MiMi - Field studies and characterisation

State-of-the-art-report

MiMi 1998:3

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Summary

Complex biogeochemical and physical processes determine the mobilisation and dispersion of contaminants from disposed untreated mining waste as well as remediated waste deposits. Field studies are necessary for understanding the problem of acid mine drainage, and to be able to predict future problems. Field studies of geochemical fluxes and massbalances of metals and other elements and compounds on carefully chosen sites are necessary to quantify the importance of key processes such as weathering, adsorption, desorption, precipitation of secondary minerals, rate of dispersion of contaminants and dispersion paths. Field studies are also necessary to quantify the release of metals and other elements from deposits of mining waste to recipients. Thorough field studies form the necessary basis for predictive modelling. You have to be able to model the present situation to be able to predict the future. Field studies are necessary to study the efficiency of various remediation methods.

There are numerous cases with field studies of acid producing mine waste, and of such waste remediated by various methods. If the MiMi program focuses on one or two sites where important technologies such as dry and wet covers are applied, with a concentrated effort of field studies, laboratory studies and modelling in such a way that a detailed understanding of key processes is achieved, it is possible to move the knowledge front considerably forwards.

To understand and quantify key processes, the MiMi field studies should be focused on detailed studies of various types of interfaces. These could be water solid interfaces such as between pore water and solids, between waste and water cover, and between suspended matter/colloids and water. Other important interfaces are those between the various layers of dry covers, between the dry cover and the waste, and between various zones of oxidising waste.

Many field studies have the character of monitoring with analyses of rather few metals and pH. The field studies in the MiMi project should have good resolution in both time and space. Best available techniques for sampling and analyses should be used. Detailed time series are necessary for many sample types, especially when sampling various types of water and when sampling pore gas.

At the chosen Mimi site, it is necessary with a thorough mineralogical, chemical and microbiological characterisation. The mineralogical and chemical composition of unweathered waste, waste effected by secondary processes, and secondary mineral precipitations, should be determined. The various important species of microbes should be identified, and their abundance quantified. The properties of the dry cover used at the MiMi site should also be carefully examined. This fundamental characterisation is the basis for all projects and subprojects in the MiMi program.

Much research effort has been spent in the Canadian MEND program and elsewhere to develop static and kinetic tests for mine waste. This should therefore not be an important project in the MiMi program. However, recommended versions of standard methods such as ABA and Humidity Cells, should be applied in the “Characterisation” project, to be able to compare the MiMi site with results obtained world-wide, and to compare with the more sophisticated studies of weathering rate that will be performed in the project “Laboratory studies of key processes”. It is important to compare these results to evaluate the usefulness of these common standard methods.

Although it is often claimed that mineralogical studies are important for characterisation, one gets the impression that the mineralogy is not used as it could be when evaluating results from static and kinetic tests. This should therefore be an important task for MiMi. This is important to be able to achieve one of the major goals with the MiMi project: to be able to predict
environmental effects from mining of a specific ore from the composition of an ore and its wall rocks already during exploration.

It can be concluded that for the MiMi program:

• research should be focused on one or two common sites
• at the chosen site, it is necessary with a thorough mineralogical, chemical and microbiological characterisation
• the field studies should be focused on detailed studies of various types of interfaces such as between pore water and solids, between waste and water cover, between suspended matter/colloids and water, and between dry cover and the waste.
• the field studies should have good resolution in both time and space.
• research during the first years should be focused on the use of dry covers, water covers, and raised groundwater tables as remediation methods.
1 Introduction

It has long been known that drainage waters from waste rock and mill tailings produced by mining often have low pH and high concentrations of metals. The Swedish biologist Carl von Linné realised in the 18th century that the Falu River in south central Sweden was polluted due to the mining at Falu copper mine (Swedish Environmental Protection Agency 1993). However, it was not until relatively recently it was understood that such acid mine drainage (AMD) in many cases may cause damage to the environment in mining areas. Jacks (1976) first described the potential problem with AMD in Sweden in a systematic way. The Swedish Environmental Protection Agency started the research program "Waste deposits from the mining industry" in 1983, and the program ran to 1988. It was followed by a later program financed by the Swedish Waste Research Council (AFR) (Gustafsson 1997). Only ten years ago, AMD was beginning to be recognised as a serious environmental problem in mining areas in Canada (Feasby et al. 1997). The situation is similar worldwide. During the last two decades, and in particular the last one, considerable efforts from mining companies, authorities and research organisations have been spent on the understanding of the fundamental processes in weathering deposits of mining waste, and to develop cost efficient technology to prevent and control AMD. The Canadian MEND (Mine Environment Neutral Drainage) program, which has been running from 1989, probably represents the most concentrated research effort with a systematic collaboration between mining industry and government, but also in for instance USA, Australia, Norway, Sweden and Germany considerable AMD-related research has been performed.

Oxidation of pyrite and/or pyrrhotite causes the formation of AMD with subsequent release of metals from sulphides such as chalcopyrite, galena and sphalerite, and also from other minerals. The fundamental reactions of oxidation of these iron-sulphides and subsequent reactions with other minerals are relatively well known, and today available also in textbooks (e.g. Jambor & Blowes 1994; Morin & Hutt 1997). The initial oxidation of sulphides needs oxygen and water. After the initial oxidation by oxygen the oxidation may continue even if oxygen is absent. Fe(III) is then the oxidant. The oxidation of pyrite is usually described by reactions 1-4:

\[
\text{FeS}_2 + 7/2\text{O}_2 + \text{H}_2\text{O} \rightarrow \text{Fe}^{2+} + 2\text{SO}_4^{2-} + 2\text{H}^+ \quad (1)
\]

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

\[
\text{Fe}^{3+} + 3\text{H}_2\text{O} \rightarrow \text{Fe(OH)}_3 + 3\text{H}^+ \quad (3)
\]

\[
\text{FeS}_2 + 14\text{Fe}^{3+} + 8\text{H}_2\text{O} \rightarrow 15\text{Fe}^{2+} + 2\text{SO}_4^{2-} + 15\text{Fe}^{2+} \quad (4)
\]

However, there are still considerable gaps in the understanding of sulphide oxidation during differing physiochemical conditions, in waste deposits with different mineralogical composition and different grain size distributions, and in waste deposits where remediation methods such as dry cover of water cover have been applied. Much also remains to be learnt about the complex relationships between sulphide mineral grain surfaces in contact with water. The sulphides as well as other minerals may during certain physiochemical conditions have considerable capability to adsorb heavy metals and other elements on their surfaces (e.g. Jean & Bancroft 1986; Lövgren 1990; Al et al. 1997). Adsorption-desorption reactions may thus be important for the long-term behaviour of remediated mine waste deposits.

Complex biogeochemical and physical processes determine the mobilisation and dispersion of contaminants from disposed untreated mining waste as well as remediated waste deposits. Many studies have shown that field studies are necessary for understanding the problem of acid mine drainage, and to be able to predict future problems (e.g. Dubrovsky et al. 1985; Blowes &
Field studies of geochemical fluxes and massbalances of metals and other elements and compounds on carefully chosen sites are necessary to quantify the importance of key processes such as weathering, adsorption, desorption, precipitation of secondary minerals, rate of dispersion of contaminants and dispersion paths. Field studies are also necessary to quantify the release of metals and other elements from deposits of mining waste to recipients. Thorough field studies form a necessary basis for predictive modelling. You have to be able to model the present situation to be able to predict the future.

Field studies are necessary to study the efficiency of various remediation methods.
2 Characterisation

The prediction of mine drainage quality within a particular geologic setting depends on an accurate characterisation and understanding of the potential of the waste to produce acidity, and to produce alkalinity. A detailed knowledge of the mineralogical composition is necessary. In the case of partly oxidised material, it is also necessary to identify secondary minerals, and to quantify their occurrence. In this section, various types of laboratory tests used for characterisation of mine waste are discussed, whereas grain-size distribution, deposit size, deposit type, hydrological characteristics etc. are discussed in the section "Field studies".

Both mining companies and regulators need reliable, relatively fast and not too expensive methods to characterise ore and wall rocks, in order to be able to predict possible AMD in potential new mines, and to be able to plan the methods needed during and after mining for environmental protection. This has been one of the major tasks of the Canadian MEND program (Feasby 1997). Both static and kinetic tests have been used at various scales. Static tests are designed to give a simple and inexpensive indication of the potential of a waste to generate AMD, whereas kinetic tests are performed to examine the oxidation and weathering characteristics as a function of time (Lawrence & Day 1997). Both static and kinetic tests should be performed on samples from a specific site.

2.1 Acid Base Accounting

Acid Base Accounting (ABA) is a common static test method to roughly determine the balance between acid producing and buffering capacity of a material. The original test procedure was suggested by Sobek et al. (1978). Total sulphur content as wt. % S is converted to maximum potential acidity (AP), expressed in units of kg CaCO3 per tonne material. The conversion is based on the assumption that all S occurs in pyrite, and that the pyrite is completely oxidised to sulphate and solid Fe(OH)3. It is also assumed that H+ produced by pyrite oxidation is incompletely neutralised by CaCO3 to pH < 6. Net neutralisation potential (NNP) is determined by treating a sample with HCl. The mixture of sample and acid is heated to 90°C. After 30 minutes the mixture is cooled to room temperature, and the amount of unconsumed acid is determined by titration with NaOH to pH 7. NNP is also expressed as kg CaCO3 per tonne material. The acid producing potential (APP) of the sample is then calculated as AP-NNP. A material with negative APP could potentially produce acid drainage water. In British Columbia, recommended procedures for static and kinetic testing have been developed (Price et al. 1997). Several parameters should be included that were not included in the original test set up suggested by Sobek et al. (1978). Complete ABA analyses should include total-sulphur, acid leachable sulphate-sulphur, acid-insoluble sulphate-sulphur, sulphide sulphur, bulk neutralisation potential, carbonate-carbon and pH (Price et al. 1997). Otherwise the interpretations of AP and NNP can be erroneous in some lithologies.

Lawrence & Wang (1997) have shown that NNP values for a sample may show large variations depending on the particular test procedure used, and on various test set ups within a specific procedure. NP is often overestimated in ABA tests. The modified ABA test referred to above is to prefer.

ABA tests are used on both waste rocks and tailings. Waste rock samples must be crushed to reduce the grain size. For an efficient waste management, ABA tests should be performed on ore and wall rock samples taken already during exploration. Static tests should always be used in combination with kinetic tests on some of the samples. Potential costs for treatment of waste and/or remediation of waste deposits should be included in the decision to open a mine or not.
2.2 **Humidity Cells test**

Humidity Cells test is the most common kinetic method for environmental impact assessment of mining waste. Previously 200 g of each sample were placed in plastic boxes with an entrance port for filtered compressed air, and an exit port for the air. The test starts with that the samples are moistened with distilled water. After that dry air flows through the boxes in three days, and then humidified air flows through the boxes in three days. On the seventh day, 200 ml distilled water is added to each sample. After one hour the water is decanted and filtered by using filters with a pore size of 0.45 µm. Amount of filtered water, pH and electric conductivity are measured. The water is sampled for eventual further analysis. It is now recommended that one kg of sample is used, and that 250 to 500 ml of water is added each cycle (Lawrence & Day 1997). Dissolved metals should be analysed, preferably both major and trace elements. Otherwise it is difficult to interpret which reactions are controlling the geochemistry of the leachates. Rock samples should be crushed to minus 6 mm. For tailings samples usually having low permeability for water and oxygen, a different design with larger diameter and shallower depths of the cells should be used to expose a greater surface area for reaction.

The number of cycles needed to characterise a specific deposit varies, but in most cases several months or even years are needed. It is often difficult, or even impossible, to scale up Humidity Cells tests to real waste deposits, but they give an understanding of the kinetic scenario that could be expected in a waste deposit tested. It is possible that Humidity Cells tests during favourable circumstances could be used to estimate the amount of sulphides oxidised and subsequent release of metals. Ljungberg & Öhlander (1996) found that this worked in a tailings deposit which is exposed to the atmosphere, which is moist the whole year round, and which is not buffered by carbonates.

Kinetic tests should always be used in combination with static tests of the same samples.

2.3 **Other methods**

ABA, the original test or modified ABA, and Humidity Cells test are the most commonly used static and kinetic tests, respectively, but several other tests have been developed (Lawrence & Day 1997). In Australia, the Net Acid Generation (NAG) static test has been developed in a project sponsored by the Australian Mining Industry Research Organisation (AMIRA), and it seems to work well (Miller et al. 1997). Hydrogen peroxide is used as an oxidant in the NAG test. A kinetic NAG test has also been developed.

Various types of column tests, accelerated kinetic leaching, bacterial oxidation test and Soxhlet extraction are other kinetic tests that have been developed (Lawrence & Day 1997). Pilot scale leaching tests are also used. Common for all kinetic tests is the scale problem.

2.4 **Mineralogical studies**

A good knowledge of the mineralogy is a foundation to all predictive and descriptive models, and is the most important characterisation of mine waste. The results of integrated mineralogical and hydrogeochemical investigations of a tailings impoundment or a waste rock deposit can lead to conclusions whose sum is greater than that derivable if the two studies were conducted independently (Jambor 1994). Oxidation of sulphidic waste gives rise to a whole variety of secondary minerals as well as producing acid- and metal-rich leachates. The mineralogy may give indication of which reactions that are regulating the dissolved concentrations of elements in the mine waste. Mineral studies also show if the minerals are affected by weathering. For example, similar secondary enrichments as those occurring in porphyry copper deposits that
have been affected by gossan forming processes and supergene enrichment reactions (e.g. Thornber 1985), are also occurring in mine waste (Boorman and Watson 1976, Blowes and Jambor 1990). Knowledge of the mineralogy is also important when deciding if the waste is a potential acid- and metal-producer. Lin (1996; 1997) has demonstrated that a detailed knowledge of the primary mineralogy as well as of secondary precipitations is necessary to be able to understand the processes occurring in mine waste deposits. In other words, to be able to model the future geochemical processes, the mineralogy must be known in detail.

The sampling for mineralogical studies must be carefully performed. Premature oxidation and crystallisation of secondary minerals due to drying of trapped porewater must be avoided. This can be avoided by careful sampling, storing the samples in an oxygen free atmosphere and cleaning of the collected samples.

Numerous techniques are available for investigating gangue minerals, ore minerals and their associated secondary minerals in mine waste. The most common technique is still the petrographic microscope, which is a simple and inexpensive method. Common microscopy gives useful information of the mineralogical composition, texture, and grain size distribution. Secondary minerals are often difficult to identify with this method, since they often form very thin coatings and often are poorly crystallised. Other methods include X-ray diffraction, which is of limited use when the samples are poorly crystallised or for studies of minor constituents (Bigham 1994), Mössbauer Spectroscopy, Infrared Spectroscopy and electron optical techniques such as TEM (transmission electron microscopy) or SEM (scanning electron microscopy).

### 2.5 Conclusions for the MiMi project

At the chosen Mimi site, it is necessary with a thorough mineralogical, chemical and microbiological characterisation. The mineralogical and chemical composition of unweathered waste, waste effected by secondary processes, and secondary mineral precipitations, should be determined. The various important species of microbes should be identified, and their abundance quantified. The properties of the dry cover used at the MiMi site should also be carefully examined. This fundamental characterisation is the basis for all projects and subprojects in the MiMi program.

Since so much research effort has been spent in the MEND program and elsewhere to develop static and kinetic tests for mine waste, this should not be an important project in the MiMi program. However, recommended versions of standard methods such as ABA and Humidity Cells, should be applied in the "Characterisation" project, to be able to compare the MiMi site with results obtained worldwide, and to compare with the more sophisticated studies of weathering rate that will be performed in the project "Laboratory studies of key processes". It is important to compare these results to evaluate the usefulness of these common standard methods.

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7
3 Field studies

The definition of field studies applied here is that various types of analyses and measurements performed in situ in the field, or performed on samples taken in the field, is considered as field studies. Studies of artificial samples, or samples which are synthesised or constructed in the laboratory, is not considered as field studies. This is perhaps not a perfect definition since several subprojects to the MiMi project "Laboratory studies of key processes" include analyses of real samples taken in the field. Studies of adsorption on mineral surfaces on real samples are an example of that. However, since the structure of the MiMi program is built upon a close collaboration between the researchers participating in the various projects and subprojects, the definition is useful despite the overlap between different MiMi projects. Most of the discussions below are devoted to biogeochemical studies and processes, but it must be emphasised that a thorough hydrological characterisation is necessary for each study site. For a mine waste deposit, the hydrological situation must be known to be able to quantify the amount of elements released by sulphide oxidation and weathering, the amount being in transport within the deposit, and the amount discharged to drainage water (e.g. Robertson 1994; Ritchie 1994).

In numerous case studies of AMD, the various types of water samples have only been analysed for pH, sulphate and metals such as Cu and Zn. However, to be able to understand which acid producing and buffering reactions that occur in oxidising sulphidic waste, it is important that both major and trace elements are analysed. Anions and TOC shall also be analysed. pH, conductivity, oxygen saturation and temperature shall be measured in field if possible. Hydrological parameters such as permeability distribution and capillary retention shall be determined on solid materials. Water chemistry shall be studied in time series to determine the importance of seasonal variations. This aspect is missing in most field studies, or the seasonal variations or effects of snowmelts and heavy storms are studied with too poor time resolution.

A fundamental requirement for the monitoring and study of mine waste is the ability to collect high-quality samples of solid, dissolved and gaseous phases. It is through these undisturbed and uncontaminated samples that the chemical and physical characteristics and the geochemical behaviour of the waste can be determined.

Solid material such as tailings and waste rock may be sampled through a number of different techniques. Core sampling performed by drilling is common (e.g. Boorman and Watson 1976, Ljungberg & Ohlander 1997). Other methods include digging of shafts or pits either by hand or by excavator. Core sampling is probably best since it enables sampling of undisturbed samples and it is possible to keep the samples from contact with oxygen, thereby avoiding oxidation. The solid phase is important to study since it may give information of the mineralogical composition and the complex reactions between solid and dissolved phase.

The dissolved phase in mine waste deposits includes porewater, groundwater and surface water, and gives information of the processes that occur within a waste dump. Dissolved phase is commonly extracted from the solid phase through a number of techniques such as centrifugation, pressurised consolidation, porewater displacement (MEND project 4.1.1). The dissolved phase may also be sampled with lysimeters, groundwater pipes such as piezometers or BAT pipes, or by peepers (dialysis membrane arrays) (e.g. Yanful & St-Arnaud 1992, Pedersen et al. 1997, Torstensson & Petsonk 1988). Which method to be used depends on the question at issue, and thus to which kind of sample to be collected. It is highly important for all methods to prevent the samples from being contaminated by oxygen. Whenever possible, suspended phase should be separated from the dissolved phase by filtering, or by some other method.
Poregas may be analysed by field-installed oxygen probes, or poregas may be sampled from tubing, pipes or pore gas lysimeters. The gaseous phase is then usually collected with a syringe for later analysis or immediately analysis with an autoanalyzer (MEND project 4.1.1).

The sampling methods described here are those commonly used for investigations of various aspects of mine waste. In basic research, also other methods are used for geochemical field studies. Relevant such methods are described in the sections below. Best available techniques will be used and further developed in the MiMi program. One example is the method to measure oxygen distribution with sub-mm resolution in sediments, as described in the section "Suspended sediment-water interactions". This will be used in studies of the oxygen distribution below the surface of water covered tailings in one of the MiMi projects.

### 3.1 Water/solid interactions

The sediment-water interactions in natural waters can be divided into two major groups. Firstly, there are interactions between dissolved elements and particles-colloids in the free water column, and secondly, there are interactions in the deposited sediment. To understand geochemical reactions in the deposited sediment and element fluxes from/to sediments it is important to understand the nature of the settling particles-colloids.

#### Suspended sediment-water interactions

Dissolved trace element concentrations in natural waters are believed to be controlled by the transfer of dissolved species into particulate forms. During this process, colloidal intermediates in the size range 1 nm-1µm play important roles (Morel & Gshwend 1987; Vezina & Cornett 1990; Dai et al. 1995). According to the "Brownian-pumping" model by Honeyman & Santchi (1989), dissolved elements rapidly adsorb to colloids, which then aggregate more slowly into particles that can be collected on 0.45 µm filters. This implies that the rate of colloid aggregation into macroparticles controls the transfer of elements into the particulate phase, before their removal from the water column by sedimentation. Knowledge of the relative importance of the various interactions between particles-colloids and the dissolved phase, and of the composition and characteristics of the particle surfaces in natural waters, is though limited. Current modelling capabilities in the water-colloid-particle system decrease rapidly in the sequence element-inorganic ligands, element-organic ligands and element-particles (Turner 1995). Since many ore related trace elements have a high affinity for the solid phase, to understand their overall fate, we need to better comprehend the dynamic colloid and large particle processes occurring in waters around mining areas.

Several papers have emphasised the role of colloidal material for the biogeochemical cycling of organic C, trace elements, and radionuclides in the oceans (Benner et al. 1992; Wen et al.1996; Moran & Buesseler 1992; 1993). Trace metals are also commonly found associated to colloidal material in estuarine and riverine waters (Sigleo & Heltz 1981; Whitehouse et al. 1990; Benoit et al. 1994; Martin et al. 1995, Viers et al. 1997). There are several studies of colloids in ground water, with special emphasis on the transport of radionuclides (Degueldre & Wernli 1986; Degueldre et al. 1989; Vilks et al. 1991; Miekeley et al. 1992). Relatively little work has been done on the cycling (particle-colloid-element interactions) of trace elements, other than Mn and Fe, in lacustrine systems (Balistrieri et al. 1994; Sigg et al. 1996). There are few (if any) studies of the colloid fraction in surface waters close to mining areas.

A major barrier to understanding the role of the suspended solid fraction in transport processes has been an inability to measure the concentration and composition of this material, particularly in the colloidal size range. There are essentially three different types of techniques that have
been used to address this problem: Filtration and ultrafiltration (Buffle et al. 1992), light scattering methods (e.g. PCS=photon correlation spectroscopy) and TEM-SEM studies combined with new non-perturbing preparation techniques for colloidal particles directly in the field (Perret et al. 1991).

Resulting from its potential to isolate significant amounts of aquatic colloids in a reasonable time, the use of cross-flow ultrafiltration, CFF (one type of FFF=Field Flow Fractionation), has accelerated over the past five years and emerged as the dominant technique employed by marine chemists and researchers in freshwater systems studying colloidal processes. CFF can be used both to measure the size distribution of the colloidal material, and to collect colloidal material of well-defined sizes for elemental analysis to establish the composition of the matrix and associated trace elements.

CFF has unfortunately not until recently been associated with a critical evaluation of the applicability to natural aquatic systems. Recent testing has shown that the results obtained are highly instrument- and operator-dependent and poor mass balances for many bulk constituents are evident (e.g., Buesseler et al. 1996; Reitmeyer et al. 1996). Mechanistic studies of specific systems using standard colloids under controlled and varying conditions have indicated that both colloid sorption and concentration polarisation may be significant and cause artificial fractionation of the colloidal pool (Gustafsson et al. 1996). It has been shown that in organic rich fresh waters much of the alkali and alkaline earth elements are retained by ultrafilters, but Viers et al. (1997) have suggested that this is an artefact caused by the filtration procedure. This indicates that CFF must be carefully evaluated in each system under investigation. If this is done, CFF must be regarded as an "important tool" for the studies of colloids in natural waters.

The photon correlation spectroscopy (PCS) technique has the advantage that it may be used to detect and characterise particulate matter with respect to particle size and size distribution in suspension without invasive procedures (Ledin et al. 1993; Ledin et al. 1994). Some PCS instruments are equipped with a variable-angle detector enabling the collection of light scattering data in the angular range 10 to 155°. The advantage with data collection at other scattering angles than 90° is the possibility of detecting small particles that may exist in suspension together with larger particle fractions. Ledin (1993) used a 90° angle PCS instrument in studies of colloids in freshwater in southern Sweden and noted a large variation in the signal to mass relationship when the PCS technique was used for quantification of colloid concentrations in natural samples. This problem is partly solved with a variable-angle instrument.

To reveal the usually complicated nature of the colloidal fraction it is recommended to integrate the use of CFF, PCS and TEM-SEM in studies of colloids in natural waters.

**Deposited sediment-water interface studies**

Diagenesis is the sum total of physical, chemical and mineralogical changes taking place in sediments (and thus also in mine tailings) during and after their deposition. In natural waters, the sharpest gradients in physical, chemical and biological properties are found at the sediment - water interface, often as a result of early diagenetic processes. At the sediment - water interface, chemical as well as microbiological transformation processes are responsible for the cycling of elements between water and sediments. Early diagenetic redox transformation reactions of carbon, oxygen, nitrogen, manganese, iron and sulphur have important consequences for trace element cycling. For example, trace metals can be remobilized slightly below the interface due to the dissolution of trace metal carrier phases such as Fe - Mn oxides (Shaw et al. 1990; Davison 1993). This may cause a rapid back-diffusion of trace metals into the overlying water
column. Such benthic efflux may be an important flux component in mass balance calculations for flooded tailings dams and lakes.

Pore water data are a sensitive indicator of early diagenetic reactions involving oxygen and trace metals, and can be used to identify reaction zones where metals are remobilized close to the sediment - water interface (Shaw et al. 1990). The precise location of reaction zones requires a spatial resolution on the millimetre scale, which is beyond the capabilities of the standard methods that until now have been used for pore water collection (Bufflap and Allen 1995). However, a new and promising technique has been developed for high-resolution measurements of trace metal concentrations in pore waters (Zhang et al. 1995).

Diffusion exerts a major control on the transport of dissolved species in the sediment- water interface region (Berner 1980; Santschi et al. 1990). The process of diffusion of dissolved species in sediments can be described by the general diagenetic equation

\[
R \cdot \frac{\partial C}{\partial t} = D_e \cdot \frac{\partial^2 C}{\partial z^2} - k \cdot R \left( C_e - C_{eq} \right) + w_0 \frac{\partial C}{\partial z}
\]

where \( C_z \) and \( C_{eq} \) are the pore water concentration at depth \( z \) and at equilibrium, respectively (mol . m\(^{-3}\)), \( D_e \) is the effective diffusivity = \( D_p \cdot \varnothing \), where \( \varnothing \) is the sediment porosity, \( D_p \) is the pore diffusivity (m\(^2\) . s\(^{-1}\)) in sediments which is calculated from the diffusivity in bulk water, \( D_0 \), according to \( D_p = D_0 / \tau^2 \), where \( \tau \) is tortuosity, \( R \) is the chemical retention factor = \( \varnothing + (1 - \varnothing) K_d \rho_s \), where \( K_d \) is the distribution factor describing the concentration ratio at equilibrium between solid and liquid phase (m\(^3\) kg\(^{-1}\)), \( \rho_s \) is the mineral density of the sediment (kg m\(^{-3}\)), \( t \) is time (s), \( z \) is depth (m), \( k \) is a first-order reaction rate constant (s\(^{-1}\)) and \( w_0 \) is the Darcy advection rate in the bioturbated layer (m\(^3\) . m\(^{-2}\) . s\(^{-1}\)) (Santschi et al. 1990). The general diagenetic equation forms the basis for mathematical modelling of early diagenetic processes (Berner 1980).

The diffusive flux, \( J \) (mol . m\(^{-2}\) . s\(^{-1}\)), of a dissolved species across the sediment - water interface can be calculated by using Fick’s first law:

\[
J = \varnothing \cdot D_p \cdot \frac{dC}{dz}
\]

where \( \varnothing \) is the sediment porosity (m\(^3\) . m\(^{-3}\)), \( D_p \) is the effective sediment diffusion coefficient (m\(^2\) . s\(^{-1}\)) and \( dC/dz \) is the pore water concentration gradient across the interface. Depending on the pore water concentration gradient across the interface, dissolved species may diffuse from the sediment into the water column (e. g. Westerlund et al. 1986) or, if concentrations in the bottom water exceed those in the pore water, downward diffusion from overlying water to the sediment may occur (e. g. Carignan & Nriagu, 1985).

The water immediately above the sediment constitutes a diffusive boundary layer (DBL) with submillimetre - millimetre thickness. In this layer ions move only by means of molecular diffusion, and the DBL is the main resistance to total flux for substances with reaction rates equal to or greater than the rate of transport across this layer. This appears to be the case for oxygen (Rasmussen & Jørgensen 1992).

The ratio of particle mass and surface area to solution volume in mine tailings and natural sediments is large. As a consequence the composition of pore waters is a sensitive indicator of the reactions of minerals with sediment pore solutions. Diagenetic processes occurring in the sediment - water interface region can be described quantitatively in terms of mathematical rate expressions (Berner 1980). Pore water data have been used to model the early diagenetic
reactions of e.g. iron (Wersin et al. 1991), manganese (Thamdrup et al. 1994) and trace elements (Smith & Comans 1996; Widerlund 1996).

In coastal sediments, oxygen microelectrode data have been used to model oxygen consumption by assuming zero-order or first-order consumption kinetics close to the interface (Rasmussen & Jørgensen 1992). Such modelling can be applied also in studies of flooded mine tailings, where oxygen consumption is a key parameter in the oxidation of sulphides.

Oxygen microelectrodes are well functioning tools with which it is possible to analyse the distribution of oxygen close to the sediment-water interface at < 100 µm resolution (Revsbech & Jørgensen, 1986). The mechanisms of oxygen consumption can thus be experimentally analysed in order to understand the regulation of oxygen close to the interface. Oxygen microelectrodes can also be used to measure the thickness of the diffusive boundary layer (e.g. Gundersen & Jørgensen 1991), which is thought to be an important parameter regulating benthic efflux of many elements.

Common techniques for studies of oxygen consumption at the sediment-water interface include oxygen uptake measurements in laboratory-incubated sediment cores or in situ measurements using benthic flux chambers. In recent years, oxygen microelectrode data have been used to determine the diffusive oxygen flux into sediments. Under certain circumstances this diffusive oxygen flux can be used as a measure of the oxygen consumption in the sediment (Rasmussen & Jørgensen 1992).

The primary concern with obtaining sediment pore water is finding an extraction technique that will produce samples that best represent the natural, in situ environment. To accomplish this goal, the technique that is used should have the lowest potential for producing sampling artefacts. The four most commonly used methods for pore water collection are squeezing, centrifugation, dialysis and suction filtration (Bufflap & Allen 1995). Each method has its advantages and disadvantages, and none of these methods permits high resolution sampling on a millimetre scale. In addition, sample contamination by fine particles (colloids) is important in all methods that are based on the use of filters (squeezing, centrifugation and suction filtration).

Trace metal concentrations in pore waters have recently been measured at high resolution (1.25 mm) by the newly developed technique of diffusive thin films (Zhang et al. 1995). Such high resolution is necessary in order to obtain reliable estimates of dissolved trace metal gradients across the sediment-water interface.

Benthic fluxes of dissolved species can be determined in a number of different ways (Table 1). The most reliable and direct method is in situ benthic chambers. In recent years, the fluxes of many chemical species have been determined in this way for a number of different environments (e.g. Elderfield et al. 1981; Westerlund et al. 1986; Devol 1987; Glud et al. 1995). Although different chamber designs are in use, all are modifications of the same basic prototype - a stirred bell jar with sampling ports.

The most up-to-date, in situ chamber technique is based on the use of autonomous vehicles or "landers" (Tengberg 1997). With this technique, an autonomous, unmanned vehicle that descends by gravity is used for the chamber deployment on the bottom. After deployment, in situ experiments are carried out by the use of various types of sensors (e.g. temperature, oxygen, pH, resistivity, sulphide and nitrate). Fluxes across the sediment-water interface are measured by time interval sampling of the water inside the chamber. The benthic chamber technique can be combined with high-resolution measurements of pore water concentrations according to the method of Zhang et al. (1995). In addition, sediment sampling can be performed within the chamber.
Table 1 Methods for determining rates of sediment-water exchange and benthic fluxes.

<table>
<thead>
<tr>
<th>Method</th>
<th>Problems</th>
</tr>
</thead>
<tbody>
<tr>
<td>Basin or lake mass balance</td>
<td>Provides limited insight into mechanisms and reaction sites in water vs. those in sediments</td>
</tr>
<tr>
<td>Basin or lake concentration profiles</td>
<td>Mechanisms and reaction sites in sediments that are responsible for concentration increases cannot be deduced</td>
</tr>
<tr>
<td>Core incubation</td>
<td>Decompression, degassing, possible changes in temperature, pH, redox and turbulence conditions in sediments, possible alteration of biological processes</td>
</tr>
<tr>
<td>Pore water gradient method</td>
<td>Difficult to obtain reliable estimates of concentration gradients across the interface. Assumptions of linear gradients near the interface may be invalid because of heterogeneous composition of surface sediments. Higher concentrations of iron and/or manganese at the sediment-water interface add another transport resistance to fluxes of reactive species. Enhancement of benthic exchange by macrofauna activities</td>
</tr>
<tr>
<td>In situ benthic flux chambers</td>
<td>Existence of diffusive boundary layer resistance above the sediment-water interface requires that turbulence levels inside benthic chambers equal those outside the chamber for the measurement of species that react within the upper few millimetres of sediments. Regulation of oxygen concentration is required inside chamber if consumption rates are high</td>
</tr>
</tbody>
</table>

3.2 **Role of microorganisms**

A full understanding of mine waste disposal environments cannot be achieved until microbial processes are included in models, theories, interpretation of results etc. This is because bacteria catalyse many reactions that would otherwise be very slow, or kinetically hindered. Microbial reduction of iron (III), manganese (IV) and sulphate to iron (II), manganese (II) and sulphide, respectively, has important implications for the speciation of metals in mining waste deposits. Obvious reasons are as follows: Fe(III) and Mn(IV) oxides strongly adsorb a wide variety of trace metals. Further coprecipitation of toxic metals with Fe(III) and, to a lesser extent, with Mn(IV) is common in mining waste drainage water.

There is substantial evidence for release of trace metals into the interstitial water of aquatic sediments, into anoxic bottom waters, or to groundwater as the results of bacterial Fe(III) and Mn(IV) reduction. This remobilization of heavy metals may also occur in anaerobic mine waste.
Sulphide production by bacteria takes place in anaerobic environments, and if chalchophile metals are present they will be immobilised as metal sulphides.

Consequently, Fe and Mn reducing bacteria tend to mobilise metals while sulphate-reducing bacteria immobilise them. Speciation of metals in mining waste deposits will then be strongly dependent on which bacterial groups are present and active.

It is, therefore, important at the main MiMi site to characterise the dominant bacteria, and to quantify their activity and importance. The achieved knowledge should be used to complete the models of metal speciations in these deposits.

Ledin & Pedersen (1996) have recently reviewed the role of microorganisms in the context of mining waste.

### 3.3 Role of organic matter

Organic matter such as humic acids may have a dominating effect on the chemical state and mobility of trace metals. If a vegetation cover is established on mining waste, e.g. tailings, or if vegetation is established on a dry cover of mining waste, or if organic matter (i.e. organic waste) is used in the construction of a dry cover, organic acids will be produced. If these come into contact with the waste material, they will affect the mobility of metals in various ways depending on the physiochemical environment.

Fundamental aspects of how organic matter affects the mobility of metals in various environments relevant for the MiMi programme, and the importance of the formation of complexes between organic matter and metals and the interactions of these complexes with minerals, should be studied in the project "laboratory studies of key processes". The role of organic matter must, however, also be studied and quantified at the specific environments of the MiMi field sites. Since the field studies aim at establishing a detailed mass balance and quantification of key processes at the key localities, the role of the organic material must be evaluated.

When mining tailings are deposited they often contain considerable amounts of flotation chemicals such as xanthates. Their significance and their importance for the near-field environment and metal mobility are not well known. These aspects should, therefore, also be included in the field project.

More details about the role of organic matter are given in the state of art report of the project "laboratory studies of key processes".

### 3.4 Exposed waste deposits

The oxidation of pyrite and/or pyrrhotite which causes the formation of AMD with subsequent release of metals, may occur in sulphide-bearing mine tailings, waste rock, walls in open pits and underground mines, and in other heaps and dumps produced during mining. There are numerous case studies of the quality of drainage waters from such deposits. There are also studies, which in more detail have focused on processes within the deposits. Blows & Jambor (1990) studied the tailings impoundment at the former Waite Amulet Zn-Cu mine in Canada. Detailed geochemical and mineralogical profiles through the tailings were compared with geochemical profiles of pore water. An important conclusion was that the tailings could vertically be subdivided into three major zones; an upper oxidation zone where sulphides are depleted, an intermediate where sulphide oxidation and acid neutralisation reactions are
occurring, and an unoxidized zone. Similar situations have been described elsewhere, and are described and summarised in textbooks (e.g. Jambor & Blowes 1994; Morin & Hutt 1997). Studies of the pyrrhotite-bearing tailings at the former Laver Cu-mine in northern Sweden showed that the oxidation front is very sharp and well defined, and that there is a zone with secondary Cu-enrichment just below the oxidation front (Holmström et al. 1997). This secondary enrichment is caused both by adsorption on mineral surfaces and by precipitation of secondary minerals.

Few studies have, however, been performed where the total geochemical massbalances have been established on well-characterised field sites, and in that way the importance of the various key processes quantified. As an example of the relevance of thorough field studies Ljungberg & Öhlander (1997) showed, by detailed geochemical field studies of the tailings at the abandoned Laver Cu mine in northern Sweden, that less than 10% of the total amount of e.g. Cu and Zn, annually released by weathering, actually leaves the tailings with drainage water. The remaining amount of metals, more than 90%, is retained in the tailings by secondary processes. Zinc is retained in a large volume of the tailings by adsorption on mineral surfaces, whereas Cu is enriched in the narrow and distinct zone immediately below the oxidation front as described above (Holmström et al. 1997). The release of Cu in outlet water is controlled by desorption and dissolution of secondary minerals when the Cu-rich zone, which is continuously moving downward along with the oxidation front, reaches the groundwater table. Thus, conventional modelling of mineral stabilities cannot predict future copper outlets at Laver, but rather modelling of how large areas of the copper-rich zone that annually reaches the groundwater table.

The Laver example illustrates the necessity of field studies when dealing with mining waste. The MiMi programme, with a coherent combination of field studies and modelling and laboratory studies on material on common sites, has the potential to significantly contribute to the understanding of the processes occurring in mine waste deposits.

The waste rock environment is in some aspects more complicated than tailings impoundments. Waste rock dumps contain large amount of coarse material and have a wide range in grain size distribution. Transport mechanisms of gas and water are thus completely different from the situation in tailings deposits (Ritchie 1994). Flow heterogeneity in for example the form of preferential flow paths must be considered when modelling field-scale transport of weathering products (Eriksson & Destouni 1994). Waste rocks dumps are often well oxygenated, and the AMD production as pronounced as in tailings. One of the most detailed studies of large waste rock deposits has been performed at Aitik in northern Sweden, the largest Cu-mine in Europe. To quantify the sulphide oxidation on field scale, oxygen transport and consumption have been measured in oxygen profiles obtained from 11 capped monitoring wells (Bennett et al. 1994). Field monitored geochemical mass balances of drainage water have been compared with laboratory results and predictive models developed (Strömberg 1997).

Due to difficulties with sampling in waste rock dumps, there are few geochemical studies of solids and water, and comparisons of results from the different sample types. Lin (1996) investigated a profile through an old waste rock dump in central Sweden, and found a rough zonation of various horizons within the dump. The geochemistry of the waste rock environment is, however, probably less known that that of tailings deposits.

The rate of oxidation of sulphidic waste deposits may be quantified with direct measurements of the oxygen flux across the tailings-atmosphere interface. This is a relatively new method for site specific determinations of the actual oxidation rate in sulphide-bearing tailings (Tibble & Nicholson 1997).
3.5 Dry covers

Barriers such as dry and wet covers are used to prevent oxygen and water transport into disposed mine waste, and thereby minimising sulphide oxidation. Long term performance of various types of barrier constructions are discussed in the state of the art report for the MiMi project "Prevention and control", where also the variety of different dry covers that potentially can be used is discussed. In this report the biogeochemical aspects are covered, and how field studies are used to investigate the efficiency of dry covers.

Most field studies of dry covers are directed towards measurements of the effect of the cover on oxygen transport and penetration, on water balance and on the quality of discharged water. The old sulphide-rich waste rock dump at Bersbo in south central Sweden was the first full-scale case of remediation in Sweden where the dry cover method was used, the capping was completed in 1989, and one of the first in the world (Lundgren 1997). Various types of cover material were tested. Water percolation through the cover was monitored in collection lysimeters. Oxygen transport was measured in thin tubes that were installed before the capping with a dry cover. The cover acts as a good water barrier, and where a clay liner was used as a part of the cover, it is also a good oxygen barrier (Lundgren 1997).

The analysis of the leachate waters, however, illustrates some problems. In one of the two drainages from the covered area, only relatively limited decreases of the concentrations of metals such as Cd, Cu and Zn were detected (Håkansson et al. 1994). The concentration of sulphate increased from 600 to 1000 mg/l, and Fe increased from 6 mg/l to more than 200 mg/l. The hypothesis was put forward by Håkansson et al. (1994) that these changes were the result of reduction of Fe(III) oxyhydroxides to Fe(II) in the mine shafts, which led to oxidation of sulphides. A possible shift from oxygen to Fe as an oxidant was observed.

The results from Bersbo illustrate that mine waste as far as possible should be unoxidized before remediation.

The Saxberget sulphide mine in central Sweden was closed in 1988. In 1995, the tailings ponds had been covered with compacted glacial till with high clay content in the sealing layer, and with unspecified till as protection layer (Lindvall et al. 1997a). Infiltration lysimeters were installed under the cover, and groundwater wells were installed within the cover. 30 oxygen probes were installed to characterise oxygen transport in the cover. In addition, oxygen lysimeters were installed just below the cover. Surface water quality is also monitored. The results obtained so far indicate that oxygen transport to the tailings is almost eliminated, and that hydraulic conductivity is considerably reduced (Lindvall et al. 1997a). In drainage waters, no significant change of metal concentrations has yet been detected, but is expected to occur later.

At Aitik in northern Sweden, a cover consisting of three layers of till has been started to be constructed on AMD-producing waste rock deposits (Lindvall et al. 1997b).

The Swedish experience of the use of dry covers is in the frontline. A description of the Swedish experiences of field studies of dry covers is a description of the state of the art, but there are of course also experiences gained in other countries. MEND research in Canada has led to the conclusion that dry covers may be effective, but also that they may be expensive to construct (Feasby et al. 1997).

Most field study cases of dry covers thus includes measurements of the effect of the cover on oxygen transport and penetration, on water balance, and on the quality of discharged water. Another possibility is to construct in situ test plots, which are instrumented for various measurements, enabling detailed studies. This has been done on the Norebec-Manitou site in
Canada, where non-acid generating tailings have been used as cover on reactive waste (Aubertin et al. 1997).

It can be concluded for the MiMi program that field studies on the main MiMi site should include more detailed in situ studies of the processes occurring in the cover as well as below the cover. Particular emphasis should be put on detailed studies of the interfaces between various cover layers and the interface between the waste and the cover. The geochemistry of pore water and lysimeter water should be studied, and the geochemistry and mineralogy of profiles with detailed subsampling should be studied of the various solid materials.

### 3.6 Water covers

Various methods have been developed or suggested to prevent the formation of acid drainage from sulphide mining waste. One method is to limit the supply of oxygen by covering the waste with water. Tailings can be deposited in natural lakes, or tailings can be flooded. Waste rock can, for instance, be deposited in pit lakes when mining activities have closed down. One major result of the MEND program is that water covers and underwater disposal is considered to be the best prevention technology for unoxidized sulphide-bearing waste (Feasby et al. 1997). This conclusion is based on both field and laboratory studies. Water covers have been shown to strongly inhibit the oxidation rates of sulphide-bearing mining wastes. Water covering has been applied also in for instance Norway (Arnesen et al. 1997) and Sweden (Broman & Göransson 1994).

Recent studies in Canada, where sulphide-rich Zn-, Cu- and Pb-bearing tailings have been deposited in natural lakes, have given encouraging results (Fraser and Robertson, 1994; Pedersen et al., 1994). In these studies low concentrations of Cu, Zn, Cd, Pb and other metals were found in the sediment pore waters. There seemed to be no or only minor diffusion of dissolved metals from the sediments and pore water to the water column in the lakes. Detailed pore water profiles illustrating the same conclusions were presented by Pedersen et al. 1997. High-resolution pore water data from a tailings pond were also presented by Pedersen et al. (1997), but the history of the pond which has received neutralised sludge, complicates the interpretation. It is stressed that it is favourable with a deposition of natural sediments upon the tailings, which leads to anoxic conditions in the tailings.

Flooding of tailings is comparable to the situation with deposition in natural lakes if the waste has not been exposed to oxidising conditions. Flooding may, therefore, have the potential to be a cost effective remediation method. The water cover act as a barrier against oxygen supply, but it does not totally stop the diffusion of oxygen to the tailings. Contributions from water cover aeration and downward infiltration may occur (Li et al. 1997). Laboratory studies indicate that sulphide oxidation may occur at a non-negligible rate in water covered waste (Payant et al., 1995; Aubé et al., 1995). Before the applicability of flooding as a remediation method is completely understood, there is a need to research how flooding affects weathering and transport processes in waste.

The tailings of the Stekenjokk Cu-Zn mine in northern Sweden were flooded in 1991 (Broman & Göransson 1991), and offer a good opportunity for field studies of the efficiency of this remediation method. Such a geochemical field study was performed in 1995, and the results were presented by Ljungberg et al. (1997). This is the so far most complete geochemical field study of flooeded mine tailings, but the pore water data should be completed with pore water profiles with higher resolution. Possible sources to the metal content of the water in the pond are weathering of the tailings, weathering of the dykes, drainage water from the surroundings, and oxidation of resuspended sulphide-rich particles. The sampling covered these various sources. The main effort of the field studies, however, has been focused on the weathering of the waste.
material, the amounts of metals and other elements released into the pore water, and possible
diffusion of the released elements to the water column in the dam. The interactions between
tailings, pore water and the water column have been studied. Profiles of the tailings, pore water,
and the water column have been sampled and analysed. The mineralogical composition of the
tailings has been studied with common microscopy and SEM-EDS.

Most lakes in northern Sweden are of so-called dimictic type, i.e. there is a water turnover
(mixing of surface and bottom waters) during spring and autumn. During winter, below the ice,
a redox gradient is often developed with anoxic or at least oxygen-poor bottom waters (Pontér et
al., 1992). During summer a so-called summer-stagnation may be developed. A situation with
warm surface water and cold bottom water can be the result, and this can lead to concentration
gradients in the water column. Sampling was performed at several occasions to cover these
seasonal variations.

It was also considered important to sample several stations in the pond since the water depth
varies, and the tailings composition may vary over the area. Sulphidic waste rock has been used
for the construction of the dykes and the break water system. Possible effects from oxidation of
sulphides in these structures were studied by sampling at several stations.

It is also important to study the local background conditions. Tributaries and the stream
Stekenjokk, which is the outlet of the raw water pond but has no connection with the tailings
pond, were studied to obtain information of the background concentrations. Small streams
draining the dykes were also sampled and analysed.

The results of the Stekenjokk study showed that the pond water is well mixed and oxic the
whole year round and has higher concentrations of Ca, S (as sulphate), Zn and Cd than local
background values. Cu-content is low. Pore waters have higher concentrations of Ca and S than
the water column, but in general have lower or similar concentrations of Zn and Cd. Only in
tailings, which have been oxidised and weathered before the flooding, the pore water
concentrations of Zn and Cd are much higher than those of the water column. This type of
oxidised tailings cover only c. 5 % of the bottom of the pond, but mixing calculations using
Ca/1000Sr ratios suggest that 60 % of the Ca in the water column originate by diffusion from
pore water in these tailings. Dissolution of gypsum in the oxidised tailing is the most probable
explanation to the high concentration of Ca and S in this pore water. The most important source
of Zn in the water column is probably diffusion from pore water in the parts of the tailings,
which have been oxidised before the flooding.

Iron sulphide oxidation and carbonate buffering probably occur in the uppermost part of the
normal tailings at Stekenjokk. These reactions in combination with remnants of process water
probably control the pore water composition in these parts of the tailings. Humidity Cells test
results indicate that in the high pH conditions prevailing in the Stekenjokk dam, sulphide
surfaces may become less reactive with time due to coatings with iron oxyhydroxides. The
sedimentation rate in the dam is very low, but it would have been favourable if it was higher.

Weathering of sulphide-bearing waste rocks used in the dyke constructions is not an important
source of metals in the water column of the pond at Stekenjokk, but it may at some mines be
important (Pedersen et al. 1994). Run off waters draining the areas surrounding the dam, and
oxidation of resuspended tailings in the dam water, are also not important metal sources to the
water column. The breakwater system in the dam works well. Otherwise oxidation of
resuspended sulphide-rich particles may be an important metal source (Li et al. 1997).
3.7 Other methods and treatments of mining waste

Dry and wet covers are the most commonly considered methods for remediation of mine waste, but there are many other methods, or variants of the cover methods, which have been considered, and in some cases, also studied. This is described in the state of the art report for the MiMi project "Prevention and control". In general the results have been followed up or monitored, often by studies of the quality of drainage waste. It can be concluded for the MiMi program, that the field studies should be focused on detailed studies of the processes occurring at interfaces of various types.

3.8 Conclusions for the MiMi project

There are numerous cases with field studies of AMD producing mine waste, and of such waste remediated by various methods. If the MiMi program focuses on one or two sites where dry and wet covers can be studied, with a concentrated effort of field studies, laboratory studies and modelling in such a way that a detailed understanding of key processes is achieved, it is possible to move the knowledge front considerably forwards.

To understand and quantify key processes, the field studies should be focused on detailed studies of various types of interfaces. These could be water solid interfaces such as between pore water and solids, between waste and water cover, and between suspended matter/colloids and water. Other important interfaces are those between the various layers of dry covers, between the dry cover and the waste, and between various redox and weathering zones in the waste.

Mass balances should be set up to quantify the amount of metals and other elements annually released by oxidation and weathering, the amounts of elements annually being in transport within the waste systems, and the amounts being released from the waste deposits. In such a way the importance of key processes such as weathering, sorption, desorption, etc can be quantified. This is necessary to put the laboratory studies in a meaningful context, and without such results, predictive modelling is not possible.

Best available techniques for sampling and analyses should be used. Detailed time series are necessary for many sample types, especially when sampling various types of water and when sampling pore gas.
4 References


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The vision of the MiMi-programme

Twenty years from today the mining industry in Sweden is still strong and flourishing, using technologies that are internationally competitive and environmentally acceptable. The environmental standards are set high, since most of the ore deposits and mining activities are situated in sparsely populated areas with a very sensitive nature of high ecological and recreational value. Applying economic methods for processing and reuse of waste products, the release of heavy metals from waste deposits is kept low, the impact on the environment is small and restricted to the close vicinity of the mining areas. Methods used for waste disposal and remediation are efficient, robust and reliable so that, when any remediation is completed, a deposit can be left without the need for supervision or maintenance.

The MiMi programme has made it possible to predict the extent of environmental impact and has provided tools and methods to control and design processes and waste treatment systems already from investigation of the mineralogical and chemical composition of the ore and the wall rocks, and the local hydrology and topography. Furthermore, it is possible to design cost-efficient treatment systems for existing deposits of mining waste.