Hydrogeological characteristics and numerical modeling of groundwater flow and contaminant transport in the Folldal mining site

Effects of acid mine drainage on groundwater

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Abstract
Mining activity has been the main support for the economy in Folldal, Hedmark, during more than two centuries due to the extraction of copper, zinc and sulfur that rebounded in the industrial and social development of the region. The depletion of the ore lead to a new direction in the activity of the village, turning into the tourism associated with the visit of the old mine and its protection to keep the original buildings, galleries and mine tailings.

However, the exposure and subsequent oxidation of the mine wastes produces acid mine drainage, which is discharging into the river Folla with high concentrations of heavy metals that are toxic and trigger serious environmental damage. The absence of life in the Folla river downstream of Folldal center has been manifested by authorities and there are projects ongoing trying to find a solution and restore the aquatic conditions in Folla.

Therefore, this study assesses the impact of acid mine drainage in the groundwater environment and its possible contribution to Folla river. The groundwater flow and quality was investigated integrating field data with the numerical models MODFLOW and PHREEQC in order to simulate the hydrological and geochemical observations and the effectiveness of a limestone drain as a possible remediation solution.

The groundwater level in the study area shows variations, ± 1m, through the hydrological year due to arid winters and humid summers. The hydrogeological numerical model reproduces these changes with a mean error of 1.4m. Heavy metals in groundwater present a concentration trend versus depth. Groundwater in the mine tailings up to 6 m deep is heavily polluted, 20 μg/L Cd, 250 μg/L Co, 1500 μg/L Cu; while at 10 m depth is HCO₃⁻ rich and has very low concentration of heavy metals, 0.02 μg/L Cd, 1.31 μg/L, Co 3 μg/L Cu. The combination of water balance and hydrochemistry indicates that currently the groundwater input of sulfate, copper and zinc into the river Folla proceeding from the Folldal mine area is 2000 t/yr, 9t/yr and 16t/yr, respectively. This represents the 30% of the total yearly contamination transport in Folla. The geochemical simulation revealed that an anoxic limestone drain would remove more than 75% of Cu and Zn from groundwater and the pH would increase from 4 to 6.5. Therefore, two limestone barriers, 100-150m long, are suggested in the main mine tailings areas, in the proximity of Folla river.
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1. Introduction

Folldal mining area, in Hedmark, Norway, was intensely mined for more than two centuries (1747-1968) and is now protected by the Cultural Heritage Law, which does not allow alteration of the historical site. The mining activity led to uncover sulfide bearing minerals in form of waste materials including, overburden, waste rock and tailings. The weathering of these sulfides exposed to air and water produced acid mine drainage (AMD), characterized by high acidity and concentrations of dissolved heavy metals and $\text{SO}_4^{2-}$. The impact and effects of the acid mine drainage discharging into river Folla have been studied for many years, but the effect in the groundwater was unknown.

Acid drainage outflowing from mines, where large amounts of sulfide minerals are exposed to weathering in tunnels, waste piles and mine tailings, is one of the main sources of soil, surface and groundwater contamination connected with anthropogenic activities in the world. Its toxicity and corrosivity cause significant degradation of the water quality, and its extreme persistence make it last for decades or hundreds of years (Aduvire and Coullaut, 2006).

The absence of life in the Folla river downstream of Folldal center has been manifested by authorities and there are projects ongoing trying to find a solution and restore the aquatic conditions in Folla. The lack of information regarding to groundwater quality and groundwater-river interaction can yield to errors and underestimations of its contribution, leading to not fully functioning or failure of the treatment solution.

Several previous studies investigated and assessed the impact of acid mine drainage in the environment surrounding abandoned and active mines around the world. In these studies detailed field data is commonly coupled with hydro-geochemical models, commonly PHREEQC and MINTEQA2, used as a tool to reproduce the physical and biogeochemical ongoing processes, in order to get a better understanding of the complex reactions taking place within the site. For instance, in the Adak mine tailings deposit in Sweden, an exhaustive environmental evaluation of the trace elements distribution in soil, sediments, plants and water was performed based on field data analysis and geochemical modeling with PHREEQC (Bhattacharya et al., 2006).
In Odiel basin (Spain) where many rivers are affected by acid mine drainage, such as Villar Creek, the toxicity of sediment and water and its environmental risk was evaluated along its course using Microtox test, field data analysis and geochemical modeling using PHREEQC and MINTEQA2 (Sarmiento et al., 2011).

Furthermore, in the P-area impoundment in Copper Cliff tailings (Canada) a reactive transport model using MIN3P was developed to show how the combination of large field data and a conceptual model of the impoundment can improve the insight and analysis of the site (Brookfield et al., 2006).

Also in Canada, in the Camp tailings impoundment in Sherridon, mineralogical, geochemical and hydrogeological analyses were performed to characterize the active oxidation zone and the principal acid-neutralization reactions within the tailings. The study also combined solid, pore water and groundwater samples and a posterior geochemical model using MINTEQA2 and PHRQPITZ (Moncur et al., 2005).

For the Erini Stream in Greece, a hydro-geochemical model, coupling SWAT and PHREEQC, was used to predict the concentration of Zn, Cd and Mn along the stream course over one hydrological year (Papassiopi et al., 2014).

Other studies focuses on techniques to provide assessment on predict the future evolution and impact of acid mine drainage. For example, in Impoundment 1 in Kristineberg (Sweden), a pyritic tailings deposit, a geochemical model was developed to assess and predict the leachate and groundwater composition in the impoundment. It was used the geochemical code STEADYQL, a mathematical code that includes slow reaction kinetics. (Salmon and Malmström., 2004). Previously Impoundment 1 case was used in a desk study to provide preliminary risk assessment of mine drainage pollution by using a simple hydrochemical model to estimate the acid-neutralization capacity based on discharge and mineralogical information (Banwart and Malmström, 2001).

Other times the objectives are to know the future effects in the surrounding water environment, like in the Lake Senftenberg, where the prediction of the impact of the water quality of the groundwater inflow into the lake was modeled based on water sampling and pyrite oxidation laboratory experiments. Based on these data the mass fluxes into the lake
were defined, allowing computer numerical models to predict the future water quality of the lake (Werner et al., 2001).

The effects can be characterized also for a long term like in Lausitz mining district. It was used to test a deterministic multi-mine life-cycle model developed to characterize the mass discharges and predict the evolution of the groundwater and river quality until 2100. The model combined HistoGis, PYROX and PHT3D (Graupner et al., 2014).

The foregoing studies are based on large and detailed field data; however this is an uncommon circumstance in old abandoned mines. In the cases where the data is sparseness the estimation, evaluation and prediction of the generation and evolution of acid mine drainage depends on the conceptual models. Thus, modeling has been use in a number of previous studies with limited data. For example, the evolution of minewater composition at Polkemmet Colliery and its impact to River Almond (Scotland) was modeled using NETPATH and PHREEQE, respectively (Chen et al., 1999). The authors concluded that geochemical model, although was based on a number of assumptions, provides a reliable preliminary estimation of minewater evolution and water quality impacts.

1.1. Objectives

The aims of this study are:

i) Characterize hydrogeologically and geochemically the groundwater at Folldal study area.

ii) Quantify the groundwater flow pattern, the water balance, and the groundwater velocity with a numerical 2D transient groundwater model with MODFLOW

iii) Implement geochemical simulations using PHREEQC as a tool for quantifying the natural contribution of pyrite oxidation to groundwater under the conditions of the study area, and the formation and evolution of acid mine drainage in the mine tailings

iv) Assess the treatment of groundwater using an anoxic limestone barrier by the integration of field data, the hydrogeological transport model and the geochemical simulations
2. Acid mine drainage

Acid mine drainage (AMD) is an acidic waste-water rich in dissolved solids, trace elements and inorganic compounds that occurs in mining areas rich in sulfur or metal sulfides. The generation of AMD occurs when the reduced sulfide minerals, S\textsuperscript{2-}, predominantly pyrite (FeS\textsubscript{2}), are exposed to atmospheric conditions (Simate and Ndlovu, 2014). In this environment, exposed to oxygen and water, the minerals may become unstable and oxidize, resulting in soluble oxidation by-products. These by-products, dissolved in water, can acidify the water if the rock contains insufficient neutralizing capacity. The increase of acidity yields the release and mobilization of heavy metals contained in the minerals which remain soluble in the AMD. The pH plays a key role on the mobility of heavy metals (Figure 2.1), at low pH metals present zero sorption to the bulk material while at neutral pH they remain fixed to the mineral (Appelo and Postma, 2005); and on the precipitation of secondary minerals, above pH 4-5.

AMD is characterized by pH below 6 (INAP, 2009) and by high content of suspended solids (order of mg/l) and sulfate and heavy metals, such as Fe, Cu, Zn, Al, Pb, Mn, Co, Cd, Ni, Cr and Hg. These elements in high concentrations are harmful to aquatic, soil and air environments (Roberts and Johnson, 1978; Solomon, 2008).

2.1. Formation of acid mine drainage

The generation of acid, and its neutralization, is the result of a large number of geochemical reactions. The main elements involved in the process are: sulfide minerals, water or humidity, an oxidant (mainly oxygen from atmosphere or from chemical origin), bacteria and temperature.
Some sulfide minerals are more reactive than others, for example iron sulfides (e.g. pyrite and pyrrhotite) are more reactive than lead, zinc or copper sulfides (respectively galena, shpalerita and chalcopyrite). This is caused by the difference in the crystalline structure stability and the solubility of the minerals (Aduvire and Coullaut, 2006). The grain size also affects the process velocity. Fine mineral texture and poorly crystallized varieties oxidizes much faster than big crystalline grains (Aduvire and Coullaut, 2006).

As water and oxygen are two essential components in the reaction, the exclusion of any of them will stop the generation of acid. Microbial activity also plays an important role in the acceleration of acid generation, some bacteria, such as *Thiobacillus ferrooxidans*, *Leptospirillum ferrooxidans* and *Sulfolobus species*, act as catalyze of the reactions (Adams et al., 2005). Its importance highly depends on the pH and temperature conditions, as well as on the concentrations of some trace elements that are toxic for bacteria (Adams et al., 2005).

The method to attenuate water acidification is called acid neutralization. It occurs naturally if neutralizing minerals are available in the environment, such as calcite, dolomite and other carbonates, iron and aluminum hydroxides, aluminosilicates and clays. The dissolution of these minerals counteracts the acidic products of sulfide oxidation, but their effectiveness to buffer the acid is determined by their reactivity or how fast they are dissolved (Brough et al., 2013). Generally, Ca-Mg carbonate minerals are the most reactive and are required in large amounts for effective neutralization (INAP, 2009).

Acid mine drainage is thus formed when the surrounding materials of the production area have insufficient capacity to neutralize the acid and maintain neutral pH conditions or when the initial capacity is depleted.

### 2.1.1. Pyrite oxidation

The oxidation of pyrite (FeS$_2$) by atmospheric O$_2$ is defined as:

$$\text{FeS}_2 + \frac{7}{2}\text{O}_2 + \text{H}_2\text{O} \rightarrow \text{Fe}^{2+} + 2\text{SO}_4^{2-} + 2\text{H}^+ \quad (2.1)$$

The presence of sulfate (SO$_4^{2-}$) is typically the first indicator of sulfide mineral oxidation (Simate et al., 2014). Dissolved ferrous iron (Fe$^{2+}$), sulfate and hydron (H$^+$) represent an increase of the total dissolved solid and water acidity, thus a decrease in the pH (unless is
naturally neutralized). If sufficient oxidant is available, ferrous iron may be oxidized to ferric iron ($Fe^{3+}$) by the reaction:

$$Fe^{2+} + \frac{1}{4} O_2 + H^+ \rightarrow Fe^{3+} + \frac{1}{2} H_2O \quad (2.2)$$

In general, the ferric ion precipitates when pH is above 3.5 by hydrolysis as hydroxide $Fe(OH)3$, what triggers a decrease in pH.

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 (solid) + 3H^+ \quad (2.3)$$

Some ferric cations which do not precipitate in the solution can continue oxidizing the pyrite additionally (indirect oxidation) and form new ferrous ions, sulfate and hydrogen.

$$Fe^{3+} + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16 H^+ \quad (2.4)$$

Based on these basic reactions, the overall process (Figure 2.2) is represented by:

$$FeS_2 + 15/4O_2 + 7/2H2O \rightarrow Fe(OH)_3 + 2 SO_4^{2-} + 4H^+ \quad (2.5)$$

The acidic products from pyrite oxidation may be counteracted by neutralization reactions. The generic reaction of acid consumption by dissolution of Ca-Mg carbonate minerals can be described as:

$$MeCO_3+ H^+ = Me^{2+}+ HCO_3^{-} \quad (2.6)$$

where Me represents calcium or magnesium divalent cation (INAP, 2009).

Figure 2.2: Model for the oxidation of pyrite (Modified from Simate and Ndlovu, 2014).
2.2. Treatment of acid mine drainage

AMD can vary in its chemistry, and thus its environmental effects, depending on the ore mineralogy, the quantity of material, the mining methods (surface or underground), the variety of mineral processing operations and the climate and hydrology conditions. Therefore multiple remediation techniques and innovation solutions have been developed to address the problem.

The treatment processes are based on the removal and isolation of metals from solution. There are numerous remediation options classified according to if they control the acid source or its migration (Johnson and Hallberg, 2005). Source control techniques prevent the formation based on the removal of oxygen or water from the system prior its contact with the sulfide material by flooding underground mines and mine tailings, coating or sealing waste heaps and tailings or revegetating the land (Aduvire and Coullaut, 2006), without require any further water treatment. On the other hand, migration remediation techniques treat the resultant acid drainage using chemical or biological treatment systems, either actively or passively (Figure 2.3). These are usually used for a short period of time, they do not work on the long term; therefore a combination of two or more of these methods is usually required to neutralize the acid waters and obtain better water quality more effectively.

The treatments are selected depending on the goal to achieve, such as neutralize, desalination, general removal of metals or specific pollutant treatment; and usually are based on: (1) metals precipitation, the most common method is to add a chemical-neutralizing agent to increase the pH and precipitate the dissolved metals; (2) ion exchange and sorption process, to remove heavy metals from the aqueous solution; and (3) use of microbial activity, either by using biocides to inhibit the activity of the bacteria which catalyze sulfide oxidation or using microorganisms able to generate alkalinity and immobilize heavy metals (Johnson and Hallberg, 2005).

In this study, the performance of an anoxic limestone drain to neutralize and remove metals from the groundwater was simulated. This passive system was selected among the others, because is the most consistently efficient passive treatment in terms of the cost per tonne of acid removed, and their efficiency is independent of the pH of the incoming water (Ziemkiewicz et al., 2003). Anoxic limestone drains treat mine water based on metals
precipitation by adding limestone as a neutralizing agent. They consist of buried and sealed cells of limestone into which water flows through. When the acidic groundwater flows through the drain he limestone dissolves, and since CO$_2$ cannot escape, an increase of bicarbonate occurs, thus adding alkalinity, precipitating metals and increasing the pH (Ziemkiewicz et al., 2003).

Figure 2.3: Generic Range of Drainage Treatment Technologies (INAP, 2009).
3. Study site

Folldal is located in Hedmark County, central Norway, and 410 km North from Oslo. It is settled at the foot of Rondeslottet Mountain on the north side of river Folla, which is the third largest tributary to river Glomma, the longest and largest river in Norway (Aanes, 1980).

The area investigated has a surface area of approximately 137.7 km$^2$ and it covers Folldal village, part of Folldal Main mine, and two main areas of mine tailings. It is bound to the South by Folla river (comprising a stretch of 1.5 km), to the North by the mining area and to the west by Gørbekken creek (tributary to river Folla) (Figure 3.1).

![Figure 3.1: Study area location. West and East mine tailings areas marked with a red rectangle.](image)

The land above the aquifer is covered by agricultural and forest areas (pine trees, grasses and lichens), the mine (comprising mine galleries, slag heaps, tips along and mine tailings along with 70 buildings) and urban infrastructures (some of them apparently built on top of mine tailings). Folldal Mines, mining area and facilities (buildings, machinery and equipment), is a cultural monument site and is protected by the Cultural Heritage Law, which does not allow alteration of the historical materials, structures and landscape.
3.1. Mining history

Folldal Main mine was opened in 1747 and the first mining operation lasted until 1878. In 1906, with foreign financial support, the mining activity begun again and mine infrastructure and production facilities were built (Figure 3.2), but after several production periods the Main mine was closed down in 1941 because there was no more extractable ore left (Folldal Gruver, 2015). Even so, the processing facilities located in the Main mine were still active and received ore to process from the other mines in Folldal area until 1968, when all the mines were closed down and the mining activity was moved to Hjerkinn.

The ore major minerals at the Main mine were pyrite ($\text{FeS}_2$), chalcopyrite ($\text{CuFeS}$) and sphalerite ($\text{ZnS}$), rich in base metals: copper, zinc, lead and iron sulfides; and trace metals such as cobalt, arsenic, bismuth, antimony and tin. In total 1.15 million tons of ore with 1.9% Cu and 1.1% Zn were extracted by underground mining, in 14 levels approximately 700 m deep (Geological Survey of Norway, 2014), and further send to Alvdal station by ropeway. Overall, 3.65 million tons of ore extracted from Folldal area mines were treated in the production facilities in the Main mine.

Once the ore was extracted, the valuable mineral or metals were obtained from it through different mechanical and chemical techniques, which changed the structure and concentration of the ore, such as crushing, washing, flotation and gravity concentration.

This mineral processing generated massive amounts of waste material and water, which were deposited uncovered in Folldal area (Figure 3.3).
3.2. Geology

Geologically the study area is formed by sedimentary material of variable thickness on top of an igneous rock basement (Figure 3.4).

![Schematic cross section of the geology of the area.](image)

Figure 3.4: Schematic cross section of the geology of the area.

The sedimentary cover consists of glaci-fluvial deposits from the last glacial period and quaternary fluvial deposits (Figure 3.5). The material transported and deposited by glaciers is poorly sorted, hard packed and varies in grain size from clay and fine sand to blocks. The fluvial material is sorted and rounded, composed of sand and gravel and some layers of organic material (Geological Survey of Norway, 2014).

![Quaternary soil map of Folldal area.](image)

Figure 3.5: Quaternary soil map of Folldal area (Geological Survey of Norway, 2014).
The bedrock consists of cambro-silurian materials of the Trondheim Nappe complex: the volcanogenic Fundsjø group is overlaying the sedimentary Gula group (Figure 3.6). The Fundsjø group comprises volcanic, volcanoclastic and minor sediments such as basalts, quartzites, greywackes and graphitic schists from base to top (Bjerkgård and Bjørlykke., 1994). The Gula group consists of psammitic-chlorite-mica schist, quartzites and semipelites, which have intercalations of conglomerates and marbles.

![Geological map of Folldal area (Bjerkgård and Bjørlykke, 1994).](image)

**Figure 3.6:** Geological map of Folldal area (Bjerkgård and Bjørlykke, 1994).

The basalts contain a Trondhjemite, which is a light-colored tonalite subvolcanic intrusion, dating from 488 Ma and source of the sulfide deposits in the Fundsjø group. The sulfide deposits are located at three different levels and correspond to Cu-Zn class of volcanic-associated massive sulfide deposits (Bjerkgård and Bjørlykke, 1996). The disseminated ore is overlain by strata-bound massive mineralization; its metal content depends on the host-rock or underlying strata composition.

The sulfide deposit at the Main mine is hosted by metabasalt (Figure 3.7), and its mineralization process took place in four separate lenses (Geological Survey of Norway, 2014). The ore, which thickness was around 112 m prior mining was richer in copper content than zinc, had low content of lead and high content of cobalt.
3.3. Hydrological setting

Folldal area has a mountain climate, characterized by cold and arid winters and humid and relatively hot summers. In the wintertime the temperature is below 0 ºC, with the lowest recorded in January around -34 ºC. The highest temperature is around 28ºC, measured in July. The mean annual temperature is 1ºC.

Its annual precipitation is among the lowest in Norway (Norwegian Meteorological Insitute, 2015a), based on available meteorological data from the observation station of Folldal, the mean annual precipitation is 470 mm while the wettest areas of Norway can have an annual precipitation above 3500 mm. Characteristic of the precipitation in Folldal area is a dry winter and high rainfall events in summer (Figure 3.8).

Figure 3.7: Schematic stratigraphic columns of host rocks and wall-rock alteration related to Folldal Main mine (Hovedgruva) deposit. The thicknesses of the ore and alteration zones are exaggerated compared to host rocks (modified from Bjerkgård and Bjørlykke, 1994).

Figure 3.8: Monthly precipitation and daily temperature measurements in Folldal meteorological station during the hydrological year 2014-2015 (Norwegian Meteorological Institute, 2015c). The black line shows the normal.
Runoff from Folldal village is discharging into the river Folla, which has a total length of 108 km. It runs from Dovre Mountains to Alvdal, where it meets river Glomma. Its catchment area covers 2110 km². The water flow is relatively high in summer (9 m³/s), has some single floods during autumn, is low in winter (1.6 m³/s) due to part of the river froze and it has large floods during spring (40 m³/s) when the snow cover melts (Figure 3.9).

![Water flow in Grimsmoen NVE measuring station in Folla river during hydrological year 2014-2015 (NVE, 2015).](image)

The main groundwater recharge comes directly from local precipitation and snow melt. Although, human activities, such as residential areas and urban infrastructures, and the steep slope of the area are reducing the water infiltration and increasing the surface runoff (Martínez de Azagra, 2006).

### 3.4. Previous works

Mining and processing ore yields environmental problems and challenges. Consequently, in the 1980's the Norwegian Environmental Pollution control Agency (SFT) established remedial actions for mining areas (mainly removal of waste material dumps) to reduce the emissions of copper to 10 μg Cu/L in the rivers by 2010 (Miljødirektoratet, 2014), to ensure that biological life is maintained (copper is an acute-acting poison to aquatic organisms). Thus, projects to control and monitor the effects of mining in Folldal area, as well as implement of measures, without alter the historical landscape; have been conducted for many years.
3.4.1. 1975-1976
During 1975-1976 a study was carried out by Aanes (1980) in the Folla River in order to evaluate the impact of the mining activity. The study covered 36 km of the river, from the mine at Hjerkkin until 11 km downstream Folldal village, using seven sampling stations along the river. The investigation was based on physical, geochemical and biological analysis.

The river section downstream of Folldal Main mine was found to be the most polluted section of the river (Aanes, 1980). The effect of the acid mine drainage was clearly observed, comparing data from the station located right before the Main mine area and the station located 11 km downstream of Folldal, there was a considerable difference in heavy metals concentration in the river water and in deposits of ferric hydroxide on the substrate along the riverbed. Moreover, it was also the section with lowest density of benthic macro-invertebrates.

3.4.2. 2006
During autumn of 2006 a trial anaerobic system was built by Ettner (2007) to remove copper and zinc by using sulfate-reducing bacteria to precipitate metal sulfides. The system consisted of four basins about 8×1.5 m and 1.5 m deep, filled with rotted hay, tree clippings, shell sand and gypsum. After constructed they were filled with water and then allowed to rest, developing an anaerobic system.

During early 2007, analysis of the basin water showed metal precipitation and a decrease of 78% in the reduction potential (Eh). After monitoring water flow was allowed again, at a constant flow rate as possible in order to effectively remove metals the water flow to the system (Ettner, 2007). Unfortunately, no further data and results about this project or experiment are available.

3.4.3. 1966-2013
The Norwegian Institute for Water Research (NIVA) has conducted surveys in Folla watercourse since 1966. Investigations have covered the entire river section from the mining area at Hjerkinn to down to Alvdal.
During 1992-1994 a number of mitigation measures were implemented, under the direction of the Norwegian Directorate of Mining (DMF). Different mine waste materials from the abandoned mining Folldal area were removed and deposited inside Tverrfjellet mine (Tverrfjellet mine closed in March 1993) and a simple drainage system was constructed.

Late autumn 2005 the DMF started works to drain the Main mine area better. New drainage ditches were constructed to prevent surface water from entering the mine during high flood periods and to prevent leakage to the area downhill.

The runoff from Folldal center was followed up with a systematic program since 1993, after the mitigation measures were finished, to 2013. The program included monthly surface water sampling at two stations in the Main mine area (Station 1 and Station 2) and one in Folla river (Fo7 Folshaugmoen), downstream of Folldal center. Over the years the monitoring was upgraded with continuous records and a new measuring station (Grimsmoen) was established by NVE at Folla to measure the water flow.

The continuous monitoring shows that the concentration of heavy metals of the drainage flowing from the mining area and of Folla River varies along the year depending on the rainfall and the climate. Episodes of intense precipitation and snow-melt lead to large runoff (Figure 3.10). Yet, apart from changes due to annual meteorological variations it seems that the annual mean is constant (Table 3.1). The water quality of the drainage and in Folla river downstream Folldal center has not changed significantly over the last 16 years (Table 3.1). Suggesting that it may take a very long time until conditions prior mining are naturally restored.

Figure 3.10: Daily transport of copper (kg Cu/d) at the two drainage measuring stations in Folldal mining area (Thyve and Iversen, 2013).
Runoff from the remaining mine waste have shown to be the main source of pollution in the area. However, there are discrepancies between the metal transport values, especially zinc, of the mine drainage and Folla River. The values from the drainage measuring stations and river Folla follow the same tendency; however values from Folla are 2-3 times higher in the snow melt and floods seasons (Figure 3.11). This disagreement may indicate that all the runoff and inflows from the mine waste area are not captured in the drainage system (Thyve and Iversen, 2013). Thus, this excess of water is flowing towards the river and in the process infiltrating the ground-surface.
3.4.4. 2014

Since 2014 the Norwegian Geotechnical Institute (NGI) has been assigned by the DMF to consider possible measures to reduce the pollution from the mine and improve the water quality in Folla. Therefore, NGI conducted field investigations in early July 2014 to identify and quantify the possible sources of contamination by sampling tailings and water (surface and groundwater) and to assess the potential impact of possible measures.

The mine tailings shown to be a mixture of different types of material in terms of mineralogy, geochemistry, grain size and weathering, the sulfur (S) content analyzed ranges from 0.2 to 50%. This heterogeneity is due to they were deposited at different times, ways and depths. The analysis of the samples shown that the tailings are acid generating, they have potential to generate acid drainage for a long time and lead to contaminated water with high concentrations of heavy metals. Although no clear trend in disposal (content independent of depth) or pollution potential was observed. Hence, the mine tailings are considered to be the main source of contamination and acid formation. A map based on the acid formation potential of the different masses was developed (Figure 3.12).

Water samples analysis revealed that surface water is very affected by acid formation; samples were classified as very heavy polluted according to the Directorate of Environment classification and had very low pH and high conductivity, 2.5 and 4 mS/cm, respectively. The groundwater from shallow wells showed that infiltration water is rapidly affected by the tailings surface layer, samples had low pH, between 4-6, high conductivity, 1.5 mS/cm, and high concentration of heavy metals (e.g. 130 mg/l Fe, 2.7 mg/l Cu, 20 mg/l Zn and 320 µg/l Ni).
Figure 3.12: Map of the acid forming potential of the mine tailings in Folldal Main mine area. Points in red are tailing samples, in dark blue are groundwater samples, and in light blue are surface water samples. White stripes show zones with few samples (NGI, 2014).
4. Methods

The hydrogeological characterization of the geometry, boundary conditions and hydrogeological parameters of the aquifer was based on field methods, as piezometer network installation, continuous groundwater level monitoring, geophysical surveys and slug test. Further, a numerical 2D transient groundwater model with MODFLOW was used as a tool for quantifying (1) the groundwater flow pattern, (2) the water balance, (3) the groundwater velocity.

The geochemical characterization was based on three data sets of new groundwater samples collected and analyzed for major ions and trace metals. Further, three geochemical models with PHREEQC were used as a tool for quantifying (1) the natural contribution of pyrite oxidation to groundwater, (2) the formation and evolution of AMD in the mine tailings, and (3) the performance of an anoxic limestone drain, as passive system to neutralize acidic groundwater.

4.1. Field studies

To determine the aquifer and groundwater characteristics a combination of several measurement methods was used in four field surveys carried out between June 2014 and April 2015. The first field work campaign, in late June 2014, lasted ten days and was part of the course “Field methods in hydrogeology” by UiO. Geophysical methods, hydrogeological techniques, single groundwater monitoring, water sampling and analysis were performed.

In the second field survey, in mid-October 2014, sensors for continuous monitoring were installed and a second water sampling and hydrochemical analysis data set was obtained. In early December a one-day field work was carried out to obtain the data recorded with the sensors and measure the groundwater and river level. The fourth and last field campaign was in late April 2015 to download the data recorded with the sensors, measure the groundwater table and obtain the third data set of water sample and hydrochemical analysis.

4.1.1. Seismic refraction

To investigate the aquifer depth, assumed to be the igneous bedrock depth, seismic refraction survey was carried out. The combination of geophysical exploration techniques and
geological survey makes possible to locate and characterize discontinuities and water-bearing structures (Sharma, 1997).

Seismic refraction investigation determines the structure and composition of the subsurface by measuring the change in velocity of elastic-seismic waves when traveling through different geologic units. Since each type of material has a specific sound propagation velocity (Table 4.1).

Refraction surveys determine the velocity distribution by generating seismic waves (sound) at a point and recording the times of arrival of the first longitudinal waves (P-waves) at various other points on the soil surface (Figure 4.1). In this method the reflected and incident angles are critical, thus capable to evaluate increasing velocity gradients and to locate layers or features with high velocities (Schlumberger, 2015), as in this case were the objective is to find the igneous bedrock depth, which has higher velocity than the sediment cover (Table 4.1).

To obtain the data from the subsoil geophones are used to convert ground motion to electrical signal. This electrical signal is recorded in the seismograph and then filtered to obtain the seismic profiles. However to analyze the profile, the corrected refraction times have to be converted into depths.

Figure 4.1: Seismic method (Assaad et al., 2004).
In a two-horizontal-layer problem, like in this study area where the igneous bedrock (layer 2) is covered by glacial-fluvial sediments (layer 1), the depth of the layers can be estimated using the intercept time method, which uses the apparent velocity of the layers and the intercept time from the critically refracted wave to calculate the layer thickness. The velocity, $V$, (L/T) is calculated by the inverse of the slope of the line fitting the first refracted arrival lines:

$$V = \frac{x_2-x_1}{t_2-t_1} \quad (4.1)$$

where $x$ is the channel distance (L) and $t$ is the arrival time (T).

The equation for the intercept time, $T_{int}$, (at $x=0$) is,

$$T_{int} = 2h_1 \sqrt{\frac{1}{v_1^2} - \frac{1}{v_2^2}} \quad (4.2)$$

where $h_1$ is the depth of layer 1 (L), $v_1$ is the velocity of layer 1 (L/T) and $v_2$ is the velocity of layer 2 (L/T).

Data acquisition

The seismic acquisition was performed during the “Field methods in hydrogeology” course in June 2014. Three seismic surveying were carried out in the area using a Geode seismic recorder (Figure 4.5). As a source of energy a sledge hammer was stroked on a steel plate on the ground (Figure 4.2). It produces 50-200Hz, enough energy to obtain good results for targets at shallow depths. Low frequency geophones were disposed along the linear array. The separation of recording points was 5 meters, and the energy source to geophone offset was 1.5 m. For every profile forward, middle and reverse shots were performed.

Figure 4.2: Seismic profile acquisition in the field in June 2014.
After the seismic data was processed by Asbjørn Breivik (UiO), the refraction profiles were interpreted using the time intercept method and the table of P-waves velocities (Table 4.1).

<table>
<thead>
<tr>
<th>Material type</th>
<th>$V_p$ (km/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Unconsolidated sediments</strong></td>
<td></td>
</tr>
<tr>
<td>Clay</td>
<td>1.0-2.5</td>
</tr>
<tr>
<td>Sand (dry)</td>
<td>0.2-1.0</td>
</tr>
<tr>
<td>Sand (saturated)</td>
<td>1.5-2.0</td>
</tr>
<tr>
<td><strong>Igneous and metamorphic rocks</strong></td>
<td></td>
</tr>
<tr>
<td>Basalt</td>
<td>5.3-5.6</td>
</tr>
<tr>
<td>Granite</td>
<td>4.7-6.0</td>
</tr>
<tr>
<td>Slate</td>
<td>3.5-4.4</td>
</tr>
</tbody>
</table>

Table 4.1: Approximate range of velocities of P-waves for typical rocks and materials of the study area (based on Mussett and Khan, 2000).

4.1.2. Electrical Resistance Tomography

The Electrical resistance tomography (ERT) is helpful to obtain an overview of the study area subsoil characteristics. This geophysical technique uses the apparent resistivity values measured on the ground surface to obtain 2D images of the real resistivity distribution of the subsurface. The presence of water, due to its dissolved salts, alters the conductivity of many rocks, permeable materials, for example, tend to have lower resistivity when are saturated than when are dry (Musseet & Khan, 2000). Based on that, this method was used to investigate the water table location in the study area and to evaluate possible sedimentological changes in the top part of the aquifer.

During electrical tomography surveys, an electric current is injected into the ground and the resulting potential difference between a pair of points of the surface is measured (Figure 4.3). The resistance of the conductive object is described by Ohm’s law as the relationship between the injected current and the potential measured (U.S. EPA, 2011):

$$ R = \frac{\Delta V}{I} \quad (4.3) $$
where \( R \) is the resistance (\( \Omega \)), \( V \) is the voltage (V) and \( I \) is the current (A). On the other hand, the resistance of a length of the material is given by:

\[
R = \frac{L}{K}
\]  
(4.4)

where \( \rho \) is the resistivity of the object (\( \Omega \)m) and \( K \) is a constant depending on the geometry of the conductor object (m).

Combining (4.3) and (4.4) the resistance of an object or material is expressed in terms of its resistivity:

\[
\rho = K \Delta V/I
\]  
(4.5)

However, this resistivity measured is fictional, and is called apparent resistivity. The variation of the real resistivity of the subsurface can be estimated by the inversion method. The result is an image of a pseudosection of the subsurface where the real resistivity values are represented.

---

**Figure 4.3:** The arrangement of electrodes for a 2D electrical survey and the sequence of measurements used to build up a pseudosection for the Wenner electrode array (from Loke, 2004).

To obtain the data electrodes connected to a multi-core cable, a resistivity meter and a laptop computer are necessary. There are different gradient arrays or electrode arrangements. Generally, they use four electrodes: two injecting electricity and two registering the potential produced. The difference between them is the electrode spacing and the position of the current electrodes regarding the potential ones. The selection of array type and sequence of measurements depends on the depth and resolution wanted (Loke, 2004).
Data acquisition

The ERT acquisition was performed during the course “Field methods in hydrogeology” in June 2014 (Figure 4.4). Three 2-D electrical imaging transects, including nine profiles were carried out in the area using a SYSCAL PRO Switch electrical resistivymeter with 48 electrodes along a straight line attached to a multi-core cable (Figure 4.5). A constant spacing of 1 m between electrodes was used. The sequence of measurements was taken using the Wenner electrode array; which gives good resolution of horizontally layered and shallow structures, such as the groundwater table.

In the Wenner array, the inner two electrodes register the potential and the outer two electrodes inject the electric current. The array spacing expands about the array midpoint while maintaining an equivalent spacing between electrodes (Figure 4.3).

Analysis

After the ERT data was processed by Helen French (NMBU), the nine profiles were interpreted using field observations, soil map and the book resistivity values for different materials (Table 4.2).
<table>
<thead>
<tr>
<th>Material type</th>
<th>Resistivity (ohm.m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unconsolidated sediments</td>
<td></td>
</tr>
<tr>
<td>Clay</td>
<td>1.0-100</td>
</tr>
<tr>
<td>Sand</td>
<td>10-1000</td>
</tr>
<tr>
<td>Igneous and metamorphic rocks</td>
<td></td>
</tr>
<tr>
<td>Basalt</td>
<td>$10^3$-$10^7$</td>
</tr>
<tr>
<td>Granite</td>
<td>$10^4$-$10^6$</td>
</tr>
<tr>
<td>Slate</td>
<td>$10^3$-$10^8$</td>
</tr>
<tr>
<td>Other</td>
<td></td>
</tr>
<tr>
<td>Fresh groundwater</td>
<td>10-100</td>
</tr>
</tbody>
</table>

**Table 4.2:** Representative values of resistivity for typical rocks and materials of the study area (based on Luke, 2004).

### 4.1.3. Piezometer network

In order to characterize the groundwater table elevation 14 new observation metallic casing open wells were drilled in June 2014, in addition to seven PVC casing closed wells that were drilled in the area in the 1990's (Forbord, 2015). The wells were distributed across the area between Folldal village and Folla, covering two mine tailings exposed areas and the forest area in between. To obtain a cross section of the groundwater flow, 2-3 piezometers were located along a perpendicular transect to the river in each area (Figure 4.6).

Moreover, in the forest area and the eastern mine tailings a piezometer nest with three piezometers at different depths was located in the river bed to determine the vertical hydraulic gradient in order to investigate the relationship groundwater-river (Figure 4.7).

![Figure 4.7: Piezometer nest CS by the river side. Conceptual figure of the different well depths (dash lines).](image-url)
The wells were installed using a jackhammer to drill the metal pipes into the ground (Figure 4.8). For the piezometer nest, were used 1m long and 3 cm wide pipes, and a 7.5 cm long well screen. On the other 8 wells, 1.5 m long and 4 cm wide (outer diameter) pipes, and a 16.5 cm long metal-over well screen. The well screen is situated just above the well bottom.
4.1.4. Groundwater level monitoring

To observe the seasonal effects in the groundwater aquifer system during the year, the groundwater level was monitored manually during the field campaigns using a water level probe, and continuously with the installation of pressure and temperature dataloggers.

Pressure dataloggers are installed at a known depth below the water table for measuring and recording time-series pressure data on groundwater level fluctuations in wells. They have a highly accurate membrane that senses the pressure changes, in the weight of the water column above the device plus the current atmospheric pressure. Therefore, the absolute pressure measurements recorded by the diver in the well have to be compensated for the air pressure variations. The resulting water column pressure (cmH20) can be expressed as height of water column above the diver or as water level with respect to the top of well casing, which can be done using (1) a manual measurement of the water level at that exact time, (2) the diver cable length (Figure 4.9).

![Figure 4.9: Water level compensation methods (Schlumberger Water Services, 2007). (a) The height of the water column above the Diver and (b) with respect to the top of well casing, if the water level is below the top of casing it has a positive value.](image)

**Divers data installation and acquisition**

Five water pressure dataloggers and one barometric pressure datalogger (Mini-diver model by Schlumberger Water Services) were placed in the PVC wells C2, C7, M1, A1 and A4 in October 2014 (Figure 4.10), and were set to record data every 6 hours. These specific wells were selected because they strategically cover all the boundaries of the area of study. The devices have a pressure range of 17 mH20, an accuracy of ± 1.0 cmH20 and a resolution of 0.4 cmH20 (Schlumberger Water Services, 2010). The temperature range is 100 °C, the accuracy ± 0.1 °C and the resolution 0.01 °C.
4.1.5. Slug test – Hvorslev’s analysis method

To estimate the hydraulic conductivity of the subsurface material slug tests were carried out in all the wells. Slug tests are suited to obtain a field-permeability value in unconfined aquifers (Watson and Burnett, 1993).

The hydraulic conductivity ($K$) is a hydraulic property of the soil that describes the capacity of the medium to transmit water, which depends on the properties of the soil matrix. It is defined by Darcy’s law as:

$$K = \frac{q}{i} \quad (4.6)$$

where $q$ is the specific discharge ($L/T$) and $i$ is the hydraulic gradient. The hydraulic gradient is the decrease in the hydraulic head divided by the distance, $i = \Delta h/\Delta L$.

The slug test is a field technique to determine in situ the hydraulic conductivity of the porous media in the immediate vicinity of the well (Kresic, 2007). It consists of measuring the rate of recovery of head in a well after a sudden change in the water level at that well (Butler, 1997). From here, the hydraulic conductivity value is estimated by analyzing the data measured. Slug test are practical in low permeability materials, such as mine tailings, clays and silt, since they assure a relatively slow rate water-level recovery which provides accurate measurements (Watson and Burnett, 1993).

Data acquisition

The tests were performed during the “Field Methods in Hydrogeology” course in June 2014 and in December 2014. Water was added to each well, raising the level one meter over the original elevation, to produce a change in the initial hydraulic head and then the decrease of the water level, as the water drained into the aquifer, was monitored (1) manually, by doing
manual water-level measurements every 5-30 s depending on the well; and (2) automatically, using pressure dataloggers. Schlumberger divers were installed at half of the distance between the depth of the pipe and the water table and were set to record every 10 s.

**Data analysis - Hvorslev’s method**

From the test response, the rate at which the water in the well falls, the hydraulic conductivity of the soil can be estimated comparing the data measured with theoretical models. In this study, the response data was analyzed by the Hvorslev method (Hvorslev, 1951), since is an analysis method for slug test data from wells that do not completely penetrate the unconfined aquifer or are not fully screened, like the wells in this study. The results were interpreted using the representative values of hydraulic conductivity for soil materials (Table 4.3).

<table>
<thead>
<tr>
<th>Material type</th>
<th>Hydraulic conductivity (m/d)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unconsolidated sediments</td>
<td></td>
</tr>
<tr>
<td>Clay</td>
<td>$10^8 - 10^3$</td>
</tr>
<tr>
<td>Sand</td>
<td>$10^1 - 10^2$</td>
</tr>
<tr>
<td>Glacial till</td>
<td>$10^7 - 10^1$</td>
</tr>
<tr>
<td>Silt, Loess</td>
<td>$10^3 - 10^1$</td>
</tr>
</tbody>
</table>

Table 4.3: representative values of hydraulic conductivity of typical soils in the study area (based on Heath 1983).

Hvorslev defined the drawdown water process as exponential – it is quick in the beginning and slows down with time when it is close to the static water level. Hvorslev proposed an analytical theoretical method where the drawdown data is plotted as head ratio (H/Ho) against time, in log an arithmetic scale respectively (Figure 4.11). And then the time associated to $H/Ho = 0.37$ ($T_o$), the time it takes for the water level to fall to 37% of the initial change, is used in the hydraulic conductivity equation:

$$K = \frac{r^2 \ln(L_e/R)}{2L_e T_o}$$

(4.7)

where $K$ is hydraulic conductivity (cm/s), $r$ is the inner radius of the well screen (cm), $R$ is the outer radius of the well screen (cm) and $L_e$ is the length of the well screen (cm). This
formula applies if the length of the well screen is more than 8 times the outer radius of the well screen \((L_e/R > 8)\), as in the case for the wells used in this study.

**Figure 4.11:** Hvorslev's method (from Fetter, 1994, based on Hvorslev, 1951).

### 4.1.6. Water sampling

The hydrochemical sampling was carried out during three surveys in late June and mid-October 2014, and in April 2015, with in situ measurements and laboratory analysis. A total of 78 water samples were collected throughout the study area (Figure 4.5). The samples included 63 from monitoring wells, 3 from drainage creek from the mine and 12 from river locations.

The wells were pumped to purge the water according to the standard procedure for groundwater sampling, which states that monitoring wells should be purged, at least, three times the volume of water stored in the well (U.S. EPA, 2013).

The water samples were filtered in the field using 0.45 µm Santorius Stedim millipore filters. Samples were collected in 50 mL centrifuge tubes as: (1) filtered samples for anions and cations analyses and (2) filtered samples for metal analyses, which were acidified to pH< 2 with 0.02 M ultrapure HNO₃.
Temperature, pH, electrical conductivity, dissolved O$_2$, Eh and alkalinity were measured in the field (Figure 4.12). These physicochemical measures were made using a VWR pHenomenal PC5000H and OX4000H instruments equipped with combination of electrodes. The pH meter was calibrated using standard solutions (pH 4, pH 7 and pH 10). The alkalinity was determined by manual titration method; using “Alazarin-sulfonic-Na- salt” as pH indicator, 0.02M HCl acid was added to 50 ml of sample through a pipette, until there was a shift of color.

![Figure 4.12: Field measurements of the physicochemical properties and alkalinity of the water samples.](image)

### 4.1.7. Water quality chemical analysis

78 water samples were analyzed in the laboratory of the Department of Geosciences by Ion Chromatography and by Inductively Coupled Plasma Mass Spectrometry. The results of the water analysis performed at the Laboratory of Department of Geosciences and the physicochemical values measured in the field are tabulated in Appendix A.

#### 4.1.7.1. Ion chromatography

Ion chromatography was used to determine major anions and cations in the water samples. The basic principle of chromatography is to separate the different ions of the sample based on their interaction with a resin (Figure 4.13).

A small volume of sample, typically 2-3 mL, is pumped and passes through the ion chromatography column, where the ions of the sample are separated due to different retention affinity with the fixed phase of the analytical columns. Once separated, the sample passes through a conductivity detector where the amount of ions in the sample is recorded as electrical conductivity (University of Oslo, 2013). Prior to running the water sample, the
system is calibrated using a reference standard solution, which the data obtained is compared with in order to identify and quantify the ions in the sample. The result is a chromatogram (a plot of the detector output vs. time) where the position of the peaks shows the identified ion and its area indicates the ion’s amount (concentration).

![Figure 4.13: Ion chromatography method (University of Oslo, 2013).](image)

**Analysis**

Prior analysis the samples were diluted ten and hundred times with deionized water to avoid high analyte concentration, if the sample is too concentrated the analytes are out of the calibration range (Dionex Corporation, 2011).

Folldal samples were analyzed for the major anions (F\(^-\), Cl\(^-\), SO\(_4\)\(^{2-}\), Br\(^-\), NO\(_3^-\) and PO\(_4\)\(^{2-}\)) and cations (Na\(^+\), K\(^+\), Mg\(^{2+}\) and Ca\(^{2+}\)) using the Dionex ICS-1000 Ion Chromatography System (Figure 4.14), which uses chemical suppression of eluent conductivity to conduct the analysis, following the standard method 4110 B. 28 mM KOH was used as eluent. The relative standard deviation (RSD) is around 0.5% for all the species, except for NO\(_3^-\) and PO\(_4\)\(^{2-}\) that is 1.5%. The minimum concentration detection limits can be as low as few ppb.

![Figure 4.14: Dionex ICS-1000 in the laboratory of the Department of Geosciences.](image)
4.1.7.2. Inductively Coupled Plasma-Mass Spectrometry (ICP-MS)

ICP-MS was used to determine the heavy metals in solution in the water samples. This analysis technique combines the inductively coupled plasma (ICP) with a mass spectrometer (Figure 4.15).

The ICP converts the atoms of the elements in the sample to singly-charged ions, which are transferred into the mass spectrometer. In the mass spectrometer the ions are separated according to their mass-to-charge ratio (m/e), i.e. an electrostatic quadruple mass filter is set up to at any given time, only allow a single mass-to-charge ratio to entrance and exit to the detector (PerkinElmer Instruments, 2011). The ions hitting the detector release electrons, which are amplified until they become a pulse. The system counts and translates the pulses into an electrical signal (intensity) that is proportional to the amount of the element in the sample. The concentration of the elements is then determined through calibration by using a reference standard solution.

![Figure 4.15: ICP-MS method (Kahrilas, 2013).](image)

**Analysis**

Prior analysis samples were first centrifuged, to avoid particles in the solution due to the low dissolved solids tolerance of the machine (TDS <0.2%) (Bruker, 2011), and then diluted with 1% HNO₃ acid to avoid high analyte concentration, if the sample is too concentrated the analytes are out of the calibration range. All the samples were diluted five times, except for the most contaminated ones that were diluted 20, 40 and up to 80 times.
Folldal samples were analyzed for Cd, Co, Sb, Pb, Ni, Cu and Zn using the Bruker Aurora Elite M-90 ICP-MS (Figure 4.16). The mass spectrometers have resolutions between 0.7-1.0 amu. The approximate instrumental detection limit of the elements Pb, Cd and Sb is <10 ppt, for Ni is <20 ppt, for Co < 40 ppt, for Cu <900 ppt and for zinc is <1200 ppt. Sensitivities > 200,000 cps / ppb.

4.2. Hydrogeological model

4.2.1. Numerical model

A two-dimensional numerical model was developed to simulate the transient groundwater flow and the particles and mass transport in the aquifer. At the model were defined, using the data collected from June 2014 to April 2015: the location of 5 observation wells; hydraulic conductivity, specific yield and porosity layer parameters; and model boundaries and individual elements of the water balance such as recharge from precipitation, drains, aquifer’s depth, Folla River and the regional inflow.

GMS (Aquaveo, 2007) graphic user interface for the USGS-MODFLOW code, which is a 3D finite-difference groundwater flow model (McDonald and Harbaugh, 1988), was used to simulate 2D groundwater flow in the aquifer. And MODPATH and MT3DMS computer codes were used to simulate particle and mass transport in the study area. MODPATH is a particle-tracking post-processing model that computes flow paths using output from MODFLOW (Pollock, 1994). And MT3DMS is a modular 3-D multi-species transport model for simulation of advection, dispersion, and chemical reactions of contaminants in groundwater systems (Zeng and Wang, 1999).

4.2.1.1. Theoretical framework

MODFLOW is based on the general equations of groundwater flow derived from the physical principles of Darcy's Law and fluid mass balance. Darcy's Law describes the
volume rate of water flow transported through the aquifer area. It is defined as the flow per unit cross sectional area of the porous medium. Can be expressed as:

\[ Q_i = -K_i A \frac{\partial h}{\partial i} = -K_i h \frac{\partial h}{\partial i} \quad (4.8) \]

where \( K \) is the soil hydraulic conductivity (m/d), \( A \) is the outflow area (m²), \( h \) is the hydraulic head or aquifer's depth (assuming the change of the hydraulic head along flow lines is very slow) (m) and \( \partial h/\partial i \) is the hydraulic gradient (m/m). \( A = 1 \cdot h \), area per unit width.

The Principle of fluid mass balance for the entire aquifer states that the amount of water into the system must equal the amount of water out of the system plus the change in amount of water over a period. Assuming steady-state, the change in amount of water inside the system is 0. In unconfined aquifers any change in the groundwater flow over the section \( dx \cdot dy \) is equal to the recharge from precipitation. And assuming the Dupuit hypothesis, which states that all equipotential lines are vertical and that the velocity is constant along the vertical, thus the hydraulic head is independent of the vertical direction, \( \partial h/\partial z = 0 \). The fluid mass balance is written as:

\[ Q - \left( Q + \frac{\partial y}{\partial x} dx \cdot dy \right) + N dx \cdot dy = 0 \quad (4.9) \]

Or

\[ \frac{\partial Q}{\partial x y} = N \]

where \( Q \) is the flow (m³/d) and \( N \) is the recharge (m³/d).

Assuming transient state, the difference between the inputs and outputs in time must be equal to the volume of water stored in:

\[ -\frac{\partial q_x}{\partial x} - \frac{\partial q_y}{\partial y} + N = S_y \frac{\partial h}{\partial t} \quad (4.10) \]

where \( S_y \) is the specific yield and \( t \) is the time (d). In unconfined aquifers the saturated thickness changes as the water table does. The specific yield is the percentage of voids within porous medium that store or release water without retention.

From the combination of (4.9) and (4.10) the 2D transient groundwater flow is described by (assuming that the velocity vector is horizontal):

\[ \frac{\partial}{\partial x} \left( K_x h \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y} \left( K_y h \frac{\partial h}{\partial y} \right) + N = S_y \frac{\partial h}{\partial t} \quad (4.11) \]
MODPATH uses the average linear groundwater velocity (Equation 4.24) to calculate the spatial and temporal evolution of the contaminants. Since the aquifer is a porous media, the water must move through the pores faster than the flux. Hence, this speed \( v \) is calculated by dividing Darcy's velocity, \( q \) (m/d), by the effective porosity of the sediments \( \theta \).

MT3DS is based on the three major classes of contaminant transport advection, dispersion/diffusion, and chemical reactions. The 3-D transport in a transient groundwater system is described by the following partial differential equation (Zeng and Wang, 1999):

\[
\frac{\partial \theta C}{\partial t} = \frac{\partial}{\partial x_i} \left( \theta D_{ij} \frac{\partial C}{\partial x_j} \right) - \frac{\partial}{\partial x_i} \left( \theta v_i C \right) + q_s C_s + \sum R_n \tag{4.12}
\]

where \( C \) is dissolved concentration of the species (M/L^3), \( D \) is the hydrodynamic dispersion coefficient tensor (m), \( v_i \) is the average linear groundwater velocity (m/d), \( q_s \) is the volumetric flow rate per unit volume of aquifer (1/d), \( C_s \) is the concentration of the source or sink flux for species (ML^3) and \( \sum R_n \) is the chemical reaction term.

### 4.2.2. Model setup

To model the groundwater flow of aquifer, one 2D grid layer covering the study area was constructed. It is oriented N-S, covering 1.37 km^2. It has 59 rows and 62 columns, being the discretization 25 m in both directions.

In the model the time simulated was from June 2014 to April 2015. The period was divided into 11 stress periods, corresponding to each month, which in its turn were divided into 10 time steps. For each month (or stress period) it was used a different data set of boundary conditions (north border and river stage) and precipitation recharge. The model outcome was compared with the head observations measured with the continuous measurements.

#### 4.2.2.1. Geometry

The topography of the area was implemented with the digital terrain model (DTM) of the area from the Norwegian mapping authority (Kvartverket, 2014) (Figure 4.17). The bottom elevation of the aquifer, interpreted as the impermeable bedrock layer, was defined using the results from the seismic refraction profiles combined with the top elevation.
The initial and end location of the three seismic profiles and their depth was plotted as 2D scatter data in the study area map. Due to the scarce data on the northern part of the area, two points were defined from the groundwater wells database (GRANADA) maintained by NGU and one point was added in the northernmost part, close to well M1, with a depth of 15 m as most probable thickness of the materials (Figure 4.18a). The data proceeding from seismic and wells were interpolated by the natural neighbor’s method. The resulting map of the thickness of the aquifer was subtracted to the top elevation map to get the bottom elevation of the aquifer (Figure 4.18b).
4.2.2.2. Boundary conditions

The contact of the western (Gorbekeken creek) and eastern borders with the aquifer was defined as impermeable, no-flow boundary (Neumann condition). The northern and southern (Folla river) borders were represented by considering the regional inflow and the river, respectively, as fixed head border conditions (Dirichlet condition).

Northern border

The hydraulic head of the northern border was defined based on the groundwater level in well M1, as it is the only well available in the northern part of the area. Assuming that the configuration of the groundwater table commonly is a smooth reproduction of the land-surface topography (USGS, 2013) the water table from M1 was subtracted to the topography and that value was used to interpolate the average monthly hydraulic head for all the points created along the northern border (Figure 4.19).

\[ \text{Hydraulic head} (\text{m}) \]

\[ 746.4 \quad 746.6 \quad 746.8 \quad 747.0 \quad 747.2 \quad 747.4 \quad 747.6 \quad 747.8 \quad 748.0 \quad 748.2 \]

\[ 30/06/2014 \quad 31/07/2014 \quad 31/08/2014 \quad 30/09/2014 \quad 30/10/2014 \quad 30/11/2014 \quad 30/12/2014 \quad 31/01/2015 \quad 28/02/2015 \quad 31/03/2015 \quad 30/04/2015 \]

**Figure 4.19:** Hydraulic head in the westernmost point during the period simulated, June 2014-April 2015.

Folla border

To simulate the monthly water level fluctuations along the hydrological year in Folla river, 16 GPS points taken along the river bed the 5\textsuperscript{th} of December 2014 to create a topographic profile of the river, in combination with the water level daily data from the Grimsmoen station (Figure 4.20) maintained by NVE 6 km downstream from Follidal were used.
To obtain the elevation of the water level in Folla, the river water level measured at Grimsmoen station on the 5th of December (0.99 m) was subtracted to the river topography to find its bottom elevation. Second, the monthly average water level was calculated from Grimsmoen station recorded data and it was summed to the bottom elevation of each GPS point (Table 4.4).

<table>
<thead>
<tr>
<th>Point</th>
<th>UTM Coordinates</th>
<th>Bottom elevation (m)*</th>
<th>Point</th>
<th>UTM Coordinates</th>
<th>Bottom elevation (m)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>550748.3 6889419.05</td>
<td>694.69</td>
<td>9</td>
<td>551000.69 6888906.03</td>
<td>692.52</td>
</tr>
<tr>
<td>2</td>
<td>550768.84 6889393.65</td>
<td>694.66</td>
<td>10</td>
<td>551040.08 6888855.71</td>
<td>692.11</td>
</tr>
<tr>
<td>3</td>
<td>550786.03 6889371.78</td>
<td>694.55</td>
<td>11</td>
<td>551118.29 6888767.72</td>
<td>691.63</td>
</tr>
<tr>
<td>4</td>
<td>550805.714 6889271.33</td>
<td>694.37</td>
<td>12</td>
<td>551163.73 6888729.25</td>
<td>691.03</td>
</tr>
<tr>
<td>5</td>
<td>550818.69 6889158.17</td>
<td>693.79</td>
<td>13</td>
<td>551229.79 6888718.97</td>
<td>690.76</td>
</tr>
<tr>
<td>6</td>
<td>550832.96 6889099.45</td>
<td>693.8</td>
<td>14</td>
<td>551373.28 6888688.64</td>
<td>689.99</td>
</tr>
<tr>
<td>7</td>
<td>550870.09 6889034.96</td>
<td>693.22</td>
<td>15</td>
<td>551551.4 6888715.08</td>
<td>688.91</td>
</tr>
<tr>
<td>8</td>
<td>550919.36 6888982.58</td>
<td>692.99</td>
<td>16</td>
<td>551656.6 6888751.42</td>
<td>688.74</td>
</tr>
</tbody>
</table>

Table 4.4: Location of the GPS river points and the bottom elevation of the river for each point.

* Obtained from subtracting the recorded water level in Grimsmoen station the 5th of December was 0.99.
Drains

The drainage system in the study area was mapped in the field. It was represented in the model with the drain boundary condition package (Figure 4.21). The drain elevation was set to topography the cell top elevation with a high conductance value of 1000 m²/d, based on Batelaan and De Smedt (1998). This way, when the water table rises above the ground surface it will be drained.

![Figure 4.21: Drain system represented in the model.](image)

4.2.2.3. Recharge

The recharge, the portion of infiltrated water that reaches the aquifer due to vertical downward flux from the unsaturated zone (Kresic, 2007), was calculated according to the following equation (Kresic, 2007):

\[
\text{Recharge} = \text{Precipitation} - \text{Surface Runoff} - \text{Evapotranspiration} \quad (4.13)
\]

The precipitation and evapotranspiration were assumed to be the constant in the whole study area but the surface runoff was considered to depend on the topographical slope. The high slopes reduce the water infiltration, increasing the surface runoff (Martínez de Azagra, 2006). Therefore, two zones with differentiate recharge were defined, one covering the flat area from the river to the road RV-29 and the second one covering the hillside from the road up to the mine.

To estimate the recharge in both areas the slope of the hillside was calculated by using two points along its profile. A slope of 9.5% was calculated. Values of surface runoff coefficient
based on the slope gradient and the type of soil/vegetation were selected based on the compilation of tables made by Martínez de Azagra (2006). An average value of 0.48 and 0.1 were estimated for zone 1 and zone 2, respectively (Table 4.5). The amount of surface runoff was calculated from applying the surface runoff coefficient to the monthly precipitation measured in Folldal meteorological station.

<table>
<thead>
<tr>
<th>Surface runoff Coefficient</th>
<th>Zone 1 Slope 9.5 %</th>
<th>Zone 2 Slope &lt; 1%</th>
</tr>
</thead>
<tbody>
<tr>
<td>No vegetation</td>
<td>0.7</td>
<td>-</td>
</tr>
<tr>
<td>Light vegetation</td>
<td>0.25</td>
<td>0.15</td>
</tr>
<tr>
<td>Dense vegetation</td>
<td>-</td>
<td>0.05</td>
</tr>
<tr>
<td>Mean</td>
<td>0.48</td>
<td>0.1</td>
</tr>
</tbody>
</table>

Table 4.5: Surface runoff coefficients used to calculate the recharge, values extracted from Martínez de Azagra (2006).

The evapotranspiration was estimated using Turc equation (Turc, 1961):

$$ET(\text{mm/yr}) = \frac{N}{\sqrt{0.094 + \frac{N}{(300 + 25T + 0.05T^3)^2}}}$$ (4.13)

where $N$ is annual precipitation (mm/yr) and $T$ is the average yearly temperature (°C). Since this equation gives an annual evapotranspiration, the % of ET related to the annual precipitation (hydrological year 2014-2015) was used to calculate the monthly evapotranspiration. The monthly surface runoff and evapotranspiration estimated were subtracted to the monthly precipitation. For the winter months December, January and February recharge was neglected, precipitation was assumed as snow (Figure 4.22).

Figure 4.22: Recharge infiltration to the aquifer calculated for the period simulated, from June 2014 to April 2015.
4.2.2.4. Hydraulic conductivity

Six zones with different hydraulic conductivity were selected (Figure 4.23) according to the soil map distribution (Figure 3.5) with the following exceptions (1) the Eastern mine tailings were delimited as one zone, (2) the anthropogenic material on the northern part was neglected and considered as glacial deposits, (3) the organic material was neglected assuming it to be a thin layer, was included as part of fluvial deposits zone.

Figure 4.23: The six zones with differentiate hydraulic conductivity. Zone 1 and 2 correspond to thick moraine, zone 3 to glacial deposits, zone 6 to fluvial deposits, and zone 4 and 6 to mine tailings.

The hydraulic values specified were found by calibrating the model. The range of values for each zone used (0.001-10 m/d) was selected from the slug test results (5.4) and the typical values for geological material (Table 4.3).

4.2.2.5. Calibration and sensitivity analysis

A sensitivity analysis was performed to test the different responses of the aquifer to changes in the parameters with higher uncertainty: hydraulic conductivity and specific yield. The see how much influence each parameter had on the model outcome, one parameter was kept constant while the other one was changed. Changes in hydraulic conductivity were made using slug test results interval (0.001-10 m/d), and in specific yield using literature range of values (0.009-0.35) from Kresic (2007).
The calibration of the model was performed manually by "trial-and-error" comparing the computed heads with the data collected from the 5 observation wells (M1, A1, A4, C2 and C7) with pressure dataloggers installed in the aquifer. The comparison was done by visual agreement and by quantifying the mean absolute error between the computed solution and the observed values. The mean absolute error ($\bar{e}$) is calculated by the following formula:

$$\bar{e} = \frac{\sum|h_o - h_c|}{n} \quad (4.14)$$

where $h_o$ is the observed hydraulic head (m) and $h_c$ is the computed hydraulic head (m).

4.2.3. Contaminants transport scenario

To obtain a preliminary knowledge of the evolution and distribution of contaminants in the aquifer in order to assess the construction of a limestone barrier, a tracking particle solution was applied and an advection-dispersion mass transport simulation was run. Three contamination focus zones were defined (Figure 3.1): the Main mine area, the West tailings and the East tailings; in order to observe the differences in contamination mobility and transport.

In the advection-dispersion mass transport simulation the diffusion coefficient was $10^{-9}$ m$^2$/s as typical value for major constituents dissolved in groundwater constituents (Kresic, 2007), the longitudinal dispersivity was defined as 10% of the length (0.25 for each cell) based on Appelo and Postma (2005), and the porosity selected was 0.38 based on the value for mine tailings from Pabst et al. (2014). In this type of simulation MT3DMS estimates the distribution and evolution of the contaminant plume from the amount of contaminant without considering the type of contaminant. Therefore, to simulate the mass transport during the transient period June 2014 to April 2015, the amount of Total Dissolved Solids (TDS), 2800 ppm, from the acid mine drainage creek sample from June 2014 was defined as initial contaminant concentration infiltrating the aquifer. The TDS of the sample was estimated from the EC using the following equation (Iyasele and Idiata, 2015):

$$TDS \ (ppm) = 0.64 \times EC \ (\mu S/cm) \quad (4.15.)$$
4.3. Geochemical simulations

From the data obtained during the field investigations to assess the ongoing conditions of acidification and metal leaching in the study area in Folldal, three geochemical and mass transport models were developed and simulated using PHREEQC code (Parkhurst and Appelo, 1999, version 3) with the standard set of thermodynamic data in the default phreeqc.dat database file.

The three models were developed to include all the three basic stages on acid mine drainage formation and treatment: (1) presence of reactant minerals and driven conditions, (2) formation and (3) remediation. To simulate the specific conditions observed in Folldal were used groundwater and soil data from the Eastern mine tailings area, as found to be the best for representing the spatial and temporal distribution differences related to acidification and heavy metals contamination in the study area.

PHREEQC is a computer program used to simulate one-dimensional calculations of different chemical reactions and processes in water (Parkhurst and Appelo, 2013), commonly used in geochemical transport modelling of acid mine drainage (Chen et al., 1999; Hoth et al., 2001; Nicholson et al., 2003; Eary and Williamson, 2006; and Papassiopi et al., 2014). In the input file, the data is specified via keywords and associated data blocks. The database contains the chemical definitions and constants needed to perform the calculations.

4.3.1. Reactant phases

To identify the natural geochemical reactions and phases involved in the formation of AMD by oxidation of pyrite in the mine tailings an inverse geochemical model was implemented. The inverse modelling PHREEQC capability was used to quantify the changes in the mineral phases involved in the process along the water flow path in the study area which caused the infiltrating precipitation water (initial solution) to change to the observed groundwater composition (final solution). An example of the input file is given in appendix B1.

4.3.1.1. Inverse modeling code

The inverse modeling is designed to determine the geochemical reactions occurring from the compositional differences between two aqueous solutions. From an initial solution the program balances the concentration of the aqueous species and quantifies the mole transfers
of the minerals and gases that yield to the composition of the second aqueous solution; resulting in one or several possible models to choose from.

4.3.1.2. Model setup

In the inverse modelling the two solutions and phases required were defined based on the soil and water chemistry data collected by NGI in July 2014 and literature data. The initial solution composition was defined from rainwater chemistry in Kiruna station, Sweden, listed in Appelo and Postma (2005) with the following exceptions:

1. the solution was brought to equilibrium with atmospheric O$_2$(g)
2. alkalinity and equilibrium with atmospheric CO$_2$(g) were neglected, in order to simulate the in situ conditions, measurements of groundwater and soil chemistry showed non presence of carbon
3. sulfate concentration was charge balanced
4. temperature was set to 2.2°C, which is the mean annual temperature measured in Folldal.

The final solution was specified using the groundwater hydrochemical concentration from well C6. It is considered to be a representative sample of in situ groundwater acidified by pyrite oxidation. The aqueous species used were only the major elements included in the phases selected: Na$^+$, K$^+$, Mg$^{2+}$, Ca$^{2+}$, Cl$^-$, SO$_4^{2-}$, Fe$^{2+}$ (Table 4.6).

The reacting phases involved in the formation of acidification in the mine tailings (Table 4.6), were based on the pyrite oxidation reactions (Section 2.1.1) and the geological study of Folldal area by Bjerkgård and Bjørlykke (1994) (Section 3.2.). Pyrite, O$_2$(g), K-mica, chlorite, anorthite and daphnite (as chlorite with iron content) were specified as dissolving species; and Fe(OH)3, secondary mineral of pyrite oxidation, as precipitating phase. Gypsum was specified with no restrains, since can be present in the soil and may precipitate as secondary mineral. Kaolinite was specified as a phase with no restrains, although is not likely to precipitate in those environments but was needed to balance the silicon. Presence of halite was assumed as possible fossil seawater in the glacial sediments. CO$_2$ and alkalinity were not considered in the model, no carbon or carbonates were measured in the groundwater (alkalinity 0) or the soil (TIC 0.6%).
Table 4.6: Solution compositions and reacting phases on the inverse modelling simulation.*Precipitation. ** Listed in Appelo and Postma (2005)*** From NGI (2014)

<table>
<thead>
<tr>
<th></th>
<th>Rainwater** (µmol/L)</th>
<th>Groundwater*** (µmol/L)</th>
<th>Phases</th>
</tr>
</thead>
<tbody>
<tr>
<td>$T$ ($^\circ$C)</td>
<td>2.2</td>
<td>7.5</td>
<td>O$_2$(g)</td>
</tr>
<tr>
<td>pH</td>
<td>5.6</td>
<td>4.5</td>
<td>Pyrite [FeS$_2$]</td>
</tr>
<tr>
<td>$Ca^{2+}$</td>
<td>16</td>
<td>4948</td>
<td>Anorthite [CaAl$_2$Si$_2$O$_8$]</td>
</tr>
<tr>
<td>$Mg^{2+}$</td>
<td>5</td>
<td>3481</td>
<td>K-mica [KAl$_3$Si$_2$O$_10$(OH)$_2$]</td>
</tr>
<tr>
<td>$Na^+$</td>
<td>13</td>
<td>255.3</td>
<td>Chlorite [Mg$_3$Al$_2$Si$_3$O$_10$(OH)$_6$]</td>
</tr>
<tr>
<td>$K^+$</td>
<td>5</td>
<td>190</td>
<td>Daphnite [Fe$_5$Al$_3$Si$_3$O$_10$(OH)$_8$]</td>
</tr>
<tr>
<td>$Cl^-$</td>
<td>11</td>
<td>145.2</td>
<td>Halite [NaCl]</td>
</tr>
<tr>
<td>$SO$_4$²⁻</td>
<td>31.1</td>
<td>10840</td>
<td>Gypsum [CaSO$_4$·2H$_2$O]</td>
</tr>
<tr>
<td>$Fe_{tot}$</td>
<td>-</td>
<td>2385</td>
<td>Kaolinite [Al$_2$Si$_2$O$_5$(OH)$_4$]</td>
</tr>
<tr>
<td>$Si$</td>
<td>-</td>
<td>3550</td>
<td>Fe(OH)$_3$</td>
</tr>
<tr>
<td>$O_2$(g)</td>
<td>865</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>

4.3.2. AMD formation

To estimate the formation of AMD from pyrite oxidation and its spatial and temporal evolution in the mine tailings, a one-dimensional (vertical) kinetic-mass transport geochemical model was established. The kinetics PHREEQC capability was used to simulate the kinetically controlled pyrite oxidation (formation of AMD) and the transport was used to quantify the changes in groundwater composition and mineral phases in a one-dimensional column with time (Figure 4.24). An example of the input file is given in appendix B2.

![Conceptual model of the acid mine drainage formation and transport in a 1D vertical column.](image)

Figure 4.24: Conceptual model of the acid mine drainage formation and transport in a 1D vertical column.
4.3.2.1. Kinetics code

The kinetics keyword permits the user to specify rate expressions for modelling a kinetically controlled reaction of a solid or solute, such as pyrite. The aqueous oxidation of pyrite (FeS$_2$) involves different kinetic reactions and can be driven by molecular oxygen (O$_2$) or ferric iron (Fe$^{3+}$), depending on the pH and bacteria presence. The two pathways are described by:

\[
\text{FeS}_2 + \frac{15}{4} \text{O}_2 + \frac{7}{2} \text{H}_2\text{O} \rightarrow \text{Fe(OH)}_3 + 2\text{SO}_4^{2-} + 4\text{H}^+ \quad (4.16)
\]

and

\[
\text{FeS}_2 + 14\text{Fe}^{3+} + 8\text{H}_2\text{O} \rightarrow 15\text{Fe}^{2+} + 2\text{SO}_4^{2-} + 16\text{H}^+ \quad (4.17)
\]

The reaction between pyrite and O$_2$ is slow compared to the oxidation by Fe$^{3+}$, which yields a low pH faster (Appelo and Postma, 2005). Both are kinetically controlled reactions which produce changes in the chemicals concentration depending on their rate expressions.

The rate expressions for pyrite ($R_{\text{pyrite}}$) were defined according to the general rate equation from Parkhurst and Appelo (1999):

\[
R_{\text{pyrite}} = \frac{r_{\text{pyrite}} A_0}{V (m_{\text{pyrite}}/m_{\text{pyrite}})^n} \quad (4.18)
\]

where $r$ is the specific rate (mol/m$^2$/s), $A_0$ is the initial surface area of pyrite (m$^2$), $V$ is the amount of solution (kgw), $m_0$ is the initial moles of pyrite, $m$ is the moles of pyrite at a given time, and $n$ is a factor to account for changes in the ratio $A_0/V$ during dissolution.

4.3.2.2. Transport code

The transport code is capable to model one-dimensional transport processes and to incorporate in the simulation all type of geochemical process, as kinetic reactions. The 1D transport of a chemical is described in PHREEQC (Parkhurst and Appelo, 1999) by the advection-reaction-dispersion (ARD) transport equation:

\[
\frac{\partial c}{\partial t} = -\nu \frac{\partial c}{\partial x} + D_L \frac{\partial^2 c}{\partial x^2} - \frac{\partial q}{\partial t} \quad (4.19)
\]
where $C$ is total dissolved concentration in water for an element (mol/kgw), $t$ is time (s), $v$ is pore water flow velocity (m/s), $x$ is distance (m), $D_L$ is the hydrodynamic dispersion coefficient (m$^2$/s) and $q$ is concentration in the solid phase (expressed as mol/kgw in the pores). $D_L = D_e + \alpha_L$, where $D_e$ is effective diffusion coefficient (m$^2$/s), and $\alpha_L$ is dispersivity (m).

### 4.3.2.3. Model setup

To model the formation and the vertically transport of AMD in the mine tailings, one-dimensional (vertical) column was defined. It has a length of 10 m and 10 cells, each of which has the same length. Rainwater was defined to entry the column, which was infilled with groundwater with pyrite and K-mica. The simulation parameters were defined using the soil and water chemistry data collected by NGI in July 2014, and the groundwater hydrochemistry properties were based on the collected in October 2014 and literature data.

**Solutions**

The inlet solution composition was defined from the rainwater mention in the section above, with the following exception, the solution was brought to equilibrium with atmospheric O$_2$ and CO$_2$ (g) (Table 4.7).

The initial groundwater solution in the column was based on well C7 groundwater chemistry from October 2014 (Table 4.7), with Na$^+$ concentration charge balanced to improve the electrical balance, considered to be representative in situ non-polluted groundwater. It was specified to be in conditions of equilibrium with the mineral phase K-mica to represent the composition of the soil (besides pyrite), and with goethite and gypsum as probable secondary mineral to precipitate. The initial amount of K-mica in the column was based on the soil sample F8 K concentration listed in NGI (2014), assuming that all the K measured comes from K-mica (Table 4.8).
Rainwater * (µmol/L) | Groundwater ** (µmol/L)  
---|---
 Temp | 2.2 | 4.4  
 pH | 5.6 | 6.8  
 Ca²⁺ | 16 | 1831  
 Mg²⁺ | 5 | 206.1  
 Na⁺ | 27 | 419.2  
 K⁺ | 5 | 177.1  
 Cl⁻ | 11 | 69.41  
 SO₄²⁻ | 21 | 290.1  
 Alkalinity | 21 | 4000  
 CO₂(g) | 49.81 | -  
 O₂(g) | 865 | -  

**Table 4.7**: Rainwater and groundwater solution compositions used for calculations.  
‘Listed in Appelo and Postma (2005) **From NGI (2014)

**Kinetic reaction**

The amount of reactant pyrite in the mine tailings was defined from the S concentration (mol/ton) of soil sample F8 (Table 4.8). The mass of pyrite dissolving in PHREEQC had to be entered as molar amount (mol/l), hence it was converted by the following equation:

\[
\text{Pyrite} \left( \frac{\text{mol}}{l} \right) = \frac{\text{mol}_{\text{pyrite}}}{t} \times \rho_b \times \frac{1 \text{ m}^3}{10^3 \text{ l}} \tag{4.20}
\]

where \(\rho_b\) is the bulk density (t/m³). \(p_b = \rho_s \times (1 - \theta)\), where \(\theta\) is porosity and \(\rho_s\) is the density of the particle (t/m³) (Table 4.8).

**Table 4.8**: Soil composition minerals concentration and related soil parameters.  
* Values from Pabst et al. (2014). **Values from NGI (2014).
The rate expressions of pyrite oxidation were defined according to following equations:

\[ r = 10^{-8.19}m_{O_2}^{0.5}m_{H^+}^{-0.11} \] (4.21)

\[ r = 6.3 \times 10^{-4}m_{Fe^{3+}}^{0.92}(1 + \frac{m_{Fe^{2+}}}{10^{-6}})^{-0.43} \] (4.22.a)

\[ r = 1.9 \times 10^{-6}m_{Fe^{3+}}^{0.28}(1 + \frac{m_{Fe^{2+}}}{10^{-6}})^{-0.52}m_{H^+}^{-0.3} \] (4.22.b)

where \( r \) is in (mol/m\(^2\)/s) and \( m \) is the concentration (molality). Equation (4.21) is oxidation by \( O_2 \) based on Williamson and Rimstidt (1994); Equation (4.22) is oxidation by \( Fe^{3+} \) when oxygen is absent (a) and when oxygen is present (b) based on Appelo and Postma (2005) (modified from Williamson and Rimstidt, 1994). Rates (A and b) are limited to pH range 0.5-3.0, since for pH >3.0 goethite (FeOOH) precipitates.

The initial surface area of pyrite, \( A_0 \), was estimated from the following equation for geometric surface area, assuming that the pyrite grains are cubic shape, from Eary and Williamson (2006) (based on Hodson, 2002):

\[ A_0 = \frac{2[(ab)+(bc)+(ac)]}{(ab)c + \rho} \] (4.23)

where \( a, b \) and \( c \) are the dimensions of the particle and \( \rho \) is the density of pyrite. It was assumed a grain size of 0.01 µm, based on Appelo and Postma (2005) for aquifer sediment containing pyrite, and 5.01 g/cm\(^3\) pyrite density.

**Transport**

The transport parameters (Table 4.9) were defined for one-dimensional column with advection-dispersion transport. As fixed values: diffusion coefficient, dispersivity, pore water velocity, cells geometry and constant boundary conditions at the first cell and flux at the last one. The dispersivity of the aquifer was defined as the 10% of the traveled distance, for each cell, based on Appelo and Postma (2005). The pore water vertical velocity \( (v) \) was calculated from the field data, using equation:

\[ v = -\frac{K}{\theta}i \] (4.24)

where \( K \) is the vertical hydraulic conductivity – specified as 1/10 of the average measured in well C5 and C6 in June 2014 (m/d), \( \theta \) is porosity of mine tailings from Pabst et al. (2014) and \( i \) is the the vertical hydraulic gradient calculated from wells C5 and C6 in the study area. In PHREEQC the velocity is defined by the length of the cell divided by the time steps, therefore the length of the cell (1m) was divided by the vertical water velocity to find the number of time steps.
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diffusion coefficient, $D_e$ (m$^2$/s) *</td>
<td>$10^{-9}$</td>
</tr>
<tr>
<td>Dispersivity, $\alpha_L$ (m) (per cell)</td>
<td>0.1</td>
</tr>
<tr>
<td>Vertical pore water velocity, $v$ (m/s)</td>
<td>$8.17 \times 10^{-7}$</td>
</tr>
<tr>
<td>Vertical hydraulic conductivity, $K$ (m/s)</td>
<td>0.561</td>
</tr>
<tr>
<td>Vertical hydraulic gradient, $i$</td>
<td>0.048</td>
</tr>
<tr>
<td>Time steps</td>
<td>$1.22 \times 10^6$</td>
</tr>
</tbody>
</table>

Table 1.9: Transport parameters. * Value from Kresic (2007).

4.3.3. Remediation

To estimate the efficiency of a limestone drain to remove heavy metals from the groundwater in the mine tailings, a simple one-dimensional (horizontal) geochemical mass transport model with presence of limestone as exchanger was established. The exchanger PHREEQC capability was used to simulate the ion-exchange reactions occurring in the limestone drain when acidified water flows through the carbonate barrier. The surface capability was used to simulate the binding of heavy metals to the surface of the hydrous ferric oxide precipitating when the contaminated groundwater is neutralized by the carbonate. Normally, ion exchange can describe most of the sorption of cations in aquifers, however surface complexation is needed for heavy metals that sorb very strongly, like Cu$^{2+}$ (Appelo and Postma, 2005). The transport was used to determine the durability of the drain system (Figure 4.25). An example of the input file is given in appendix B3.

![Figure 4.25: Conceptual model of the limestone drain system in the mine tailings.](image-url)
4.3.3.1. Exchange code

The exchange code is design to model ion-exchange equilibria reactions, which entail substitution of one chemical for another one at the solid surfaces in the soil. Neutralization of acid waters occurs when carbonate minerals, such as calcite (CaCO$_3$), are present in the system due to their high capacity to buffer acidity, according to the reaction:

$$\text{CaCO}_3 + 2\text{H}^+ \rightarrow \text{Ca}^{2+} + \text{CO}_2 \text{(aq)} + \text{H}_2\text{O} \quad (4.25)$$

The ion-exchange equilibria, among the acidified groundwater and calcite, is simulated in PHREEQC by mass-action and mole-balance equations for exchange sites. The activity of the exchange ($a_{ie}$) is calculated using the equation (Pakhrust and Appelo, 1999):

$$a_{ie} = \gamma_{ie} \frac{b_{e,ie} n_{ie}}{T_e} \quad (4.26)$$

where $\gamma_{ie}$ is activity coefficient, $b_{e,ie}$ is the number of equivalent of exchanger $e$, $n_{ie}$ is moles of exchange specie $i_e$ (calcite) and $T_e$ is the total number of exchange sites for the exchanger (in equivalents).

4.3.3.2. Surface code

The surface complexation code is design to simulate the sorption of metal ions on hydrous ferric oxide. The sorption in PHREEQC is simulated by using defined constant surface complexation reactions of the elements for two sites on hydrous ferric oxide, a strong and a weak site. For calculating the surface charge it is required to define the amount and composition of each surface: the surface area of the solid (m$^2$/kg), the number of sorption sites (sites/nm$^2$) and the sorption density (mol/kg) (Pakhrust and Appelo, 1999).

4.3.3.3. Model setup

To model the effectiveness of a limestone drain system in the mine tailings, one-dimensional (horizontal) column was defined. It has a length of 20m and 20 cells, each of which has the same length. Cell number 10 was defined as the drain. AMD effluent was defined to entry the column, which was infilled with groundwater in equilibrium with pyrite and K-mica. The simulation parameters were defined using the soil and water chemistry data collected by NGI in July 2014, the groundwater hydrochemistry collected in October 2014 and literature data.
Solutions

The inlet solution composition was defined from the hydrochemistry of the acid mine drainage stream sample V4 analyzed from NGI (2014) (Table 4.10), as representation of the direct leaching effluent from the mine tailings. The Ca$^{2+}$ was charge balanced to improve the electrical balance. Heavy metals Cu$^{2+}$ and Zn$^{2+}$ were added to the aqueous species specified, to investigate how different compositions and contaminants in water affect the exchange process.

The initial groundwater solution infilling the column was the same as in the previous section (Table 4.7), considered being representative in situ non-polluted groundwater. It was specified to be in conditions of equilibrium with the mineral phases pyrite and K-mica to represent the composition of the soil. Goethite and gypsum were defined as secondary minerals phases in equilibrium with the column, due to limestone drains use to increase the pH and yield iron (goethite) and sulfate (gypsum) precipitation.

<table>
<thead>
<tr>
<th></th>
<th>AMD (µmol/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temp</td>
<td>7</td>
</tr>
<tr>
<td>pH</td>
<td>2.6</td>
</tr>
<tr>
<td>Ca$^{2+}$</td>
<td>4200</td>
</tr>
<tr>
<td>Mg$^{2+}$</td>
<td>9089</td>
</tr>
<tr>
<td>Na$^+$</td>
<td>190.5</td>
</tr>
<tr>
<td>K$^+$</td>
<td>49.83</td>
</tr>
<tr>
<td>Cl$^-$</td>
<td>101.7</td>
</tr>
<tr>
<td>SO$_4^{2-}$</td>
<td>31780</td>
</tr>
<tr>
<td>Cu$^{2+}$</td>
<td>11159</td>
</tr>
<tr>
<td>Zn$^{2+}$</td>
<td>648.4</td>
</tr>
<tr>
<td>Fe$_{tot}$</td>
<td>14190</td>
</tr>
</tbody>
</table>

Table 4.10: Acid mine drainage effluent composition from NGI (2014).
Calcite was defined as exchanger and was brought to equilibrium with the specified groundwater in cell number 5. The amount of moles of calcite used was estimated using the following equation:

\[ \text{moles Calcite} = \%_{\text{calcite}} \times \rho_{\text{limestone}} \times M_{\text{calcite}} \]  \hspace{1cm} (4.27)

where \( \%_{\text{calcite}} \) is the fractional content of calcite in the limestone drain (g\text{pureCaCO}_3/g\text{limestone}), \( \rho_b \) is the limestone bulk density (g/ dm\(^3\)) and \( M \) is the molar mass of calcite (moles\text{CaCO}_3/g\text{CaCO}_3).

The fractional content and the bulk density values (Table 4.11) were estimated as the mean value of the values described in Cravotta and Watzlaf (2003) study, where the authors evaluated the design and efficiency of limestone drains for acid mine drainage remediation from construction characteristics and composition of influent and effluent data at 13 limestone drains in USA.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>% calcite (g\text{pureCaCO}_3/g\text{limestone})</td>
<td>88 %</td>
</tr>
<tr>
<td>Limestone bulk density, ( \rho_b ) (g/cm(^3))</td>
<td>1.756</td>
</tr>
<tr>
<td>Calcite molar mass, ( M ) (moles\text{CaCO}_3/g\text{CaCO}_3)</td>
<td>100.087</td>
</tr>
</tbody>
</table>

Table 4.11: Limestone drain composition parameters from Cravotta and Watzlaf (2003).

Surface complexation

The composition of the surface assemblage was defined by specifying the surface assemblage sites related to the goethite in the equilibrium phases in the limestone drain cell. Hence, the moles of goethite precipitating in the drain were used to determine the number of moles of surface sites. The surface area of the solid was defined as 600 m\(^2\)/g based on literature values from Appelo and Postma (2005).

Transport

The transport parameters dispersivity and diffusion coefficient were defined for one-dimensional column with horizontal advection-dispersion transport as in the previous section (Table 4.9). The pore water horizontal velocity used (1.97x10\(^{-6}\) m/s) was obtained from the results of the hydrogeological model (Table 5.17).
5. Results

5.1. Field studies

5.1.1. Seismic refraction

The processing of the profiles shows two distinct layers, the uppermost corresponding to the direct ray (in red) and the layer below it to the refracted ray (in green) (Figures 5.1, 5.2 and 5.3). Their interpretation, by the intercept time method, revealed that the upper layer is the sedimentary cover and the layer underneath is the igneous-metamorphic bedrock. The potential presence of a thin low-velocity layer on top, which is not detected in the refraction seismic directly, has been neglected; hence this gives a slight overestimation of the depth (Breivik, 2015).

![Seismic refraction profile](image)

**Figure 5.1**: Seismic refraction profile 1, the location is found in Figure 4.5. Note the sketch of the intercept time method, where $x_i$ and $t_i$ are used to calculate the velocity of the layer and $t_{int}$ is the intercept time.

According to the calculations, the velocity of the uppermost layer sediments is 1.7 km/s and the velocity of the layer below is 6.3 km/s in profile 1, with an uncertainty of ±0.2 km/s manually calculated by using different $x_i$ and $t_i$ values.
In profile 2, the velocity of the uppermost layer sediments is 1.4 km/s and the velocity of the layer below is 3.1 km/s, with an uncertainty of ±0.5 km/s.

In profile 3, the velocity of the uppermost layer sediments is 1.35 km/s and the velocity of the layer below is 3.9 km/s, with an uncertainty of ±0.3 km/s.
Based on the table of typical velocities for different materials (Table 4.1), the uppermost layer is interpreted as saturated sand and the lower layer to basalt, in profile 1, and slate, in profiles 2 and 3. Thus, the uppermost layer is the sedimentary cover and the lower layer is the bedrock. The results are in agreement with the geology of the area (Figure 3.4 and 3.6) and field observations, the surface was soil material and no exposed bedrock was seen.

The thickness of the bedrock varies between 30-42 m in the different profiles, being profile 2 the one with a thicker cover layer. The estimations of the sediment cover thickness were implemented into the numerical model geometry.

**5.1.2. Electrical Resistance Tomography**

The analysis and interpretation of the ERT profiles based on the real values of resistivity of the sediments shows similar range of resistivity between them but with differences in its distribution. In transect a, profiles 1, 2, 3 and 4 show a decrease in the resistivity of the sediments with depth and towards the river (from north to south) (Figure 5.4). The northern profiles (1, 2 and 3) present two different heterogeneous layers, one on top with resistivity values ranging from 200 to 1500 Ohm.m and one below with values from 5 to 70 Ohm.m. While the southern profile (4) only presents the lower resistivity layer. According to the table of typical resistivity values of the materials (Table 4.2) and the water table elevation (Figure 5.7), the low resistivity layer corresponds to saturated sediments while the medium resistivity layer corresponds to drier sediment. Moreover, looking at the location and depth of the wells in profiles 3 and 4, it seems realistic that the layer of lower resistance is caused by larger content water.

Transect b, profiles 1, 2 and 3, presents two distinctive layers and show an increase of resistivity towards the river (Figure 5.5). The top layer is homogenous and has resistivity values ranging from 2000 to 6000 Ohm.m. The layer below is heterogeneous and has values in the range of 70-1500 Ohm.m. According to the table of typical resistivity of the materials (Table 4.2) and the results in transect a, the high resistivity layer might correspond to dry sediments while the low-medium resistivity layer to more saturated sediments. Moreover, the groundwater level measured in well B3 is at 693.8 m.a.s.l. which fits with the elevation of the saturated layer.
Figure 5.4: ERT profiles 1, 2, 3 and 4 in transect A.

Figure 5.5: ERT profiles 1, 2 and 3 in transect B.
Profile 2, from transect c, also shows two distinct heterogeneous layers with resistivity decreasing with depth (Figure 5.6). The top 3m layer resistivity values range from 600 to 2000 Ohm.m and in the layer below the values are ranging from 20 to 600 Ohm.m. However, the crust layer with high resistivity disappears around 10 m close to the river, where the material presents low resistivity values. On the other hand, profile 1 presents only one heterogeneous layer with low resistivity values. According to the table of typical resistivity of the materials (Table 4.2) and the water table elevation, around 690 m.a.s.l., the high resistivity layer might correspond to dry sediments while the low-medium resistivity layer corresponds to saturated sediments.

These results highlight the heterogeneous distribution of the soil, not seen in the seismic results, and were further used to verify the hydraulic properties of the materials and the location of the water table.

5.1.3. Piezometer network
The wells were located creating a cross section in each of the two mine tailings exposed areas, with the previous PVC installed (Table 5.2), and the forest area in between (4.5). The distance between cross sections A and B was 390m and between B and C 670 m. During the drilling the water table was reached for each well, and many fine sediments were flushed in order to clean the pipes, however due to the medium size of the screen, fine particles infiltrated and deposited around and within the well screen.

The depth of the wells in transect A was 3 m, and the distance between the wells was 60 m. In transect B the depths of the wells ranged from 8.5 to 2.5, and from 3.5 to 1.5 for the wells.
in the piezometer nest BS. The distance between wells was 130m, and 1.5 and 0.3m for the wells in the piezometer nest. The depths of the wells in transect C ranged between 5.5 and 4.5m, and from 4 to 1.5 for the wells in the piezometer nest CS. The distance between wells was around 80m, and for the wells in the piezometer nest was 0.3 m (Table 5.1).

<table>
<thead>
<tr>
<th>Wells</th>
<th>UTM Coordinates</th>
<th>Z</th>
<th>Pipe length (m)</th>
<th>Well</th>
<th>UTM Coordinates</th>
<th>Z</th>
<th>Pipe length (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>A2</td>
<td>550909.2 688967.2</td>
<td>700.07</td>
<td>3.2</td>
<td>BS3</td>
<td>550985.3 6888921.1</td>
<td>693.7</td>
<td>2.08</td>
</tr>
<tr>
<td>A3</td>
<td>550862.9 6889432.1</td>
<td>698.65</td>
<td>3.2</td>
<td>CI</td>
<td>551609.7 6889058.6</td>
<td>697.8</td>
<td>6.2</td>
</tr>
<tr>
<td>B1</td>
<td>551122.0 6889138.3</td>
<td>706.52</td>
<td>9.2</td>
<td>C3</td>
<td>551658.0 6888975.7</td>
<td>693.84</td>
<td>4.7</td>
</tr>
<tr>
<td>B2</td>
<td>551054.1 6888995.5</td>
<td>699.15</td>
<td>6.2</td>
<td>C4</td>
<td>551660.0 6888896.3</td>
<td>692.76</td>
<td>4.7</td>
</tr>
<tr>
<td>B3</td>
<td>550991.1 6888930.2</td>
<td>695.77</td>
<td>3.2</td>
<td>CS1</td>
<td>551687.2 688753.1</td>
<td>690.43</td>
<td>4.1</td>
</tr>
<tr>
<td>BS1</td>
<td>550983.9 6888922.7</td>
<td>694.02</td>
<td>4.08</td>
<td>CS2</td>
<td>551655.3 6888753.0</td>
<td>690.38</td>
<td>3.1</td>
</tr>
<tr>
<td>BS2</td>
<td>550984.1 6888921.5</td>
<td>693.93</td>
<td>3.08</td>
<td>CS3</td>
<td>551655.0 6888752.9</td>
<td>690.75</td>
<td>2.1</td>
</tr>
</tbody>
</table>

Table 5.1: Coordinates and depth of the wells installed in June 2014.

<table>
<thead>
<tr>
<th>Wells</th>
<th>UTM Coordinates</th>
<th>Z (m)</th>
<th>Pipe length (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>M1</td>
<td>551607.5 688900.3</td>
<td>700.07</td>
<td>4.02</td>
</tr>
<tr>
<td>A1</td>
<td>550975.4 6889664.6</td>
<td>698.65</td>
<td>2.43</td>
</tr>
<tr>
<td>A4</td>
<td>550813.9 6889428.4</td>
<td>706.52</td>
<td>1.89</td>
</tr>
<tr>
<td>C2</td>
<td>551598.0 6889018.2</td>
<td>695.13</td>
<td>5.02</td>
</tr>
<tr>
<td>C5</td>
<td>551685.4 6888815.0</td>
<td>692.43</td>
<td>3.6</td>
</tr>
<tr>
<td>C6</td>
<td>551686.0 6888812.3</td>
<td>692.37</td>
<td>2.85</td>
</tr>
<tr>
<td>C7</td>
<td>551687.2 6888801.2</td>
<td>692.62</td>
<td>10.8</td>
</tr>
</tbody>
</table>

Table 5.2: Coordinates and depth of the PVC wells prior installed.

The cross section of the potential head distribution across the three different transects shows that the groundwater is flowing towards river Folla in a similar pattern; the hydraulic gradient is 0.03, 0.04 and 0.01 in cross section A, B and C, respectively (Figures 5.7). The differences between the water level profiles are due to the different topography of the areas with a higher slope in area A and a flat extension of more than 200 m for area C.
5.1.4. Groundwater level monitoring – seasonal changes

Punctual measurements

The punctual groundwater level measurements show, in general, a decrease of the water table elevation for the period from June until December 2014, and an increase tendency from December until April 2015. The water table in wells A1 and A3 in transect A, follows the general tendency observed, with ± 30 cm variation in A1, and ±1m in A3. While in well A2 the water table only decreased, in April was too low and it was not possible to measure it (Figure 5.8).

Figure 5.8: Change in groundwater level in cross section A. The water table shows a general tendency to decrease in autumn and winter, and increased in spring. Single manual measurements taken in June, October 5th, December 15th 2014 and April 22nd 2015.
The water table values measured in the wells in transect B (Table 5.3) revealed that the groundwater level has been decreasing since June in all the wells, ± 1m variation. Two data sets (December and April) of groundwater elevation from the piezometer nest wells is missing due to the wells were either covered by snow or frozen (Figure 5.9). The hydraulic head in the piezometer nest indicates that groundwater flows upwards, is discharging into Folla. The hydraulic head in the deeper piezometer (Bs1) is higher than in the shallower piezometer (Bs3).

<table>
<thead>
<tr>
<th></th>
<th>B1</th>
<th>B2</th>
<th>B3</th>
<th>BS1</th>
<th>BS2</th>
<th>BS3</th>
</tr>
</thead>
<tbody>
<tr>
<td>June</td>
<td>700.47</td>
<td>696.5</td>
<td>694.84</td>
<td>693.11</td>
<td>693.1</td>
<td>693.09</td>
</tr>
<tr>
<td>October</td>
<td>700.4</td>
<td>695.98</td>
<td>94.95</td>
<td>693.13</td>
<td>693.12</td>
<td>693.11</td>
</tr>
<tr>
<td>April</td>
<td>699.85</td>
<td>695.32</td>
<td>694</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

The water table in the wells in transect C seems to change differently according to the location and type of well (Figure 5.10). The hydraulic head in the northern wells C1 and C2 decreased following the same pattern, ±70cm variation. The water table in the central metal wells C3 and C4 decreased similarly, ±30cm, although it was not possible to measure the water table in well C3 in April. The groundwater level presents minimal changes, ± 20cm, and the same tendency in wells C5, C6, but in the deep well C7 the decrease during winter is larger, ± 40 cm variation. Likewise, in the shallower wells of the piezometer nest the hydraulic gradient changes with the same pattern, but in the deeper well, CS1, shows a higher decrease from June to October, ±10 cm variation. The hydraulic head in the piezometer nest also indicates that groundwater is flowing upwards, discharging into Folla; there are no seasonal measured changes on the flow direction.
Continuous monitoring

The continuous monitoring shows a decrease in the water table during autumn and winter and an increase in spring, except for well A4 where the groundwater level started to increase in mid-December (Figure 5.12). The groundwater level data from well C7 revealed two periods of decrease-increase. The water level increased a 5-10 cm in late December beginning of January, and then decreased constantly up to 20 cm until March when it rises again. The hydraulic head mean value is 689.91 m, being the maximum 690.1 m (28th December) and the minimum 689.79 m (7th March). The mean groundwater temperature is 4.33 °C, ranging from 4.8 °C (October) to 3.5 °C (April).

The groundwater level data from well C2 shows a constant linear diminish of the groundwater level from October until early April, when the level increases. The hydraulic head mean value is 692.22 m, being the maximum 692.81 m (16th October) and the minimum 681.93 m (6th April). The mean groundwater temperature is 3.98°C, ranging from 7.5°C (October) to 2.1°C (April).
The groundwater level data from well A1 presents higher daily variability than the other wells, and it has high peaks in November and in early April. The water level decreased from November to the beginning of February, and it started to increase smoothly in mid-February. These results may suggest that the well is connected to Gorbekken stream; the water level fall could be connected to freezing conditions, the level rise to positive temperatures and the high peaks to intense rainfall (Figure 5.12). The hydraulic head mean value is 709.82 m, being the maximum 710.14 m (10th April) and the minimum 709.82 m (28th January). The mean groundwater temperature is 2.16 °C, ranging from 5.8 °C (October) to 0.95°C (April).

The groundwater level data from well A4 shows the opposite behavior to the general trend. The hydraulic head is close to constant from October until mid-December, when it starts increasing constantly until mid-March and then slowly decreases. This strange behavior is due to that the water froze in mid-December and the diver installed measured the weight and variations of the ice. During the field survey in April, the first 60-80 cm of the water in the well were frozen (Figure 5.11). The mean groundwater temperature is 1.05°C, ranging from 5.5°C (October) to 0.28°C (March).

The groundwater level data from well M1 shows a constant linear decrease of the groundwater level from October until early February, when the level increases in a series of three steps. The hydraulic head mean value is 758.34 m, being the maximum 758.99 m (27th October) and the minimum 757.86 m (10th April). The mean groundwater temperature is 4.5°C, ranging from 6.5°C (October) to 2.9°C (April).
Figure 5.12: Seasonal changes in the groundwater level in wells M1, A1, A4, C2, C7 recorded from October 2014 until April 2015 by the divers installed, compared to the precipitation (mm) and temperature (°C).
5.1.5. Slug test-Hvorslev’s analysis method

The analysis of the manual and automatic slug test data, shows that both measurement techniques are comparable, which indicates accurate experimental procedure (Figure 5.13). In the well B2, for example, both measuring techniques show a slow start drawdown in the first 200 s (approximately 3 and a half minutes) and then a faster decrease. As time increases there is a minimal discrepancy (9.5µm) between the manual data and the data recorded by the diver, probably caused due to higher time gaps between manual measurements.

![Figure 5.13](image)

Figure 5.13: Plot of the logarithmic head $h/h_0$ versus the time since slug test initiation in well A2 and B2.

The hydraulic values estimated from the tests by Hvorslev’s method span three orders of magnitude, ranging from 0.01 to 10 m/day (Table 5.4). There is no relationship between hydraulic conductivity value and depth or location of the well, which indicates heterogeneity of the materials as seen in the soil map (Figure 3.5). The mean value is 1.04 m/d for whole the study area, and 0.43 m/d, 0.25m/d and 1.74 m/d for materials in the transects A, B and C, respectively. Results are subjected to error due to the particles accumulated and packed around the well screen, reducing the measured K of the surrounding area.

According to the table of typical hydraulic conductivity for different materials the hydraulic conductivity values found are in the ranges defined for glacial till, silt and sand. These values were used further in the numerical model.

<table>
<thead>
<tr>
<th>Well</th>
<th>M1</th>
<th>A1</th>
<th>A2</th>
<th>A3</th>
<th>A4</th>
<th>B1</th>
<th>B2</th>
<th>B3</th>
<th>BS</th>
</tr>
</thead>
<tbody>
<tr>
<td>K (m/d)</td>
<td>0.15</td>
<td>1.19</td>
<td>0.02</td>
<td>0.09</td>
<td>-</td>
<td>0.23</td>
<td>0.15</td>
<td>0.05</td>
<td>0.57</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Well</th>
<th>C1</th>
<th>C2</th>
<th>C3</th>
<th>C4</th>
<th>C5</th>
<th>C6</th>
<th>C7</th>
<th>CS</th>
</tr>
</thead>
<tbody>
<tr>
<td>K (m/d)</td>
<td>0.02</td>
<td>0.86</td>
<td>0.04</td>
<td>0.58</td>
<td>4.42</td>
<td>6.8</td>
<td>0.09</td>
<td>1.51</td>
</tr>
</tbody>
</table>

Table 5.4: Hydraulic values estimated from the slug tests using the Hvorslev’s method.
5.1.6. Groundwater discharge

From the values of the hydraulic conductivity, $K$, obtained with the slug test and the values of the hydraulic gradient from the hydraulic head measurements, the horizontal groundwater discharge to Folla River was theoretically estimated using Darcy’s Law equation (Equation 4.8).

In the discharge calculations, the mean hydraulic conductivity and hydraulic gradient of each transect were used. The cross sectional horizontal area was calculated as the aquifer depth estimated from the seismic profiles per unit width (Table 5.5). The discharge calculated from the hydraulic head in transect A and C, located in the Western and Eastern mine tailings, respectively, is two times higher than in transect B, located in the forest central area.

<table>
<thead>
<tr>
<th></th>
<th>Transect A</th>
<th>Transect B</th>
<th>Transect C</th>
<th>Mean Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic conductivity, $K$ (m/d)</td>
<td>0.6</td>
<td>0.2</td>
<td>1.79</td>
<td>0.86</td>
</tr>
<tr>
<td>Area, $A$ (m$^2$)</td>
<td>35</td>
<td>42</td>
<td>30</td>
<td>36</td>
</tr>
<tr>
<td>Hydraulic gradient, $i$</td>
<td>0.032</td>
<td>0.035</td>
<td>0.011</td>
<td>0.026</td>
</tr>
<tr>
<td>Discharge, $Q$ (m$^3$/d)</td>
<td>0.67</td>
<td>0.29</td>
<td>0.59</td>
<td>0.8</td>
</tr>
</tbody>
</table>

Table 5.5: Groundwater discharge to Folla River estimated from data collected in June 2014.

The seasonal changes of horizontal groundwater discharge were estimated, comparing values from June and December 2014 (Table 5.5 and 5.6). There is a decrease in both transects A and C, although the change in C is more abrupt than in A. Hydraulic head values in wells in transect B were not collected.

<table>
<thead>
<tr>
<th></th>
<th>Transect A</th>
<th>Transect C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic conductivity, $K$ (m/d)</td>
<td>0.6</td>
<td>1.79</td>
</tr>
<tr>
<td>Area, $A$ (m$^2$)</td>
<td>35</td>
<td>30</td>
</tr>
<tr>
<td>Hydraulic gradient, $i$</td>
<td>0.029</td>
<td>0.007</td>
</tr>
<tr>
<td>Discharge, $Q$ (m$^3$/d)</td>
<td>0.61</td>
<td>0.38</td>
</tr>
</tbody>
</table>

Table 5.6: Groundwater discharge to Folla River estimated from data collected in December 2014.
The mean discharge for the study area was also estimated in order to compare it with the computed value in the hydrogeological model. Considering that the study area comprises a section 1.5 km of river, and a mean discharge of 0.8m$^3$/d (Table 5.5), the total discharge from groundwater into Folla River is 1200 m$^3$/d.

5.1.7. Water quality chemical analysis

The major ions and heavy metals concentration and the physico-chemical properties vary considerably along the study area but are quite constant during the year. For example, SO$_4^{2-}$ varies between 7 (in well B3) and 12000 mg/L (in well M1), and Cu varies between 1 mg/L (in well CS2) and 120000 mg/L (in well M1). The pH values range from 2.2 to 8.8, the average EC is 1.4 mS/cm, the temperature is changing from 6.5 in summer to 0.5°C in spring. The water chemistry in the area can be classified into two chemical types: Ca-SO$_4$ type in the mine tailings wells and Ca-HCO$_3$ type in the forest area. Heavy metals concentration in groundwater is above drinking levels in almost all the wells in the mine tailings, and concentration of Cu in the river is above the limit established (10 µg/l) by the Norwegian Environment Directorate.

5.1.7.1. Physicochemistry properties

pH

Surface water

The pH in Folla River is ranging from 6.8 to 7.4 without a trend in its distribution along the river section or evolution during the sampling events. The pH in the AMD stream is very acidic, around 2.7 in June and October; AND 3.85 in April 2015. This increase is probably caused by the dilution of the acid drainage water and the snowmelt water (Figure 5.14).

Figure 5.14: Acid mine drainage flowing towards the river crossing area A covered by melting snow in April 2015.
**Groundwater**

The groundwater pH measured in the field average 6.5 in June and 6.1 in October and April. The minimal value is in well M1 (pH around 2.3), and the maximum values are in transect B wells, where pH is neutral (6.9-7). The plot of pH versus depth shows a clear trend for pH to increase with depth in transect A and C (Figure 5.15). The groundwater in the shallower wells in the mine tailings is acid while water in the deeper wells is circumneutral, expected since the shallower groundwater is directly affected by the sulfide oxidation process in the unsaturated zone.

![Figure 5.15: pH values versus depth. Data from October 2014.](image)

**Electrical conductivity**

The electrical conductivity (EC) of water is a measure of the capacity of water to conduct electrical current, which is related to the amounts of ions present in the water (the greater the ions concentration, the greater the conductivity), and therefore to the Total Dissolved Solids (TDS) in the water. Consequently, water with high EC and low pH is likely to contain heavy metals in high concentrations.

The groundwater EC measured in the field averages 1.37 mS/cm in June and 1.7mS/cm in October and April, EC values for groundwater range from 0.03 to 2 mS/cm (Kresic, 2007). The minimal value was measured in well A1 (<0.1), and the maximum value in well M1 (around 10.7mS/cm). The EC measured in the river averages 0.11 mS/cm and is close to constant along the river section and during the year. On the other hand, the EC in the acid mine drainage was 4.4 mS/cm in June and decreased to 2.4 mS/cm in April, most probably due to the mix with water from snowmelt (Figure 5.14).
The plot of EC versus pH (Figure 5.16) reveals that all the groundwater samples from the mine tailings, transect A and C, have high EC (>1.8 mS/cm) independently of the pH with the following exceptions. Groundwater sample from well A1 has a very low EC (<0.1 mS/cm), therefore is less affected by the mine tailings. Samples from the deepest wells in transect C, well C1 and well C7, also present low EC (<0.5 mS/cm), showing that the contamination plume from sulfides oxidation is in the shallow groundwater. The reason why high EC values show to be independent of the pH is that some heavy metals, like Ni and Zn are also very soluble near neutrality (Figure 2.1).

Figure 5.16: Electrical conductivity versus pH for all the water samples collected in October 2014.

Temperature
The temperature in shallow groundwater is expected to change during the year. Due to low reliability of the field temperature measurements (the two multiparameter meters used gave different temperature) the temperature distribution has been investigated from the divers installed (Figure 5.17). The groundwater in the shallower wells A1 and A4, 1.9 and 2.4 m deep, respectively, decreased from 5.5 °C in October to 0.5 °C in December. The groundwater in well M1, which is 4 m deep, decreased from 6.5 to 2.8. Similarly, in well C2, 5 m deep, decreased from 7.5 to 2.1. In the deeper well C7, 10 m deep, the temperature decrease was smoother; it changed from 4.7 to 3.5.

The deeper groundwater seems to be nearly constant during winter. The shallower water is almost at freezing temperature in winter and spring. The relatively higher temperature in wells M1 and CB4 in summer-autumn might be caused by high TDS content.
Figure 5.17: Changes in groundwater temperature in the wells monitored with dataloggers from October 2014 until April 2015. Wells ordered by depth: C7 > C2 > M1 > A4 > A1.

5.1.7.2. Water characterization

Surface water

Compiling chemical analyses of Folla River and the acid mine drainage on the Piper diagram (Figure 5.18) shows that the fresh water is mainly of Ca-HCO$_3$ type, with some influence of SO$_4$, and the acid mine drainage is Mg-SO$_4$.

Figure 5.18: Piper diagram plotting the bulk chemical composition of surface water sampled in the study area using average concentration values. The proportions of cations and anions are expressed in meq/L.
Groundwater

Compiling groundwater chemical analyses on the Piper diagram (Figure 5.19) shows a variable groundwater composition. The groundwater samples can be grouped in two water types, Ca-SO$_4$ and Ca-HCO$_3$, characterized by high concentrations of Ca, SO$_4$/HCO$_3$. Except groundwater in well M1 which composition is Mg-SO$_4$. The two water types confirm the presence of igneous-metamorphic rocks rich in sulfides and Ca-Mg silicates. The waters rich in SO$_4$ are affected by sulfides oxidation, and considered as contaminated groundwater, while the ones rich in HCO$_3$ can be classified as cleaner groundwater.

Groundwater from wells in transect A is Ca-SO$_4$ rich and has similar major ions composition, except for well A1 that is rich in HCO$_3$. Groundwater from wells in transect B is Ca-HCO$_3$ rich, except for well B2 that has elevated concentration of sulfate. Wells B1 and B3 show to be less influenced by Ca. Groundwater from wells in transect C is rich in Ca-SO$_4$, with some being influenced by Mg, with the following exceptions. Well C1 has elevated concentration of K and well C7 is Ca-HCO$_3$ rich. The K, may come from other sources, such as the industry located in the area.

![Piper diagram](image)

*Figure 5.19: Piper diagram plotting the bulk chemical composition of groundwater sampled in the different wells of the study area using average concentration values. The proportions of cations and anions are expressed in meq/L.*
Accuracy of chemical analysis

The reliability of the chemical analysis of ions can be estimated by means of the electrical balance (E.B.). Theoretically, the electro-neutrality of water is zero; hence the sum of cations should be equal to the sum of anions. However, discrepancies are inevitable due to the analytical accuracy. The percentage deviation in E.B. can be estimated by the following equation (Appelo & Postma, 2005):

\[
E.B. (\%) = \left( \frac{\sum \text{cations} + \sum \text{anions}}{\sum \text{cations} - \sum \text{anions}} \right) \times 100
\] (5.1)

where cations and anions are expressed as meq/L and inserted with their charge sign. The sums are usually calculated over the major ions: Na\(^+\), K\(^+\), Ca\(^{2+}\), Mg\(^{2+}\), Cl\(^-\), HCO\(_3\)\(^-\), SO\(_4\)\(^{2-}\) and NO\(_3\)\(^-\). However, in some conditions, such as acid waters, other ions like H\(^+\), Fe\(^{3+}\) and Fe\(^{2+}\) can also be important. Analyses are considered reliable when the E.B. deviation is < 5%.

The calculations of the electrical balance of the groundwater chemical analyses were made using the major ions analyzed by the IC, the cations Na\(^+\), K\(^+\), Ca\(^{2+}\), Mg\(^{2+}\) and the anions Cl\(^-\), F\(^-\), HCO\(_3\)\(^-\), SO\(_4\)\(^{2-}\) and NO\(_3\)\(^-\). To obtain accurate calculations the estimation was done using PHREEQC. The results show a general large negative error (Table 5.7), with a maximum value of almost 75% for groundwater sample M1 and a minimum value of 6% in well B2. These great errors are due to not measure and use other cations that also contribute significantly, e.g. Fe\(^{2+}\). The cations were analyzed without adding HNO\(_3\) acid to the sample, therefore Fe\(^{2+}\) precipitated.

<table>
<thead>
<tr>
<th>Well</th>
<th>E.B.</th>
<th>Well</th>
<th>E.B.</th>
</tr>
</thead>
<tbody>
<tr>
<td>M1</td>
<td>-75%</td>
<td>C1</td>
<td>-66%</td>
</tr>
<tr>
<td>A1</td>
<td>-35%</td>
<td>C2</td>
<td>-8%</td>
</tr>
<tr>
<td>A2</td>
<td>-39%</td>
<td>C3</td>
<td>-43%</td>
</tr>
<tr>
<td>A3</td>
<td>-63%</td>
<td>C4</td>
<td>-13%</td>
</tr>
<tr>
<td>A4</td>
<td>-55%</td>
<td>C5</td>
<td>-29%</td>
</tr>
<tr>
<td>B1</td>
<td>-9%</td>
<td>C6</td>
<td>-30%</td>
</tr>
<tr>
<td>B2</td>
<td>-6%</td>
<td>C7</td>
<td>-11%</td>
</tr>
<tr>
<td>B3</td>
<td>-9%</td>
<td>C81</td>
<td>-10%</td>
</tr>
<tr>
<td>BS1</td>
<td>-14%</td>
<td>C82</td>
<td>-18%</td>
</tr>
<tr>
<td>BS2</td>
<td>-15%</td>
<td>C83</td>
<td>-24%</td>
</tr>
<tr>
<td>BS3</td>
<td>-25%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 5.7: Electrical balance (%) of the groundwater chemical analyses of the samples from October 2014.
5.1.7.3. Heavy metals

Surface water

The concentration of heavy metals (Cd, Co, Ni, Cu, Zn, Pb and Sb) is heterogeneous along the Folla river section but quite constant during the year (Table 5.8). The exceptions are Co and Ni that seem to increase in April, from 0.6 to 2.5 µg/L and from 0.6 to 80 µg/L, respectively for each element, showing that they are mobile around pH 7. The average concentration of Cu is 5 times higher than the 10 µg/L limit established by the Norwegian Environment Directorate.

The acid mine drainage stream has very high concentration of heavy metals except for Sb, which is 0.01 µg/L. The concentrations are nearly constant during the year (Table 5.8), except for Pb which decreased from 8 to 1 from 2014 to 2015.

<table>
<thead>
<tr>
<th></th>
<th>Cd</th>
<th>Sb</th>
<th>Pb</th>
<th>Co</th>
<th>Ni</th>
<th>Cu</th>
<th>Zn</th>
</tr>
</thead>
<tbody>
<tr>
<td>River</td>
<td>0.21</td>
<td>0.02</td>
<td>0.14</td>
<td>0.62</td>
<td>23.8</td>
<td>53.48</td>
<td>53.14</td>
</tr>
<tr>
<td>AMD stream</td>
<td>58.8</td>
<td>0.01</td>
<td>6.1</td>
<td>849.4</td>
<td>365.5</td>
<td>37366.8</td>
<td>27165.2</td>
</tr>
</tbody>
</table>

Table 5.8: Average concentration (µg/L) of heavy metals in Folla river and the acid mine drainage stream.

Groundwater

The results of the chemical analyses of heavy metals (Cd, Co, Ni, Cu, Zn, Pb and Sb) shows an heterogeneous composition of those elements in the groundwater with minimal variations during the year. Plotting the concentrations of the heavy metals on the study area map reveals a similar spatial pattern distribution for all them. Groundwater in well M1 is the one containing higher concentration of all the elements analyzed, except for Pb and Sb. The groundwater in the mine tailings is contaminated but the concentration of heavy metals varies with depth, being the shallower wells more contaminated than the deeper wells. Groundwater in the forest area, transect B wells, and in well A1 shows to be clean with concentrations around 0.1-5 µg/l. Concentration of Sb and Pb are an exception to this general distribution, their concentration is very low in all the groundwater samples (<0.3 µg/l), only well A4 shows very high concentration of Pb (70 µg/l). The low concentration of Sb could be explained by very low content of the heavy metals in the tailings. For Pb is due
to its lower mobility and strongly sorption above pH range 3.5-4.5 (Appelo and Postma, 2005).

Cadmium concentration in groundwater in the study area ranges from 0.01 to 250 µg/l (Figure 5.20). The maximum value is found in well M1. Groundwater in the western mine tailings has low concentration of Cd except for well A4 (70 µg/l). In the eastern mine tailings the groundwater in the shallower wells, C2, C3, C5 and C6, has similar high Cd concentration, among 15 and 30 µg/l (Figure 5.21). The groundwater sampled in the shallower well in the piezometer nest CS also has high concentration of Cd (15 µg/l), meaning that contaminants are also flowing to Folla through groundwater.

Figure 5.20: Distribution of cadmium concentration (µg/L) in groundwater in the study area.

Figure 5.21: Plot of cadmium concentration (µg/L) versus depth for the groundwater samples from wells in transect A and C.
Cobalt concentration in groundwater in the study area ranges from 0.07 to 5,700 µg/L (Figure 5.22). The maximum value is found in well M1. Groundwater in the western mine tailings has low concentration of Co except for wells A3 and A4, where is 200 and 600 µg/L, respectively. In the eastern mine tailings groundwater in well C3 has the highest concentration (490 µg/L), followed by wells C5 and C6 (250 µg/L). The groundwater sampled in the medium well in the piezometer nest CS also has high concentration of Co (200 µg/L), meaning that contaminants are also flowing to Folla through groundwater.

Nickel concentration in groundwater in the study area ranges from 0.6 to 2,200 µg/L (Figure 5.23). The maximum value is found in well M1. The Ni concentration in groundwater in the western mine tailings is increasing towards the river. In the eastern mine tailings Ni has a similar distribution as Co, the groundwater in well C3 has the highest concentration (570 µg/L), followed by wells C5 and C6 (200 µg/L). The groundwater sampled in the medium well in the piezometer nest CS also has high concentration of Ni (100 µg/L), meaning that contaminants are also flowing to Folla through groundwater.
Copper concentration in groundwater in the study area ranges from 0.6 to 120,000 µg/L (Figure 5.24). The maximum value is found in well M1. Groundwater in the western mine tailings has low concentration of Cu except for well A4, where is very high 21,500 µg/L. In the eastern mine tailings groundwater in wells C5 and C6 is rich in Cu, about 2,000 µg/L. The groundwater sampled in the shallower well in the piezometer nest CS also has high concentration of Cu (750 µg/L), meaning that contaminants are also flowing to Folla through groundwater.
Zinc concentration in groundwater in the study area ranges from 7.33 to 97,000 µg/L (Figure 5.25). The maximum value is found in well M1. Groundwater in the western mine tailings has low concentration of Zn except for wells A3 and A4, where is very high 25,000 µg/L. In the eastern mine tailings groundwater all the wells is very rich in Zn (among 10,000 and 30,000 µg/L) except for the deeper wells C1 and C7. Anomalously, the groundwater sampled in the piezometer nest wells BS and CS also revealed high concentration of Zn (from 2,000 to 15,000 µg/L). Therefore, an experiment in the laboratory at the Department of Geosciences was carried out to investigate if the metallic pipes used were the source of the high content in zinc.

To test the zinc contribution of the pipes, the two well screens used in the study area were placed in a crystal bucket filled with mili-Q water from December 2014 until April 2015 (Figure 5.26). Water from each bucket was sampled 6 times (approximately every 3 weeks), and afterwards it was analyzed in the ICP-MS. The resulting analyzed concentration showed the maximum value contribution from the pipes is 1300 µg/L (measured 3 weeks after the experiment set up). Therefore, the pipes contribute on the amount of zinc dissolved in water but are not the main source of the element.
Figure 5.26: Experiment setup with the two sizes of well screens used in the study area.

5.1.6.3.1 Groundwater quality

Many of the elements concentration in groundwater and the river are exceeding the limit for a regular drinking water supply, based on that, the water samples were classified with respect to contamination using the Norwegian Environment Directorate classic-classification system for freshwater (SFT, 1997) (Table 5.8).

According to the classification system, despite the large differences in heavy metals concentrations along the area, the groundwater in the study area is very heavily polluted in well M1 and the wells in the mine tailings, and it has marked contamination in the forest area and well A1 (Table 5.9). So no groundwater with negligible contamination is found in the area. Water in the Folla river section investigated is also heavily polluted.

<table>
<thead>
<tr>
<th>Element (µg/L)</th>
<th>I Negligible contamination</th>
<th>II Moderately polluted</th>
<th>III Marked contamination</th>
<th>IV Heavily polluted</th>
<th>V Very heavily polluted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cu</td>
<td>&lt;0.6</td>
<td>0.6-1.5</td>
<td>1.5-3</td>
<td>3-6</td>
<td>&gt;6</td>
</tr>
<tr>
<td>Zn</td>
<td>&lt;5</td>
<td>5-20</td>
<td>20-50</td>
<td>50-100</td>
<td>&gt;100</td>
</tr>
<tr>
<td>Cd</td>
<td>&lt;0.04</td>
<td>0.04-0.1</td>
<td>0.1-0.2</td>
<td>0.2-0.4</td>
<td>&gt;0.4</td>
</tr>
<tr>
<td>Ni</td>
<td>&lt;0.5</td>
<td>0.5-2.5</td>
<td>2.5-5</td>
<td>5-10</td>
<td>&gt;10</td>
</tr>
<tr>
<td>Pb</td>
<td>&lt;0.5</td>
<td>0.5-1.2</td>
<td>1.2-2.5</td>
<td>2.5-5</td>
<td>&gt;5</td>
</tr>
</tbody>
</table>

Table 5.8: Contamination classic-classification system from the Norwegian Environment Directorate (modified from SFT, 1997).
### Table 5.9: Classification of the groundwater sampled in the study area according to the location and the heavy metal concentration.

<table>
<thead>
<tr>
<th>Well</th>
<th>Cu</th>
<th>Zn</th>
<th>Cd</th>
<th>Ni</th>
<th>Pb</th>
</tr>
</thead>
<tbody>
<tr>
<td>M1</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>IV</td>
</tr>
<tr>
<td>A1</td>
<td>V</td>
<td>III</td>
<td>III</td>
<td>I</td>
<td>I</td>
</tr>
<tr>
<td>A2</td>
<td>II</td>
<td>V</td>
<td>II</td>
<td>V</td>
<td>I</td>
</tr>
<tr>
<td>A3</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>I</td>
</tr>
<tr>
<td>A4</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>V</td>
<td>V</td>
</tr>
<tr>
<td>B1</td>
<td>V</td>
<td>III</td>
<td>II</td>
<td>IV</td>
<td>I</td>
</tr>
<tr>
<td>B2</td>
<td>IV</td>
<td>II</td>
<td>II</td>
<td>III</td>
<td>II</td>
</tr>
<tr>
<td>B3</td>
<td>IV</td>
<td>II</td>
<td>I</td>
<td>V</td>
<td>I</td>
</tr>
<tr>
<td>BS1</td>
<td>III</td>
<td>V</td>
<td>I</td>
<td>III</td>
<td>I</td>
</tr>
<tr>
<td>BS2</td>
<td>II</td>
<td>V</td>
<td>I</td>
<td>II</td>
<td>I</td>
</tr>
<tr>
<td>BS3</td>
<td>II</td>
<td>V</td>
<td>I</td>
<td>IV</td>
<td>I</td>
</tr>
<tr>
<td>River</td>
<td>V</td>
<td>IV</td>
<td>III</td>
<td>II-V</td>
<td>I</td>
</tr>
</tbody>
</table>

Table 5.10: Daily (kg/d) and yearly (Tons/yr) pollution discharge from groundwater to Folla, along the 1.5 km section of the river included in the study area.

#### 5.1.8. Groundwater contamination contribution to Folla River

From the calculated total daily groundwater discharge to Folla, 1200 m³/d along the 1.5 km section of river, and the mean concentration of heavy metals (Sb, Pb, Cd, Co, Ni, Cu and Zn) and SO₄²⁻ in groundwater, the daily and yearly mass load contribution of contaminants from groundwater to Folla can be estimated (Table 5.10). SO₄²⁻, Zn and Cu are the elements discharged in higher amounts, followed by Ni, Co and Cd. The amount of Pb and Sb is 4 and 6 times lower than Zn.
5.2. Hydrogeological model

5.2.1. Aquifer parameters

The sensitivity analysis of the parameters shows that the model is very sensitive to changes in the northern border. (Table 5.11). Different water table depth below the ground surface were tested, lower north border head give higher global heads than the deeper ones.

<table>
<thead>
<tr>
<th>North border head (m)</th>
<th>Mean absolute error</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1.41</td>
</tr>
<tr>
<td>2</td>
<td>No convergence</td>
</tr>
<tr>
<td>3</td>
<td>2.75</td>
</tr>
<tr>
<td>4</td>
<td>No convergence</td>
</tr>
</tbody>
</table>

Table 5.11: Mean absolute error of the simulations with variable distance between the ground surface and the water table in the north border.

The presence or absence of the interpolated thickness point shows minimum changes in the global head distribution. Different thickness ranging from 10 to 20 m were tested, but the model show its limitations when the thickness was <15 m (Table 5.12).

<table>
<thead>
<tr>
<th>Aquifer thickness – interpolated point (m)</th>
<th>Mean absolute error (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>15</td>
<td>1.41</td>
</tr>
<tr>
<td>20</td>
<td>2.23</td>
</tr>
</tbody>
</table>

Table 5.12: Mean absolute error (m) of the simulations with variable aquifer thickness for the interpolated point.

Changes in the hydraulic conductivity and specific yield are noticeably affecting the head distribution; the eastern part of the study area is easily flooded. Simulations with lower hydraulic conductivity values produced elevation of the water table until reaching the topographic surface and over it, which would be observed in the field site as a flooded or swampy area. When the hydraulic conductivity value is increased the gradient decreases (Table 5.13). Although, the changes in the mean error are small due to there are few observation points.
<table>
<thead>
<tr>
<th>Hydraulic conductivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>(m/d)</td>
</tr>
<tr>
<td>0.01</td>
</tr>
<tr>
<td>0.1</td>
</tr>
<tr>
<td>1</td>
</tr>
<tr>
<td>5</td>
</tr>
<tr>
<td>10</td>
</tr>
</tbody>
</table>

Table 5.13: Mean absolute error of the simulations with variable hydraulic conductivity.

Changes in the specific yield are more gradual and less abrupt. For high values (0.1-0.3) values of specific yield value the phreatic zone increases flooding some zones of the aquifer, specially the eastern part (Table 5.14).

<table>
<thead>
<tr>
<th>Specific yield</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.01</td>
</tr>
<tr>
<td>0.05</td>
</tr>
<tr>
<td>0.1</td>
</tr>
<tr>
<td>0.3</td>
</tr>
<tr>
<td>Mean absolute error</td>
</tr>
<tr>
<td>1.41</td>
</tr>
<tr>
<td>1.45</td>
</tr>
<tr>
<td>1.5</td>
</tr>
<tr>
<td>1.91</td>
</tr>
</tbody>
</table>

Table 5.14: Mean absolute error of the simulations with variable specific yield.

In the model the hydraulic conductivity zones and the specific yield were calibrated until the best fit between the computed and the collected data was achieved (Table 5.15). The best distribution heads solution has a mean absolute error equal to 1.41m, for all stress periods and observation wells. The fit reached is considerably good except for well A1, where there is up to 3 m of difference between computed and observed head (Figure 5.27). The hydraulic conductivity of the materials varies between 0.7 and 10 m/d, in the range of the values obtained in the field investigations. The specific yield is 0.005, lower than the general range for unconfined aquifers. But according to Kresic (2007), lower values are possible in cases of finer and less uniform material, such as in finer grained and less uniform material and mine tailings.
### Table 5.15: Aquifer parameter values adopted and range of values used in the sensitivity analysis and the calibration process.

<table>
<thead>
<tr>
<th>Zone 1</th>
<th>Zone 2</th>
<th>Zone 3</th>
<th>Zone 4</th>
<th>Zone 5</th>
<th>Zone 6</th>
<th>Specific yield, $S_y$</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.8</td>
<td>0.8</td>
<td>0.9</td>
<td>0.7</td>
<td>7.5</td>
<td>10</td>
<td>0.01</td>
</tr>
</tbody>
</table>

#### 5.2.2. Water balance

The monthly water budget is close to constant from June 2014 to April 2015, except for the winter months. The precipitation recharge represents 3% of the total inflow (1850 m$^3$/d), the % stored is < 0.0003% and the 97% of is from regional groundwater, i.e. groundwater entering the system from the north border (Figure 5.28). The drainage system represents 19% of the total outflow (1850m$^3$/d), while the stored water out-flowing is < 0.00025E% and the 81% left is flowing out of the system through the southern border (Folla river). The only change during the year is produced in the total input during winter, when precipitation recharge was neglected and the regional inflow is 99% of the input. However, the drainage system and the regional flow maintain a constant proportion of the outflow water.
Figure 5.28: Pie chart of the water balance in the aquifer for the period June 2014 to April 2015. Groundwater inflow refers to the groundwater entering the system from the north border and outflow to the water leaving the system through the Folla river border.

5.2.3. Contaminants transport scenario

The preliminary long-term path-line evolution of the contaminants was estimated with MODPATH using a steady state model. From the Main mine site, the particles travel towards the W mine tailings, being the minimum time for the particle to reach Folla River less than 23 years (Figure 5.29). The minimum travelling time for particles from the furthermost part of the mine tailings zones to reach Folla is around 11 and 4 years for the We and E mine tailings, respectively.

Figure 5.29: Plan view of the tracking particles path in the steady state simulation.
Since the aquifer materials are heterogeneous, different porosity values were tested to examine how this parameter is affecting the particle traveling time (Table 5.16). Higher porosity increases the time, while lower porosity values decreases it. When the porosity is 0.15 the minimum traveling time from the Main mine area is less than 9 years. If the porosity increases a 12% (0.38), the traveling time is three times larger (23 years) and if the porosity is 0.5 the traveling time are 30 years. Therefore, it seems that the big change in the results occurs when porosity is between 0.15 and 0.35, and when is larger than 0.35 the increase of time is not that abrupt. The values used were the ones obtained with 0.38 porosity, based on the porosity of mine tailings from Pabst et al. (2014).

<table>
<thead>
<tr>
<th>Porosity</th>
<th>Main mine</th>
<th>E mine tailings</th>
<th>W mine tailings</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.15</td>
<td>8.8</td>
<td>1.5</td>
<td>4.63</td>
</tr>
<tr>
<td>0.38</td>
<td>22.5</td>
<td>3.94</td>
<td>10.65</td>
</tr>
<tr>
<td>0.5</td>
<td>29.6</td>
<td>5.2</td>
<td>15.44</td>
</tr>
</tbody>
</table>

Table 5.16: Minimum traveling time of the particles for different porosity values.

A more accurate mean velocity of the particles from each contamination focus site was estimated using the transient model. Particles from the Main mine are the fastest ones, they travel 3 times faster than particles in the W mine tailings and 2 times than in the E mine tailings (Table 5.17) where the hydraulic gradient is larger. Although, according to the pathline evolution observed in the steady state (Figure 5.29) the velocity of the particles from the Main mine decreases when the they reach the flat lowland area.

<table>
<thead>
<tr>
<th>Porosity</th>
<th>Main mine</th>
<th>E mine tailings</th>
<th>W mine tailings</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.15</td>
<td>0.73</td>
<td>0.44</td>
<td>0.23</td>
</tr>
<tr>
<td>0.38</td>
<td>0.29</td>
<td>0.17</td>
<td>0.09</td>
</tr>
<tr>
<td>0.5</td>
<td>0.22</td>
<td>0.13</td>
<td>0.07</td>
</tr>
</tbody>
</table>

Table 5.17: Mean traveling velocity of the particles for different porosity values.
5.2.3.1. Mass transport

The distribution and evolution of the contaminants during the period June 2014 to April 2015 was estimated with MT3DMS. The contamination advection-dispersion mass transport simulation shows different distribution and evolution of the plume of contamination (Figure 5.30), in the mine tailings zones the contamination plume is more concentrated while in the Main mine zone it spreads and dilutes faster. Adding the longitudinal dispersion and the molecular diffusion increases the spreading of contamination, although the traveled distance of the plume from the Mine area agrees with the MODPATH distance simulated in the transient model.

Figure 5.30: Plan view of the spatial distribution of the contaminant plume from the three different focus sites (Main mine, W mine tailings and E mine tailings) in the study area at the beginning and at the end of the simulated period.
5.3. Geochemical simulations

5.3.1. Reactant phases

The inverse modeling calculations revealed that 6 different possible models (Table 5.18) can describe the production of the chemistry of analyzed groundwater in well C6 from rain water as result of pyrite oxidation, and silicates mineral dissolution within the tailings. In order to obtain possible models, the solutions were balanced increasing the uncertainty limit to 40%.

<table>
<thead>
<tr>
<th>Phase</th>
<th>Model 1</th>
<th>Model 2</th>
<th>Model 3</th>
<th>Model 4</th>
<th>Model 5</th>
<th>Model 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pyrite</td>
<td>2.39e-03</td>
<td>2.39e-03</td>
<td>4.76e-03</td>
<td>6.3e+01</td>
<td>1.02e+02</td>
<td>1.27e+02</td>
</tr>
<tr>
<td>O₂(g)</td>
<td>7.92e-03</td>
<td>7.92e-03</td>
<td>1.68e-02</td>
<td>2.36e+02</td>
<td>3.83e+02</td>
<td>4.78e+02</td>
</tr>
<tr>
<td>Anorthite</td>
<td>1.87e-04</td>
<td>1.85e-04</td>
<td>4.93e-03</td>
<td>1.26e+02</td>
<td>2.04e+02</td>
<td>2.55e+02</td>
</tr>
<tr>
<td>K-mica</td>
<td>1.85e-04</td>
<td>-</td>
<td>1.85e-04</td>
<td>1.311e-04</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Halite</td>
<td>1.13e-04</td>
<td>1.402e-04</td>
<td>1.40e-04</td>
<td>-</td>
<td>-</td>
<td>-1.06e-04</td>
</tr>
<tr>
<td>Gypsum</td>
<td>4.76e-03</td>
<td>4.932e-03</td>
<td>-</td>
<td>-1.26e+02</td>
<td>-204e+02</td>
<td>-2.55e+02</td>
</tr>
<tr>
<td>Kaolinite</td>
<td>-8.81e-04</td>
<td>-7.32e-04</td>
<td>-5.63e-03</td>
<td>-1.26e+02</td>
<td>-2.04e+02</td>
<td>-2.55e+02</td>
</tr>
<tr>
<td>Fe(OH)$_3$</td>
<td>-</td>
<td>-</td>
<td>-2.37e+03</td>
<td>-6.3e+02</td>
<td>-1.02e+02</td>
<td>-1.27e+02</td>
</tr>
</tbody>
</table>

Table 5.18: Six possible models and phases identified on the inverse modelling simulation in PHREEQC from groundwater sample C6 and rainwater from Appelo and Postma (2005). Positive mole transfers indicate dissolution, negative mole transfers indicate precipitation. In italics the valid model chosen.

In general, the results for the dissolution/precipitation fraction of the phases are similar. The differences between the models are in the amount of pyrite dissolving (of about 4-5 orders of magnitude), the presence of dissolution of K-mica, dissolution/precipitation of halite and gypsum, and dissolution of Fe(OH)$_3$. Daphnite is not dissolving in any of the possible models; hence all the Fe dissolving comes from pyrite.

In the models where the amount of pyrite dissolving is about $10^{-3}$ orders of magnitude the amount of kaolinite precipitating comes from the dissolution of anorthite, K-mica and chlorite. And there is Fe(OH)$_3$ precipitation when the amount of dissolved oxygen increases to $10^2$. However, in the models where the amount of pyrite dissolving is about 10 or $10^2$ orders of magnitude, the amount of anorthite is also increased and then all the kaolinite and gypsum precipitating come from anorthite.
Model 3 is chosen as the plausible model based on: (1) the presence of weathering of K-mica, K-mica mineral phase was specified to include all the weathering from potassium micas and pyroxenes and amphiboles present in the geology of the area (Section 3.2.); and (2) precipitation of Fe(OH)$_3$, in the field investigations precipitated red-yellowish ferric hydroxide particles were observed when pumping water in the mine tailings wells (Figure 5.31), and several bands of precipitating iron were found in the river bed after its confluence with acid mine drainage, caused by the strong gradient in the water pH.

![Figure 5.31: Ferric hydroxide particles in groundwater when pumping well C6. Ferric hydroxide crust on the river bed when the acid mine drainage discharges to Folla river.](image)

### 5.3.2. AMD formation

The simulation of the kinetic controlled oxidation of pyrite in the mine tailings reveals that is a very slow process in terms of dissolving and depleting the amount of pyrite in the soil (Figure 5.32); probably due to the high water content limiting the oxygen flux. A water exchange equal to the total pore volume of the 10 m column lasts around 5 months. After twenty years only a 5.4% of the initial amount of pyrite in the first meter of soil (1.32 mol/l) is dissolved. K-mica is weathered and its initial amount (0.1 mol/l) is decreased by 12% in twenty years. Goethite is precipitating (from 1 to 2.5 µmol/l) when the acidified solution enters each cell for the first time (Figure 5.33). On the other hand, Fe(OH)$_3$ and gypsum are not precipitating.

![Figure 5.32: Evolution of pH and pyrite and K-mica calculated concentrations(M) over 20 years of pyrite oxidation in the first meter (first cell) of the column.](image)
However the effects of this process in the pH are very fast. In only 1 month the pH decreased from 6.8 (initial groundwater pH) to 4.5 in the first meter of column (Figure 5.34). And after 7 months, the pH was 4.5 in the whole column. The simulation is in agreement with the field data pH distribution, where the pH range 4-7 is observed.
5.3.3. Remediation

The simulation of the efficiency of a 1m limestone to remove groundwater of heavy metals, as Fe, Zn\(^{2+}\) and Cu\(^{2+}\), revealed that in the conditions specified (Section 4.3.3.) limestone effectively retards Fe, Zn\(^{2+}\) and Cu\(^{2+}\). The results were interpreted versus pore volumes; a water exchange equal to the total pore volume of the 20 m column lasts around 5 months. With constant inflow of acid mine drainage water, the initial concentration of copper (1.16 mmol/L), zinc (0.65 mmol/L) and iron (14.19 mmol/L) in the groundwater will be decreased 96.5%, 99.7% and 99.9%, respectively, after water flows through the limestone drain (Figure 5.35). Copper and zinc are found dissolved as Cu\(^{+2}\), Zn\(^{+2}\) and in the form of XCO\(_3\), XHCO\(_3^+\), where X represents the metal. Fe is found as ferrous iron (Fe\(^{2+}\)). However, despite the concentration of the copper is decreasing, its amount after passing through the barrier (17.1 µmol/L) still classifies groundwater as heavily polluted according to the limit permitted by the Norwegian Environment Directorate (0.09 µmol/L) (Table 5.8).

After 2 years (equivalent to 5 pore volumes flushed) the concentration of copper and zinc will still be decreased 73 and 75%, respectively. However, the concentration of zinc (163.3 µmol/L) is above the concentration limit (1.53 µmol/L) of very heavily polluted groundwater according to the Norwegian Environment Directorate classification. No changes on the iron concentration retained are observed, due to it will be precipitating until calcite may be depleted.

![Figure 5.35: Zn\(^{2+}\) and Cu\(^{2+}\) concentration in groundwater after (a) 5 monthst (1 pore volume) and (b) 2 years (2 pore volumes). Note that the barrier (black bold line) is located at 45 m distance.](image-url)
The amount of available calcite in the drain system is smoothly decreasing, after 2 years (equivalent to 5 pore volumes) 89% of the initial amount of calcite (15 moles) will still be available. Goethite and gypsum precipitate linearly after ion exchange among calcite and the heavy metals starts. After 2 years of contamination inflow, 1.2 and 1.4 mols of goethite and gypsum, respectively, would have precipitate within the limestone drain. The precipitation of goethite generates iron hydroxide surface where the heavy metals can be sorbed. Gypsum also shows to be precipitating in the cells after the limestone drain in a lower amount, around 20 mmol. The pyrite in equilibrium with the system is not affected by the entry of acid mine water in the column, and its amount remains constant over time. The limestone drain also shows to be effectively increasing the groundwater pH, it changes the pH from 4 to 6.5 (Figure 5.36) after water flows through the drain system.

![Figure 5.36: Calcite dissolution and goethite and gypsum precipitation in the limestone drain for the period corresponding to 2 years (equivalent to 5 pore volumes).](image)

Various drain widths and limestone purity were tested to investigate the effectiveness of the limestone drain for different geometries and compositions. If the drain is 2m wide it retains 10% more of heavy metals, and if it is 0.5 m wide retains 10% less. Thus, a wider barrier increases the life-time of the limestone drain. When the initial limestone purity (88%) is changed to 95% the drain retains heavy metals 9% more effectively than if the purity is decreased to an 80%, when it retains a 12% less. Therefore, a more pure limestone composition increases the life-time of the drain.
5.4. Limestone drain application

Based on the results from the tracking particles simulation two limestone drains, perpendicular to the groundwater flow, are proposed to remove the contaminants from the groundwater before it discharges to the river. One in the western mine tailings area of around 150 m of longitude and another one in the eastern mine tailings, of around 100 m of longitude (Figure 5.38).

The barriers should be 10 m deep, based on the hydrochemistry results from the groundwater in the east mine tailings. Water at 6m depth is very heavily polluted while at 10 m was considered moderately polluted. No information of groundwater deeper than 3m is available for the western mine tailings, but since the mine tailings are very similar the same pattern of contamination versus depth is inferred.
Figure 5.38: Close view of the mine tailings with the locations of the two limestone drains (yellow lines) suggested.
6. Discussion
This section will integrate the results obtained from field surveys, numerical hydrogeological and geochemical models to characterize the groundwater of Folldal old mining site. The objective is to provide an overall view of the hydrogeological and hydrogeochemical functioning and processes in the study area as well as propose potential solutions and further improvements for future studies.

6.1. Hydrochemistry
Folldal Main mine and its tailings cause acid formation and high content of heavy metals in the groundwater. The groundwater composition type (Figure 5.19) and the heavy metals distribution (Figures 5.20-22-23-24-25) shows that the groundwater in the upper part of the mining area (well M1) and in the proximities of the mine tailings is highly polluted. While the groundwater in the forest central area has relatively good quality (Table 5.9). This is due to there are no mine wastes directly contaminating the forest area and contaminated groundwater flowing towards that area (Figure 5.29).

The groundwater analyses in the different sampling points within the mine tailings also revealed that the shallower wells have higher concentration of contaminants; therefore the groundwater chemistry is controlled by surface infiltration. In those areas where the top soil consists of tailings, the rainwater infiltrating oxidizes the sulfides present and forms AMD, which percolates downwards to the groundwater. When it is mixed with the groundwater it will start moving with the groundwater flow direction, changing its spatial distribution.

The geochemical processes that take place between the mine area in the higher part of the study area to the low lands in the proximity of the river are unknown due to the limited piezometer network (Figure 4.5). Although, since the groundwater in the central area has relatively low content of heavy metals, probably the acidic water is partially diluted and neutralized in its path to the river proximity.

The differences observed in the groundwater of the mine tailings areas for the range of concentration values of the elements, the spatial distribution of physicochemistry properties and composition can be explained by the heterogeneity of the materials. NGI analysis of soil samples (Figure 3.12) have shown that the tailings are very heterogeneous, consisting of a
mixture of different types of material, in terms of mineralogy, weathering and texture; due to they were collected from the different mines in the area and deposited at different times. Therefore, the mineralogical composition of the tailings strongly influences the vertical transport of metals from soil to groundwater. More content of sulfides will generate more acid formation and more content of neutralizing elements, as aluminosilicates, clays or iron hydroxides, will decrease it. Changes in the soil, as oxidation or weathering, will alter their properties enhancing vertical metal mobility (Maclean and Bledsoe, 1992). These metals released from the oxidation of sulfides are dissolving in to the groundwater, changing the chemistry of the water. Two groundwater types are found in the area, Ca-SO$_4$ and Ca-HCO$_3$, the water rich in SO$_4$ are affected by sulfides oxidation and the HCO$_3$ can be considered as the original water.

The dissolved concentration values measured along the study area of Cu, Cd, Zn, Ni and Co are directly connected with the pH of the groundwater. Groundwater in wells with more acidic pH, M1, A4, C3, C5 and C6, generally have higher concentration of heavy metals. The lower the pH value the more metal can be found in solution, and thus more metal is mobilized (Appelo and Postma, 2005). And when the pH increases the adsorption and precipitation of metals increases (Figure 2.1). Cd, Zn and Ni are sorbed above pH range 6-8, which explains the similar pattern of wells (M1, A4, C3, C5 and C6) with high concentration of these elements along the area (Figure 5.20-23-25).

The low concentration of Sb could be explained by very low content of this metal in the tailings. The low content of Pb in the groundwater seems to be related to low content of this metal in the original ore (Bjerkgård and Bjørlykke, 1996) and to its lower mobility and strongly sorption above pH range 3.5-4.5 (Figure 2.1). The anomalous high content of Pb in well A4 (70 µg/l) might be due to the combination of local high Pb concentration in the soil composition and the groundwater pH, around 4.3, in which Pb can still be as dissolved metal.

Although it seems that there is a trend of concentration of heavy metals to decrease with depth, there is no trend for wells at similar depth in the mine tailings. For example, wells C3 and C4, both 4.85m deep, have pH 4.8 and 6.93, EC 2.15 and 2, and 65 and 607 µg/l Co concentrations, respectively. The difference between the wells could be explained by the local geochemical processes, due to the mineralogical variety composition of the tailings and
soil, i.e. its content of sulfides and neutralizing minerals. Based on the soil samples analyzed by NGI, the sulfur content in the mine tailings where the wells are located ranges from 0.2 to 9.9 %S. According to the lithology, the area contains chlorite and other micas (gangue minerals), which are aluminosilicates that can contribute to acid-neutralization reactions.

The inverse geochemical simulations corroborated that the silicates present in the tailings are also dissolving and counteracting the acid formation from sulfides oxidation. However, the estimated amount of pyrite dissolving (4.76e-03 mols/L) in the inverse model takes into account the amount of iron from chlorite [(Mg,Fe)₃(Si,Al)₄O₁₀], pyroxenes and amphiboles present in the lithology of the area. This means that the actual amount of pyrite dissolving would be slightly lower than the modelled. Nevertheless, the iron contribution from chlorite dissolving is one order of magnitude smaller, which implies minimal effects on the iron contribution from pyrite.

The vertical movement of the heavy metals in the groundwater column is affected by the hydrogeological properties of the soil. Soils with greater porosity, voids or fractures, or high hydraulic conductivity contribute more to the mobility and transport of heavy metals (Sherene, 2010). According to this, the AMD formation and vertical mobility residence time estimated with the geochemical simulation would be much longer. The vertical velocity used in the model was estimated from the hydraulic head of two wells (C5 and C6) with the highest hydraulic conductivity in the study area (5.4), therefore in the zones of the aquifer where the hydraulic conductivity is one order of magnitude lower (less mobility), the residence time would increase a 15%.

The transport of heavy metals and SO₄²⁻ from groundwater to the river estimated shows that groundwater produces a high contaminant load to Folla River, especially for SO₄²⁻, Zn and Cu. From the yearly pollution transport in Folla river data measured by NIVA (Table 3.1), and the yearly pollution discharge from groundwater calculated (Table 5.10) it can be estimated that the contribution of contamination from groundwater is 25-35% of the total contamination in Folla River. Moreover, the daily contribution of Zn from groundwater, 16 kg/days, explains the discrepancy of daily zinc transport between the acid mine drainage and Folla concentration stated by NIVA (Figure 3.11). The majority of the contamination would come from the western and eastern mine tailings, where the heavy metals concentration in
water is much higher than in the forest area, and where the daily discharge, 0.6 \( m^3/d/m \), is twice the one in transect B, 0.29 \( m^3/d/m \).

### 6.2. Hydrogeology

The groundwater level monitoring revealed that changes in level and temperature are highly related to climatic conditions like precipitation and seasonal changes (Figure 5.12 and 5.17). The general trend of the water table to decrease from summer to winter is explained due to (1) low recharge in autumn, (2) snow precipitation and accumulation in winter (temperatures below zero), which freezes and waterproofs the soil impeding the possibility of recharge and (3) snow-melt during spring increasing the recharge. This pattern can be considered annual since the hydrological year used (April 2014-2015) follows similar trend to the normal historical data of temperature and precipitation (Figure 3.8).

Considering the punctual measured head in the wells and in the two piezometer nests located in the river bed the discharge of groundwater to Folla river takes from late spring to autumn. Based on Darcy’s law, it was estimated, 0.8\( m^3/d/m \) (Table 5.10). However, during the winter months when the river and the river bed are frozen, the river bed and the ice in the border of the river act as an impermeable barrier to the movement of shallow groundwater, because the pore space of the soil within the zone of saturation is filled with ice. The groundwater below the frozen layer will move slower due to the high density of water when temperature is ranging from 0° to 5 °C (Kresic, 2007). This decrease is seen in the groundwater discharge results from December 2014, for example in transect C, decreases to 0.4\( m^3/d/m \) (Table 5.11).

One of the uncertainties on the hydrogeological model is associated with the north boundary geometry and conditions due to limited data on that part of the study area. The aquifer thickness was interpolated from 9 points (6 from the seismic profiles, 2 from the groundwater wells database (GRANADA, and 1 interpolated) mainly covering the southern and eastern part of the area (Figure 4.18). The boundary condition along the north border was defined from the water table depth from one single well, M1. This restricted range of information data of the north boundary limited the accuracy of the model. The model shown limitations when the aquifer thickness was less than 15 m and when the head was below 2 and 4 m the topographic surface. This means that it requires a high flow proceeding from the
upper part of the study area that can be reached either with an increase flow area and/or with high heads in the boundary conditions.

The uncertainty in the northern border may overestimate the groundwater inflow from the north border. The seasonal changes from the continuous monitoring show that the groundwater level is highly affected by the climatic conditions (temperatures below zero and impede recharge during winters), the water table variation is up to ± 1m (Figure 5.12). Therefore, the precipitation recharge, except for the winter months, is the main source of inflow. The study area is located at the foot of the mountain; the large groundwater regional inflow estimated by the model (97%) is explained as the flow proceeding from the precipitation recharge in the mountain. This is corroborated when the mean precipitation recharge inflow is estimated for the mountain area feeding the study area (5.5 km²); provides 1500 m³/day of groundwater regional inflow into the aquifer, similar to the value estimated by the model (1795 m³/day). And the 3% precipitation recharge estimated in the model corresponds to the amount directly infiltrating in the study area. Moreover, the groundwater outflow estimated by the model, around 1795 m³/day, is very similar to the mean groundwater discharge calculated using Darcy’s Law from the field hydraulic head data, around 1200 m³/day.

To calibrate the model, the study area was divided into six zones with different hydraulic conductivity (Figure 4.23). Most of the computed values fit the observed hydraulic head with discrepancies lower than 1 m, except for well A1. This is probably caused by the location of the well; it is situated very close to the Gorbekken creek border, simulated as non-flow boundary. The continuous monitoring indicates that this well might be connected to the creek.

One of the main problems during the calibration of the model was the presence of flooded cells in the eastern part of the area when increasing the specific yield and decreasing hydraulic conductivity. This is caused by the rise of the water table over the topographic surface until 1 m over it. The cause can be, as in the case of Gorbekken creek, to define the eastern boundary as no-flow. This no-flow border is stopping the flow that can spread out of the modelled area causing an excess of water in the surrounding western cells. This may indicate that probably is not a non-flow boundary and that there is a percentage of the flow
moving out of the limits that were selected for this model. A cone shape of the study area would be more appropriated in further studies for the modelling of the study area.

The hydrogeological model showed to be sensitive to changes in (1) the northern border, (2) the specific yield and (3) the hydraulic conductivity, which was only known punctually in the wells location (Table 5.11-12-13-14). In addition, it was only possible to use 5 of the 21 wells (M1, A1, A4, C2 and C7) for continuous monitoring of the groundwater level. Even with that, the transient groundwater model matches the dynamics of the processes observed in Folldal with a mean absolute error of 1.41m, for all stress periods and observation wells.

The model represents very well the general operation of the aquifer, and although the five wells selected for continuous monitoring are located in the boundaries of the model, their strategic location proves a realistic general pattern and water balance of the aquifer. Moreover, the model was calibrated with an average of the aquifer properties, that it is considered a good approach in this study area since the different geological materials are similar and it would not be expected to observe abrupt changes in the water table.

6.3. Contaminants remediation simulation

Based on the integration of results from field work, the particle-mass transport hydrogeological simulation and the limestone barrier geochemical simulation, it was possible to estimate the optimal life-time, geometry and location of a limestone drain as a possible remediation technique.

The effectiveness of the limestone barrier is subjected to the uncertainties produced by the scarce mineralogical and geochemical data of the mine tailings. The values used for defining amount of exchanger and the thermodynamic and rates of sorption processes in the geochemical model were obtained from literature data. For example, the amount of exchanger depends on the bulk density and its purity, which range from 1600 to 2350 kg/m$^3$ and 82-97%, respectively. However, the results of the byproducts precipitating and effluent properties agree with results of real limestone drains performance described by Ziemkiewicz et al. (2003). The authors found that in anoxic limestone drains the pH was raised after the drain to around 6.3 and there was precipitation of ferric hydroxide inside the drains, and this causes retard in the water flow leading to premature failure (Ziemkiewicz et al., 2003). In this study limestone drain simulation, the pH was raised to around 6.5 after the drain, there
was precipitation of ferric iron oxyhydroxide (goethite) and the life-time of 1m drain was about 2 years. The longevity of the drain was defined in this study as the time when the limestone will not produce enough alkalinity to maintain the concentration of contaminants below the concentration limits of very heavily polluted groundwater according to the Norwegian Environment Directorate classification (SFT, 1997).

From the tracking particles simulation, the minimum traveling time of the contaminants from each contamination focus site would highly depend on the porosity, for contaminants it will take from 9 to 30 years to cross the study area, from 4 to 15 years to cross the western mine tailings and from 1 to 5 years to cross the eastern mine tailings area. The groundwater horizontal velocity estimated from the tracking particles simulation, and used in the geochemical simulation, is in the same range, 0.04-0.35 m/d, as the calculated from the field measurements using Equation 4.24. This makes reliable the integration of field and computed data used to obtain the life-time of the limestone barrier.

Based on the hydrochemistry results, the deepest well, 10m, in the eastern mine tailings (well C7) is rich in HCO$_3^-$, which indicates that is minimally affected by pyrite oxidation. While the second deepest, 6 m is heavily polluted. This suggests that contaminants probably reach less than 9 m of depth in the aquifer. Therefore a 10m deep barrier would be enough to remove most of the contamination, but copper, which at that depth is still slightly above the permitted limit. Although no data of groundwater at that depth is available in the west mine tailings, conditions could be expected to be similar since the materials and the flow conditions are alike in both mine tailings areas.

On the other hand, a 1m barrier was simulated, although a sensitivity analysis of the width showed that a wider drain increases its effectiveness 10% per m wider, and therefore its longevity. Moreover, a wider drain implies that the groundwater has more time to flow through the drain, and therefore more time of ion-exchange and sorption process to remove the contaminants.

The proposed location of the drains in the lower part of the western and eastern mine tailings is based on the particles path-line obtained with the hydrogeological model, which shows that particles from the Main mine flow to the eastern mine tailings, and particles in the mine tailings areas flow directly to the Folla River. Therefore, the limestone drains will be needed.
before the groundwater reaches the river but closer to the river bed. If the treatment system is located before or in the middle of the mine tailings, the precipitation infiltrating the areas behind the drain will enhance the oxidation of sulfides which, in less quantity, will continue generating acid formation discharging into the river Folla. Based on the cartographical map, the longitude of the barrier, covering the mine tailings section, should be around 150 m in the west mine tailings and around 100 m in the east mine tailings (Figure 5.38).

6.4. Further work

The results obtained during this study present conclusions concerning hydrogeological flow, geochemical processes and potential remediation process with the use of an anoxic limestone barrier. To complete the study the following data or surveys would be necessary to improve the points where the models simulated presented greater uncertainty:

1) Obtain more data of the aquifer depth and geometry in the north and west area, by doing seismic surveys and drilling more wells. This possibly enables more representative regional inflow by the north border.

2) Observation wells in the north and in center of the study area. To obtain more hydraulic conductivity data (from slug test) and groundwater seasonal changes monitoring.

3) Studies of the vadose-saturated zone interactions, to obtain the actual amount of AMD forming and percolating to the aquifer in the mine tailings areas to validate the AMD formation simulation.

4) Mineralogical data of the tailings, such as the composition and amount of sulfides and silicates minerals present, so that the actual phases and amounts involved in the reactions can be simulated in the geochemical models. For example, doing a column test. From this the acid formation life-time of the mine tailings could be estimated.

5) Improve the geochemical simulation of the limestone barrier by adding all the contaminants present in groundwater, to obtain a better estimation of the limestone drain characteristics needed to successfully remove the heavy metals.
7. Conclusions

- The aquifer in the study site has a thickness in the range 30-40 m in the proximities of the river. The aquifer consists of two distinct geological layers, a heterogeneous sedimentary cover of glacial-fluvial origin on top of igneous-metamorphic bedrock.
- The hydraulic conductivity estimated from the slug test ranges from 0.02 to 6.8 m/d, with a mean value of 1.04 m/d for the whole study area. The values fit literature estimations for the glacial and fluvial deposits of the study area. There is no relationship between hydraulic conductivity value and depth or location of the well, which indicates heterogeneity of the materials.
- The groundwater flows from the Main mine area towards river Folla, NE-SSW direction. The mean hydraulic gradient in each transect is 0.03, 0.03 and 0.01. The study area receives recharge from the rain and groundwater regional inflow whose origin is the precipitation, around 1500 m$^3$/d, which is discharged to Folla river section.
- The seasonal changes in groundwater level and temperature are related to climatic conditions. As a general trend the groundwater level decreases from summer to winter and increases in spring, from ± 1m to ±0.2m. The temperature of groundwater decreases up to 6°C, from 6.5°C in summer to 0.5°C in spring, in the shallower wells. The temperature of the deeper groundwater is nearly constant during the period October 2014 – April 2015, only varied ± 1°C.
- The two-dimensional groundwater model is satisfactorily simulating the hydrogeological trends and changes. It has a mean absolute error of 1.41, for all stress periods and observation wells. The discrepancies between computed and observation data are generally less than one meter, except for well A1. It seems, however, that the no-flow boundaries in the East and West could not be a non-flow boundary.
- The results of the chemical analyses confirm that the weathering of sulfides is contaminating groundwater, and that the mine tailings are the source of contamination. The geochemical simulations estimated that 4.76e-03 mols/L of pyrite are dissolving. The groundwater chemistry in the study area can be classified as Ca-SO$_2$ type in the mine tailings and as Ca-HCO$_3$ type in the central forest area.
- The physiochemical measurements show variable distribution along the study area with poor correlation between parameters. The pH values range from 2.2 to 2.8, the
minimal value is in the Main mine area (pH around 2-3) and the maximum values are in the forest area, where the pH is neutral. pH shows a clear trend to increase with depth in transect A and C. The groundwater EC measured in the field averages 1.4 mS/cm, which indicates high content of TDS. The minimal value was measured in the forest area, <0.5mS/cm, and the maximum value in the Main area, 10.7mS/cm.

- The concentration of heavy metals varies considerably within the study area with minimal variations during the year. There is a concentration trend to decrease with depth. For instance, the concentration of \( \text{SO}_4^{2-} \) ranges between 7 and 12000 mg/L; Cu concentration ranges from 1 to 120000 µg/L. The distribution of the heavy metals reveals a similar pattern distribution for all of them, shallow wells M1, A3, A4, C2, C3, C5 and C6, located in the Main mine area and the mine tailings, are the most contaminated.

- Heavy metals concentration in groundwater is above drinking levels in all the wells in the mine tailings, except well A1. This well seems to be hydraulically connected with Gorbekken creek and therefore it has input of surface water that is proceeding from areas not affected by mining activity. Groundwater in the forest area presents low concentration of heavy metals, from 0.01 to 4 µg/L, except for zinc, >70 µg/L.

- The concentration of Cu in the river, 57 µg/L, is above the limit established by the Norwegian Environment Directorate. Groundwater is discharging, 0.8 m\(^3\)/d/m, to Folla river and based on the data collected during this study, 30% of the yearly contamination transport in the river comes from the groundwater in Follidar area.

- The geochemical simulation proved that an anoxic limestone drain would be a possible passive treatment to remove heavy metals from groundwater. More than 75% of Cu and Zn will be reduced from groundwater and the pH will increase from 4 to 6.5.

- Possible treatment systems should be located and focused in the two mine tailings zones, which are the main source of contamination discharging into the river. A 10 m deep and 150 m and 100 m long barriers in the west and east mine tailings, respectively, are suggested.

- Overall, the integration of field methods, the two-dimensional transient flow and mass transport model and the geochemical simulations demonstrate to be a good tool to characterize Folldal aquifer. And it shown to have convenient and functional applications to predict pollution distribution and evolution.
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APPENDIX A: Hydrochemistry
### A1-Table of physiochemical properties

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**Notes:**
- AMD: Anoxic Dissolved Oxygen
- T: Temperature
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### Notes
- AMD: AMD stream
APPENDIX B: PHREEQC simulations
B1 – Inverse modeling

Database file: phreeqc.dat

SOLUTION 1 # rainwater (from Appelo and Postma)
  temp 2.2
  pH 5.6
  units umol/l
  Na 13
  K 5
  Mg 5
  Ca 16
  Cl 11
  S(6) 31.1
  O(0) 865
  -water 1 # kg

SOLUTION 2 # groundwater well C6
  temp 7.5
  pH 4.5
  units mg/l
  Na 5.86
  K 7.42
  Mg 84.5
  Ca 198
  Cl 5.14
  S(6) 1040
  Fe 133
  Si 21.3
  Alkalinity 0.0

INVERSE_MODELING 1
  -solutions 1 2
  -uncertainty 0.4 0.4
  -phases
    Pyrite dis
    O2(g) dis
    Anorthite dis
    K-mica dis
    Chlorite(14A) dis
    Daphnite-14A dis
    Halite
    Gypsum
    Kaolinite
    Fe(OH)3(a) pre

  -multiple_precision true

PHASES
Halite
  NaCl = Na+ + Cl-
  log_k 0.0

Daphnite-14A
  Fe5Al8Si13O30(OH)8 +16.0000 H+ = 2.0000 Al+++ + 3.0000 H4SiO4
  + 5.0000 Fe++ + 6.0000 H2O
  log_k 52.2821

END
B2 – Kinetic pyrite oxidation

Database file: phreeqc.dat

SOLUTION 0 # rainwater (from Appelo and Postma)
  temp          2.2
  pH            5.6
  units         umol/l
  density       1
  Alkalinity    21
  Ca            16
  Cl            11
  K             5
  Mg            5
  Na            27
  S(6)          21
  O(0)          1  O2(g) -0.68
  C             1  CO2(g) -3.4
  -water        1  # kg

SOLUTION 1-10 # Initial groundwater in the column (C7)
  temp          4.4
  pH            6.8
  units         mol/l
  Ca            0.00183
  K             0.000177
  Mg            0.000206
  Na            0.0001775  charge
  Cl            6.9388e-005
  S(6)          0.00029
  Br            6.2576e-006
  N(5)          1.0806e-005
  Alkalinity    0.004

EQUILIBRIUM_PHASES 1-10
  Goethite      0.0 0.0
  Fe(OH)3(a)    0.0 0.0
  K-mica        0.0 0.1  # mols/l
  Gypsum        0.0 0.0

RATES # from appelo p. 456
Pyrite
  -start
  1  A = 120 * m0
 10 if SI("Pyrite") > 0 then goto 100
 20 fH = mol ("H+")
 30 fFe2 = (1 + tot("Fe(2)"))/1e-6
 40 if mol("O2") < 1e-6 then goto 80
 50 rO2 = 10^-8.19 * mol("O2")^0.5 * fH^-0.11
 60 rO2_Fe3 = 6.3e-4 * tot("Fe(3)")^0.92 + fFe2^-0.43
 70 goto 90
 80 rem
 81 rFe3 = 1.9e-6 * tot ("Fe(3)")^0.28 + fFe2^-0.52 + fH^-0.3
 90 rate = A * (m/m0)^0.67 * (rO2 + rO2_Fe3 + rFe3) * (1 - SR("Pyrite"))
100 save rate * time
  -end
KINETICS 1-10
Pyrite
   -formula FeS2 1
   -m0 1.32246 #mols of Pyrite/L
   -tol 1e-008
-step_divide 1
-runge_kutta 3
-bad_step_max 500

TRANSPORT
   -cells 10
   -shifts 10
   -time_step 1223990.208 # seconds
   -boundary_conditions constant flux
   -lengths 1
   -dispersivities 10*0.1 #1/10 length
   -diffusion_coefficient 1e-09
   -thermal_diffusion 2 1e-09
   -punch_frequency 2

USER_GRAPH
   -heading distance(m) pH
   -axis_titles "Depth (m)" "pH"
   -chart_title
   -init false
   -connect_simulations false
   -plot_concentration_vs_x
   -start
   10 graph_x dist
   20 GRAPH_Y EQUI("Goethite")
   -end
B3 – Limestone drain

Database file: phreeqc.dat

SOLUTION 0  # Acid mine drainage stream (V4)
  temp    7
  pH      2.6
  pe      4
  redox   pe
  units   mg/l
  density 1
  Ca      201  charge
  K       1.94
  Mg      220
  Na      4.36
  S(6)    3040
  Cu      73.3
  Fe      789
  Zn      42.2
  Cl      3.59
  -water  1  # kg

SOLUTION 1-20  #Initial groundwater in the column (C7)
  temp    5
  pH      6.8
  units   mol/l
  Ca      0.00184
  K       0.000177
  Mg      0.000206
  Na      0.000177  charge
  F      1.57908e-006
  Cl      6.9388e-005
  S(6)    0.000294
  Br      6.25751e-006
  N(5)    1.08056e-005
  Alkalinity 0.004

EQUILIBRIUM_PHASES 1-9  #Soil
  Pyrite  0 1.3
  K-mica  0 0.1
  Goethite 0 0.0
  Gypsum  0 0.0

EQUILIBRIUM_PHASES 10  #Limestone drain
  Calcite 0 15.44
  Gypsum  0 0.0
  Goethite 0 0.0

EXCHANGE 10
  CaX2 Calcite equilibrium_phase 1
  -equilibrate with solution 2

SURFACE_SPECIES
  Hfo_sOH + Zn+2 = Hfo_sOZn+ + H+
    -log_k  0.99
  Hfo_wOH + Zn+2 = Hfo_wOZn+ + H+
    -log_k -1.99
  Hfo_sOH + Cu2+ = Hfo_sOCu+ + H+
    -log_k  2.89
  Hfo_wOH + Cu2+ = Hfo_wOCu+ + H+
    -log_k  0.6  # table 10.5
SURFACE 10
  Hfo_wOH Goethite equilibrium_phase 0.1 1e5 #Hfo_wOH  0.1     600
200
  Hfo_sOH Goethite equilibrium_phase 0.001 #Hfo_sOH  0.01
  -cd_music
  -donnan 1e-008

EQUILIBRIUM_PHASES 11-20 #Soil
  Pyrite   0 1.3
  K-mica   0 0.1
  Goethite 0 0.0
  Gypsum   0 0.0

TRANSPORT
  -cells   20
  -shifts  21
  -time_step  666666.6667 # seconds
  -boundary_conditions constant flux
  -lengths  1
  -dispersivities  0.1
  -diffusion_coefficient  1e-09
  -thermal_diffusion   2 1e-09
  -punch_frequency    21

USER_GRAPH
  -headings dist pH
  -axis_title Distance(m) pH
  -plot_concentration_vs x
  -connect_simulations false
  -start
      10 graph_x dist
      20 graph_y -la("H+")
  -end