump. A front-end loader and a dozer were also used to mix and push the material down the face.
Figure 4.1 Sketch of the trial dump
Figure 4.2 Simplified plan view of trial dump
Alternate truck dumping was applied to panel 5 where three (3) truck loads of blue waste and one (1) truck load of limestone were tipped at the crest and allowed to freely flow or ravel down the tip face. This method of tipping and dumping was alternated until the panel was completed. Finally, panel 6 was built with distinct layers consisting of 3 meters of blue waste followed by 1 meter of limestone that were sequentially placed along the length of the panel. The stacker built panels (7 and 8) were constructed using a crusher and stacker. The crusher was fed using a front-end loader, and alternatively loaded 12 cubic meters of blue waste and 4 cubic meters of limestone. This ratio was developed according to an estimation of the layer thickness at 400 m of a full-scale stacker built dump. It assumed approximately 10 cm progress daily. The overall construction of the trial dump which began in 1999 was completed in April 2002.

4.3.2 Blending Ratio of Limestone and Acid Waste Rock

As previously mentioned, the trial dump was a scale up version of the laboratory scale experiments that involved 2 kg and 35 kg leach columns, and the meso-scale experiment with 500-tonne test pads. Several tests were set up with different blending ratios to establish the appropriate blending ratio. These tests included blending 5%, 10%, and 25% of the limestone with the blue waste. Miller et al. (2003a) reported that blending 25% of limestone with 75% blue waste provided effective performance in preventing ARD in both leach columns and within the meso-scale test pads. The blending of 10% limestone and 90% of the blue waste only showed satisfactory performance for the leach column alone. On the other hand, the blending of 5%
limestone with 95% blue waste failed in both the leach columns and test pads. In summary, according to these tests results, blending 25% limestone with 75% blue waste was selected for application to the trial dump.

4.3.3 Sampling and Monitoring

Waste-rock samples were collected from various locations within the trial dump during construction. The samples were obtained at three different times as the over-dumping on lysimeters continued. The samples were collected at the crest, mid-slope, and toe above each lysimeters and analyzed for Net Acid Generation (NAG), Acid Neutralizing Capacity (ANC) and total sulfur following the methods described at IWRI and EGI (2002).

To establish baseline drainage and collection patterns for each lysimeter, leachate sampling commenced in October 2000 while construction was still in progress. Leachate samples were collected every two weeks and then continued monthly, starting June 2001. Leachate temperature, pH and electrical conductivities (EC) were measured in the field, while TSS, TDS, alkalinity, acidity, major cation and anion, and dissolved metals were measured in the laboratory. Monthly monitoring of oxygen concentration and temperature was also carried out during the trials.

4.3.4 Dump Excavation

In December 2004, the trial dump was decommissioned, and a post-mortem study was conducted at the same time before the trial dump was over-dumped. Since
the present study was focused on panels built with limestone blending and limestone covers, panels 5, 6, and 7 were selected for post-mortem excavation after 4 years in 2004. An excavator and a bulldozer were used to cut trenches in the panel in order to expose the weathered materials immediately below the surface of the panels. Each trench was 6 m to 15 m wide and approximately 2 m deep, as shown in Figure 4.3.

![Trench excavated in panel 5](image)

**Figure 4.3 Trench excavated in panel 5 for post-mortem study**

Once the panels were exposed, several zones were selected for sampling according to their visual appearance based on the existence of oxidation products or alkaline solution migration. Bulk samples of 5 to 15 kgs were collected from the side walls of the excavations within designated zones using a hand shovel. The samples were placed in large plastic bags and sealed to avoid loss of moisture content. There were
approximately 30 samples collected and subsequently tested for grain size distribution, ABA and NAG tests.

4.4 Results and Discussion

The results obtained from the geochemical tests for the waste rock samples collected from the construction of the trial dump, as well as the excavation during the post-mortem study, are presented here. In addition, results of chemical tests on leachate samples collected during the trial dump experiment are presented. These tests concentrated on key constituents of ARD and metal leaching such as acidity, sulfate and dissolved copper. The field measurements of temperature for the interior of the trial dump are also presented. This section presents only the data from selected thermistors and oxygen ports. The complete temperature and oxygen data for all thermistors and oxygen ports, as well as the chemistry of leachates are included in the Appendix of this thesis.

The quality of seepage captured from the lysimeters along with temperature data is used to determine the effectiveness of the blending methods used at the trial dump.

4.4.1 Evolution of Leachate Chemistry

This section presents the leachate quality data collected for panel 4 between August 2001 to October 2004, and from May 2001 to October 2004 for panels 5 and 6. The leachates of panel 7 were sampled from February 2002 to May 2004. All blended
panels had already been covered by the waste material for between three and four weeks when the first sampling was conducted.

The results presented here exclude the quality of leachate from panel 8 because of the influence of seepage from the adjacent area. The potential and possibly frequent interception water that came from the adjacent area near panel 8 (as explained in Chapter 2 of this thesis) made it difficult to evaluate whether this observation was caused solely by the blending or was also due to external water contributions from other locations.

4.4.1.1 pH Trend

Figure 4.4 shows the pH measured in the drainage from the lysimeters for each of the limestone blending panels. The initial pH measurement of the leachate from each panel was obtained after the lysimeter was covered by the waste rock material for approximately two to four weeks. At that time, the pH of the leachate for the panels built with the truck dumping methods (panels 4, 5, and 6) was already starting to decrease (lower than 6), while the panel that was constructed by the crusher and the stacker system (panel 7) still maintained neutral pH. As pH measurements continued, the pH for the leachate for the panels built with the truck dumping method decreased further and the pH dropped to below 4 within three months. The minimum pH of 2.5 for panels 5 and 6 was reached after 3 months. However, it took approximately 12 months for the pH of the leachate from panel 4 to decrease to 2.5. Subsequently, the pH of the
Figure 4.4 pH trends in leachates from blended panels

Leachate from these three panels continued to fluctuate between 2.5 and 3 until Year 4 of the experiment. On the other hand, the pH of all the leachate from the panel 7, built by the crusher and the stacker system, behaved strikingly differently. The pH for panel 7 was maintained above 6.5 in the leachate for approximately three years, until the end of the experiment.

4.4.2.2 Acidity and Sulfate

Figure 4.5(a) shows the measured acidity concentrations in the leachates from all panels, except leachate from panel 7 which had a neutral pH. The magnitude and the pattern of acidity concentration for the leachates from panels 5 and 6 were similar, although the decline was delayed for panel 6. The maximum acidity of the leachates
from both panels 5 and 6 was approximately 70,000 mg CaCO$_3$/l and the peak concentration occurred within approximately 18 months after placement of the waste rock. The acidity for the leachate collected from panel 4 was much lower compared to the leachates from panels 5 and 6. The maximum acidity for panel 4 was 14,000 mg CaCO$_3$/l or only 20% of the peak acidity in the leachate collected from panels 5 and 6.

The sulfate concentrations for the leachates collected from the blended panels are shown on Figure 4.5(a). The trend in the sulfate concentrations are similar with the trend observed in acidity. The leachate for panels 5 and 6 had the higher maximum sulfate concentration compared to the leachate from panel 4. The maximum sulfate concentrations in the leachates from panels 5 and 6 were 92,000 mg/l and 89,600 mg/l, respectively, which occurred approximately 18 months after placement of the waste rock. On the hand, the maximum sulfate concentration in the leachate from panel 4 was 19,600 mg/l, which was approximately five times lower than those ones from panels 5 and 6. Although the pH in the leachate from the panel built by the crusher and stacker system (panel 7) was maintained above 6, sulfate concentrations were still detected in the leachate. It can be seen in Figure 4.5(a) that the sulfate concentration for panel 7 varied between 780 ppm and 1,720 ppm. These values were typically two orders of magnitude lower than the sulfate concentration for the leachate produced from the blended panels constructed by the truck dumping method (panels 4 to 6).

The release rates of constituents in the leachate outflows were estimated according to the concentration of the constituents and the volume of the leachate, and
subsequently were divided by the waste rock mass. The release rates for the leachate from the panel dump was estimated by assuming that leachate was generated only by the waste rock mass that was present directly above the area of lysimeters, and these values are reported as gram/ton-day. Figures 4.5(c) and 4.5(d) show the sulfate and acidity release rates for the leachates from the various blended panels, respectively. At the beginning of the experiment, the acidity and sulfate release rates from the alternated truck dumping panel (panel 5) was higher than any other method. After one year, the acidity and sulfate release rates from both the alternated truck dumping blend (panel 5) and the layered truck dumping blend (panel 6) were comparable. The acidity and sulfate release rates for both panels 5 and 6 reached the peak 20 months after placement of waste rock. The maximum release rates in panel 5 were 37.7 g/ton per day for acidity and 56.6 g/t per day for sulfate, and the maximum acidity and sulfate release rates for panel 6 were 50 g/t per day and 54 g/t per day, respectively. The acidity and sulfate release rates for the pre-mixing truck dumping method (panel 4) was much lower compared to the acidity and sulfate release rates for the other truck dumping method (panels 5 and 6). The peak sulfate and acidity release rates for the panel 4 was only 5.7 g/ton per day and 8.1 g/ton per day, respectively. The sulfate release rate from the panel 7, which was constructed by the stacker method, was much lower compared to the sulfate release rates from the blended panels constructed by truck dumping (i.e. panels 4 to 6). The peak sulfate release rate from panel 7 was only 1.3 g/t per day (Figure 4.5c).
Figure 4.5 (a) Acidity concentration trends, (b) sulfate concentration trends, (c) acidity release rate trends, and (d) sulfate release rate trends from blended panels
4.4.3 Temperature Profile

Thermistors strings were placed along the base and exposed tip face of the panels during construction allow for monitoring of temperature within the blended panels, as illustrated in Figure 4.6. Figure 4.7 shows the measured temperature versus time at the base of the blended and non-blended panels. The measuring points were located at the base of the panels 5 meters and 15 m from the toe of the dump where there was no restriction on the oxygen supply. The data in Figure 4.7 does not include measurements at panel 6 at the base since the data could not be recorded due to thermistor damage during construction.

As shown in Figure 4.7(a), the temperatures for the blended panels located 5 meters from the trial dump toe tended to increase during the experiment and reached the maximum after 12 to 18 months, then tended to maintain a plateau until the end of the experiment. This pattern was quite different than the temperature pattern for the non-blended panels which reached the peak at the same time period, but then started to decline through to the end of the experiment, except for panel 3 which the temperature had not yet decreased. The increases in temperatures for the blended panels were also not as high as the non-blended panels. For example, the maximum temperature for the blended panels was 18° C compared to 58 °C for the non-blended panels. In general, the range in temperature recorded at the blended panels constructed by truck dumping (i.e. panels 4 and 5) was similar to the temperature range recorded at panel 7, which was constructed with the crusher and stacker system.
Figure 4.6 Location of instrumentation within the blended panels
Figure 4.7 Temperature versus time at the base of the non-blended and blended panels for thermistors located (a) 5 m and (b) 15 m from the toe of the trial dump
In contrast, higher temperatures and different temperature patterns were recorded from thermistors in the blended panels located 15 meter from the toe of the trial dump, particularly for the panels constructed by truck dumping as shown in Figure 4.7(b). Temperatures as high as 45°C were recorded in the thermistors in panels 4 and 5. The temperatures recorded at panels 4 and 5 were comparable with temperatures recorded at panels 1 and 3, which were non-blended panels. Furthermore, the temperature patterns from panels 4 and 5 were also similar to the patterns of temperature at the non-blended panels (i.e. panels 1 to 3), with the maximum temperature being recorded after 12 months and 18 months, respectively. On the other hand, the temperatures at the thermistor located at panel 7 (15m from the toe of the dump), and constructed by the crusher and stacker system, was much lower compared to temperatures at panels 4 to 6 which were built with the truck dumping. Temperatures that ranged from 8°C to 15°C were measured at panel 7 during the experiment.

High temperatures were also recorded in the thermistors that were placed at the face of panels 4 to 7, as shown in Figures 4.8, 4.9 and 4.10. The thermistors were located approximately 8 m and 16 m from the toe of the dump. The maximum temperatures recorded at thermistors on the face of the dump located at 8 m and 16 m from the toe of panel 4 were 36°C and 47°C, respectively (Figure 4.8). Similar maximum temperature were also recorded at thermistor located at the face of panel 6 approximately 8 m from the toe of the dump, which was 48°C. Higher temperatures
Figure 4.8 Temperature versus depth with time at the face of pre-mixed truck dumping blended panel (panel 4) for thermistor located (a) 8 m and (b) 16 m from the toe of the trial dump
Figure 4.9 Temperature versus depth with time at the face alternating truck dumping blended panel (panel 5) for thermistor located (a) 8 m and (b) 16 m from the toe of the trial dump
Figure 4.10 Temperature versus time at (a) the face alternating truck dumping blended panel (panel 6) for thermistor located 8 m and (b) crushed and stacker dumping blended panel (panel 7) 15 m from the toe of the trial dump
temperatures were measured at panel 5 for both thermistors located 8 m and 16 m from the toe of panel 5. The maximum temperature, as high as 68°C was recorded for these thermistors (Figure 4.9). This value is comparable with temperature recorded at the unblended panels, as presented and discussed in Chapter 3 of this thesis. Elevated temperatures were also measured in panel 7 (Figure 4.10b), but high temperatures, ranging from 30°C to 47°C were only attained near surface at less than 8 m vertical depth, while temperature at the lower depths remained around 20°C.

As can be seen in Figures 4.8 to 4.10, temperatures at different elevations from the same thermistor were not always similar. Since panels 4 to 7 were constructed from limestone as well as blue waste, the heterogeneity of materials contributed to the temperature difference in thermistors. The geochemical results for the samples collected from the panels, is illustrated in Figure 4.11 and is described in the subsequent section to support this observation.

4.4.4 Evaluation of Blending Method

Table 4.1 shows the bulk ANC/MPA ratio for each blend of the trial dump based on samples collected from the slope of the dump during construction. Approximately 9 samples were collected from each panel associated with the different methods of blending. These samples were then analyzed for NAG, ANC, and total sulfur content. The average bulk ANC/MPA ratio varied from 2.1 to 534. The minimum bulk ANC/MPA ratio occurred in panel 7. This same panel had the least variability in the ANC/MPA ratio as compared to the other panels.
Table 4.1 Summary of limestone blending methods and performance

<table>
<thead>
<tr>
<th>Panel</th>
<th>Blending Methods</th>
<th>ANC/MPA ratio</th>
<th>Blending Performance</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Average</td>
<td>Minimum</td>
</tr>
<tr>
<td>4</td>
<td>Mixed – truck dumping</td>
<td>2.4</td>
<td>0.04</td>
</tr>
<tr>
<td>5</td>
<td>Alternated - truck dumping</td>
<td>6.0</td>
<td>0.02</td>
</tr>
<tr>
<td>6</td>
<td>Layered – truck dumping</td>
<td>534</td>
<td>0.08</td>
</tr>
<tr>
<td>7</td>
<td>Crushed – stacker dumping</td>
<td>2.1</td>
<td>0.8</td>
</tr>
</tbody>
</table>

Figure 4.11 shows the plot between sulfur content and ANC of each sample from panels 4 to 8 that were used to examine the distribution of the ANC/MPA ratio. This plot also includes lines that show the ANC/MPA ratio at different values. If the ANC/MPA ratio is less than 1, it indicates that ARD is potentially generated. A higher ANC/MPA ratio will increase the possibility of preventing ARD. Considering the size variability of the waste rock particle, an ANC/MPA ratio of 2 was considered adequate to prevent acid rock drainage (Miller, 2003a). As can be seen in Figure 4.11 and Table 4.1, the ANC/MPA ratio for each individual sample varied within each panel. For example, several samples from panels 5 and 6 had ANC/MPA ratios ranging from near zero to greater than 5. An ANC/MPA ratio above 5 occurred in samples having high ANC and low sulfur values, which indicated the blended samples contained mostly limestone. On the other hand, the ANC/MPA ratios of near zero were shown in samples containing high sulfur and low ANC value, thus indicating that the blended samples contained primarily acid waste rock. The data demonstrated that adequate blending was not achieved across panels 5 and 6. Although the ANC/MPA ratios for individual
was not achieved across panels 5 and 6. Although the ANC/MPA ratios for individual samples from panels 4 and 7 were not as high as from panels 5 and 6, the ANC/MPA ratio of samples from panels 4 and 7 were less variable than the ANC/MPA ratio of panels 5 and 6 (Table 4.1). These data suggest that better blending was achieved for panels 4 and 7 compared to panels 5 and 6.

The blended limestone and acid rock panels were designed with a high bulk ANC/MPA ratio in order to ensure that there was a sufficient alkaline source to prevent the acid seepage produced from the acid waste rock. According to the measured sulfur
content of the acid waste rock (i.e. 2% S) and the ANC of the limestone, a blending ratio of 25% limestone to 75% blue waste was expected to provide a bulk ANC/MPA ratio of greater than 5, as shown by the meso-scale 500-tonne test pad, (Miller et al. 2003a). This ratio was considered adequate to prevent the acid generation, as shown by results from the previous smaller experiments for the columns tests and meso-scale test pads. The results in Table 4.1 show the average ANC/MPA value for samples collected from each panel to be greater than 2; however, the ANC/MPA ratio of the individual samples for each panel varied in a wide range, particularly for panels 5 and 6. The results for some of the samples indicated that blending between the limestone and the acid rock did not occur. The high variability of the ANC/MPA ratio that ranged between 0.02 and 1537 for the samples showed that thorough mixing was difficult to achieve over the entire region of the panels.

As shown in Figure 4.11 and Table 4.1, the panels that were constructed using the truck dumping method (panels 4 to 6) failed to prevent acid drainage, even within a short period of time, while the panel built with the crusher-stacker system (panel 7) performed very well. The performance of various blending methods was likely determined not only by the bulk ANC/MPA ratio, but also by the intimacy of blending, which can be measured in the ANC/MPA for each individual sample. For example, panel 7, which had the smallest bulk ANC/MPA ratio and the smallest variation in individual ANC/MPA ratio, maintained neutral pH throughout the experiment. On the other hand, panel 6 which had the highest average ANC/MPA ratio but also had the highest
variation in ANC/MPA ratio continued to produce low pH leachate. Scharer et al. (2000) also reported bulk ANC/MPA ratios as high as 5 produced acidic drainage in the long-term.

Although it was shown not to work effectively over the long term, the pre-mixing treatment for panel 4 had the best performance among the truck dumping methods. Furthermore, the ANC/MPA ratio of samples from panel 4 were less variable compared to those from panels 5 and 6 which were constructed by truck dumping. It is considered that there was likely greater contact between the alkaline solution from the limestone with the acid seepage from the waste rock in the mixed panel 4, thus producing a gradual increase in pH along with lower acidity and sulfate concentration. The pre-mixing method involved blending approximately 500 m$^3$ of material (i.e. three trucks of acid waste rock and one truck of limestone). Although heavy equipment such as a front loader, a dozer, and an excavator were utilized, complete intimate blending was still difficult to achieve. Performance of the blended material might have been improved if smaller amounts of materials had been involved. However, it would not have been practical for the day-to-day operation, when considering the load from one truck splits into several piles, since it would be time consuming and require more equipment.

The end dumping technique was used for placing the waste rock in the alternating and layering methods in panels 5 and 6. Since the waste rock consisted of materials with different particle sizes, segregation occurred while the truck dumping
method was deployed. Coarser particle material tended to accumulate at the dump toe while the finer particle material tended to be retained at the crest of the dump (Poisson et al., 2009; Nichols, 1986). The ROM limestone and the acid waste rock materials used for this experiment tended to have different particle size distributions due to the variation in hardness between the limestone and the acid waste rock. As a result, the distribution of particle size for the limestone and the acid rock varied along the face of the dump. The observed variability of the ANC/MPA ratio along with the identification of the unblended conditions between the samples collected at different elevations of the dump face support the argument for segregation.

Panel 6 was designed to have a three-meter thick acid waste rock layer followed by a one-meter thick layer of limestone. The segregation that occurred during implementation of the end dumping method made it difficult to maintain a uniform thickness along the layers of each material, thus an adequate ANC/MPA ratio might not have been achieved at every location along the profile of the panel. Some samples collected from panel 6 indicated they were either pure limestone or acid waste rock. The method of truck dumping that produced the layered system in panel 6 isolated the limestone from the acid waste rock. Layer arrangements with 1 meter thick limestone and 3 meter thick acid waste rock did not provide adequate contact between the limestone and acid waste rock, which is a key element in achieving effective blending. The application of a thin layer was found to be the most favorable for the blending of limestone and acid rock as suggested by Kempton et al. (1997).
Figure 4.12 shows the particle size distribution for the samples collected from panels 4 to 7. The distribution of grain size for samples collected from panel 7, which were constructed with stacker dumping, was more uniform compared to samples collected from the panels constructed with truck dumping (i.e. panels 4 to 6). The waste rock materials placed in the panel 7 were crushed prior to the dumping, therefore both the limestone and the blue waste had similar particle size. In contrast, the limestone and the blue waste dumped at panels 4 to 6 were Run of Mine (ROM). Due to the difference in hardness between the limestone and the blue waste, particle sizes from the limestone tended to be coarser than those for the blue waste. Although the ANC/MPA ratio for the different size fraction was not assessed in the samples, the waste rock that had a relatively uniform size distribution in panel 7 promoted the blending across the different size fractions. As a result, the overall drainage of panel 7 remained neutral with low dissolved metal concentration and moderate sulfate concentration. Mehling (1997) classified a dump with this quality of drainage as a near ideally blended dump.

NAG test results for blue waste samples collected from the blended panels indicated that NAG values of several samples were higher than the upper limit of NAG values of typical blue waste (e.g. 35 kg H₂SO₄/ton). As a result, the actual bulk ANC/NAG ratio for the blended panels was lower than the intended design since the blended panel was constructed based on a mass ratio between the blue waste and the limestone (e.g. 3 parts of blue waste to 1 part of limestone). The bulk ANC/NAG ratio that was
Figure 4.12 Particle size distribution for samples from blended panels constructed with (a) pre-mixed truck dumping (panel 4), (b) alternated truck dumping (panel 5), (c) layered truck dumping (panel 6), and (d) crushed stacker dumping (panel 7)
lower in the built panels compared to the intended design means that there was not sufficient excess of neutralization capacity in the blended panels. The limited excess of the neutralization capacity, together with incomplete blending between the acid rock and the limestone, caused a failure of the blended panels that were built by truck dumping to prevent acid generation.

4.4.5 Oxidation Rate in the Limestone Blended Panels

The elevated temperature and sulfate content in the effluent clearly indicated that sulfide mineral oxidation took place inside the blended dump. Results of oxygen measurements, as shown in Chapter 3, indicate that oxygen concentrations between the blended and the non blended panels was not significantly different. However, the observed temperatures, concentrations and release rates for acidity and sulfate in a particular blended panel (i.e. panels 4 and 7) were low. The oxidation rate (IOR) for the blended panels were assessed in order to determine whether the lower constituent release rates and temperatures observed in the blended panels were due to a lesser quantity of acid waste rock mass involved in the blended panels, or caused by a low sulfide mineral oxidation rate. As described previously, the IOR is estimated based on the data for sulfate release rates from the waste rock. The following equation was used to calculate IOR and oxygen consumption rates equal to 0.583 times the sulfate release rate based on the stoichiometry of reaction (Bennett et al., 2000).

\[
\begin{align*}
\text{FeS}_2 + \frac{7}{2} \text{O}_2 + \text{H}_2\text{O} &\rightarrow \text{Fe}^{2+} + 2 \text{SO}_4^{2-} + 2 \text{H}^+ \\
(4.3)
\end{align*}
\]
Table 4.2 shows the results of the oxidation rate calculations for the blended panels based on the mass of blue waste rock. This table also includes the oxidation rate for the panels that contained only blue waste (unblended), for which the detailed results are presented in Chapter 3. The calculation for the blended panels was normalized according to an assumption based on the quantity of blue waste involved, which was 25% less than that of the unblended panels. As can be seen from Table 4.2, the oxidation rate for the blended panels constructed with alternating and layered truck dumping (i.e. panels 5 and 6) are comparable with the oxidation rate for the unblended panels. The oxidation for pre-mixed truck dumping (i.e. panel 4) was approximately two times lower than the oxidation rate of the unblended panels, while the oxidation rate of panel 7, which was constructed by crusher and stacker system, was approximately one order of magnitude lower than the oxidation rate of the unblended panels. The calculations for oxidation rate for waste rock materials from panels 4 and 7 are consistent with the results shown by Mehling et al. (1997) who stated that the sulfide oxidation rate may be reduced when highly reactive materials such as limestone are applied.

Sulfide minerals such as pyrite initially react with oxygen to produce dissolved ferrous, sulfate and hydrogen ions, as shown in Equation 4.3. The ferrous ion generated is furthered oxidized to ferric ion as shown in Equation 4.4. This reaction can be accelerated in the presence of microorganisms such as *T. ferrooxidans*. The activity of *T. ferrooxidans* is pH dependent, with optimal conditions occurring around pH 3 (Singer
Table 4.2 Oxidation rates of the blended and unblended panels

<table>
<thead>
<tr>
<th>Period</th>
<th>Blended panels</th>
<th>Unblended panel</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Pre-mixed Truck dumping (panel 4)</td>
<td>Layered Truck dumping (panel 6)</td>
</tr>
<tr>
<td>Year 1</td>
<td>1.4 X10(^{-11})</td>
<td>3.4 X10(^{-11})</td>
</tr>
<tr>
<td>Year 2</td>
<td>5.8 X10(^{-11})</td>
<td>2.9 X10(^{-10})</td>
</tr>
<tr>
<td>Year 3</td>
<td>4.9 X10(^{-11})</td>
<td>3.5 X10(^{-11})</td>
</tr>
<tr>
<td>Overall</td>
<td>4.0 X10(^{-11})</td>
<td>1.3 X10(^{-10})</td>
</tr>
</tbody>
</table>

and Stumm, 1970; Evangelou et al. 1985; Taylor et al. 1984; Blowes et al. 2004). The presence of limestone in the blended panels can be expected to increase the pH to a value greater than 3, thus reducing the catalytic effect of the microorganism and further decreasing the rate of pyrite oxidation (Evangelou, 1995; Nicholson et al. 1990).

\[
\text{Fe}^{+2} + \frac{1}{4} \text{O}_2 + \text{H}^+ \rightarrow \text{Fe}^{+3} + \frac{1}{2} \text{H}_2\text{O} \quad (4.4)
\]

The high precipitation at the site may have a direct impact on the oxidation process within the blended panels. The continuous rainfall and associated flushing of the waste rock throughout the year affects the presence of the secondary minerals within the system. Although the waste rock is flushed almost continuously throughout the year, the results of the mineralogy test conducted on the samples collected during the excavation of panel 7 showed the existence of secondary minerals. Miller (2007) reported the presence of jarosite as a secondary mineral attached to the surfaces of
pyrite particles. The same mineral was also observed in the samples collected from the 500-tonne test pad which had the limestone blending and limestone cover as reported by Miller et al. (2003a).

Neutral pH also affects the presence of oxidation products such as Fe$^{+3}$, which acts as an oxidizing agent and reacts with pyrite to produce more ferrous ion, (i.e. shown in Equation 3.3). Fe$^{+3}$ concentrations in solution are also controlled by pH. At a pH higher than 3, Fe$^{+3}$ will precipitate and the oxidizing agent will be eliminated from the system, slowing the rate of pyrite oxidation. This precipitation iron can form an armouring or coating layer on the surface of the pyrite particles, which in turn, can significantly slow oxygen diffusion for the surface of the pyrite particle resulting in a reduction of the oxidation rate (Lapakko et al. 1997; Nicholson et al. 1990; Caldeira et al. 2003). The neutralization of the acidic leachates with limestone produces high calcium and sulfate concentrations. If the concentrations of both species are sufficiently high, gypsum will be precipitated as well. The presence of the gypsum in the system also potentially reduces acid generation (Filipek et al. 1995).

### 4.4.6 Internal Structure of Blended Panels

The excavation conducted at the end of the experiment provided additional information regarding the internal structure of the blended panels. The excavation conducted on panels 5, 6, and 7 revealed that the internal structure of the panels constructed using the truck dumping method was different as compared to the panel constructed with a crusher and stacker system as shown in Figure 4.13.
Figure 4.13 Excavation of blended limestone panels built with a) truck dumping (panel 5), and b) crusher and stacker dumping (panel 7)
Layering of the acidic and limestone materials was observed within panels 5 and 6, as shown on Figure 4.13(a), which were constructed with the truck dumping method. Pockets and layers of different material were formed when the blending of truck dumping method was utilized. Relatively fine layers of weathered acidic rock, coarse layers of limestone, and free matrix zone acidic rock or limestone layers were distinguished along the walls of the trenches in both panels 5 and 6. The widths of these layers were generally 0.1 m to 1 meter thick. Visual observation also indicated that some acid rock layers had limited contact with the limestone layers. Most of the acid waste rock encountered in panels 5 and 6 was found to be in an oxidized condition as identified by the presence of secondary oxidation products. The pH of the samples collected from the different oxidation zones in panels 5 and 6 ranged from approximately 3 to 5.5, as shown in Figure 4.14, with only two of the eleven samples having a pH greater than 5.

In contrast, the layering of acidic waste rock and limestone was not found in panel 7, as shown on Figure 4.13(b). Neither the limestone nor the acid waste rock was distinguishable due to the relatively intimately blended conditions achieved. Furthermore, the oxidized acid waste rock was not observed on panel 7. The paste pH of samples collected at all locations in panel 7 had a neutral pH as shown in Figure 4.14. The close contact between the limestone and acid waste rock will produce the circumneutral pH pore water that will remove the ferric ion from solution and then will
Figure 4.14 Paste pH of excavated samples from panels 5 to 7

slow the oxidation rate and allow for the formation of the armouring layer (Miller et al. 2009).

The existence of coarse and fine layers in the blended dump is also believed to affect the flow path of both alkaline and acidic leachate generated from every layer. This hypothesis will be furthered discussed in Chapters 5 and 6. As high precipitation occurs at the site, the infiltrating water may flow through layers preferentially. When a preferential flow path develops in a coarse or free matrix zone, the alkaline leachate flow path might be different from the flow path for acidic leachate, particularly when a specific waste rock type such as limestone is coarser than the acid waste rock.
The coarse layers also created pathways for transporting oxygen and water vapor. The monitoring results showed high temperatures were recorded most frequently for the thermistors buried along the slope of the dump for blended panels constructed with the alternate truck dumping method as illustrated in Figure 4.9, where the thermistor was buried on the face of panel 5 and located close (i.e. 8 meter and 16 meters) to the toe of the dump. High temperatures were also measured at different depths with a maximum temperature as high as 68°C recorded.

4.4.7 Implications of Dump Design

The failures observed in the blended panels illustrates that the design of blended dumps based solely on bulk ANC/MPA ratio criteria is not sufficient for the prevention of ARD. The level of intimate blending of the limestone with the acid waste rock must also be considered.

Morin and Hut (2000) introduced the concept of the discrete mixing zone for the truck dumping method in order to prevent acid generation. This concept is based on applying sufficient ANC material zones among the PAF zones, such that the acid leachate from the PAF material zone can mix with the alkaline/neutral leachate from the ANC material zone and then be neutralized. Since the peak acidity concentration in leachate from the acid rock was typically as high as 90,000 mg/l, and the alkalinity in the limestone leachate was only between 50 mg/l and 100 mg/l, the number of required discrete mixing zones is very high given the quantity of acid waste rock at the Grasberg Mine. Although a significant quantity of limestone is available at the mine, the amount
of limestone that can be produced is limited by the mining sequence. In addition, considering that the height of full-scale dumps at Grasberg Mine can reach several hundred meters with surface areas measuring thousands of hectares, implementing a discrete mixing zone would be extremely challenging and likely not be practical given the large volume of waste rock to be managed.

Some segregation also occurred within the blended panel constructed by the crusher and stacker system; however, it was less pronounced compared with the blended panels deposited with truck dumping. The crusher and the stacker system produced material that was relatively uniform in particle size; therefore segregation was reduced during deposition and improved contact between the acidic rock and the limestone was achieved. When the crusher and stacker method is scaled up to a full-scale stacker-built dump that has 400 m faces, further segregation may occur and thus may lead to preferential flow paths. Thus, the impact of preferential flow on the quality of leachate should be considered when developing an adequate ANC/MPA ratio to be implemented for the full-scale, crusher-stacker built dump.

The mining sequence plays an important role in the success of blending for a large-scale dump built by the crusher and stacker system. Lack of limestone production during certain time periods will affect the availability to achieve an adequate ANC/MPA ratio. Considering the 400 m height of the dump and the short time period of no limestone production might occur, stockpiles should be considered mandatory. In
addition, developing ANC/MPA ratios based on short time periods such as a week or a month may be more suitable than applying bulk ANC/MPA ratios for the whole dump.

4.5 Conclusions

Blending methods for ARD mitigation were evaluated for a large-scale trial dump at the Grasberg Mine to assess the efficiency of different blending methods for limestone and acid waste rock, including truck placement with pre-mixing, alternate dumping, and layering methods, and the crusher-stacker system. The safe ANC/MPA ratio that was expected to prevent acid generation was applied in the design of this trial dump according to the results obtained from the previous smaller scale experiments.

The trial dump results demonstrated that complete blending could not be achieved in the panels built with the truck dumping method, as shown by wide variability in the ANC/MPA ratio of the samples. Although the average bulk ANC/MPA ratio was high for the panels built by truck dumping with the alternating and layering methods (panels 5 and 6), these panels were not effective in preventing acid drainage. The wide range of ANC/MPA ratios from individual samples (from 0.02 to 1,537) suggested that adequate blending was not achieved across these two panels. On the other hand, improved intimate blending was achieved when a crusher and stacker system was used to construct the blended panel. The particle size for the limestone and acid waste rock materials in this panel promoted adequate blending as shown by the narrow range of ANC/MPA ratio for individual samples (0.8 to 6). Although the average
bulk ANC/MPA ratio was only approximately 2.1, it was found to be quite effective in minimizing acid drainage.

The elevated temperatures that were recorded within the blended panels, particularly from thermistors placed in the face of the panels indicated sulfide oxidation occurred in the blended materials including the panel built with a crusher and stacker system (panel 7), which achieved nearly ideal blending condition. However, the measured sulfide oxidation rate in panel 7 was approximately two orders of magnitude lower compared to the oxidation rate observed for acid waste rock in the unblended panels. Thus, it was shown acid drainage could be prevented or at least greatly reduced in the crusher-stacker built panel. Calculations for the sulfide oxidation rate indicated that a low oxidation rate was only achieved when effective blending occurred since the sulfide oxidation rate was not reduced in the blended panels constructed with the truck dumping method.

When applying the crusher and stacker system to a large-scale dump operation, it is necessary to maintain an adequate ANC/MPA ratio in the waste rock pile throughout the operation. Since the mining sequence plays an important role in blending performance, establishing an adequate ANC/MPA ratio for a short period of time is recommended to ensure that the potential acid material receives adequate neutralizing capacity at the whole range of particle size.

Segregation that occurs on the face of a large-scale dump built with the crusher-stacker system might create multiple layers. The high rainfall rate at Grasberg mine may
also promote preferential flow within particular layers. Therefore, further study was conducted in order to understand the occurrence of preferential flow within potential layers for the high precipitation environment, as well as to observe the influence of preferential flow on the quality of the drainage. This aspect of the study will be addressed subsequently in Chapters 5 and 6.
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Chapter 5 Behavior of Water Flow in Layered Mine Rock Stockpiles: A
(RMeso-Scale Laboratory Experiment)

5.1 Introduction

Mining involves the removal of significant quantities of waste rock to provide access to ores within open pits. The waste rock is almost always stored in engineered stock piles and the height of the rock piles can reach several hundred meters with surface areas covering thousands of hectares (Ritchie, 1994). Either haul trucks or a boom stacker is used to construct the waste rock piles. When the stacker boom is used, large deposits are constructed as a series of smaller cone piles. Conversely, a pile with one or more benches is constructed as a result of a truck dumping method.

Several different dumping methods, such as end-dumping, push-dumping, and free-dumping may be employed when haul trucks are used. Each different method of dumping creates waste rock piles with distinguishable physical characteristics (Morin et al. 1991; Nichols, 1986). For example, segregation of the various sizes of rock material tends not to occur when the dump is constructed with the free-dumping method since the waste rock does not ravel down a tip face. In contrast, piles that are built by push-dumping and end-dumping methods promote the coarser waste rock particles to ravel

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4 A version of this chapter will be submitted for publication. Andrina, J., Wilson, G.W. and Miller, S., Behavior of Water Flow in Layered Mine Rock Stockpiles: A Meso-Scale Laboratory Experiment
down slope to deeper locations while the fines stay located near the top of slope. The end-dumping method causes more extensive segregation than the push-dumping method (Nichols, 1986).

The structure becomes even more complex as the waste rock pile develops. Herasymuik (1996) and Fines et al. (2003) showed the existence of highly segregated, inclined, coarse and fine grain layer textures within the internal structure of waste rock piles that were excavated at mine sites in North America. Similar layers were also observed during the excavation of the trial dump at the Grasberg Mine in Indonesia, although the layers were relatively thin since the height of the trial dump was only 20 meters (Andrina et al. 2006). The geophysical investigation that was conducted by Possion et al. (2009) which were followed by the hydrogeological and chemical analyses also confirmed the existence of two different layers in the the test pit at Laronda Mine. The existence of the inclined coarse and fine layers in the waste rock pile influences the flow path of water that infiltrates at the surface of the waste rock and this subsequently affects the quality of the effluent from the pile.

Water flows in the waste rock pile due to a hydraulic head gradient. The hydraulic gradient consists of the gravitation head and the hydrostatic pressure, as shown in Equation 5.1.
\[ h = z + \frac{u_w}{\rho \cdot g} \]

where: \( h \) = total head, m

\( z \) = gravitational head, m

\( \frac{u_w}{\rho \cdot g} \) = pressure head, m

\( u_w \) = pore-water pressure, kg/m²

\( \rho_w \) = density of water, kg/m²

\( g \) = gravitational acceleration, m/s²

The gravitation head occurs due to a difference in elevation between two locations resulting in a gravitational flow, while the hydrostatic pressure is governed by capillary forces within the waste rock materials that leads to capillary flow.

Waste rock piles are stored under unsaturated conditions in which unsaturated flow within the pile is governed by grain size distribution, the porosity and the texture of the pile (Smith and Beckie, 2003). Both the grain size and distribution of the particles affect matric suction and hydraulic conductivity. The soil-water characteristic curve (SWCC) defines the relationship between matric suction, water content and unsaturated hydraulic conductivity for the waste rock. The SWCC of fine materials is usually measured using a small pressure plate (or a Tempe cell) in which the samples are exposed to several increments of suction (Fredlund and Rahardjo, 1993). Since it is difficult to measure the SWCC for coarse particles with a Tempe cell, predictive methods are often used to estimate the SWCC of the coarse materials (Aubertin et al.
2003; Fredlund et al. 2002; Arya et al. 1999). The hydraulic conductivity of unsaturated material depends on the water content in the pores (Lu and Likos, 2004). The unsaturated hydraulic conductivity of materials can be predicted based on the SWCC for the material together with the saturated hydraulic conductivity (Mbonimpa et al. 2006; Leong and Rahardjo, 1997; van Genuchten et al. 1980; Mualem, 1976). Under saturated conditions, the coarse material has the capacity to transfer water at much higher rates than the fine material. At higher values of suction, or under less saturated conditions, the finer material may transfer more water faster than the coarse material.

A review of the literature revealed that previous studies for coarse and fine layers in waste rock piles primarily emphasized flow behavior. For example, most of the published works regarding coarse and fine layer structures have been directed at soil cover systems and capillary barriers to reduce infiltration and oxygen entry into waste dumps (Bussière et al. 2007; Parent and Carbal, 2006; Bussière et al. 2003b; Khire et al. 2000; Stormont and Anderson, 1999; Nicholson et al. 1989) or to improve the stability of soil slopes (Tami et al. 2004). Hence, most of the record has been directed at layers of material at the ground surface such as cover systems. A limited number of laboratory experiments that investigated flow behavior in vertical layers of coarse and fine material have been completed (Wu et al. 2007; Li, 2000; Newman, 1999;). The laboratory results confirmed the role of rainfall intensity in determining the water flow path during infiltration. It was shown that water tends to flow mainly through the coarse layer at high rainfall intensities. The effect of macropores with non-capillary flow
behavior is thus superimposed on the finer/coarser matrix zone. On the other hand, the water flows through the fine layer at low rainfall intensities.

The flow behavior of waste rock piles that have inclined coarse and fine layers within their internal structure have been investigated using numerical models. Several models have been developed for analyzing the flow in waste rock piles that contain single or multiple dipping, interbedded layers (Fines, 2006; Fala et al. 2005; Weeks et al. 2004; Wilson et al. 2000). These models are based on the assumption that capillary flow controls flow in both layers. The model results indicated that the flow was affected by the angle of inclination, the contact length between the layers, the hydraulic properties of the waste rock and the geometry of the dump. Although all of these models provided information regarding the parameters that influence flow behavior in a pile, the implications for drainage quality was still not completely understood.

The laboratory experiment for this study was designed to develop a fundamental understanding of the flow path within inclined coarse and fine layers of waste rock, as well as the effect of the flow path on drainage quality. This laboratory experiment was set up because it was difficult to conduct the investigation at the waste rock dump without knowing the location of the existence of coarse and fine layers in the existing dump. The apparatus for the experiment was built specifically to mimic the internal structure of a waste rock dump that was constructed by truck placement using the end-dumping method. Both acid waste rock and limestone were used to construct
three panels with different layer arrangements. Different rates of rainfall were used in
the experiment to simulate different climates at different locations.

The present chapter focuses on flow behavior within the layers of rock piles with the
specific objectives of:

- Understanding the fundamental flow mechanism in the inclined coarse and fine layers
  of waste rock;
- Understanding the effects of the inclined coarse and fine layers on the flow path;
- Assessing the effects of rainfall intensity on the flow path.

The associated analyses regarding the chemistry of the leachates collected from the
experiment are presented in Chapter 6.

5.2 Experimental Design

Three meso-scale panels with varying dimensions were built using 10 mm-thick
acrylic sheets. The first panel was 1,750 mm long, 1,500 mm high, and 250 mm wide.
The second panel had the same length and width as the first panel, but its height was
increased to 2,000 mm. The third panel had the same width and height as the first
panel, but its length was increased to 2,250 mm. Figure 5.1 shows a photograph of a
meso-scale panel prior to filling it with waste rock.
Figure 5.1 Meso-scale panel 2

Holes were drilled in the base of the panels to collect drainage from the individual layers within the panels and rigid PVC tubes with 25-mm diameters were installed in each hole. Prior to the installation of the PVC tubes, a piece of acrylic sheet that consisted of several small drain holes was glued to the PVC tubes to promote better drainage. A flexible hose was attached to each tube, and the other end was connected to a plastic container. Acrylic strips were installed at the base of each panel to serve as dividers between each layer and to prevent any cross-over flow between layers during high rainfall application rates. Before placing the waste rock layers into the panels, a filter material with particle sizes of approximately 6 mm was installed at the
base of the panel at a depth of approximately 30 to 50 mm to prevent the fine particles from clogging the drain holes. Drainpipes were also installed along the bottom edge of the slope of each panel to collect any leachate that might flow onto the slope. Figure 5.2 shows the drainage system in the panel.

![Basal drain and Basal slope drain in the meso-scale panels](image)

**Figure 5.2 (a) Basal drain and (b) basal slope drain in the meso-scale panels**

The waste rock materials used for this laboratory experiment were shipped from the Grasberg Mine, and were comprised of two different geochemical rock types. The first type was an acid rock, namely red waste, which contained about 5% sulfur. It was fresh rock obtained directly from the pit blast and had various particle sizes, ranging up to a maximum size of approximately 250 mm. The second rock was limestone that came from the limestone quarry. It had been crushed prior to shipment and thus had relatively smaller and more uniform particle sizes compared to the red waste. The
maximum particle size for the limestone was about 50 mm. To obtain the particle size required for the laboratory experiment, both the red waste and the limestone were crushed with a jaw crusher and a cone crusher, and then a sieve was used to obtain the desired particle size.

The fine-grained material was prepared such that its maximum particle size was less than 10 mm. The majority of the material passed through the 4.75 mm sieve. Furthermore, the particle size distribution of the fine grain was prepared such that it would be similar to the fine portion of the typical Run of Mine (ROM) material, for example, the waste rock used in the trial dump that was described in Chapter 2 of this thesis. The coarse-grained material contained particle sizes ranging between 10 mm and 25 mm. The maximum size of the coarse-grained particles was determined based on a minimum 10:1 ratio between the layer thickness and the maximum particle size. Figure 5.3 shows the grain size distribution for fine-grained ROM material from the trial dump, as well as the coarse-grained and fine-grained red waste and limestone that were used for the experiment.

The panels were assembled with the fine and coarse waste rock layers adjacent to each other. The layers were approximately 250 mm thick and placed at an angle of 37° (i.e. angle of repose). This angle reflects the majority of current practice for constructing waste rock piles at mine sites worldwide. The first panel consisted of seven layers, alternating between coarse-grained and fine-grained acid rock, as shown in Figure 5.4. The same arrangement of layers was used for the second panel, as shown in
Figure 5.3 Grain size distribution of coarse and fine materials

Figure 5.4 Layer arrangement at panel 1
Figure 5.5, with the addition of a 500-mm-deep horizontal layer of coarse limestone placed on the top of each acid rock layer so that the effect of the alkaline solution (drainage from the limestone layer) on leachate quality could be assessed. The third panel had nine layers, alternating with limestone and acid rock. The fine and coarse layers in the third panel were arranged to accommodate a different combination of coarse and fine layers, as shown in Figure 5.6. In general, the limestone coarse layers were placed within the constructed panel since coarse layers of limestone were dominant in the internal structure of the limestone blended panels in the trial dump, as observed during the excavation of the blended panel in the trial dump (Chapter 4).

![Diagram of layer arrangement at panel 2](image)

**Figure 5.5 Layer arrangement at panel 2**
A rainfall simulator was designed to provide a range of uniform and constant fluxes (i.e. rainfall intensity) to every layer. The system consisted of a water supply tank, multi-channel peristaltic pumps, distribution hoses, and application sponges, which were placed on top of each layer to promote an even distribution. Figure 5.7 shows the water distribution system. The top layer of the panels was covered with aluminum foil to limit water loss due to evaporation. Several holes were also drilled on the side walls of the panels at the selected layers to ensure a supply of oxygen into the panel and these were kept open for the duration of the experiment.
Figure 5.7 Water distribution system

The water was applied to the top of the panels (excluding the top of layer 1) and along the outer slope of layers 7 for panels 1 and 2, and layer 9 for panel 3. Since a uniform flux application for every layer was crucial to the experiment, the rainfall application was calibrated to confirm the uniformity of flux application. For each flux application rate, the calibration was run for 30 minutes after steady state was achieved.

Instrumentation, including thermocouples, tensiometers, and polyethylene tubes for measuring oxygen were installed at heights of 500 mm and 1,000 mm for all panels. In addition to these locations, instrumentation for the second panel was placed at a height of 1,500 mm. Elbow tensiometers, with 1-bar ceramic tips, were installed in the panels to monitor matric suction, and a digital read out was used to read the suction. The suction measurement of tensiometer ranged from 0 kPA to 80 kPa with
accuracy approximately 0.2 kPa. The precision of the digital read out was around 0.01 kPa. All of the instruments were installed during placement of the waste rock layers. The installation was carefully controlled to ensure that all of the instruments maintained good contact with the waste rock material while preventing any damage to the equipment. Once installed, all of the instruments remained in the panels until the experiment was completed. Figures 5.8, 5.9 and 5.10 show the instrumentations at the panel 1, 2, and 3, respectively.

Figure 5.8 Instrumentations at panel 1
Panel 2

\[\text{NOT TO SCALE}\]

- = Tensiometer
○ = Thermocouple
△ = Gas tubing

150 cm

100 cm

50 cm

Figure 5.9 Instrumentations at panel 2

Panel 3

\[\text{NOT TO SCALE}\]

- = Tensiometer
○ = Thermocouple
△ = Gas tubing

110 cm

50 cm

Figure 5.10 Instrumentations at panel 3
There was a possibility that water could be transferred from the fine layer to the coarse layer at the base part of the panel due to the build-up of pore water pressures within the fine layer and low suction in the coarse layer (i.e. cross over flow or drip surface breakthrough). Therefore suction lysimeters, which were made of 1-bar ceramic material, were inserted into the panels near the base of each layer and connected to leachate collection containers. Suctions to each lysimeters were controlled by placing the discharge point of each container at specific elevations below the base of each layer, as shown in Figure 5.11. The suctions were varied between 0 kPa, 1 kPa, 2 kPa and 4 kPa by changing the elevation of the collection containers.

Figure 5.11 Suction lysimeters
A series of infiltration tests was subsequently conducted by applying different fluxes to represent different climate types. For each infiltration rate, the lysimeters were also set at different bottom boundary conditions by setting the suctions of the lysimeters in the range of 0 kPa to 4 kPa. The volumes of leachates collected were monitored and recorded daily. Matric suction was measured at regular intervals in the profile of each layer at 500 mm and 1,000 mm above the base of panels 1 and 3, and at 500 mm, 1,000 mm and 1,500 mm above the base of panel 2, depending on the experimental program.

The representative samples of coarse and fine grain materials was analyzed for grain-size, saturated hydraulic conductivity and a soil water characteristic curve according to methods described in ASTM-D422-63 (ASTM, 2007), ASTM-D2434-68 (ASTM, 2006), and Fredlund and Rahardjo (1993), respectively.

5.3 Results and Discussion

This section provides the results obtained for laboratory tests conducted on the waste rock samples used in the meso-scale experiment. The laboratory tests included physical characterization of samples including grain size, hydraulic conductivity and Soil Water Characteristics Curve (SWCC). The leachate quantity collected from each layer for the different rates of rainfall are documented in this section along with measurements of matric suctions. Data for the leachate quantity together with matric suctions measured in each layer of the panels are further examined in order to provide insight
into the flow mechanism within the coarse and fine layers, and also to help determine the key factors that control the flow of water in these layers.

5.3.1 Characteristics of Materials

The particle size distributions for both the acid waste rock (i.e. red waste) and limestone were presented previously in Figure 5.3. In general, the grain size distributions between red waste and limestone are similar. Most of the coarse-grained particles had diameters ranging between 10 mm and 25 mm. The coefficients of uniformity of the coarse grains were 1.6 and 1.8 for the red waste and the limestone, respectively, which confirmed that the materials were well sorted. The grain size distribution of the coarse material showed that the particles were comprised of 98% gravel and 2% sand for the red waste, and 99% gravel and 1% sand for the limestone. The gravel size particles of red waste were dominated by medium gravel size (69%) and coarse gravel size (29%), and only 2% of the gravel was fine. The gravel size particles for limestone mainly contained medium gravel size (57%), and coarse gravel size (42%), and only 1% of fine gravel. The particle size for most of the fine materials in the red waste and limestone was less than 4.75 mm in diameter. The fine grained material of red waste contained 54% sand, 35% fine gravel, and 11% silt and clay size. The sand consisted of 48% coarse sand, with 28% medium sand, and 24% fine sand. The fine grained size of limestone contained more fine gravel compared to the red waste. The coarse limestone contained 49% fine gravel, 46% sand, and 5% silt and clay. The sand consisted of 59% coarse sand, 27% medium sand, and 14% fine sand. The uniformity
coefficients for the fine grained sample of red waste and limestone were 45 and 42, respectively, which indicated that the material was poorly sorted.

The soil water characteristic curve (SWCC) for the coarse and fine materials are shown in Figure 5.12(a), which represented the drying curve for each material. The SWCC of the fine grain material was obtained in the laboratory using a Tempe cell, while the SWCC of the coarse material was generated from the SoilVision model (Soil Vision System Ltd, 2005) that uses the equation developed by Fredlund et al. (2002). The Air Entry Value (AEV) of the fine material was approximately 1 kPa for both red waste and limestone. In contrast, the AEV of the coarse-grained material for both acid rock and limestone was only 0.03 kPa. The low AEV of the coarse material indicated that it had no capillary capacity to hold water. However, the capillary barrier would be potentially developed in the coarse layer when it was placed underneath the fine layer.

The saturated hydraulic conductivity ‘Ksat’ for each material was measured in the laboratory using a constant head and falling head permeameter. The diameter of the falling head permeameter was around 25 cm. The saturated hydraulic conductivity of the coarse-grain acid rock material and limestone were approximately between 1 x $10^{-2}$ m/s and 1.5 x $10^{-2}$ m/s, and the saturated hydraulic conductivity of the fine-grain acid rock material and limestone varied between 3.9 x $10^{-4}$ m/s and 8.0 x $10^{-4}$m/s. The unsaturated hydraulic conductivity functions for these materials were estimated based on the measured saturated hydraulic conductivities, Ks, and the SWCC, according to the function developed by Fredlund and Xing (1994). As shown in Figure 5.12 (b), when the
Figure 5.12 (a) Soil Water Curve Characteristic Curve, and (b) Unsaturated hydraulic conductivity for the coarse and fine waste rock used in the meso-scale panel experiment.

Suction exceeds the AEV for either the coarse or fine grained material of 0.03 kPa and 1 kPa, respectively, the unsaturated hydraulic conductivity for both materials decreases rapidly by several orders of magnitude.

5.3.2 Measured Outflow from Layers

The installation of drainage outlets at the base of each layer allowed for assessing the variability of outflow among the layers in response to different fluxes or rainfall rates. The applied rainfall rates were 2 mm/day ($2.3 \times 10^{-5}$ mm/s), 5 mm/day
(5.8 \times 10^5 \text{ mm/s}), and 10 \text{ mm/day} (1.2 \times 10^{-4} \text{ mm/s}) for each cycle of the experiment. Each rainfall rate represented different climate types. The selected rainfall intensities were lower than the saturated hydraulic conductivity of the materials used in the experiment. Each cycle of the experiment ran from 2 weeks to 6 weeks, depending on how much time was required to achieve pseudo steady state.

In order to confirm that every layer in the panel received the same rainfall application rate, the Christiansen formula was used to calculate the coefficient of uniformity of the rainfall application system (Christiansen, 1942). The uniformity coefficient is calculated according to the difference between a finite number of rainfall intensities measuring points in the layer and the average value of measurement from all layers in the panel. The uniformity coefficient approaches 100 as the rainfall approaches complete uniformity. The uniformity coefficient of the rainfall rate of the experiment varied between 97\% and 99\%, indicating that the rainfall was distributed evenly on each layer.

Figures 5.13(a) and 5.13(b) show the volume of outflow and percentage of outflow distribution, respectively, for each layer in panel 1 following the application of different rainfall rates with the base boundary condition set at a constant suction value of 4 \text{ kPa}. The results demonstrated that the distribution of flow depended on the intensity of the rainfall. When the rainfall rate was set equal to 2 \text{ mm/day} or 2.3 \times 10^{-5} \text{ mm/s}, approximately 89\% of the total water outflow occurred within the middle of the panel region, mainly in layers 2 to 5. The largest amount, approximately 31\% of the
Figure 5.13 (a) Volume of leachate and (b) leachate distribution from each layer for panel 1
total outflow, was recorded for layer 2 (fine grain). When the rainfall rate increased to 5 mm/day or $5.8 \times 10^{-5}$ mm/s, the flow was distributed more evenly among the layers, and the measured outflow quantities from each layer were closer together, particularly for layers 1 to 4. In addition, the amount of the water flow distributed to the middle of the panel decreased from 89% to 68%. When the rainfall rate was further increased to 10 mm/day or $1.2 \times 10^{-4}$ mm/s, the amount of water that moved in parallel with the profile of layers decreased, and vertical flow had a more dominant flow mechanism. Consequently, the percentage of water collected in layers 3 (fine grain) and 4 (coarse grain) decreased. Approximately 70% of the leachate was collected from the outlets located underneath the surface rainfall simulators (i.e., the drain points located on the slope section and also from layers 1 (coarse grain) and 2 (fine grain). The coefficient of variation in relation with the percentage of water collected from each layer also decreased from 102% to 66% when the rainfall rate increased from 2 mm/day to 10 mm/day. The amount of leachate collected from layer 6 (fine grain) was relatively small, and no leachate was collected from layer 7 (coarse grain).

A similar trend to panel 1 was also observed in panel 2 when different rainfall rates were applied, as shown in Figures 5.14(a) and 5.14(b). It was also observed that as the rainfall rate increased, less water moved according to the profile of the layers. The distribution of water flow in panel 2 was slightly different from panel 1 due to geometrical differences. In the case of panel 2, the surface rainfall simulator application points were shifted laterally due to the increased height (i.e. 500 mm) of the panel. In
Figure 5.14 (a) Volume of leachate and (b) leachate distribution from each layer for panel 2
other words, two-thirds of the application points were positioned directly above the lower slope boundary and collection drains. Therefore, more leachate was collected from the lower layers L1 and L2 and also from the outlets along the bottom slope of panel 2 than was collected in panel 1. At a rainfall rate of 2 mm/day or \(2.3 \times 10^{-5}\) mm/s, about 63% of the total outflow was distributed evenly in layers 2 and 5. A high quantity of outflow was recorded in layer 5 (coarse grain), which was located at a relatively short distance from the rainfall simulator application points at the outer slope of the panels. This layer produced almost 25% of the total outflow for an applied rainfall rate of 2 mm/day. When the rainfall rate increased to 5 mm/day or \(5.8 \times 10^{-5}\) mm/s, the largest outflow shifted to layer 1 (coarse grain) as a result of increased vertical flow, and it accounted for approximately 40% of the total outflow. When the rainfall rate was further increased to 10 mm/day or \(1.2 \times 10^{-4}\) mm/s, more water was directed vertically to the bottom and basal slope drains. The highest percentage, about 31% of the total outflow, was recorded from the lowest outlet drainage point (i.e. S4) situated along the basal slope, located next to layer 1 (coarse grain). Similar to panel 1, only a small volume of leachate was collected in layer 6 (fine grain), while layer 7 (coarse grain) had no outflow. The coefficient of variation from the percentage flow distribution was 82% and 90% when the rainfall rates were 2 mm/day and 10 mm/day, respectively.

Figures 5.15(a) and 5.15(b) show the volume of leachate and the distribution of the leachate for each layer in panel 3, respectively. Similar to panels 1 and 2, the leachate distribution was concentrated in the middle of the panel when the rainfall rate
Figure 5.15 (a) Volume of leachate and (b) leachate distribution from each layer for panel 3
was 2 mm/day (2.3 x 10^{-5} mm/s). The highest leachate quantity was reported at layer 4 in the middle of panel 3, which contained fine-grain material. This leachate accounted for approximately 50% of the total outflow for the panel. When the rainfall rate was increased to 5 mm/day (5.8 x 10^{-5} mm/s), the leachate distribution shifted to the left side (i.e. interior) of the panel, which clearly illustrates the influence of the vertical flow mechanism. The maximum quantity of leachate, approximately 31% of the total outflow, was recorded in the fine-grain material for layer 2. When the rainfall rate was further increased to 10 mm/day, leachate flow rates became more evenly distributed among layers 1 (coarse grain), 2 (fine grain), and 3 (coarse grain) which were located directly underneath the area where the rainfall was applied. The coefficient of variation in relation with the percentage of water collected from each layer also decreased from 131% to 47% when the rainfall rate increased from 2 mm/day to 10 mm/day. The contribution to vertical (downward) flow in panel 3 was not as pronounced as was observed in panels 1 and 2. The geometry of panel 3, which was wider than panels 1 and 2 (i.e. 2,250 mm versus 1,750 mm) appears to have affected the flow path. Furthermore, the arrangement of the layers in panel 3, which were different from the arrangements in panels 1 and 2, also influenced the flow path. No leachate was collected from slopes (S1 to S3), layers 8 (coarse grain) and 9 (coarse grain) during the experiment.

Changing the rainfall rate was observed to change the amount of leachate that flowed through the adjacent coarse and fine layers, particularly for bottom layers.
These changes were observed clearly in panel 1 between layer 1 (coarse grain) and layer 2 (fine grain), as shown in Figure 5.16. However, the changes between layer 3 (coarse grain) and layer 4 (fine grain) was not observed due to the influence of the vertical flow. For example, when the rainfall intensity was increased from 2 mm/day to 5 mm/day, the amount of leachate that was collected from the fine material of panel 1 decreased.

Figure 5.16 Distribution outflow in adjacent coarse and fine layers in panel 1
Figure 5.17 summarizes the relative combined quantities of leachate collected from the layers of coarse material and fine layers in panels 1, 2, and 3 at the different rainfall simulation rates. The combined quantity of leachate collected in the coarse layers also included the quantity of leachate collected from the basal slope. Similar trends were observed in the fine layers for all panels in that the percentage of the outflow decreased as the rainfall rate increased. The decrease in total outflow from the fine layers ranged between 9% and 29% as the rainfall application rate was increased. The minimum and maximum decreases were recorded in panels 2 and 3, respectively. Increasing the rainfall rate promoted water flow through the coarse layer with downward gravitational flow becoming the dominant flow mechanism. The percentage of outflow from the coarse layers of panels 1, 2, and 3 increased from 52% to 70%, from 61% to 79%, and from 29% to 59%, respectively, when the rainfall application rate was increased from 2 mm/day to 10 mm/day.
Figure 5.17 Distribution of leachate volume from the combined outlets for the course layers (including basal slope) and fine layers for (a) panel 1, (b) panel 2, and (c) panel 3

5.3.3 Flow Path

The experimental results demonstrated that the flow path of water within the layers was related to the rainfall rate, as shown previously in Figures 5.13 to 5.17. At low rainfall rates, water tended to flow laterally, following the inclined profile of the layers due to the capillary forces of the fine material that was able to retain more water. However, some water was also transported through the coarse layers, and
leachate was also distributed to the outlets of the bottom slope, suggesting that the water had percolated directly downward. The amount of water reported at the outlets for the fine layers was significant, and in some cases, the amount of water collected from the fine layers tended to be greater than the amount of water collected from the coarse layers that were situated next to the fine layers. In panel 3, preferential flow occurred within layer 4 (fine material), and was found to be about 50% of the total outflow collected from all the layers.

Although the quantity of outflow collected from each layer increased as the rainfall intensity increased, the increments were not similar in all layers. Increasing the infiltration rate was found to change the distribution of water among the layers. When the rainfall rate was increased, the water content in the fine layer also increased and reached the point such that the matric suction in the fine layers became very low. Thus, breakthrough occurred and the underlying coarse layer became the preferential pathway for flow. In this case, the outflow from the coarse layer was higher than the outflow from the overlying fine layer. These observations are in agreement with the results reported by Wu et al. (2007), Nichol et al. (2003), and Newman (1999), which indicated that more water was transferred through the coarse layer during higher rainfall rates. However, as the rainfall rate was increased, gravitational (i.e. vertical/downward) flow became the dominant factor, resulting in shifting lateral water along the coarse layers directly to the base of the panel.
Drainage collected from each individual layer began to decrease after termination of the rainfall application. However, it took approximately one day before the drainage stopped completely. The time required for the drainage to stop depended on the initial rate of rainfall and the materials in the layers. Figure 5.18 shows the daily outflow volume and outflow distribution from the layers in panel 1 during the drainage period following the application of the 2 mm/day rainfall rate. Drainage from the coarse layers stopped in less time than the drainage from the fine layers. Breakthrough drainage from the fine to coarse layers decreased rapidly due to the reduction in the water content within the fine layers. The fine layer became the preferred path for the flow of water after the application of rainfall was terminated. The placement of a fine layer over a coarse layer creates a capillary barrier at the interface, thus the water in the fine layer does not flow downward to the coarse layer, and it tends to be diverted laterally along the profile of the layers as capillary flow (Kung, 1990; Hillel and Baker, 1988).

During pseudo steady state, very low values of matric suction were measured in the layers that were close to the base of the panel, such as layers 1 to 3. The matric suction values were found to vary between 0.1 and 0.2 kPa, while the matric suction measured within the outer layers were slightly higher. Similar low values of matric suction were also measured in the excavation in the trial dump that was exposed to the high rainfall environment. Figure 5.12(b) shows the relationship between unsaturated hydraulic conductivity and matric suction; the effective hydraulic conductivities for both
Figure 5.18 (a) Daily outflow volume and (b) outflow distribution in panel 1 during the drainage period

materials is the same when the matric suction was equals around 0.2 kPa. When the matric suction was less than 0.2 kPa, the unsaturated hydraulic conductivity of the coarse material exceeded the unsaturated hydraulic conductivity of the fine material. On the other hand, the opposite condition occurs when the matric suction is higher than 0.2 kPa. As a result, since the unsaturated hydraulic conductivity of coarse material was similar to or higher than the effective hydraulic conductivity of the fine
material, the quantity of leachate collected from the coarse layers was significant during
the pseudo steady state phase of the experiment.

When rainfall simulation was applied to the panels, no discharge was observed
at the outlet for layer 7 in panels 1 and 2 or for layers 8 and 9 in panel 3. Matric suction
measurements in the lower sections of these layers remained high, ranging up to 13
kPa. Referring to the SWCC for the coarse material shown in Figure 5.8(a) and the
measured suction, the corresponding water content for this value of matric suction is
near to 0%. This value is considerably lower than the value reported by El Boushi and
Davis (1969), who stated that at least 3% water content is required in coarse materials
to generate water flow. In addition to the high values of suction, the unsaturated
hydraulic conductivity of the coarse material approaches zero. Even though the water
flow occurred in the coarse material, the major flow mechanism appears to be
gravitational drip or film flow. Therefore, water flowed to the outlets located directly
below the coarse layers instead of flowing to the outlet for each layer. In summary, the
flow in the coarse layer is considered to be non-Darcian.

Low outflow was also recorded for the fine layers that were immediately
adjacent to the top coarse layers, such as layer 6 in panels 1 and 2. Although some
water was collected from these layers, the amount was relatively low compared to the
outflow for other layers. Since a significant amount of water from this layer had already
flowed into the underlying layers, only a relatively small amount of water was retained
and it flowed laterally through this fine layer. The measured values of matric suction in
the lower region of these layers (i.e. layer 6 in panels 1 and 2) varied from 0.3 kPa to 0.6 kPa, which were slightly higher than the matric suction values measured for layers 1 to 5. This relatively higher value of matric suctions caused a reduction in the effective hydraulic conductivity, thus the water flowed more slowly in the fine material in layer 6 compared to the underlying layer.

5.3.4 Relationship between the Evolution of Matric Suction and Outflow

Matric suction was measured within the profile of the panels at the different stages of rainfall application, such as at the beginning of the rainfall application when the system was still relatively dry, as well as at pseudo steady state when the panels showed relatively stable outflows following the application of rainfall for a particular period. Matric suction values were also measured during the drainage period after the application of rainfall had been terminated for a period of 12 days. Although great care was taken with the installation of the instrumentation, monitoring results indicated that not all tensiometers provided reliable readings, as in the case of the tensiometer located at a lower part of layer 1 for panel 1. For example, it was found readings frequently fluctuated at relatively high suction values, between 5 and 15 kPa, while at the same time a significant amount of outflow was collected from the layer suggesting the water content in the coarse material was well above the residual value and corresponded to a suction of 10 kPa (i.e. Figure 5.12a). The poor readings may have been caused by poor contact between the ceramic tip and the coarse rock, allowing air that occupied the pores to influence the reading.
The flow path for infiltration water was found to be influenced by the changes in the water content within the layers. This phenomena has been reported by others (Fala et al. 2005; Sililo and Tellam, 2000; Schroth et al. 1998). Although water content was not measured directly in this experiment, the water content in a particular layer can be estimated on the basis of matric suction measurements and the SWCC for coarse and fine materials presented in Figure 5.12(a). The matric suction values and outflow data were assessed in order to understand how changes in the water content related to the quantity of outflow collected from the individual layers. The data was collected during application of the 2 mm/day rainfall for panel 1. Measurements for discharge rates (i.e. flow volumes) and matric suction were obtained at the same time.

Figure 5.19 presents the evolution of matric suction in each layer of panel 1 from the beginning of the cycle until a condition of pseudo steady state was achieved for the rainfall rate of 2 mm/day. Prior to the application of infiltration water (or the system was drained for 12 days after the previous infiltration test), relatively high matric suction values were recorded in each layer in panel 1 at elevations of 0.5 m and 1 m. The matric suction was found to vary between 1.2 kPa to 7 kPa, except for the tensiometer in L1 which reached 23 kPa. In general, it was observed through the drainage phase prior to the application of infiltration water that matric suction values varied between 1.2 kPa and 2.8 kPa in the fine layers, while the matric suction values in the coarse layers were seen to be much higher, ranging between approximately 2 kPa and 23 kPa. These high matric suction readings were usually recorded within the coarse
Figure 5.19 Matric suction measurements for panel 1 during a rainfall rate application equaled to 2 mm/day at (a) 0.5 m elevation in layer 1, (b) 0.5 m elevation in layers 2 to 7, (c) 1 m elevation in layers 1 to 7, and (d) Leachate volume for each layer in panel 1

layers that remained dry most of the time since the flow did not occur at locations near the perimeter of the panel.

The first matric suction measurement in panel 1 was obtained six (6) hours after the application of rainfall began. The matric suction measurements indicated that a gradual decrease in suction started to occur, particularly for the tensiometers located at
the 1 m elevation. After six (6) hours of rainfall application, the matric suction for
tensiometers located at 1 m elevation varied from 0.8 kPa to 2.7 kPa and the matric
suction in the tensiometers located at 0.5 m elevation ranged from 0.8 kPa to 3 kPa,
except for the tensiometer in L1 which reached 16.8 kPa. When the measurements
continued between the 8 and 12 hour intervals, all tensiometers at elevations of 0.5 m
and 1 m registered a continual decrease in matric suction, indicating that the layers
were getting wetter, including the tensiometer located in layer 1, which had no rainfall
application at the top of the layer.

The first leachate collected from the base of the layers in panel 1, as shown in
Figure 5.19(d), occurred after forty-eight (48) hours of rainfall application and the
leachate was collected only from layer 5 (coarse grain). The position of layer 5 was
relatively close to the rainfall simulators located along the upper slope of the panels.
The matric suction values in layer 5 were 0.2 kPa and 0.7 kPa for tensiometers located
at elevations of 0.5 m and 1.0 m, respectively. The matric suction at elevation 0.5 m in
layer 5 was slightly lower compared to the matric suctions recorded in the other layers
for the same elevation (i.e. from 0.5 kPa to 1.7 kPa). On the other hand, the matric
suction at 1.0 m elevation in layer 5 was comparable with the matric suctions in the
other layers at the same elevation, which varied between 0.6 kPa and 0.9 kPa.

After 72 hours of rainfall application, all outlets in panel 1 produced leachate
flows, except for outlets for layer 1 and along the basal slope. Relatively high volumes of
leachate were observed for layers 4 and 5, as shown in Figure 5.19(d). The range of
matric suction values at an elevation of 1 m varied from 0.2 to 0.5 kPa, except for layer 7 (coarse grain), which remained at approximately 0.9 kPa. Low matric suction values of near the precision of the tensiometer (i.e. 0.2 kPa) were measured at the tensiometers located at an elevation of 0.5 m in layers 2, 4, 5, and 6, and continued to fluctuate within a low range as pseudo steady state was approached.

Each layer in panel 1 was observed to produce leachate progressively. The layer that produced leachate first (i.e. layer 5 in panel 1) did not have the highest volume of leachate at the end of the cycle. After several days, the amount of leachate produced for some layers increased significantly, such as the outflows for layers 1 and 2, as can be seen in Figure 5.19(d).

Once pseudo steady state was reached or the fluctuation of outflow volume was maintained within a consistent range, matric suction values among the layers did not vary significantly, except at locations that had minimum contact with the infiltrated water, such as layer 7 (coarse grain) in panel 1, which continued to record high values. Fluctuations in the volume of leachate, as well as in matric suction, were detected during pseudo steady state. However, the changes in matric suction were not always followed by similar changes in the volume of leachate collected from the same layer. For example, increases in leachate volume from layers 1 and 2 were not apparent with decrease in matric suctions for both layers 1 and 2, as shown in Figure 5.19(b) and 5.19(c). The development of a vertical flow component associated with breakthrough may have occurred at elevations below the tensiometer locations, and thus could have
contributed to these discrepancies. In addition, the fluctuations in ambient air pressure that occurred during the experiment may also have played a role in the fluctuations of the measurements.

The average values of matric suction measured during pseudo steady state for rainfall rates of 2 mm/day (2.3 x 10^{-5} mm/s) and 10 mm/day (1.2 x 10^{-4} mm/s) are presented in Figure 5.20 for panels 1 and 3 at elevations of 1 m and 0.5 m. For improved clarity of presentation in the Figure 5.20, the plot for panel 1 does not include the data for the tensiometer located at the bottom of layer 1 and the plot for panel 3 does not include the data for the tensiometer located in layer 9 (i.e. due to the abnormally high matric suction values measured in these tensiometers). The impact of the two different rainfall rates (i.e. a factor of 5) on the magnitude of matric suction was not strikingly apparent for each of the individual layers. In addition, higher infiltration rates were not always associated with the lower values of matric suction. These results are slightly different from the results obtained by previous researchers who showed a strong correlation between matric suction measurement and the irrigation rate (Tami et al. 2004). The difference in the properties of the materials used for this experiment may have affected the values that were measured. The waste rock materials used in the panel experiment had low AEV’s and steep SWCC, hence were highly non-linear, as shown in Figures 5.8(a) and 5.8(b). Changes in water content, resulted in small changes in matric suction when the system was near saturation conditions, (i.e., pseudo steady state conditions). Due to the limited accuracy of the tensiometers, these small
Figure 5.20 Matric suction values measured during pseudo steady state at rainfall rates of 2 mm/day and 10 mm/day in layers at (a) Panel 1, 1 m elevation, (b) Panel 1, 0.5 m elevation, (c) Panel 3, 1 m elevation, and (d) Panel 3, 0.5 m elevation.
changes strong correlation between matric suction measurement and the irrigation rate (Tami et al. 2004). The difference in the properties of the materials used for this experiment may have affected the values that were measured. The waste rock materials used in the panel experiment had low AEV’s and steep SWCC, hence were highly non-linear, as shown in Figures 5.12(a) and 5.12(b). Changes in water content, resulted in small changes in matric suction when the system was near saturation conditions, (i.e., pseudo steady state conditions). Due to the limited accuracy of the tensiometers, these small changes were not always detected in the measurements. Furthermore, possible changes in the flow path that resulted from the application of rainfall at different intensities could have influenced the measured results within the same tensiometer.

5.3.5 Breakthrough Flow in Layers

In order to help understand the breakthrough or transfer flow on the layers, an experiment was conducted in panel 1 where a precipitation rate of 10 mm/day (1.2 x 10^4 mm/s) was applied to the outer layers 6 and 7 for 12 days and then continued to be applied to the layers 4 to 7 for 7 days. Within the first 24 hours, breakthrough was observed as the water collected in all layers, except layer 7. It is considered likely that the fine grain material in layer 6, which contained primarily coarse and medium sands, was not able to transfer all the flow applied to the surface of the layers 6 and 7. It can be seen in Figure 5.21 that flow was transferred as far as layer 1. Furthermore, approximately 60% of the outflow reported to the coarse layers (layers 1, 3, and 5).
Figure 5.21  (a) Volume of leachate and (b) outflow distribution from each layer with rainfall applied on layers 6 and 7 in panel 1, and (c) volume of leachate and (d) outflow distribution from each layer with rainfall applied to layers 4 to 7 in panel 1
However, the highest percentage of outflow was observed in the outlet for the fine material in layer 2, and it accounted for about 31% of the total outflow. When the rainfall was also applied to layers 4 and 5, in addition to layers 6 and 7, outflow was collected further back within the panel (i.e. discharge points S2 and S3). This indicated that breakthrough also occurred from layer 2 and that the water was not only diverted into layer 1, but also to the outlets located along the bottom slope of the panel.

The large quantity of leachate that was collected from layers 1 and 2 in panel 1 during the breakthrough test confirmed that interlayer flow had already begun to occur in the upper section of the profile. In addition, the small amount of water that was collected from layer 6 also suggested that partial breakthrough had occurred. Baker and Hillel (1990) stated that breakthrough occurs when the matric suction at the interface of the two layers exceeds the water entry value of the underlying layers. Therefore, increasing the rainfall rate increased the amount of breakthrough due to the increased water content and decreased matric suction, as well as changes to the diversion length. The diversion length is a distance between the top of the slope from the fine layer and the point where the breakthrough or the water infiltration into the underlying coarse layer becomes significant.

Ross (1990) introduced an equation to calculate the diversion length based on a quasi-linear approximation for the hydraulic conductivity of materials that can be applied to the capillary barrier system. This equation was later extended by Steenhuis et al. (1991), who estimated the hydraulic conductivity of a material as a function of matric
suction. The equation shows that the diversion length depends on infiltration rate, the angle of the slope, and the water entry value of the coarse grain, as shown in Equation 5.2.

\[
L \leq \tan \Phi \left[ \alpha \left( K_s \right)_i + K_s \left( h_a - h_{w^*} \right)_i \right] \quad (5.2)
\]

where : 
- \( L \) = diversion length
- \( \Phi \) = dip of angle of the interface from horizontal
- \( \alpha \) = sorptive number related to pore size
- \( K_s \) = saturated hydraulic conductivity of fine grain
- \( i \) = infiltration flux
- \( h_a \) = air entry value of fine grain
- \( h_{w^*} \) = water entry value of coarse grain

The test for panel 3 at the 2 mm/day rainfall rate showed that approximately 50% of the leachate was collected from layer 4 that contained fine grained material. The quantity of leachate collected from layer 4 was higher than the quantity of irrigation water applied to this layer. This data demonstrates that for a rainfall rate of 2 mm/day, the fine material was capable of transferring all the irrigation water within the layer. Therefore, on the basis of this observation, it was estimated the diversion length would be equal to or greater than the length of the layer, which was approximately 2,500 mm. Equation 5.2 can also be used to estimate the diversion length for the panel experiment with assumption the steady state condition is achieved as condition for equation 5.2.
The purpose of this assessment was to understand the influence of rainfall rate to the breakthrough in the fine layer. Since the air entry value of fine grained material and the water entry value of coarse grain material are shown in Figure 5.12(a), the saturated hydraulic conductivity for fine material is known from Figure 5.12(b), and the sorptive number depends on the grain size of the material, the diversion length can be computed as a function of infiltration rate. The calculated results presented in Figure 5.22 show the diversion length decreases with increasing infiltration and thus the breakthrough point occurs at a higher elevation as the rainfall rate increases.

![Figure 5.22 Diversion length for the fine layer as a function rainfall rate](image-url)
Experiments performed by Walter et al. (2000) and numerical model results from Bussière et al. (2003a), and Parent and Cabral (2006) confirmed the decrease of capillary diversion when the rainfall rate increased. Although Ross’s equation cannot be used to calculate the diversion length for a case such as the multiple coarse and fine layers in this experiment, the influence of the rainfall rate on the breakthrough point was considered similar for the panel experiment. As the breakthrough point potentially shifted to a higher elevation when higher rainfall rates were applied, more water could potentially flow to the lower layers. Therefore, when the rainfall rates were increased, gravitational flow became a more dominant flow mechanism, resulting in the collection of more leachate from the layers that were located in the interior of the panel as compared to discharge points from the layers located close to the outer slope.

5.3.6 Influence of Lower Boundary Suction Values on Outflow

In order to evaluate the effect of boundary conditions on the flow path of the infiltrated water, suction lysimeters were installed at the base of each layer in panels 1 to 3. Different base boundaries with respect to matric suction conditions were established for the different rainfall rates applied by raising or lowering the discharges for the suction lysimeter installed in each layer.

The effects of the base boundary conditions on the outflow distributions for panels 1 and 3 are presented in Figure 5.23, for rainfall rates of 2 mm/day (2.3 x 10^{-5} mm/s) and 10 mm/day (1.2 x 10^{-4} mm/s). For panel 1, the suction in the lysimeter was
Figure 5.23 Combined outflow volume distribution for coarse and fine layers in panel 1 at (a) 2 mm day rainfall rate and suction equal to 2 kPa to 4 kPa, (b) 10 mm/day rainfall rate and suction equal to 0 kPa to 2 kPa, and in panel 3 at (c) 2 mm day rainfall rate and suction equal to 2 kPa to 4 kPa, (d) 10 mm/day rainfall rate and suction equal to 0 kPa to 4 kPa
increased from 0 kPa to 2 kPa at the 10 mm/day rainfall rate, and from 2 kPa to 4 kPa at the 2 mm/day of rainfall intensity. Similar changes were applied at panel 3 for the 2 mm/day rainfall rate, but in the case of the 10 mm/day rainfall rate, the suction was increased from 0 kPa to 4 kPa. The observed response was similar for all panels. The total outflow from the coarse layers decreased, and the total outflow from the fine layers increased when the value of the suction of lysimeter increased. The changes varied from 1% to 9% and they were less pronounced for panel 3, particularly at low rainfall intensity. Newman (1999), who assessed the influence of the base boundary condition on the outflow of a vertical layer of coarse and fine grained materials in a vertical column with 1400 mm height and a 150 mm by 300 mm rectangular interior area, also showed an increase in the amount of leachate that was collected from the fine-grained material when the suction was increased at the base boundary.

Although changing the base boundary condition affected the total amount of leachate collected from either the combined coarse layers or the combined fine layers, a similar response was not always apparent in the individual layers, as shown in Figure 5.24. Increasing the suction was not always followed by increased outflows from individual fine layers or by decreased outflows from individual coarse layers. For example, when the rainfall rate was 2 mm/day and the base boundary condition was changed from 2 kPa to 4 kPa in panel 1, only the outflow from the fine material at layer increased, while the outflow from the other fine layers (i.e. layers 4 and 6) decreased.
Figure 5.24 Outflow volume distribution in each layer in panel 1 at (a) 2 mm day rainfall rate with suction changed from 2 kPa to 4 kPa, (b) 10 mm/day rainfall rate with suction changed from 0 kPa to 2 kPa, and in panel 3 at (c) 2 mm day rainfall rate with suction changed from 2 kPa to 4 kPa, (d) 10 mm/day rainfall rate with suction changed from 0 kPa to 4 kPa
A similar observation was made for panel 3 when the suction increased from 0 to 4 kPa at 10 mm/day rainfall intensity. Alternatively, the observed changes in individual layers in panel 3 were different at rainfall rates of 2 and 10 mm/day. When the base boundary increased from 0 kPa to 4 kPa at a 10 mm/day rainfall rate, the outflow of layer 2 (fine grain) decreased, and the discharges from layers 6 and 7 (coarse grain) increased. An interesting response was observed for the fine layers located within the interior of panel 3 when the rainfall rate was set equal to 2 mm/day and the base boundary condition was increased from 2 kPa to 4 kPa. In this case, the outflow from the fine material in layer 5 that dissipated, appeared to be transferred to the adjacent fine material in layer 4.

The base boundary condition mainly influenced the head pressure gradient between the material in the layer and the lysimeter, which could direct the water out to the lysimeter. The amount of water collected by the lysimeter was determined by the water content within the layer, particularly in the lower region of the panel. As a result, changing the basal boundary setting had a localized influence on the flow path. These results were in agreement with the model developed by Newman (1999), which showed that the change of the flow path, termed cross-over flow, occurred only in the lower region of the panel.

The flow of water within the layers is a function of total hydraulic head which includes the pressure head (i.e. depends on the water column) and the elevation. Since the hydraulic heads were not always the same in every layer, the magnitude of the
gradient as a driving force was also different among the layers. Layers with a higher pressure head due to a higher water content lead to higher gradients. As both coarse and fine materials drain under gravitational force, layers with higher water contents drained more leachate than the layers with lower water contents. Therefore, for the most part, positive correlations were exhibited by the fine layer located near the basal slope in panel 1 and by the fine layers located at the interior of panel 3 because these are the locations where the water content tended to be higher. Similar results were also observed for the outer layers when high rainfall rates were used.

5.3.7 Influence of Layer Geometry and Layer Arrangement on Flow Path

The influence of gravitational flow on the flow path appears to be the dominant mechanism for the flow through the layers at high infiltration rates. Referring to the locations for the rainfall simulators and the outflow data in Figures 5.13 to 5.14 (and Figures 5.4 to 5.6), the water flux applied at the top of the panels was likely transferred to the layers immediately above the basal slope, such as layers 1 and 2 on panels 1 and 2, and to the lowest outlet at the slope (i.e. S3 for panel 1 and S4 for panel 2). Some water from the rainfall simulators that were located along the outer slope, particularly the ones that were located at the upper part, also contributed to the outflow of those layers. On the other hand, the leachate collected from the outer layers, such as layers 5 and 6, originated mainly from the rainfall simulator along the outer slope of the panel. Although the breakthrough test indicated that the water that was applied at the surface of layers 6 and 7 could reach layers 5 and 6, it only accounted for less than 10% of the
rainfall applied, as shown previously in Figure 5.21(b) and (d). The quick response observed in layer 5 to rainfall events along with the relatively dilute concentration of constituents in this leachate (as discussed in Chapter 6) indicated that the water collected from layer 5 came from a source that was relatively close to this layer.

Vertical flow was less pronounced in panel 3. In addition to the panel geometry, the layer arrangements, which were different from the other two panels, determined the flow distribution. The existence of the adjacent two fine layers (L4 and L5) in panel 3 affected the flow path in the panel. These two layers likely acted as a single layer with double thickness, which could have led to the increased water storage capacity. Consequently, the amount of breakthrough to the underlying layers could have been reduced. Therefore, the amount of water distributed to the coarse layers located near the bottom slope was less in panel 3 than it was in panels 1 and 2. Furthermore, layer 4 (fine grain) became the preferential flow path at low rainfall rates. When two fine layers were combined, water tended to accumulate at the bottom layer. This behavior was observed at the low rainfall rate during which more water drained from layer 4 than from layer 5, as illustrated in Figure 5.15. A similar response was also obtained when the matric suction in the suction lysimeters at the lower boundary was increased during the application of the low rainfall rate, resulting in the shifting of water from layer 5 to layer 4.
5.4 Conclusions

Laboratory experiments have been performed in order to achieve a better understanding of the flow behavior in a rock pile that contains multiple inclined coarse and fine layers. Although the detailed flow path of the water from the top to the bottom of the panels was not clearly observed, the data for the distribution of the leachate that was collected from the outlet points, together with the measurement results of the matric suction provide insight on the characterization of the flow of water through the layers. The properties of the coarse and fine materials used in this experiment established the behavior of flow within the pile.

The infiltration tests conducted on the panels clearly demonstrated that the applied rainfall rate determines the distribution of flow within a waste rock pile. The fine layers were found to be the preferred flow path at low rainfall intensities, although some flow occurred through the coarse layers and thus gravitational flow was also observed. When the applied rate of rainfall was increased, more water moved through the coarse layers than was moving through the fine layers, and gravitational flow had a more pronounced effect on the system. At the highest rainfall intensity, the gravitational flow became the dominant flow mechanism, resulting in water moving straight downward through the profile. When the rainfall rate was equal to 2 mm/day (2.3 x 10^{-5} mm/s), the water flow in the combined coarse layers and fine layers, respectively, was approximately 52% and 48% for panel 1, 71% and 29% for panel 2, and 29% and 71% for panel 3. As the rainfall increased to 5 mm/day (5.8 x 10^{-5} mm/s), the
water flow through the combined coarse layers and fine layers changed to 63% and 37% for panel 1, 75% and 25% for panel 2, and 45% and 55% for panel 3. When the rainfall was further increased to 10 mm/day (1.2 x 10⁻⁴ mm/s), the water flowed through the combined coarse layer and fine layers at 70% and 30% for panel 1, 81% and 19% for panel 2, and 59% and 41% for panel 3.

Gravitational flow occurred in part due to the partial breakthrough flow from the fine layer. This was likely caused by the difference in hydraulic properties between the coarse and fine materials used in this experiment that did not create a strong capillary barrier effect along the interface of the fine and coarse materials, and thus produced significant breakthrough from the fine layers, particularly at a high rainfall rate. The trial showed that partial breakthrough occurred, for the most part, in the upper regions of the fine layer. The calculation of diversion length in the fine layer demonstrated that the diversion length changed from 2,500 mm to 1,000 mm when the rainfall rate increased from 2 mm/day to 5 mm/day. As the rainfall further increased to 10 mm/day, the diversion length reduced to 500 mm.

Matric suction measured in the layers at pseudo steady state showed that the matric suction measurement varying around 0.2 kPa to 0.5 kPa. At this range of suction, the effective hydraulic conductivities between coarse and fine materials were not so different. As a result, significant quantities of leachate outflows were observed from both coarse and fine layers.
The layer geometry and the layer arrangements affected the distribution of water among the layers. Layers that were located near the basal slope received water mostly from the top of the panel and tended to be wetter. On the other hand, the rainfall applied to the outer slope of the panel tended to flow through the outer layers. Any mitigation treatment that will be applied in the waste rock pile such as a limestone cover should be directed at the flat surface as well as at the slope of the waste rock pile to ensure that the alkaline solution flow from the limestone cover distributes evenly in the waste rock pile.

Although the layers that exist in the actual waste rock pile are more heterogeneous and have more variations than the pattern of layers used in the present experiment, the results obtained from this study provide a better understanding of the mechanisms that control flow in the coarse and fine layers, which will be valuable in practical applications. This understanding will be particularly important to the success of treatments that are intended to minimize acid outflows from the waste rock pile by relying on alkaline transport throughout the pile, i.e., treatments such as blending limestone with acid rock and using a limestone cover. The experimental results show that the water was distributed more evenly to each layer at the higher rainfall rate due to gravitational flow as indicated by a decreasing of the coefficient of variation from the leachate distribution in each layer at panels 1 and 3. The coefficient of variation decreased from 102% to 66% for panel 1 and from 131% to 47% for panel 3. The
alkaline solution from a limestone cover will therefore be leached into the waste rock pile and be distributed more evenly in a high rainfall environment.
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Chapter 6 A Meso-Scale Laboratory Experiment to Investigate Geochemical Mixing in Layered Mine Rock Stockpiles

6.1 Introduction

Inclined coarse and fine layers that form the internal structure of a waste rock pile are found in many dumps. The existence of the inclined layers influences the flow path of infiltrated water within the waste rock. The different hydraulic properties of the coarse and fine materials can create hydraulic barriers, capillary barriers, or anisotropic intra-layers in the pile that lead to the occurrence of preferential flow (Alfres, 2004a). The resulting preferential flow may affect the quality of the effluent from the waste rock pile. Eriksson and Destouni (1997) reported that preferential flow reduced the peak load and affected the long-term distribution of contaminant loading from the waste rock pile. Murr (1979) observed that preferential flow reduced the contact area between the water and the rock resulting in decreased solute recovery in the leachate. The influence of inclined coarse and fine layer structures on flow and solute transport has also been studied by soil scientists due to the concern of pesticide contamination of groundwater (Ju and Kung, 1997; Kung, 1993; Kung, 1990) and also petroleum contamination from the leaking of underground storage tanks and oil spills (Alfres et al.).

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5 A version of this chapter will be submitted for publication. Andrina, J., Wilson, G.W. and Miller, S., A Meso-Scale Laboratory Experiment to Investigate Geochemical Mixing in Blended and Layered Mine Rock Stockpiles
2004b; Søvic and Aagaard, 2003; Schroth et al. 1998). These studies consisted of both laboratory studies and field experiments that involved extensive monitoring of water content in the soil, the application of dye tracers and the development of numerical simulations in order to predict the dispersion of contaminants. The results of these experiments showed the influence of preferential flow paths in soils, which accelerated the release of contaminants because of reduced travel time and contact area (Søvic et al. 2002; Heilig et al. 2003).

Previous studies based on numerical modeling for inclined coarse and fine layers in waste rock piles show flow behavior is controlled by structure (Wilson et al. 2000; Fines et al. 2003; Weeks et al. 2004; Fala et al. 2005). However, the implication of the inclined coarse and fine layer structure on drainage quality is not comprehensively understood. Furthermore, the resulting preferential flow can be expected to influence the performance of limestone layers and limestone covers that are applied to mitigate acid generation from the waste rock piles. The success of these treatments relies on the behavior of flow within the waste rock pile. As previously discussed in Chapter 4, although the geochemical criteria for mitigating ARD generation in the blending system may appear adequate, the effluent produced by the layering system can be acidic when there is limited contact between the limestone and the acid rock. Regarding limestone covers, the occurrence of preferential flow may prevent the alkaline solution from being evenly distributed into the piles, thus reducing the effectiveness of the limestone cover
system in minimizing acid generation. This may also improve the performance, however, when the acidic water generated is less.

A meso-scale laboratory experiment was constructed for the present study to investigate flow paths within inclined coarse and fine layers of waste rock and to examine the influence of these layers on the quality of the drainage from the rock piles. The equipment was purpose-built for the laboratory study, allowing the construction of multiple coarse and fine layers to represent the internal structure of typical waste rock piles. Three meso-scale panels were set up for the experiment. The first panel, considered a base case, designed to investigate the flow path in coarse and fine layers constructed with acid waste rock. The second panel was established to investigate the effect of the alkaline solution generated from a limestone cover on the quality of the leachate from underlying coarse and fine layers of acid waste similar to panel 1. The third panel was designed to observe the impact of inter-bedded limestone within the waste rock and the potential improvement (i.e. layers of limestone sequenced within the acid rock) on the quality of the leachate. During the experiment, different intensities of rainfall were applied to each of the panels.

This chapter presents the measured results for the quality of the leachates that were collected from the individual layers for each of the panels. The specific objectives of this chapter are:

- To assess the influence of rainfall intensity on the quality of the effluents from different layers,
• To assess the effects of preferential flow on the effectiveness of limestone blending and limestone covers, and

• To evaluate the progress of the oxidation within the different types of layers.

The measured quantities of leachate flows, along with the description of flow behavior that were discussed in Chapter 5 provide the information necessary for the analyses of the influence of limestone blending and limestone cover treatments on leachate quality.

6.2 Experimental Design and Set-up

Two geochemically different rocks were selected for the meso-scale experiments. The first rock was an acid rock, namely red waste obtained from the Grasberg Mine, Indonesia, and the second rock was limestone. The red waste is high capacity-potentially acid forming (PAF) material that has a Net Acid Generation (NAG) value greater than 35 kg H\textsubscript{2}SO\textsubscript{4}/ton waste, and it contains approximately 5% to 10% pyrite. Both rocks were crushed and sieved to obtain the desired grain size distribution for the coarse and fine materials. The fine material had an average particle size of less than 10 mm, and the majority of the material passed through a 4.75 mm sieve. The coarse material contained particle sizes between 10 mm and 25 mm. Figure 6.1 shows a photograph of the layered red waste-limestone system in panel 3.

The experiment was conducted using three panels that had different geometries and layer arrangements. The first panel contained seven layers of alternated coarse and fine acid rock (i.e. red waste) as described in Chapter 5. The second panel had the same
Figure 6.1 Layered red waste and limestone at panel 3

number of layers and the same layer arrangement as the first panel, but an additional 500 mm thickness of coarse limestone placed over the inclined layers. The third panel contained nine layers of alternating acid rock and limestone, but the layer arrangement of this panel was different from that for the previous two panels. Each layer was 250 mm wide, 250 mm deep, and 1,500 mm high. The layers were inclined at an angle of repose equal to 37°. The arrangements of the layers in panels 1, 2, and 3 are shown in Figures 6.2, 6.3, and 6.4, respectively.
Figure 6.2 Layer arrangement in panel 1

Figure 6.3 Layer arrangement in panel 2
Figure 6.4 Layer arrangement in panel 3

Drainage points were installed at the bottom of each layer to collect the individual leachate from every layer. In addition, several drainpipes were also attached along the bottom edge of the slope to collect any leachate that might flow downward to the back slope of the panels. The leachates were collected in plastic containers that were placed under the individual drainpipes. A suction lysimeter constructed of a 1-bar ceramic cup was installed at the base of every layer to avoid water ponding and potential transfer from the fine layers to coarse layers due to breakthrough flow.

A water irrigation system was used to simulate rainfall. It consisted of a water supply tank, multi-channel peristaltic pumps, distribution hoses, and soaking sponges. Deionized water was used to simulate rain in panels 1 and 3 and it was applied on the top of every layer except layer 1, and was also applied along the top of the sloped panels. The water used in panel 2 was prepared by passing the deionized water through
a limestone bed to increase alkalinity. The alkalinity of the effluent from the limestone bed varied approximately from 40 to 70 ppm as CaCO$_3$, which was within a range of typical alkalinity values in the leachate from the limestone cover used in the field experiment (Miller, 2007). Similar to application at panels 1 and 2, the alkaline infiltrate was also applied on the top of every layer except layer 1, and along the top of the slope.

All panels were equipped with instrumentation, including thermocouples (type 5TC-PVC-T-24-180 from Omega with accuracy 0.5 °C), tensiometers (elbow tensiometer with accuracy 0.2 kPa), and polyethylene tubes for measuring oxygen. The instrumentation was installed at heights of 500 mm and 1,000 mm for every layer. In addition to these locations, the instrumentation for the second panel was placed at a height of 1,500 mm since the panel 2 was higher than the other panels. The locations for instrumentation are shown in Figures 5.8, 5.9 and 5.10 in Chapter 5.

A series of infiltration tests were performed by applying different rainfall rates to represent the different climate types. Various lower boundary conditions were also set by changing the matric suction in the lysimeter. Leachate volumes were monitored daily, while temperature, oxygen, and matric suction were monitored bi-weekly. The leachate collected from each layer was analyzed for pH, sulfate, alkalinity, and acidity. A sub-sample of leachate was also sent to a commercial laboratory for additional testing to determine the concentration of major cations, major anions, and dissolved metals.

The coarse and fine materials from each layer were sampled and then analyzed for NAG, Acid Neutralization Capacity (ANC), and sulfur, copper, and iron content. The
limestone was also tested by the Acid Buffering Characteristic Curve (ABCC) test to determine the effective ANC that is available. The ABCC test involves a slow titration of a sample with acid while pH of sample is monitored continuously. All analytical procedures followed the test methods described at IWRI and EGI (2002). A small bench scale kinetic test was also set up to gather information on the leaching behavior of the waste rock used in the panel. The test was set up according to the methods developed by Miller et al. (2003a). Approximately 2 kg of waste rock passing the 4.75 mm sieve was placed in a large funnel. The column was irrigated once a week, and the leachate collected from the bottom of the funnel was analyzed for pH, electrical conductivity (EC), acidity, and sulfate.

6.3 Results and Discussion

This section provides the results obtained from laboratory tests conducted on the waste rock samples used for the meso-scale experiment. The laboratory tests consisted of a geochemical analysis of the waste rock samples for sulfur and Acid Neutralization Capacity (ANC), and also Net Acid Generation (NAG). The data obtained from the meso-scale panels are presented in this section including leachate quality collected from each layer within the meso-scale panels when the different rates of rainfall were applied. The test results for the key constituents for ARD and metal leaching such as pH, acidity, sulfate and dissolved copper contained in the leachate sample are presented here.
Data for the leachate quality from every layer of the meso-scale panels are further examined in order to understand the influence of coarse and fine layers on the quality of drainage.

6.3.1 Characterization of Materials

The first type of rock used in this experiment was red waste that contained approximately 5.3% sulfur with pyrite as the main sulfur bearing mineral. The bulk acid rock was found to have a NAG of 76 kg H$_2$SO$_4$/ton and a low ANC of 31 kg H$_2$SO$_4$/ton. This material is considered to be a highly reactive material and can be expected to generate acidic leachate within a short time (Miller et al. 2003a). The bulk sample was also found to contain 84 mg of copper/kg. The second type of rock used was limestone with approximately 85% CaCO$_3$ or 850 kg H$_2$SO$_4$/ton of ANC. The limestone offered strong buffering capacity, as shown in Figure 6.5. The ABCC test indicated that 83% of the measured ANC was readily available for acid neutralization.

Representative samples were also collected from the coarse and fine grain waste rocks layers at every 500 mm vertical height increment for each layer. In total, three samples of coarse grained red waste and three samples of fine grained red waste were collected from each panel. In addition, one sample of coarse limestone and one sample of fine limestone were collected from panel 3. Figures 6.6 (a), 6.6 (b), and 6.6 (c) show the plots of NAGpH versus Net Acid Producing Potential (NAPP) for the coarse and fine red waste samples collected from panels 1, 2, and 3, respectively, based on the
Figure 6.5 Acid Buffering Characteristic Curve of the Limestone

Figure 6.6 Geochemical plots of red waste rock samples from (a) panel 1, (b) panel 2, and, (c) panel 3
plot presented by Miller et al. (1997). The NAPP value was calculated according to total sulfur (%S) and ANC of the sample. The plot is divided into quadrants representing samples that were Non-Acid Forming (NAF) materials, Potential Acid Forming (PAF) materials, Uncertain (UC) materials with low NAPP values and Uncertain (UC) materials with high NAPP values. A sample is classified as NAF materials when the NAPP of the sample is negative and its NAGpH is the same or greater than 4.5. On the other hand, a sample is classified as PAF material when the NAPP of the samples is positive and its NAGpH is less than 4.5 Samples are classified uncertain if there is disagreement between the value of NAPP and NAGpH such as NAPP is negative and NAGpH is less than 4.5 or NAPP is positive and NAGpH is equal or greater than 4.5. The plots suggested that a variability in geochemical composition was still detected as shown in Figure 6.6, although the coefficient of variation was not significant, which varied from 2.5% to 22.8%. The greatest variation in NAPP values was detected in the coarse rock samples for panel 1.

6.3.2. Leachate Quality

All leachates collected during the experiment were analyzed for their chemical constituents. The sample collection periods were divided among the three different stages of the experiment. During the first or initial stage, water was applied to the panels to simulate a high rainfall rate, which was 25 mm/day for panel 1 and 10 mm/day for panels 2 and 3. Leachate was collected and analyzed for pH to evaluate whether acid was being generated or not. The second stage consisted of prolonged
steady-state operation, in which a rainfall rate of 10 mm/day was applied continuously to the panels for about three months. Throughout the three-month irrigation period, all three panels ran most of the time, and were only shut down briefly when there was equipment problems. In addition to pH and conductivity, the samples were analyzed for acidity, alkalinity, sulfate, major cations, and dissolved metals during the second phase.

The third and final stage was the variable infiltration test period. During this stage, different rainfall rates and different bottom boundary conditions were applied within specific cycles of the experiment, with each cycle lasting between three and six weeks. Similar to the previous stage, chemical analyses for the leachates included pH, conductivity, acidity, alkalinity, sulfate, major cations, and dissolved metals.

6.3.2.1 pH Trend

Figure 6.7 shows pH trends in panel 1 during the experiment. Application of the rainfall simulator on panel 1 commenced in August 2006 at a precipitation rate of 25 mm/day. During the first month of the experiment, the pH of the leachate from every layer was measured daily to monitor acid evolution. The pH values for the initial leachates from layers 3, 4, 5, and 6 were between 3.1 and 3.3, while the pH values of the first leachates from layers 1 and 2 were approximately 5.4 and 4.2, respectively. It took approximately three weeks for the pH of the leachate from layer 2 to reach 3.3, while the pH of the leachate from layer 1 took almost four weeks to decrease to
approximately 3.4. After a month, the pH values of the leachates that were collected from the outlets located at the slope still varied from 3.5 to 5.5. These values were relatively higher than the pH values of the leachates collected from the layers. After almost two months of operation, the pH values of the leachates from most of the layers were around 3, while the pH from leachate collected along the basal slope still varied between 3.5 and 5.5. In April 2007, 8 months after the first application of the irrigation water, the pH of the leachate collected from the layers varied between 2.3 and 2.9,
while the pH of the leachates from the basal slope was slightly higher, and varied from 3.1 to 4.4.

A different pH trend was observed in the leachate collected from panel 2, as shown in Figure 6.8. At the beginning of the test, the applied rainfall rate was approximately 10 mm/day, which was lower than the initial rainfall rate applied on panel 1. Thus, the leachate was considered to be more concentrated in panel 2 than that for panel 1. The pH values of the leachates from every layer in panel 2 immediately became acidic at the beginning of the experiment. The range of pH values for the leachates for all the layers was 2.4 to 3.4. The pH values for the leachates collected along the basal slopes were in the range of 2.9 to 4.9, which were higher than pH values of the leachates collected from the individual layers.

Figure 6.8 pH trend in the leachates from panel 2
The pH values of the leachates collected from most of the layers in panel 3 were relatively neutral, as shown in Figure 6.9, due to the existence of the limestone layers. At the beginning of the experiment, relatively low pH values were measured in the leachates from layers 2 and 6, where the pH values were 4.5 and 4.9, respectively. As the experiment continued, the pH value of the leachate from layer 2 tended to become neutral, while the pH value of the leachate from layer 6 was around 4.6.

![Figure 6.9 pH trend in the leachates from panel 3](image_url)
6.3.2.2 Trends in Concentration of Acidity, Sulfate and Dissolved Metals

A rainfall rate of 10 mm/day with a base boundary condition of 0 kPa was applied to all three panels between May 2007 and October 2007 and the infiltration tests were conducted from December 2007 to April 2008.

Figures 6.10 and 6.11 shows trends in acidity and sulfate concentration in the leachate collected from panels 1 and 2, respectively. As can be seen from these graphs, the acidity and sulfate concentrations were relatively similar during the pro-longed steady state operation, from May 2007 to October 2007. The average concentration of acidity and sulfate in the leachates was much lower during this cycle of the experiment compared to the period for the infiltration test (i.e. December 2007 to April 2008) for the same rainfall rate. It was likely that the continual application of the simulated rainfall continuously flushed out the oxidation products. On the other hand, the irrigation test included the drainage period, where no rainfall was applied during this period. Due to the low pH, the acidity measured in the leachate collected from panels 1 and 2 was found to range between 48 mg CaCO₃/l and 59,286 mg CaCO₃/l for panel 1, and between 10 mg CaCO₃/l and 40,005 mg CaCO₃/l for panel 2. The leachate from panels 1 and 2 had sulfate concentrations between 640 mg/l and 95,000 mg/l, and 340 mg/l and 95,000 mg/l, respectively. The lowest acidity and sulfate concentrations were measured in the leachate collected from the outlet located at the top of the basal slope for both panels. When comparing the quality of the leachate that was collected from the individual layers, the lowest concentrations for acidity and sulfate were recorded in
Figure 6.10 Acidity and sulfate concentration trends in leachates from panel 1
Figure 6.11 Acidity and sulfate concentration trends in leachates from panel 2
the leachates from layer 5 of panels 1 and 2. The acidity and sulfate in the leachate from layer 5 of panel 1 ranged from 1,172 mg CaCO$_3$/l to 29,443 mg CaCO$_3$/l, and 1,950 mg/l to 38,000 mg/l, respectively. The acidity in the leachate from layer 5 of panel 2 varied from 600 mg CaCO$_3$/l to 5,668 mg CaCO$_3$/l, and the sulfate concentration varied from 3,000 mg/l to 11,000 mg/l. The maximum concentrations of acidity and sulfate were detected in the leachates from layer 1 in panel 1 and acidity and sulfate concentrations were approximately 60,000 mg CaCO$_3$/l and 95,000 mg/l, respectively. The maximum sulfate concentration was also measured in layer 4 of panel 2 to be approximately 95,000, while the maximum acidity measured at layer 4 from the panel 2 was approximately 40,005 mg CaCO$_3$/l.

The leachates from panel 3 had sulfate concentrations that were much lower than the sulfate concentrations in the leachates from panels 1 and 2, as shown in Figure 6.12. The sulfate concentration in the leachate from panel 3 varied between 150 mg/l and 2,300 mg/l. Acidity was detected only in the leachates from layers 4 and 6 of panel 3 and the concentration ranged from 5 mg CaCO$_3$/l to 35 mg CaCO$_3$/l, which was much less than the acidity of the leachates from panels 1 and 2. The highest alkalinity was measured in the neutral leachate of panel 3, and it varied between 5 mg CaCO$_3$/l and 64 mg CaCO$_3$/l.
Figure 6.12 Sulfate concentration trends in leachate from panel 3

The acidity and sulfate concentrations in the leachate from individual layers were quite varied, particularly for panels 1 and 2. The coefficients of variation for acidity and sulfate concentration were 48% and 49%, respectively, for panel 1. The highest coefficient of variation, which was 66% for both acidity and sulfate concentrations, was calculated for panel 2. Variability was also observed in the leachates from each layer of panel 3, but the coefficient of variation (i.e. 72 %) was higher compared to panels 1 and 2 (i.e. 48 %)
Elevated concentrations of dissolved metals, including Cu, Al, and Fe, were observed in the leachate from panels 1 and 2, as shown in Figures 6.13 and 6.14, respectively. Copper concentrations in the leachate from panel 1 varied from 0.2 mg/l to 56 mg/l, while aluminum and iron concentration ranged from 4 mg/l to 3,459 mg/l, and from 0.03 mg/l to approximately 11,500 mg/l, respectively. Similar to acidity and sulfate concentrations, the highest copper, aluminum and iron concentration were observed in leachate from layer 1 of panel 1, and the lowest concentration for these constituents was in the leachate from the highest elevation at the basal slope (i.e. S1) of panel 1. The range of concentration of copper, aluminum and iron in leachate from panel 2 was similar to the range of concentration in leachate from panel 1. The copper concentration varied between 0.7 mg/l and 63 mg/l, and the aluminum and iron concentration varied between 0.7 mg/l and 2,498 mg/l, and between 2.6 mg/l and 7,025 mg/l, respectively. The maximum concentration of these constituents in panel 2 was measured in the leachate from layer 4 and the minimum concentration of these constituents was observed in the leachate that was collected from the outlet at the highest elevation from the basal slope (i.e. S1).

Since the concentration of these dissolved metals was pH dependent, the acidic leachate from panels 1 and 2 contained relatively high metal concentrations. On the other hand, the leachates from panel 3 showed substantially lower dissolved metal concentrations (i.e. copper, aluminum and iron) compared to those observed in the leachates from panels 1 and 2, as shown in Figure 6.15. For example, the copper
Figure 6.13 Copper, aluminum and iron concentration trends in leachates from panel 1
Figure 6.14 Copper, aluminum and iron concentration trends in leachates from panel 2
Figure 6.15 Copper, aluminum and iron concentration trends in leachates from panel 3
concentration detected in leachate from layer 4 and 6, in panel 3 varied from 0.2 mg/l to 0.8 mg/l. The aluminum and iron concentrations in the leachate from panel 3 ranged from 0.2 mg/l to 13 mg/l, and from 0.03 mg/l to 0.8 mg/l, respectively. The highest concentration of these constituents was recorded in the leachate from layer 6 which had a relatively low pH compared to the pH of leachate from the other layers.

6.3.3 Concentrations and Loadings of Acidity and Sulfate at Different Rainfall Rates

The relationship between rainfall rate and leachate chemistry was examined by applying different irrigation rates to the panels. Before applying a particular infiltration rate, the panels were allowed to drain for 7 to 10 days to minimize the influence of the previous rainfall. One exception to the length of the drainage period was that the panels were allowed to dry for almost six weeks prior to the application of the rainfall rate of 5 mm/day. Each cycle of the experiment was continued until a pseudo steady state was achieved (i.e. outflow equal to infiltration), which generally required 1 to 3 weeks, depending on the rainfall rates. At the end of each experiment cycle the leachates were sampled for pH, acidity, alkalinity, and sulfate.

Figures 6.16(c) and 6.16(d) show the acidity and sulfate concentrations in the leachates collected from the outlets in panel 1 when the application rates were 2 mm/day, 5 mm/day, and 10 mm/day. As shown in these figures, the acidity and sulfate concentration in every layer changed when the rainfall rate was increased, however the concentration did not always decrease when the rainfall rate was increased. For example, the range in acidity was from 1,241 mg/l to 40,260 mg/l when rainfall
Figure 6.16 (a) Volume distribution, (b) acidity loading distribution, (c) acidity concentration, (d) sulfate concentration, (e) acidity loading, (f) sulfate loading from each layer in panel 1 for application rates of 2 mm/day, 5 mm/day and 10 mm/day
was 2 mm/day, whereas the range was 876 mg/l to 32,208 mg/l for a rainfall rate of 10 mm/day. Similarly, sulfate concentrations for a rainfall rate of 2 mm/day varied from 3,450 mg/l to 95,000 mg/l, whereas they ranged from 2,000 mg/l to 68,000 mg/l for a rainfall of 10 mm/day. At all rainfall rates investigated, the leachate collected from the outlet at the highest elevation of the basal slope (i.e. S1) always had the lowest concentration. When comparing the quality of the leachates for the different rainfall events at individual outlets for each layer, the acidity and sulfate minimum concentrations, which were varied from 6,670 mg/l to 29,443 mg/l, and from 15,500 mg/l to 21,000 mg/l, respectively, were detected in the leachate from layer 5. The maximum acidity and sulfate concentration were measured in the leachate from layer 1, in which the acidity and sulfate acidity concentration varied between 29,544 mg/l and 59,286 mg/l, and between 60,000 mg/l and 95,000 mg/l, respectively.

Figures 6.17(c) and 6.17(d) show acidity and sulfate concentration, respectively for the leachates that were collected from panel 2 at the different rainfall rates. The influence of rainfall on the concentrations were similar to that observed for panel 1, which the increase of rainfall rates did not always reduce the acidity and sulfate concentrations in the leachate from every layer. There was also a slight change in the range of concentrations. For example, the acidity concentrations ranged from 1,100 mg/l to 40,000 mg/l when the rainfall rate was 2 mm/day and the sulfate concentrations ranged from 2,400 mg/l to 95,000 mg/l. For the 10 mm/day rainfall rate,
Figure 6.17 (a) Volume distribution, (b) acidity loading distribution, (c) acidity concentration, (d) sulfate concentration, (e) acidity loading, and (f) sulfate loading from each layer in panel 2
the acidity and sulfate concentrations ranged from 1,127 mg/l to 32,332 mg/l and from 2,400 mg/l to 50,000 mg/l, respectively. Similar to panel 1, the lowest concentration of acidity (i.e. from 521 mg/l to 1,127mg/l) and sulfate (i.e. from 1,200 mg/l to 2,400 mg/l) were detected in the leachate collected from the outlet located at the highest elevation of the basal slope (i.e. S1). When comparing the concentrations among the layers for the different rainfall rates, the lowest acidity and sulfate concentrations, ranging from 3,284 mg/l to 5,668 mg/l for acidity, and from 7,520 mg/l to 18,750 mg/l for sulfate were measured in the leachate from layer 5. The maximum acidity and sulfate concentrations in panel 2 were observed in the leachate from layer 4, where the acidity varied from 30,945 mg/l to 40,005 mg/l, and the sulfate concentration ranged between 50,000 mg/l and 95,000 mg/l.

The sulfate concentrations for each layer in panel 3 are presented in Figure 6.18(c). The sulfate concentrations were relatively similar in leachate from each layer, particularly among the first five layers of panel 3. It can be seen that the rainfall rate did not have a significant influence on the sulfate concentrations. The sulfate concentrations ranged from 400 mg/l to 1,750 mg/l when the rainfall rate was 2 mm/day, and between 350 mg/l and 1,625 mg/l when the rainfall rate was 10 mm/day. The highest concentration of sulfate was detected in the leachate from layer 4 when the rainfall rates were 2 mm/day and 10 mm/day, and in the leachate from layer 5 when the rainfall rate was 5 mm/day. The lowest sulfate concentration was consistently observed in the leachate collected from layer 7 at all rainfall rates.
Figure 6.18 (a) Volume distribution, (b) sulfate loading distribution, (c) sulfate concentration, (d) sulfate loading from each layer in panel 3
The distributions of outflow for each layer in panels 1, 2 and 3 at the different rainfall rates were presented on Figures 6.16(a), 6.17(a), and 6.18(a), respectively. As shown in these figures, the role of water volume in diluting the contaminant concentration was not consistent, because when a given layer produced a greater volume of leachate, it also produced a more concentrated leachate. These results suggested that the amount of infiltrated water was not the only parameter that affected the leachate quality. The results of chemical analyses for the leachates from the individual layers as well as the analysis of flow behavior within these panels (as presented in Chapter 5) suggested that the constituent concentrations in the leachates were mainly controlled by the flow path of the leachate, as well as by the contact time between the infiltrated water and the waste rock. Thus, the combined effects of the volume of water, the flow path, and contact time determined the concentration of the contaminants in the leachates. Similar results were reported by Gish et al. (2004), who suggested that a different flow mechanism and different flow paths might be involved in contaminant transport within a soil when different irrigation rates are applied. However, Wu et al. (2007), who conducted an experiment to study solute transport from a leaching dump using a column that contained a vertical layer of coarse ore material and a second layer of fine material, indicated different results. Wu et al. (2007) showed that the solute concentrations within the layers were controlled by the leachate volumes since the higher concentrations were detected in the layer that produced the lowest volume of leachate.
The general trend for panel 1 that showed decreases in acidity and sulfate content for the leachate collected from layers near the basal slope (layer 1) towards the outer layers (i.e. layer 6) illustrate the influence of flow path on the quality of the leachate. As discussed in Chapter 5, the origin of the water was different for the layers L1 and L2 located near the basal slope compared to outer layers such as L6 to L9. The leachate collected from the layers located near the basal slope (i.e. lower boundary of the panel at ports S1, S2 and S3 in Figure 6.2) came mainly from the rainfall applied at the upper surface of the panel. The breakthrough test conducted on panel 1 with the application of 10 mm/day of rainfall only at the surface of layers 6 and 7 demonstrated that the water traveled as far as layers 1 and 2. A portion of the leachate that was collected in layer 1 of panel 1 during the 10 mm/day rainfall cycle originated from the top of layers 6 and 7. Although the actual flow paths of the water collected in the leachates from layers 1 and 2 were unknown, it was obvious that the flow paths of infiltrated water were long enough to allow the water to pick up a significant amount of solute and dissolved metal along the pathway. As a result, high solute and dissolved metal concentrations were observed in the leachates from layers 1 and 2.

Since most of the water that was applied to the top of layers 6 and 7 traveled vertically downward into the underlying layers, as proved by the breakthrough test which was previously explained in Chapter 5, the water that reported to the outer layers, such as layers 5 and 6, likely originated from the rainfall application along the slope, particularly from the lower part where the water tended to flow to the outer
layers. The early response for leachate that was typically collected in layer 5 at the beginning of an experimental cycle also suggested that the water traveled a relatively short distance. A similar explanation could also be offered for the low concentrations seen in the leachate collected from the outlet at the top of the basal slope in S1. The leachate collected from this outlet most likely originated from breakthrough flow for the rainfall application on layers 2 and 3 that were located relatively close to the S1 outlet. The flow path of leachate collected from the lower layers such as layers 1 and 2 could be half as long as the flow path for the leachate collected from the layers located nearer to toe of the slope in layers 5 and 6. The relatively short flow path and the short residence time limited the water’s capacity to extract the oxidation products, resulting in relatively low acidity and solute concentrations in the leachate.

There may have been a different mechanism that contributed to the high concentrations of solute and dissolved metals from the leachate collected from layer 4 of panel 2, which was located in the middle of panel. The volume of leachate collected from layer 4 in panel 2 was considerably lower for all rainfall application rates compared to the volume of the leachates collected in the other layers within the same panel. This low flow suggests that the leachate collected may have had a relatively longer residence time through capillary flow. Layer 4 contained fine material and water flowing according to a capillary flow mechanism with a long residence time would have had longer contact times between the aqueous phase and a solid phase in the waste rock. Thus, more
solute and dissolved metals could be extracted resulting in higher solute and dissolved metal concentrations in the leachate.

As previously described, the concentration of constituents generally decreased when the rainfall rate increased, although this trend was not always consistent in every layer. It was likely that changes in the flow path along with the increased volume of water when the rainfall rate was increased, affected the concentration of constituents. At the high rainfall rate, the water was transported downward through a vertical flow mechanism, therefore the flow path was shorter compared to the flow path for water that was transferred through the inclined layer profile. An increase in rainfall rate also lead to shorter residence times because the water flowed faster, as also observed by Sililo and Tellam (2000).

The geochemical speciation code PHREEQC (Parkhurst and Appelo, 2000) was used to confirm whether the concentrations of species, such as sulfate and dissolved metals in the leachates of panels 1 and 2, were high enough to precipitate minerals, which could reduce the sulfate and dissolved concentrations in the leachate. The concentrations of constituents found in the leachates produced from the panels were used as inputs in order to calculate the saturation index (S.I.) of minerals. A S.I. equal to zero indicates an equilibrium condition. An S.I. greater than zero indicates a supersaturated solution in which precipitation has likely occurred. On the other hand, an S.I. less than zero implies that the material is under-saturated, and therefore precipitation is unlikely. The results of the speciation model indicated that the concentration in the
leachate samples were less than saturated with respect to most of the major dissolved metals, but they were over-saturated with gypsum. The input of Ca\(^{2+}\) originated from the carbonate minerals that were contained in the acid rock, as indicated by the ANC value of the waste rock samples. The precipitation of gypsum can limit the sulfate concentration in the leachate. The SO\(_4^{2-}\) balance indicated only about 2% to 4% of the sulfate ions were in the form of gypsum. Therefore, the attenuation of sulfate by gypsum is not considered to be significant, particularly when high concentrations of sulfate were being flushed continuously from the panel.

Since panel 3 contained both acid rock and limestone, the pH values and sulfate concentrations of the leachates from each layer were influenced by the neutralization between the acid leachate and the alkaline leachate. The relatively similar pH and sulfate content for the leachates, particularly for the leachates from layers 1 to 5, suggested that mixing occurred along the pathways of the leachates. Furthermore, the residence time was adequate for water to obtain alkalinity from the limestone.

Relatively low sulfate concentrations were measured in the leachates collected from layers 6 and 7 when compared with the concentrations observed in the leachates from the other layers in panel 3 (i.e. Figure 6.12). The layer arrangement for layers 6 to 9 seemed to control the flow path of the leachates, thereby affecting the effluent quality from both layers 6 and 7. Layers 6 to 9 consisted of coarse materials; layers 6 and 8 contained the acid rock materials, while layers 7 and 9 were limestone. The fact that no leachates were collected from the bottom of layers 8 and 9 indicated that these
overlying layers did not retain the irrigation water that was applied along the outer
slope of panel 3. Hence, the flow of water was predominantly vertical (i.e. gravitational
flow) into layers 6 and 7 instead of traveling according to the inclined profile of the
layers. Since the contact time and flow path were not sufficiently long for the infiltrated
water to accumulate alkalinity from the limestone in layers 7 and 9, the infiltrating
waters that passed through layer 7 did not have adequate alkalinity to neutralize the
acid leachate that was already present in layer 6 and the acid generation rate exceeded
the alkalinity load. Therefore, relatively low pH values were detected in the leachate
from layer 6, and the leachates from both layers 6 and 7 tended to have low sulfate
concentrations. These results demonstrated that the coarse limestone and coarse acid
rock that were placed adjacent to each other can still potentially produce acid.

Figures 6.16(e) and 6.16(f) show the loading of acidity and sulfate, respectively,
for each layer in panel 1. In general, the maximum acidity and sulfate loading increased
as the rainfall rate increased. For example, the maximum acidity and sulfate loading
rose from 3.9 g/day to 28.6 g/day and from 9.6 g/day to 60.4 g/day, respectively, as the
rainfall rate changed from 2 mm/day to 10 mm/day. The minimum loading was
constantly recorded in layer 6 for all rainfall rates. Alternatively, the maximum release
rate was recorded for different layers depending on the rainfall rate. For example, when
the rainfall rate was 2 mm/day, layer 2 (fine) in panel 1 had the highest loading.
However, when the rainfall rate was increased to 5 mm/day and 10 mm/day, the
maximum loading was observed in layer 1 (coarse). The maximum acidity and sulfate
loadings were 4.4 g/day and 10.2 g/day, respectively in layer 2, at 2 mm/day, which were much lower than the maximum acidity loading of 23.2 g/day and sulfate loading of 52.6 g/day in layer 1 at the 10 mm/day rainfall rate.

Similar performance was observed in panel 2, as shown in Figures 6.17(e) and 6.17(f), with the maximum acidity and sulfate loadings being higher at the higher rainfall rates. For example the maximum acidity and sulfate loadings were observed in Layer 1 (coarse) at 4.4 g/day for acidity and 10.2 g/day for sulfate when the rainfall rate was 2 mm/day. As the rainfall rate increased to 10 mm/day, the maximum acidity and sulfate loading also increased to 23.28 g/day for acidity and 52.6 g/day for sulfate in Layer 1. The minimum acidity and sulfate loadings were observed in layer 6 (fine), which acid and sulfate loadings were varied from 0.6 g/day to 1 g/day, and from 1.2 g/day to 1.7 g/day, respectively, for different rainfall rates.

As shown in Figure 6.18(d), the changes in loadings due to changes in the rainfall rates in panel 3 were not as pronounced as observed in panels 1 and 2. In addition, the changes were not consistent in each layer due to the influence of the neutralization process. The maximum sulfate loading in layer 4 of panel 3 increased slightly from 0.92 g/day to 1.03 g/day as the rainfall intensity was increased from 2 mm/day to 5 mm/day. However, the sulfate loading decreased to 0.69 g/day when the rainfall intensity was further increased to 10 mm/day. This was because of a transfer of the maximum sulfate release rate from layer 4 (fine) at rainfall rates of 2 mm/day and 5 mm/day to layer 3 (coarse) at rainfall rate equal to 10 mm/day.
The loadings from each layer in the panel varied because the loading rate was determined by both the concentration of solute or dissolved metals, as well as by the volume of the leachate. However, it was likely that the leachate volume had a stronger positive correlation than the concentration of contaminant. Thus, the layers with the highest or lowest leachate volumes tended to have the maximum or minimum loading of constituents. For example, although the minimum acidity or sulfate concentration was measured in the leachate of layer 5, the minimum acidity or sulfate loading was recorded for layer 6 in panels 1 and 2, which had a lower volume compared to layer 5. The maximum loading was recorded for layer 1 of panels 1 and 2, since the leachate from layer 1 did not only have a relatively high concentration, but it also produced a high volume of leachate.

The contribution of the leachate from preferential pathways to the total loading of constituents was dominant, as shown in Figures 6.16(b), 6.17(b) and 6.18(b). For example, the leachate from layer 1 of panel 1 contributed between 37% to 43% of total acidity loading from panel 1 and about 26% to 51% of the total acidity loading from panel 2. Similar to panels 1 and 2, leachate from layer 4 of panel 3, which was a preferential pathway for the water flow, accounted for about 31% to 54% of the total sulfate loading. The percentage of acidity or sulfate loading from every layer in the panels 1 to 3 (i.e. Figures 6.16(b) 6.17(b) and 6.18(b)) were comparable with the percentage of outflow leachate distribution from every layer which are shown in Figures 6.16(a), 6.17(a), and 6.18(a).
6.3.4 Statistical Analyses of Acidity and Sulfate Concentrations and Loading Rates

Statistical analyses were conducted on the acidity, sulfate concentration and loadings data in order to evaluate if the changes in concentration and loading constituents with rainfall rate are significant. For this purpose, the acidity and sulfate concentrations in leachates from layers in the panels are plotted as a box-whisker plot to show the distribution of the concentration at the different rainfall rates. The dot in the box indicates the mean value of the outflow flux from the lysimeter. The 75 and 25 percentiles are indicated at the upper and lower edge of the box, respectively, and the horizontal line in the middle of the box shows the median. In addition, an f-test was also performed on acidity and sulfate concentrations and loading rates. The f-test is a statistical analysis to test the null-hypothesis where the variance of the two populations are equals, or $S_1^2 = S_2^2$. The f-value is calculated as a ratio of two variances. The calculated f-value is then compared to the expected value from the f-distribution table based on the degree of freedom (i.e. number of population $(n) - 1$). When the calculated F-calculated is lower than the F-table, the difference between two population is not significant.

Figure 6.19 shows box-whisker plots for acidity and sulfate concentration from panel 1. As can be seen from this plot, the acidity distribution at 2 mm/day rainfall rate was similar to the acidity distribution at 10 mm/day. On the other hand, acidity distributions for 5 mm/day were slightly different from the acidity distribution for the 2 mm/day and 10 mm/day rainfall rate. The sulfate concentration distribution was also
Figure 6.19 Panel 1 box whisker plots for (a) acidity, and (b) sulfate concentration

different for each rainfall rates. The range of sulfate concentrations at 2 mm/day of rainfall rate was much wider than the range of sulfate concentration for the other rainfall rates. Although, the distribution of acidity concentration was different from the distribution of sulfate concentration, the results of the f-test indicated the differences in acidity and sulfate concentration in the leachates from the different rainfall rates was generally not significant (i.e. F-calculated = 1.4 - 2.5; F-table= 2.58). Only the sulfate concentration in leachate from the 5 mm/day rainfall rate was significantly different from the sulfate concentration in the leachate from 2 mm/day and 10 mm/day rainfall rates.
The influence of rainfall rates on the acidity and sulfate concentrations was not significant for the leachates from panel 2. Distributions of acidity and sulfate concentration in leachates from panel 2 were almost similar at the different rainfall rates, as shown in Figure 6.20. The results of f-tests also suggested that the difference in concentrations for the different rainfall rates was not significant (i.e. F-calculated = 1.2 – 1.76, F-expected 2.44).

![Box whisker plots for acidity and sulfate concentration in leachates from panel 2.](image)

**Figure 6.20** Panel 2 box whisker plot for (a) acidity, and (b) sulfate concentration

Figure 6.21 shows a box-whisker plot for sulfate distribution in leachates from panel 3. As previously discussed, the influence of the rainfall rate on sulfate concentrations in the leachate from panel 3 was not pronounced compared to the other
Figure 6.21 Panel 3 box whisker plot for sulfate concentration

panels. The box-whisker plot and the result of the f-test indicated the changes in sulfate concentration, when the rainfall rates increase, was not significant (i.e. F-calculated = 1.07 – 1.5, F-expected 2.78).

When the f-test was performed on the acidity and sulfate loading rates of the leachate from panels at the different rainfall rates, the calculated f-values for acidity and sulfate loads in the leachates from panels 1 and 2 at the different rainfall rates were much higher than the expected f-value. This result suggested that the difference in acidity and sulfate load at the different rainfall rates was significant. This result is
consistent with the analysed leachate distributions in the layers, which indicated that
the flow distribution showed a significant difference when the different rainfall rates
were applied as discussed in Chapter 5. On the other hand, the difference in sulfate
loading in the leachate from panel 3 at the different rainfall rates was not significant
since the calculated f-value was lower than the expected f-value. However, the
coefficient of variation for the sulfate loadings from panel 3 largely decreased from
147% to 94% for 2 mm/day and 10 mm/day rainfall rates, respectively.

The difference in acidity and sulfate concentration in the leachate which was not
significant for the different rainfall rates suggested that increasing the amount of
rainfall could change the flow path which leads to a change in concentrations in the
leachate outflow, but increasing amount of the rainfall was not enough to make
significant changes in concentrations of the constituents. However, increasing the
amount of rainfall was enough to significantly change the loadings of constituents due
to the significant change in the volume of outflow. The results, which are presented in
Chapter 5, show that the leachate distribution in the layers was significantly different
when the different rainfall rates were applied.

6.3.5 Oxidation Rate of Waste Rock at Meso-scale Panels

As discussed in Chapter 3, the similar acid waste rock (i.e. red waste) had been
tested in the 2 kg and 35 kg leach columns, the 500 tonne test pad, and also in the full-
scale trial dump. The sulfate concentration and the sulfate release rate from the
leachate generated from these tests and the layers of meso-scale panel are compared
for the first year period in Table 6.1. The acidity and sulfate release rate for the leachate from the meso-scale panel was calculated based on the mass of the waste rock for the whole panel instead of the mass of the waste rock for each layer.

**Table 6.1 Comparison of the acidity and sulfate concentrations, sulfate release rates and Oxidation Rates of red waste from the different scale experiments**

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Maximum Acidity concentration</th>
<th>Maximum Sulfate concentration, mg/l</th>
<th>Sulfate release rate, mg/kg per day</th>
<th>Average oxidation rate, O₂ kg/kg material-second</th>
</tr>
</thead>
<tbody>
<tr>
<td>Leach column</td>
<td>7,156</td>
<td>6,800</td>
<td>27 - 41</td>
<td>3.4 x 10⁻¹⁰</td>
</tr>
<tr>
<td>Test pad</td>
<td>92.812</td>
<td>85,400</td>
<td>40</td>
<td>4.6 x 10⁻¹⁰</td>
</tr>
<tr>
<td>Full-scale field trial</td>
<td>15,000-83,000</td>
<td>16,830-101,600</td>
<td>2.8 – 7.2</td>
<td>7.5 x 10⁻¹¹</td>
</tr>
<tr>
<td>Meso-scale panel</td>
<td>208 – 6,659*</td>
<td>1,700 – 2,250*</td>
<td>28.9*</td>
<td>3.3 x 10⁻¹⁰</td>
</tr>
<tr>
<td>During long term 10 mm rainfall</td>
<td>139 – 5,309**</td>
<td></td>
<td>24.4**</td>
<td></td>
</tr>
<tr>
<td>Meso-scale panel 1</td>
<td>1,241– 40,260*</td>
<td>3,450 – 95,000*</td>
<td>11.9*</td>
<td>1.4 x 10⁻¹⁰†</td>
</tr>
<tr>
<td>During 2 mm rainfall test</td>
<td>5.668–0,005**</td>
<td>2,400–95,000*</td>
<td>12.7**</td>
<td>1.5 x 10⁻¹⁰**</td>
</tr>
<tr>
<td>Meso-scale panel 1</td>
<td>821 – 29,554*</td>
<td>2,700 – 60,000*</td>
<td>22.8*</td>
<td>2.6 x 10⁻¹⁰†</td>
</tr>
<tr>
<td>During 5 mm rainfall test</td>
<td>521– 30,945**</td>
<td>1,200–70,000*</td>
<td>18.7**</td>
<td>2.2 x 10⁻¹⁰**</td>
</tr>
<tr>
<td>Meso-scale panel 1</td>
<td>876 – 32,208*</td>
<td>2,000 – 68,000*</td>
<td>14.5*</td>
<td>1.7 x 10⁻¹⁰†</td>
</tr>
<tr>
<td>During 10 mm rainfall test</td>
<td>1,127– 2,332*</td>
<td>2,400–50,000*</td>
<td>56.7**</td>
<td>6.5 x 10⁻¹⁰**</td>
</tr>
<tr>
<td>Meso-scale during experiment</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(approximately 1 year)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>22.2*</td>
<td>2.6 x 10⁻¹⁰†</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>24.3**</td>
<td>2.8 x 10⁻¹⁰**</td>
</tr>
</tbody>
</table>

Note: * is panel 1; ** is panel 2
In general, the range in the concentrations for acidity and sulfate in the leachates collected from the meso-scale panel experiments were comparable with the range of acidity and sulfate concentration from the leach column. In contrast, the concentration of these constituents in the leachates from the meso-scale panel were still lower than the ranges for the same parameters in the leachates collected from the test pad and full-scale dump, as discussed in Chapter 3 and Miller et al. (2003a). Exceptions are for the concentrations of constituents in leachate from particular layers such as layer 1 of panel 1 or layer 4 of panel 2 which were recorded as maximum concentrations. For example, the maximum sulfate concentrations in the leachate from the full scale dump and the test pad were 101,600 mg/l, and 85,400 mg/l, respectively, which were similar to the maximum sulfate concentrations from layer 1 of panel 1 and layer 4 of panel 2, which had sulfate concentrations of 95,000 mg/l. The mass of waste rock that was involved in the meso-scale experiment was much smaller than the test pads and the full-scale trial dump. This was the main reason for the low concentrations of the constituents in the leachates from the meso-scale panels.

The sulfate release rate for each panel was determined from the concentration of sulfates with assuming the sulphate precipitation is negligible, and the total flow volume of the leachate collected in each of the drains (i.e. by multiplying the concentrations by the total volume of discharge to acquire the mass). It was then subsequently divided by the mass of the waste rock in the panel, and the number of days to give a rate which was expressed as mg/kg-day. Due to the difference in the test
setting, the sulfate release rate was calculated separately for the prolonged steady state operation, and the irrigation test period, and also according to the whole time of the panel operation. The sulfate release rate from the panels for approximately one year of operation was 22.2 mg/kg-day for panel 1 and 24.3 mg/kg-day for panel 2. The sulfate release rate from these panels was approximately one half of the rate of sulfate release from the leach columns or the test pad trials, but almost four to eight times of the sulfate release rate from the full-scale trial dump. The sulfate release rate from the irrigation test was quite varied for the different rainfall rates. The highest sulfate release rate was recorded during the 5 mm/day rainfall test application in panel 1. The drainage period, which took approximately 5 weeks, was likely due to the accumulation of the oxidation products which were then flushed during the 5 mm/day rainfall rate test. A relatively high sulfate release rate was also recorded during 10 mm/day rainfall rate on panel 2. The sulfate release rate from panel 2 during the 10 mm/day rainfall rate was almost three to four times the sulfate release rate from the 2 mm/day and 5 mm/day rainfall rates. The reason for this high sulfate release rate during the 10 mm/day rainfall rate test was not clear, but it might be due to the limited flushing of oxidation products during the 2 mm/day irrigation rate test that was conducted before the 10 mm/day rainfall rate test.

The oxidation rate for the meso-scale panel was calculated according to the sulfate release rate. The pyrite equation reaction which was presented in Reaction (3.1) of Chapter 3 was used to calculate the oxidation rate and oxygen consumption rates
equal to 0.583 times the sulfate loadings based on this stoichiometry of reaction (Bennett et al. 2000). Overall, the oxidation rates for panels 1 and 2 were 2.6 X 10^{-10} to 2.8 X 10^{-10} kg/kg per second, respectively, which are slightly lower (i.e. approximately 25% to 43%) than the oxidation rate from the leach column and the test pad, but the oxidation rate from the meso-scale panel was higher (i.e. approximately 70%) than the oxidation rate from the full-scale waste rock dump trial. The high oxidation rate from the meso-scale panel during the first year indicated that the acid waste rock in the panel was fully oxygenated.

6.3.6 Temperature Observation within the Panels

Temperatures within the layers of the panels varied from 8°C to 23°C. The observed temperature changes were generally associated with changes in the ambient temperature. The highest temperatures were recorded during the summer months of July and August, while the temperatures recorded in December were relatively low. These temperatures were much lower than the temperatures recorded in the field-scale dump trial containing a similar type of waste rock that attained temperatures up to 60°C (Andrina et al. 2006) as well as those for other sulfide minerals waste rock in the literature (e.g. Harries and Ritchie, 1981; Sracek et al. 2004). While elevated temperatures were not observed, based on the quality of leachate that became acidic, it is clear sulfide mineral oxidation occurred within the panel and it can be assumed heat was generated as a result of oxidation. However, high temperatures were not
recorded within the layers since the panels were not insulated and the heat generated did not balance the heat lost to the surrounding environment.

All the layers were observed to have similar temperatures with the maximum temperature difference among the layers at any given time to be 1°C. Also, there was no significant difference between the temperatures in the fine layers and the coarse layers. Furthermore, temperatures were also similar between the layers of limestone compared to the acid waste rock layer. It was initially expected that the temperature in the acid waste rock would be higher than the limestone layer due to the reactivity of the acid waste rock. In summary, these results support the conclusion that the internal temperatures of the panels were dominated mainly by the ambient air temperature.

6.3.7 Oxygen Profile on Layers

Oxygen was allowed to flow freely into the panels through the top of the panels and also through the holes or vents in the side wall located near the base of the panels. Oxygen concentration remained near ambient concentration, and the values varied from 19.8% to 20%. The high oxygen concentration indicated that there was no restriction in the oxygen supply. The height of the pile was not considered deep enough to limit oxygen diffusion through the top of the panel. This mechanism, together with the lower venting holes, maintained uniform atmospheric oxygen concentrations throughout the entire region of each panel.

The difference in oxygen concentration between the layers was relatively insignificant, which was indicated by the low coefficient of variation that varied
between 1% and 3%. The data also did not show that the oxygen concentration in the fine layer was less than in the coarse layer. The results demonstrated that the air permeability between the fine layer and the coarse layer was not different enough to create a different oxygen profile.

6.3.8 Oxidation Path

The presence of secondary minerals as oxidation products were observed accumulating on the surface of the rocks within the panels. Figure 6.22 is a photograph that shows the progress of the appearance of oxidation products in panel 1. Visual observations in panel 1 showed that the oxidation products first appeared in the lower part of the coarse layer about two weeks after the first application of rainfall and then progressed to the upper part of the coarse layers. The appearance of oxidation products was much slower in the fine layers than in the coarse layers. The oxidation products were not observed on the fine layer until more than a month after the rainfall application began as shown in the photograph in Figure 6.22(b). Similar to coarse layers, the oxidation products started to appear much later at the upper part of the layers. The appearance of oxidation products was more pronounced in the lower part compared to the upper part of the fine layers for approximately 18 months until the end of the experiment. A similar sequence with regard to the appearance of secondary minerals was also observed in panel 2. The presence of oxidation products in panel 3 occurred mostly on the coarse layers of the acid rock, as shown in Figure 6.1. After one year of
Figure 6.22 Oxidation progress in layers at panel 1
operation, the appearance of oxidation products was still not obvious on the fine layers, and was visible only at isolated “hot spots.”

The lag in the appearance of oxidation products within the fine layers can be attributed to the combined effects of the difference in water retention characterization (i.e. SWCC), degree of saturation, oxygen diffusion coefficient, hydraulic conductivity and surface area between the coarse and fine material. These factors could have led to differences in the oxidation rates. The larger exposed surface area of the fine grain materials due to unsaturated conditions can accelerate oxidation rates (Strömberg and Banwart, 1999; Janzen et al. 2000; and Hollings et al. 2001). Alternatively, higher water content in the fine layer can reduce the oxygen availability, since oxygen diffusion in water is orders of magnitude less than in air (Weast, 1985), however the low oxygen concentration was not observed in the fine layers of the panel. Molson et al. (2005) developed a conceptual model to describe the oxidation of pyrite in unsaturated waste rock piles. His model results showed that high oxidation rates can still occur in the fine layers although the moisture content is relatively high. The measured values for matric suction and oxygen concentration for both the coarse and fine layers in the panel were almost similar. Based on these matric suction measurement results, it was not expected that the oxidation at the fine layer would be delayed due to low oxygen concentration.

Geochemical testing of the samples showed that there was a variability in the NAG and ANC values in the samples from the coarse versus the fine materials, although it could not be concluded that the NAG values were higher in the coarse materials
compared to the fine materials, or that the ANC values were higher in the fine materials as compared to the coarse materials. However, the variability of the materials may have contributed to the differences in the oxidation rates between the coarse and fine materials.

The presence of neutralization minerals in both materials could also affect this delay. When the waste rock was crushed to become fine grained, the sulfide minerals and other minerals, such as carbonate, were also liberated. The presence of carbonate minerals in the waste rock was shown in the ANC value. As a result, the acid produced was neutralized by the carbonate mineral. Once the carbonate minerals were depleted, the secondary minerals started to appear. For comparison, a delay in the production of acid products was also observed in the leachate collected from the experiment in the small-scale column, which was described in Section 6.2. The size of the material used for this test was relatively similar to the size of the fine grain material used in the panels, in which the particle size was smaller than 4.75 mm. For the first three months at the beginning of the experiment, pH values of approximately 4.5 to 5 were recorded. The pH of the leachate from the column test then started to decrease, as shown in Figure 6.23.

The delayed appearance of oxidation products had no observable implications regarding the quality of the leachates that were collected from the fine layers. Since the leachates collected from the fine layers did not infiltrate solely through the fine layers,
low pH values were still reported in the leachate from fine layers, as described in section 6.3.2.1.

6.3.9 Performance of Limestone Layering and Limestone Cover

The effectiveness of limestone blending was determined by geo-chemical criteria, such as the Acid Neutralization Capacity (ANC)/Maximum Potential Acid (MPA) ratio and the effectiveness of blending, as discussed in Chapter 4. The existence of coarse and fine layers within the internal structure of the waste rock dump can affect
the quality of the drainage due to preferential flow that may alter or restrict the contact between the neutral leachate and the acid leachate. The similarity in quality among the leachates collected from layers 1 to 5 of panel 3 suggested that mixing of the acid and alkaline leachates occurred along the flow path.

A speciation model was performed using PHREEQC to examine the presence of Calcium minerals in the leachate as an indicator for the influence of limestone on leachate quality. The concentrations of constituents in the leachates from the panels were used as inputs in order to calculate the saturation index (S.I.) of the minerals.

The results of the speciation model for the leachates collected from the individual layers in panel 3 showed that the S.I. of gypsum was super-saturated in leachates collected from layers 1 to 5, which indicated the precipitation of gypsum. On the other hand, the S.I. of the leachates from layers 6 and 7 were always under-saturated with respect to gypsum or with no precipitation of gypsum due to a low calcium concentration. The precipitation of gypsum was controlled by the input of Ca$^{+2}$ from the limestone (Sracek et al. 2004). The PHREEQC calculated results suggested that the amount of Ca$^{+2}$ in the leachate in these two layers (layers 6 and 7) was not high enough to precipitate gypsum. As previously explained, the leachates from layers 6 and 7 were likely isolated from the other leachates, and mixing seemed to occur mainly between the leachates of these two layers and the overlying layers, such as layers 8 and 9. When the infiltrated water originated from rainfall simulators located at the slope,
the short flow path and the short contact time did not allow the water to dissolve much Ca$^{2+}$ from the limestone in layers 7 and 9.

As previously described, most of the leachate produced in panel 3 was neutral. Although the leachate that was produced from layer 6 tended to be acidic, the total drainage from panel 3 still maintained a neutral pH, because the amount of leachate from layer 6 was only around 7% to 11% of the total outflow. As discussed in Chapter 4, the limestone layering method that was constructed in the full-scale trial dump was not able to prevent acid drainage. The thickness of the layers in the full-scale trial dump (i.e. 3 m acid rock and 1 m limestone) were much thicker as compared to the thickness of the layers at the meso-scale panel (i.e. 0.25 m) and this contributed to the difference in performance between the full-scale trial dump and the meso-scale panel.

The existence of the limestone cover did not improve the effluent quality in panel 2 until the end of the experiment. At that time, the panel had been operating for approximately one year. When the experiment was set up, it was considered likely the shutdown of acid generation probably would not be identified in the first few years of the experiment. Referring to results that were reported by Miller et al. (2003b) for a 2-meter limestone cover that was placed on the 500 tonne test pad, it was found that it required approximately sixteen (16) months for the pH to increase from 2 to 4. The pH became neutral after thirty (30) months.

The processes involved in passivation in limestone covers are different from limestone blending. In the blended limestone and acid rock, it is critical that contact
exists between the calcite minerals from the limestone, and the pyrite minerals from the acid rock. The results that are presented in Chapter 4 show that the close contact between the acid waste rock and limestone in panel 7 could minimize acid generation, while the blended panel built with the truck dumping method failed to prevent acid generation due to limited contact between the limestone and the acid waste rock. The close contact between the limestone and acid waste rock will produce the circumneutral pH pore water that will remove the ferric ion from solution and then will slow the oxidation rate and allow for the formation of the armouring layer (Miller et al. 2009). On the other hand, the limestone cover works by providing alkalinity and dissolved calcium into the infiltrating water that progressively neutralizes the pore water as it slowly progresses downward. Therefore, the limestone cover will work effectively when the acid release rate from the acid waste rock is already in a low steady rate. The contact between the limestone and the acid waste rock in the waste rock pile underneath the limestone cover can accelerate the minimization of acid generation due to the slow oxidation rate as previously explained.

Miller (2007) suggested that the alkalinity supply produced by the limestone cover potentially promoted passivation of the pyrite and neutralization when the acid release rate from the acid rock material was in the range of 1 to 2 mg H₂SO₄/ kg-day. The acidity release rate of the acid rock in panel 2 varied from 11 to 42 mg H₂SO₄/ kg-day, which was much higher than the reference point. In the case of the acid rock used in the panel experiment, the low acidity rate of 1 to 2 g H₂SO₄/ton-day is expected to
occur when the acid rock has been leached for eight (8) to ten (10) years at a high rainfall rate such as 10 mm/day (Miller, 2007).

Due to the time constraints in operating the present experiment, the influence of the limestone cover on the drainage of the pile could not be assessed according to the quality of leachate. However, since the effectiveness of the limestone cover relied on the distribution of alkaline solution into the pile, it could be assessed according to this key information. When the water could be distributed evenly into the entire pile, such as with a high rainfall rate, the alkaline solution from the limestone cover was able to contact most of the acid rock in the pile, as shown by the flow distribution presented in Section 5.2 in Chapter 5 in which a lower coefficient of variation in the leachate distribution was obtained at a higher rainfall rate. Hence, the limestone cover could potentially passivate the sulfide mineral and neutralize the generated acid when the rainfall rate is high.

The influence of the limestone cover on the leachate quality could also be detected in the release rate of contaminants from panel 2. The ranges of concentrations of constituents (i.e. acidity and sulfate) in the leachate from individual layers were similar between panels 1 and 2, and the total release rate of contaminants from panel 2 was also similar with the release rate from panel 1. The alkalinity that was produced from the limestone cover appeared to have a limited impact in neutralizing a significant fraction of the generated acid due to a high acid release from the acid waste rock.
6.4 Summary and Conclusions

The influence of the inclined coarse and fine layers that form the internal structure of most waste rock piles on the quality of drainage was investigated in a meso-scale laboratory experiment. In addition, this study also evaluated the behavior of the inclined coarse and fine layers, as well as the performance of inter-bedded acid rock and limestone and finally a limestone cover. These two techniques are applied to minimize acid generation from waste rock piles. The experiment used a panel that contained acid rock layers only, a panel with acid rock layers and a limestone cover, and a panel that contained alternated acid rock and limestone layers. Different rates of rainfall were applied at the surface of the experimental panels in order to simulate various climate types.

Analyses of the leachate chemistry suggested that flow path and contact time between the infiltrating water and the rock mainly controlled the effluent quality from the individual layers. The concentrations were highest in the leachates that had the longest flow paths, such as layers that were located near the basal slope (i.e. layer 1 and 2), as well as the layers through which flow was mainly governed by the capillary flow mechanism.

As the intensity of rainfall increased, the concentration of constituents in the leachate from the layer generally decreased. At the higher rainfall rates, when the water tended to move downward by gravitational flow along a relatively short flow path, the concentrations of contaminants reporting to the base were found to be
reduced. Furthermore, the larger volume of water for the higher rainfall rates reduced
the concentrations due to dilution. In addition, the range of concentrations also
changed when the rainfall rates were increased, particularly for panel 1. However, the
statistical analyses indicated that the difference in the range of concentration at the
different rainfall rates was not significant.

On the other hand, the influence of the rainfall rate on the loading rate was
significant. The loading rates of constituents such as sulfate and acidity had a positive
correlation with the rainfall rate due to the influence from the increasing leachate
volume. The contribution of the leachate from preferential flow paths to the total
loading of contaminants such as acidity and sulfate was quite dominant and was shown
to produce as much as 26% to 51% of the total loading.

The calculated oxidation rate of waste rock based on sulfate loads, indicated
that the oxidation rate of waste rock from the meso-scale panels was approximately 2.6
X 10⁻¹⁰ kg/kg per second, which was slightly lower (i.e. approximately 45%) than the
oxidation rate of waste rock from the leach columns (i.e. 2 kg and 35 kg) and 500 tonnes
test pad, but the oxidation rate in the meso-scale panel was higher (i.e. approximately
70%) than the oxidation rate in the full-scale waste rock dump trial. The high oxidation
rate recorded during the first year of the meso-scale panel experiment indicated that
the acid waste rock in the panel was fully oxygenated.

The inter-bedded limestone and acid layers in the meso-scale panel worked
effectively in preventing acid drainage. This performance was different from the results
shown by limestone layers at the full-scale field trial. The larger thickness of the layer in
the full-scale field trial seemed to reduce the effectiveness of the limestone layer in the
geochemical mixing of the leachate. The hydraulic properties of the adjacent coarse
limestone and acid waste rock layers may restrict the mixing between the alkaline and
acid leachate, therefore the leachate that was produced from the acid waste rock
coarse layer could still be acidic as demonstrated by results from the panel that
contained acid waste rock and limestone layers.

The effectiveness of the limestone layering and the limestone cover depended
on the distribution of water within the pile. In the layered limestone and acid rock
panel, high rainfall rates appeared to best promote mixing and chemical interactions
between the acid leachate and the alkaline leachate from the different layers. This was
evidenced in a reduction of the coefficient of variation between the release rates from
the individual layers (i.e. from 147% at 2 mm/day of rainfall rate and 94% at 10 mm/day
rainfall rate). Although the performance of the limestone cover could not be completely
assessed based on the quality of effluent from the layers, it is possible to make an
assessment on the basis of the distribution of limestone leachate into the panel since
the contact between the limestone leachate and the acid rock was the key parameter in
this treatment. The highest rainfall rate of 10 mm/day is considered to be the most
favorable for the limestone cover treatment because the alkaline solution is most
evenly distributed in the layers.
The profiles of the coarse and fine layers in the large-scale waste rock piles are potentially different from the profiles of the layers that were used in the meso-scale panel experiment. However, the results of this laboratory experiment provided a fundamental understanding of the water flow and geochemical mixing for inclined coarse and fine layers within the internal structure of a waste rock pile. In summary, the present study showed that limestone layering and limestone covers function most effectively in high rainfall environments. When the infiltration of water into the waste rock piles with limestone layers or limestone covers was limited, the effectiveness of these two approaches with respect to acid mitigation was decreased.
References


Proceedings of the Sixth International Conference On Acid Rock Drainage, Cairns, Queensland, 111-121.


Chapter 7 Summary and Conclusions

7.1 Introduction

This thesis presents results from the research comprising a field study at a full-scale trial dump and a meso-scale laboratory experiment. The full-scale trial dump was constructed to investigate the kinetics of the chemical reactions within a waste rock pile and also to improve the current understanding of the hydrological, physical and geochemical characteristics for waste rock piles. Various treatments to minimize acid generation were also applied and assessed in the trial dump including the blending and layering of limestone with the acid rock, the placement of a low-permeability cover, and the placement of a limestone cover. The meso-scale laboratory study was set up to investigate the influence of coarse and fine layers as the internal structure within waste rock piles with respect to flow and geochemical behavior. The laboratory experiment consisted of three meso-scale panels. The first panel contained coarse and fine layers of acid waste rock, and the second panel consisted of the coarse and fine layers of acid rock, covered by a horizontal limestone bed on the top of the layers. The third panel contained inter-bedded acid rock and limestone. The volume and chemistry of leachate, together with the temperature and oxygen data collected from the field trial and laboratory experiment, provided the required information to achieve the objectives of this thesis.

The construction of the full-scale waste rock dump was a scale-up from the smaller scale experiments (i.e. 2 kg and 35 kg leach columns and 500 tonne test pads)
that used a similar type of acid waste rock. This scale up program provided the unique opportunity to understand the influence of the scale experiment with respect to the oxidation rate of waste rock at the high rainfall environment. In addition, the application of blending limestone with the acid rock in the full-scale field experiment provided information regarding the method of blending that was practical and also effective in minimizing acid generation at the large-scale operation.

The application of a low-permeability cover for minimizing acid generation from a waste rock pile or tailings impoundment has been widely tested (e.g. Marcoline, 2008; Adu-Wusu and Yanful, 2006). However, most of these tests were conducted at locations where the precipitation was much lower than the rainfall intensity at the Grasberg Mine (i.e. 3,000 mm), the location where the full-scale field experiment was constructed. Furthermore, a natural material such as clay soil (as used in a previous experiment) for the cover was not available at the Grasberg Mine. The purpose of placement of the low-permeability cover on the trial dump was to assess if the materials at Grasberg Mine are able to reduce the water infiltration into the waste rock dump when it is exposed to the high rainfall environment.

The previous laboratory investigations conducted on coarse and fine layers within the internal structure of the waste rock dump were conducted for the vertical layers (Newman, 1999; Wu et al. 2007), while the studies on inclined layers were limited to numerical simulations (Fines, 2006; Fala et al. 2005; Wilson et al. 2000). Most of these works focused only on the water flow issue. The meso-scale laboratory
experiment that was set up for the present research provided an opportunity to improve the understanding regarding the influence of inclined coarse and fine layers not only on the flow mechanism but also on the quality of leachate generated from the waste rock pile. The coarse layer contained waste rock sizes ranging from 10 mm to 25 mm, while the fine layer contained the size of waste rock that was less than 10 mm, although the majority of the particle sizes were less than 4.75 mm. The set up of meso-scale panels also included a horizontal limestone cover and inter-bedded acid rock layers with limestone layers which provided the ability to assess the geochemical mixing between acid leachates from the acid waste rock layers with alkalinity leachates from the limestone layers, thus determining the final quality of the drainage from the panel.

This chapter summarizes the results from the full-scale trial dump and the meso-scale laboratory experiments regarding the behavior of flow and geochemical processes in the waste rock pile. The performance of limestone blending, limestone layering, a limestone cover and a low-permeability cover are also discussed in this chapter. Finally, discussions regarding the implication of results from the research to the management of the waste rock systems are presented.

### 7.2 Flow Behavior in Waste Rock Piles

As presented in Chapter 2, four-years of monitoring outflow data from the trial dump demonstrated that the leachate outflow measured from twenty-four (24) basal lysimeters varied temporally and spatially. The cumulative net infiltration among lysimeters ranged from 20% to 163% of total precipitation. The cumulative net
infiltration defines as the total water collected from the lysimeter during the whole experiment period which compared to the total rainfall during the whole experiment period which represented the footprint of the lysimeter area. The variability in the outflow volume was mainly due to the heterogeneity of the physical characteristics of the waste rock pile, such as the surface dump condition and the internal structure of the pile.

The estimated water balance for the full-scale trial dump indicated about 82% of rainfall was collected as net infiltration. This value is higher than the cumulative net infiltration for waste rock dumps in low rainfall, semi-arid/arid climates as reported by Williams and Rohde (2008) where the cumulative net infiltration varied from 50% to 60%. This estimated net infiltration value from the trial dump was lower compared to the net infiltration from the 500 tonnes test pads at the same location as the trial dump, which had approximately 90% of the rainfall as net infiltration. The difference in the construction method used for the trial dump and the test pad lead to different physical characteristics between these two trials, contributing to the difference in the net infiltration. The trial dump was constructed with end-dumping by truck placement, while the test pad was built with random placement using an excavator. Therefore, the waste rock segregation was more pronounced for the trial dump which led to a more complex internal structure in the trial dump compared to the test pad. In addition, the trial dump had been exposed to traffic during construction making the surface of the
trial dump less permeable as compared to the test pad. As a result, the net infiltration was lower in the trial dump.

The monitoring of outflow volume from the trial dump demonstrated that the dump geometry affected the infiltration of rainwater. In general, the net infiltration was higher for the lysimeter located beneath the front slope of the trial dump compared to the lysimeters located at the middle and rear section of the panels. The daily outflow of the lysimeter located under the front slope varied from 6.6 mm/day to 16.3 mm/day which was higher compared to the daily outflow produced from lysimeters located under the flat surface that ranged from 1.1 mm/day to 12.3 mm/day. The front slope, which had not been exposed to traffic and the cover treatment, tended to be more permeable compared to the flat surface.

Preferential flow, which refers to rapid flow that may follow a macropore pathway, was observed in fast-flowing lysimeters in the trial dump, particularly during rainfall events. Although only approximately 30% of seventeen (17) lysimeters were identified as fast flowing lysimeters, the contribution of these fast-flowing lysimeters reached 58% of the total leachate outflow collected from the trial dump.

A cumulative net infiltration of higher than 100% was recorded at several lysimeters in the trial dump (i.e. L1 lysimeter for panels 5, 6 and 7, and L2 lysimeter in panel 2). This data indicated that the infiltration did not only occur as a vertical flow, but also occurred through sub-lateral flow. This sub-lateral flow was induced by the
heterogeneity of the waste rock material pile, such as the presence of coarse and fine layers that could lead to anisotropy within the pile.

The results of the meso-scale laboratory experiment, which are presented in Chapter 5, indicated that the presence of layer-parallel flow in the coarse and fine layers was determined by the rainfall rate and the effective hydraulic conductivity of the fine layer. Layer-parallel flow through the fine layers tended to be significant when the low rainfall intensity (i.e. 2 mm/day) was applied; approximately 48% and 71% of the total outflow was collected from the combined flow for the fine layers in panel 1 and panel 3, respectively. As the rainfall rate increased (i.e. 10 mm/day), layer-parallel flow through fine layers decreased, and more water was collected from the coarse layers through gravitational flow. As a result, the combined water flowed through the fine layers decreased to 30% for panel 1, and 41% for panel 3 when the rainfall rate increased to 10 mm/day.

The presence of water flow through coarse layers as gravitational flow occurred in part due to breakthrough flow from the fine layer as shown from the results of the breakthrough test. It is likely the relatively small difference in hydraulic properties between the coarse and fine materials used in the meso-scale experiment did not create a strong enough capillary barrier along the interface between the fine and coarse material, and thus produced significant breakthrough from the fine layers, particularly at higher rainfall rates. The equation introduced by Ross (1990) and later modified by Steenhuis et al. (1991) regarding the diversion length, or the maximum length that a
fine layer could retain water without break through, was used to calculate the diversion length in the panel for different rainfall rates. The calculation of diversion length in the fine layer demonstrated that the breakthrough point changed from 2,500 mm to 500 mm when the rainfall rate increased from 2 mm/day to 10 mm/day. This calculation result indicated that the breakthrough occurred at the higher elevation when the rainfall rate increased, therefore water tended to be collected at the bottom layers such as layers 1 (coarse) and 2 (fine), and also at the basal slope during high rainfall.

7.3 Geochemical Processes in the Waste Rock Pile

The characteristics of the waste rock kinetics in the large-scale experiment were found to be consistent with the performance observed in the smaller-scale experiment. The waste rock kinetics were characterized by fast initial kinetics, attaining some maximum value within one to two years, followed by a rapid decrease with a longer term decay and a slower downward trend. The concentration of key constituents such as acidity, sulfate and dissolved metal generally tended to be higher in the trial dump than in the small-scale experiment. In contrast, the release rate of these key constituents was higher in the smaller experiment than the trial dump.

The oxidation rates of the waste rock from the leach columns, 500 tonne test pads, and meso-scale panels were comparable. This result was different compared to other studies that indicated the oxidation rate measured in meso-scale experiments such as large columns and piles were lower compared to the oxidation rate of waste rock from smaller scale experiment such as the humidity cell (Frostad et al. 2005;
Stromberg and Banwart, 1999). The continuous high rainfall environment that was exposed to the test pad was considered to be comparable with the irrigation rates applied to the leach column and meso-scale panels which continuously flushed the oxidation products resulting in a similar oxidation rate between these two different scales of experiments. When the waste rock was exposed for three to four years, the oxidation rate from the trial dump was lower, up to one order of magnitude, than the oxidation rate from the smaller scale experiment (as shown in Table 7.1).

**Table 7.1 Oxidation rate of waste rock from the different scale experiments**

<table>
<thead>
<tr>
<th>Waste rock type</th>
<th>Oxidation rate, O\textsubscript{2} kg/kg material-second</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Leach column</td>
</tr>
<tr>
<td>Blue waste</td>
<td>1.3 X 10\textsuperscript{-10}</td>
</tr>
<tr>
<td>Red waste</td>
<td>4.7 X 10\textsuperscript{-10}</td>
</tr>
</tbody>
</table>

The lower oxidation observed in the trial dump is considered to be due to the scale up factor which is determined by the difference in dump hydrology and particle size distribution between the full-scale trial dump and the small-scale/meso-scale experiments. As previously mentioned, the trial dump had a lower net infiltration and a larger particle size of waste rock compared to the smaller scale experiment, therefore lower oxidation rates and lower release rates of oxidation products were obtained from the trial dump.
As a result of sulfide mineral oxidation, temperatures as high as 56° C were recorded at the base of the trial dump, while higher temperatures, approximately 72° C, were measured in the face of the dump. The high temperature readings continued to be recorded when the waste rock had been placed for four years, with temperatures of 40° C to 50° C still being measured in thermistors at various locations.

High oxygen concentration, as high as atmospheric value (i.e. 20%) were measured at most of the locations at the trial dump. Low oxygen concentrations that were near the anoxic condition were also detected at some dump locations, but they were measured temporarily. However, the implication of the temporary low oxygen concentration and the sulfide mineral oxidation was relatively negligible since the high temperature continued to be recorded at most locations and the high release rate of key constituents were still being measured in the leachates from the trial dump.

The high oxygen concentration measured along the base and the face of the dump and also near the surface of the dump demonstrated oxygen was transported into the pile through both advection and diffusion mechanisms. The oxygen was detected in the probes at the base of the panel in the trial dump located 65 m from the toe dump, as well as at the probes located at the face of the panel. These measurements indicated the advection from the front slope was not limited, at least until 65 m from the toe of the dump.

As presented in Chapter 6, the internal coarse and fine layers in the waste rock pile affected the quality of the drainage because this internal structure influenced the
flow path and residence time of the infiltrated water. When the flow path and the residence times were longer, the leachate collected from the layers had a higher concentration. Therefore, the leachates from the inner or bottom layers (i.e. layers 1 and 2) tended to have higher concentrations compared to the leachates from the outer layers (i.e. layers 5 and 6).

Since the rainfall rate determined the flow mechanisms and flow paths in the pile, the concentrations and the loading rates of the key constituents in the leachates were also influenced by the rainfall rates. Short pathways together with the increasing amounts of rainwater resulted in the leachate produced from the higher rainfall event having a lower concentration. Although the concentration was decreased when the rainfall rate increased and the range of concentration was reduced, the influence of the rainfall rates on the change in concentration of key constituents was not significant. On the other hand, the change in loading rates due to the increase of rainfall rates was significant. In general, the loading rates increased when the rainfall rate was increased. The coefficient of variation for loading rates among the layers decreased as the rainfall increased, particularly for panels 1 and 3.

The loading rate from the panel was mainly due to leachates that were generated through the preferential flow that may follow macropore pathways. The loadings from these preferential pathways could contribute up to 51% of the total loadings for key constituents.
7.4 Blending of Limestone with Acid Rock and Limestone Cover to Minimize Acid Generation

The effectiveness of the blending of limestone and acid waste rock was influenced by the blending ratio, which is represented as Acid Neutralization Capacity (ANC) to Maximum potential acid (MPA) ratio, and also by the intimacy of blending. As discussed in Chapter 4, the results of the trial dump proved that the ANC/MPA ratio that successfully prevented acid generation in the small-scale experiment did not always perform well when it was applied to the large-scale waste rock pile because the blending did not occur across the whole ranges of particle size.

The piles that were constructed by truck dumping including pre-mixing, alternated dumping, and layering as blending methods failed to prevent acid generation. The segregation that occurred during the dumping and the differences in particle size between the limestone and the acid rock tended to isolate the acid rock from the limestone. The wide range of the ANC/MPA ratio (i.e. 0.02 to 1,537) from individual samples collected from the panels constructed with alternated truck dumping and layering methods suggested the blending was not achieved across these three panels. On the other hand, samples from the panel that was constructed by a crusher and a stacker system had the narrow range of ANC/MPA ratio for individual samples (i.e. 0.8 to 6). Pre-treatment that involved crushing of the acid waste rock and the limestone prior to dumping produced acid rock and limestone that had a relatively close range of particle size. Hence, the segregation between the acid rock and limestone that
occurred during the dumping was less pronounced in the crusher and stacker system which promoted blending. Although the average bulk ANC/MPA ratio was only approximately 2.1, the crusher and stacker system was effective in minimizing acid drainage, at least during the testing period.

One of the meso-scale panels from the laboratory experiment was set up with alternating acid waste rock and limestone layers in the form of coarse and fine particles. The quality of leachate measured from the limestone layering panel indicated that acid drainage could be prevented during a one year duration. This performance was different from the performance shown by the limestone layering system at the full-scale trial dump. The relatively thinner layer thickness in the panel (i.e., 0.25 m thick acid rock or limestone layer) compared to the layers in the full-scale trial dump (i.e., 3 m thick acid rock layer and 1 m thick limestone layer) improved the performance of the limestone layering regarding minimizing acid generation.

As previously mentioned, the rainfall rate influenced the flow path for infiltrating water into the coarse and fine layered system. Consequently, the quality of leachate from a waste rock dump with coarse and fine layers was also affected by rainfall rates. The infiltration test conducted on the acid waste rock and limestone layers showed that mixing and chemical interactions occurred between the acid leachate and the alkaline leachate within the layers. As a result, relatively similar sulfate concentrations were measured in the leachate from all layers, except for the leachates from the adjacent coarse limestone and coarse acid rock layer. The results from the
meso-scale panel showed that acid drainage will still be generated from the coarse layer of acid rock when coarse limestone is placed next to coarse acid rock. The acid generation rate from the acid rock layer exceeded the alkaline load from the limestone layer and produced acid drainage. However, overall the total drainage from the panel with inter-bedded limestone and acid rock layers remained neutral.

The rainfall rate affected geochemical mixing between the leachates from the acid rock layers and limestone layers. The results from the meso-scale panel (i.e. panel 3) indicated that the coefficient of variations in the sulfate loadings was smaller when the rainfall rate was higher (i.e. 147% at 2 mm/day rainfall rate and 94% at 10 mm/day rainfall rate). These values indicated that the discrepancies in the release rates among individual layers were less at the higher rainfall rate. Therefore, the highest rainfall rate environment (i.e. 10 mm/day) is considered to be most favorable for promoting mixing and chemical interactions between the acid leachate and the alkaline leachate in the inter-bedded acid rock and limestone layers.

Similar to the acid rock and limestone layers, the performance of a limestone cover also relied on the flow path of the infiltrated water that carried the alkaline solution from the limestone cover. Although the performance of the limestone cover could not be completely assessed based on the quality of effluent from the layers, the assessment was based on the distribution of limestone leachate within the panel since the contact between the limestone leachate and the acid rock was the key parameter for the success of this treatment. The highest rainfall rate environment of 10 mm/day,
which had a lower coefficient of variation for the leachate distribution in layers compared to the lower rainfall rate (2 mm/day and 5 mm/day), was considered to be the most favorable for the limestone cover. The alkaline solution was evenly distributed in the layers during the 10 mm/day rainfall rate.

7.5 Low-permeability Cover Placement to Minimize Acid Generation

Evaluation of the volume of leachates collected from the basal lysimeters in the trial dump suggested that the application of low-permeability covers was not effective in reducing infiltration into the waste rock. The mixture of road mud and limestone cover and the weathered waste rock cover reduced the cumulative net infiltration by 11% and 14%, respectively. Difficulties in compacting the material in order to meet the required specification during the wet weather likely affected cover performance. Analyses for the chemistry of leachate also indicated that the low-permeability cover had little impact on the oxidation of sulfide minerals. Furthermore, the oxidation rate of waste rock from the covered panels (i.e. $4.0 \times 10^{-11}$ to $8.7 \times 10^{-11}$ kg/kg-second) was comparable with the oxidation rate of waste rock from the uncovered panels (i.e. $2.2 \times 10^{-11}$ to $5.1 \times 10^{-11}$ kg/kg-second). The excavation conducted beneath the HDPE cover revealed abundant oxidation products present in the rock underneath the cover. Oxidation continued to occur because the oxygen was transferred continuously through advection from the toe of the uncovered face of the panel, which was clearly shown by high oxygen concentration measurements.
7.6 Implications on Management of Waste Rock Dump

The results obtained from the full-scale trial dump and the meso-scale panels regarding flow behavior and geochemical processes have implications regarding the design, construction, and management of a waste rock pile, as follows:

- The calculation of oxidation rates based on sulfate release rates showed that the oxidation rate of waste rock at the full-scale trial dump was lower by up to one order of magnitude compared to the smaller scale experiment. Hydrological factors may slow the release of the oxidation products at the full-scale waste rock dump. Therefore, oxidation rates that are lower than the oxidation rates measured in the trial dump can be expected in larger scale waste rock piles where heights and areas are many times larger than the size of the trial dump, particularly for the waste rock pile at the high rainfall environment. The lower oxidation rate leads to the implication that the acid generation period will be much longer in full scale systems as compared with those measured in the small scale experiments and the trial dump.

- The results from the trial dump indicated that the geometry of the dump affected the infiltration process. For example, the front slope of the dump tended to have the highest net infiltration. Therefore, in order to reduce acid generation, it is recommended not to place reactive materials on the front slope of the dumps.
• The results of the trial dump demonstrated the failure of limestone blending done by means of the truck dumping. Failure occurred because the blending did not achieve thorough mixing across all size fractions of the waste rock, which was indicated by the wide range of ANC/MPA ratio from the individual samples. Pre-treatment such as crushing of the acid waste rock and the limestone prior to dumping can improve the intimacy of blending, as was shown in the results from the panel in the trial dump that was constructed with a crusher and stacker system.

• The effectiveness of limestone blending and the limestone cover depended on the distribution of water within the pile. The application of these two treatments is favorable in high rainfall environments where the water tended to flow downward through the pile. Combining limestone blending and the installation of a limestone cover along with other efforts that are intended to prevent infiltration into the waste rock pile, such as an impervious cover system, will not be effective with respect to reducing acid generation. When the infiltration is limited, the alkaline solution from the limestone does not efficiently come into contact with the acid rock.

• Since vertical flow appears to be the dominant flow mechanism for the waste rock pile located in high rainfall environments, the limestone cover needs to be placed along the slope of the dump to ensure the entire profile of the waste rock pile receives alkaline leachate from the limestone cover.
• The results from the trial dump and also from the previous experiments such as the leach columns and the 500 tonne test pads indicated that the waste rocks tested in these experiments were reactive materials. The acid and dissolved metal was leached starting at the very beginning of construction of the full-scale waste rock dump. Therefore, it will be most beneficial if the mitigation efforts are performed as early as possible.

7.7 Recommendations for Future Research

Future research is recommended for several topics associated with the investigations presented in this thesis. These are summarized in point form as follows:

• **Tracer tests on the field trial and/or meso-scale experiments.**
  
The results for the current studies from the meso-scale and the field trial do not describe the actual pathway of the water collected from the layers/lysimeters. A tracer test conducted at the macro-scale/field test would provide further information regarding the flowpath and improve the understanding for the behavior of water flow flow in layered mine rock piles.

• **Post-mortem study of the meso-scale experiment.**
  
A post-mortem study on the meso-scale experiment should be completed along with mineralogy tests in order to characterize intrinsic oxidation rates and evolution of the waste rock.
• **Water transport and geochemical numerical modeling.**
  
The development of unsaturated water flow and geochemical numerical modeling to further understand the results obtained from the field trial and the meso-scale experiment.

• **Geotechnical studies to characterize mechanical properties of the waste rock.**
  
The studies complete in this thesis only cover the geochemical aspects of the waste rock. Since both the geotechnical and the geochemical aspects must be considered in the design of the waste rock dump, testing associated with geotechnical properties also needs to be carried out.
References


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APPENDIX E: Leachate Chemistry Data from L1 Lysimeter in Panel 3 of the Trial Dump for PHREEQC Input
<p>| Date      | pH Field | EC Field | Tot. Acidity as CaCO3 | Al, ppm | Ca, ppm | Cd, ppm | Cu, ppm | Fe, ppm | K, ppm | Mg, ppm | Mn, ppm | Na, ppm | Ni, ppm | Zn, ppm | Cl, ppm | SO4, ppm | Br, ppm | Carbonate, ppm |
|-----------|----------|----------|-----------------------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|---------|----------------|
| 04-Apr-01 | 3.73     | 1.475    | 300                   | 22.6    | 132     | 0.054   | 92.2    | 0.569   | 22.8    | 33.7    | 2.42    | 3.44    | 0.174   | 13.0    | &lt;0.1   | 1,070  | &lt;0.1   | &lt;2     |
| 20-Apr-01 | 3.2      | 1.533    | 440                   | 32.2    | 168     | 0.109   | 116     | 3.98    | 21.5    | 37.4    | 3.42    | 7.54    | 0.206   | 13.6    | &lt;0.1   | 1,052  | &lt;0.1   | &lt;2     |
| 02-May-01 | 3.18     | 2.378    | 780                   | 56.3    | 256     | 0.158   | 172     | 10.2    | 26.9    | 55.9    | 5.35    | 13.3    | 0.295   | 22.2    | &lt;0.1   | 1,429  | &lt;0.1   | &lt;2     |
| 17-May-01 | 3.19     | 3.150    | 1,100                 | 82.3    | 334     | 0.224   | 199     | 30.5    | 31.8    | 89.9    | 8.37    | 23.9    | 0.411   | 37.9    | &lt;0.1   | 2,054  | &lt;0.1   | &lt;2     |
| 30-May-01 | 2.69     | 3.790    | 1,600                 | 145     | 385     | 0.255   | 265     | 43.7    | 41.3    | 87.5    | 10.2    | 18.3    | 0.587   | 51.2    | &lt;0.1   | 2,935  | &lt;0.1   | &lt;2     |
| 15-Jun-01 | 3.17     | 4.120    | 1,400                 | 108     | 570     | 0.589   | 276     | 47.4    | 42.3    | 109     | 13.8    | 29.5    | 0.631   | 72.3    | 7.5    | 2,857  | &lt;0.1   | &lt;2     |
| 04-Jul-01 | 3.1      | 5.340    | 3,300                 | 250     | 669     | 0.838   | 470     | 148     | 41.3    | 163     | 20.9    | 32.6    | 1.07    | 116     | 6.5    | 4,427  | &lt;0.1   | &lt;2     |
| 18-Jul-01 | 2.96     | 6.660    | 3,800                 | 312     | 615     | 0.934   | 548     | 208     | 35.0    | 200     | 24.4    | 36.8    | 1.22    | 163     | &lt;0.1   | 6,510  | &lt;0.1   | &lt;2     |
| 01-Aug-01 | 2.82     | 6.680    | 5,100                 | 362     | 570     | 1.12    | 622     | 227     | 43.6    | 197     | 28.9    | 38.2    | 1.51    | 178     | 1      | 5,827  | &lt;0.1   | &lt;2     |
| 13-Aug-01 | 2.83     | 6.300    | 6,600                 | 395     | 549     | 1.59    | 815     | 154     | 46.4    | 193     | 35.9    | 39.7    | 1.57    | 228     | 2      | 6,480  | &lt;0.1   | &lt;2     |
| 31-Aug-01 | 2.6      | 8,530    | 8,700                 | 582     | 516     | 1.75    | 786     | 684     | 29.6    | 253     | 43.3    | 49.1    | 2.10    | 249     | &lt;0.1   | 9,413  | &lt;0.1   | &lt;2     |
| 12-Sep-01 | 2.09     | 8,430    | 10,000                | 789     | 541     | 2.01    | 1,010   | 852     | 32.2    | 307     | 51.7    | 44.8    | 2.38    | 327     | 2.7    | 9,943  | &lt;0.1   | &lt;2     |
| 25-Sep-01 | 2.35     | 14,110   | 10,000                | 790     | 492     | 1.59    | 869     | 595     | 33.3    | 344     | 65.9    | 51.8    | 1.92    | 282     | 9.6    | 10,265 | &lt;0.1   | &lt;2     |
| 09-Oct-01 | 2.21     | 10,870   | 14,000                | 1,050   | 481     | 2.54    | 1,390   | 1,640   | 36.0    | 355     | 68.4    | 58.8    | 3.00    | 383     | &lt;0.1   | 12,968 | &lt;0.1   | &lt;2     |
| 25-Oct-01 | 2.36     | 12,560   | 22,000                | 1,750   | 505     | 2.41    | 1,230   | 2,160   | 33.3    | 655     | 88.3    | 3.33    | 58.6    | 433     | &lt;0.1   | 19,774 | &lt;0.1   | &lt;2     |
| 07-Nov-01 | 2.61     | 17,370   | 34,000                | 2,530   | 500     | 2.83    | 2,580   | 3,260   | 18.7    | 1,450   | 102     | 40.8    | 5.20    | 494     | &lt;0.1   | 32,176 | &lt;0.1   | &lt;2     |
| 20-Nov-01 | NA       | 16,450   | 37,000                | 2,460   | 501     | 3.04    | 2,880   | 3,160   | 23.2    | 1,290   | 136     | 76.6    | 5.59    | 680     | 24     | 37,900 | &lt;0.1   | &lt;2     |
| 04-Dec-01 | 2.42     | 26,900   | 45,000                | 3,680   | 533     | 3.40    | 4,070   | 2,750   | 20.1    | 2,180   | 168     | 53.5    | 7.66    | 794     | 3.8    | 46,169 | &lt;0.1   | &lt;2     |
| Date       | pH Field | EC Field | Tot. Acidity as CaCO3 | Al, ppm | Ca, ppm | Cd, ppm | Cu, ppm | Fe, ppm | K, ppm | Mg, ppm | Mn, ppm | Na, ppm | Ni, ppm | Zn, ppm | Cl, ppm | SO4, ppm | Br, ppm | Carbonate, ppm |
|------------|----------|----------|-----------------------|---------|---------|---------|---------|---------|--------|---------|---------|---------|---------|---------|---------|---------|---------|---------|----------------|
| 18-Dec-01  | 2.23     | 28,700   | 47,000                | 3,370   | 556     | 3.74    | 4,470   | 4,140   | 8.39   | 2,600   | 207     | 54.5    | 7.58    | 867     | 40      | 45,326 | &lt;0.1   | &lt;2     |
| 08-Jan-02  | 2.75     | 21,650   | 46,000                | 3,370   | 518     | 4.22    | 5,370   | 2,090   | 7.00   | 2,920   | 230     | 33.9    | 7.39    | 708     | 144     | 45,483 | &lt;0.1   | &lt;2     |
| 21-Jan-02  | 2.07     | 31,700   | 48,000                | 3,920   | 494     | 3.58    | 6,540   | 2,220   | 6.01   | 4,030   | 287     | 36.4    | 9.08    | 892     | &lt;0.1   | 49,324 | &lt;0.1   | &lt;2     |
| 06-Feb-02  | 2.29     | 20,950   | 57,000                | 3,940   | 520     | 3.30    | 6,350   | 2,810   | 4.38   | 4,460   | 314     | 31.7    | 8.29    | 877     | &lt;0.1   | 51,017 | &lt;0.1   | &lt;2     |
| 21-Feb-02  | 2.11     | 37,500   | 57,000                | 4,510   | 431     | 2.68    | 6,130   | 2,550   | 3.68   | 5,260   | 273     | 27.2    | 8.31    | 754     | 79      | 63,750 | &lt;0.1   | &lt;2     |
| 08-Mar-02  | 2.1      | 14,120   | 59,000                | 5,640   | 400     | 2.88    | 5,950   | 2,590   | 3.33   | 5,150   | 294     | 29.5    | 9.46    | 696     | 9.9     | 65,043 | &lt;0.1   | &lt;2     |
| 21-Mar-02  | 2.09     | 32,400   | 56,000                | 5,580   | 365     | 2.79    | 5,950   | 2,520   | 3.02   | 5,360   | 292     | 26.8    | 9.36    | 639     | 66      | 64,780 | &lt;0.1   | &lt;2     |
| 04-Apr-02  | 1.78     | 34,100   | 54,000                | 6,060   | 387     | 2.81    | 5,890   | 2,460   | 0.79   | 4,680   | 267     | 14.0    | 9.33    | 654     | 77      | 65,698 | &lt;0.1   | &lt;2     |
| 16-Apr-02  | 1.91     | 34,900   | 55,000                | 6,060   | 409     | 2.56    | 5,750   | 2,650   | 1.89   | 4,860   | 292     | 13.7    | 8.54    | 701     | 58      | 62,332 | &lt;0.1   | &lt;2     |
| 02-May-02  | 2.17     | 31,900   | 54,000                | 5,090   | 442     | 2.70    | 6,020   | 2,600   | 1.71   | 4,640   | 239     | 28.6    | 9.12    | 635     | &lt;0.1   | 58,077 | &lt;0.1   | &lt;2     |
| 16-May-02  | 2.14     | 32,400   | 57,000                | 5,380   | 517     | 2.24    | 5,820   | 2,290   | 1.35   | 5,200   | 272     | 19.6    | 7.80    | 641     | 76      | 62,710 | &lt;0.1   | &lt;2     |
| 13-Jun-02  | 2.31     | 30,800   | 53,000                | 5,720   | 474     | 2.46    | 5,630   | 2,600   | 1.10   | 4,950   | 294     | 23.9    | 8.48    | 697     | 67      | 63,272 | &lt;0.1   | &lt;2     |
| 11-Jul-02  | 1.9      | 30,900   | 52,000                | 5,320   | 343     | 2.48    | 5,330   | 2,460   | 0.85   | 5,020   | 264     | 18.0    | 8.59    | 593     | 3.6     | 60,887 | &lt;0.1   | &lt;2     |
| 13-Aug-02  | 2        | 30,800   | 53,600                | 5,450   | 524     | 2.39    | 5,040   | 2,700   | 0.52   | 5,160   | 255     | 21.6    | 9.03    | 559     | &lt;0.1   | 57,600 | &lt;0.1   | &lt;2     |
| 11-Sep-02  | 2.05     | 34,400   | 53,500                | 5,280   | 505     | 2.21    | 4,780   | 2,870   | 0.24   | 5,160   | 236     | 19.9    | 8.22    | 525     | 256     | 62,300 | &lt;0.1   | &lt;2     |
| 09-Oct-02  | 1.95     | 29,600   | 51,600                | 5,530   | 598     | 2.25    | 4,700   | 2,940   | 0.49   | 5,330   | 243     | 18.5    | 8.51    | 488     | &lt;0.1   | 62,700 | &lt;0.1   | &lt;2     |
| 13-Nov-02  | 2.09     | 27,100   | 48,700                | 5,760   | 423     | 2.08    | 4,140   | 2,730   | 0.40   | 5,140   | 246     | 17.8    | 7.96    | 461     | 62.2    | 63,600 | &lt;0.1   | &lt;2     |
| 11-Dec-02  | 2.13     | 25,400   | 57,100                | 6,190   | 446     | 1.74    | 2,490   | 3,680   | 0.40   | 5,690   | 193     | 14.0    | 7.34    | 273     | &lt;0.1   | 64,600 | &lt;0.1   | &lt;2     |
| 08-Jan-03  | 2.09     | 22,600   | 53,900                | 7,120   | 534     | 1.64    | 2,310   | 2,410   | 1.07   | 5,900   | 219     | 19.5    | 7.51    | 283     | 51.2    | 67,700 | &lt;0.1   | &lt;2     |</p>
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<th>Date</th>
<th>pH Field</th>
<th>EC Field</th>
<th>Tot. Acidity as CaCO₃</th>
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<th>Ca, ppm</th>
<th>Cd, ppm</th>
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