MiMi
Mitigation of the environmental impact from mining waste

The MISTRA-programme MiMi
Mitigation of the environmental impact from mining waste

MiMi - Prevention and control of pollution from mining waste products

State-of-the-art-report

MiMi 1998:2

Pär Elander,
Miljöteknik Bo Carlsson AB, Repslagaregatan 19, 582 22 Linköping, Sweden

Manfred Lindvall,
Boliden AB, 936 81 Boliden, Sweden

Karsten Håkansson
Swedish Geotechnical Institute, 581 93 Linköping, Sweden

1998

ISSN 1403-9478
ISBN 91-89350-02-2

ETT MISTRA FINANSIERAT FORSKNINGSPROGRAM

MiMi
Mitigation of the environmental
impact from Mining waste

MiMi - Prevention and control of pollution from mining waste products

State-of-the-art-report

MiMi 1998:2

Pär Elander,
Miljöteknik Bo Carlsson AB, Repslagaregatan 19, 582 22 Linköping, Sweden

Manfred Lindvall,
Boliden AB, 936 81 Boliden, Sweden

Karsten Håkansson
Swedish Geotechnical Institute, 581 93 Linköping, Sweden

1998

ISSN 1403-9478
ISBN 91-89350-02-2

Publisher: MiMi Print

ISSN 1403-9478
ISBN 91-89350-02-2

Contents

1 Introduction

Sulphide weathering in mine waste rock and mill tailings often generates drainage waters with low pH and high concentration of metals. The phenomenon is known as Acid Mine Drainage (AMD) and may pose a long-lasting threat to the surrounding environment. The processes that generate AMD involve oxidative dissolution of iron containing sulphide minerals, e.g. pyrite, and mobilisation of potentially hazardous metals due to release from primary minerals (Destouni et al 1997).

Different processes are involved in the oxidation of sulphides, including microbially catalysed as well as purely chemical reactions. In principle, both oxygen and water supply are required for the reactions. Consequently, prevention of oxygen and water transport into the mine waste should limit the sulphide oxidation. However, oxidation can proceed also in the absence of oxygen, provided Fe(III) is available to serve as an oxidant (Öhlander et al 1997). This is the case with partly oxidised waste, as Fe(III) is generated in the initial oxidation of iron with oxygen.

Another important factor for the release of metals from the mine waste is the pH of the pore water. The mobility of heavy metals is strongly increased at acidic pH, due to the higher solubility, and the lower tendency for sorption with decreasing pH. Hence, a high enough content of buffering substances such as calcite in the waste dump will effectively prevent the development of acidic drainage, thereby reducing the release of heavy metals as the oxidation processes proceed.

Regarding the fundamental processes governing the generation of acid mine drainage and metal release from mine waste, methods for prevention and control of AMD in general aims at either:

- changing the chemical properties of the waste prior to disposal by e.g. separation of pyrite or addition of buffering substances,
- restricting the transport of oxygen and/or water into the waste dump or
- treating the leachate.

Regarding their interaction with the operation of a site, prevention and control methods can be divided into four different groups:

1. treatment of waste which includes altering of the chemical properties of the waste but also altering of physical properties of the waste in order to limit the transport of oxygen or air into waste dumps,
2. flooding of the waste, i.e. a water table is established above the disposed waste as a barrier that limits transport of oxygen into the waste
3. dry covering of the waste in order to limit the transport of oxygen and/or water into the waste dump
4. treatment of leachate with the purpose to reduce the metal concentrations in the water that is discharged from the disposed waste.

2 Dry covers

2.1 General

Covering of sulphidic mine waste mainly aims at reducing the transport of oxygen into the waste dump, thereby preventing the oxidation of sulphides and the generation of acidic drainage and the release of metals. By covering of mine waste dumps, the rate of oxygen diffusion into the dump normally becomes the determining factor for the weathering of pyrite (Magnusson and Rasmuson 1982, MEND 1994). Another goal, that sometimes is considered equally important to limit the oxygen transport, is limiting the infiltration of precipitation into the waste dump. Such a reduction also leads to a reduction of the formation of leachate, and thereby limits the quantity of contaminated effluent from the dump.

In general, dry covers can be classified according to their function as follows:

Cover type	*Primary function*
1. Oxygen diffusion barriers	To limit the transport of oxygen by acting as a barrier against the diffusion of oxygen to the waste.
2. Oxygen consuming barriers	To limit the transport of oxygen by consumption of oxygen which penetrates into the cover.
3. Low permeability barriers	To limit the transport of oxygen and the formation of leachate by acting as a barrier against the diffusion of oxygen as well as the infiltration of precipitation.
4. Reaction inhibiting barriers	To provide a favourable environment to limit reaction rates and metal release

2.2 Oxygen diffusion barriers

Primary function

If a cover should perform as a barrier against oxygen transport from the atmosphere into a deposit, a layer with a low effective diffusivity for oxygen must be included in the cover. The effective diffusivity in a porous media is strongly dependent on the degree of water saturation (Millington and Shearer 1971). In water saturated soils, the diffusivity generally is low, and any saturated soil theoretically should be an excellent barrier against oxygen transport. However, the effective diffusivity increases very rapidly as the soil dries and the water content decreases which, depending on the climatic conditions, periodically happens in most superficial soil layers. Consequently, the barrier material must be able to withstand drying and to retain water in the cover to act as an effective barrier against oxygen transport.

In brief, the water retention capacity of a material is a function of the pore size and is generally enhanced by fine pore size (Nicholson et al 1991, Tremblay 1995). The degree of saturation is affected by capillary suction, which is a function of the elevation above the groundwater table. With increasing depth to the groundwater surface the higher capillary suction acts to withdraw water from the cover. Moreover, evaporation at the ground surface leads to a capillary transport of water upwards and can affect the water content to a considerable depth during dry periods.

Regarding this, materials that shall be used in oxygen transport barriers must be able to maintain a high degree of saturation also at high capillary suction. Furthermore, the best efficiency of an oxygen transport barrier can be expected at sites where the capillary suction that can act on the barrier is limited, naturally or by artificial means.

Single layer covers

To meet the requirements on an oxygen transport barrier, a single layer cover should be constructed of a material with mainly fine pores, which is generally true for fine grained materials (clays, clayey silts) and well graded materials with a high content of fines (clayey and silty moraines). A single layer cover can be an efficient oxygen transport barrier particularly at sites where the ground water table is close to the ground surface where the conditions are favourable to maintain a high degree of saturation in the cover. Depending on the geohydrology, such conditions can be found at certain tailings deposits. One example is Kristineberg, Sweden, where two tailings ponds have been reclaimed with single layer covers of till ensuring a high degree of saturation (Lindvall et al 1995).

At locations with greater depth to ground water, there is an obvious risk that the cover will be drained and the transport of oxygen will increase during dry periods. However, depending on the character of the waste, the requirements on the cover efficiency can vary and single layer covers still can be applied.

Within the research program ”Deposits of waste from mining industry” financed by the Swedish Environmental Protection Agency, it was estimated that a single layer cover constructed with 1.0 m till should result in a reduction of pyrite weathering rate (and metal release) at about 80 % in an ordinary tailings deposit (Naturvårdsverket 1993). For a cover constructed with 1.5 m till, the corresponding reduction was calculated at about 90 %. For advanced multiple layer covers including an effective low permeability barrier, the calculated reduction was estimated at about 99 %. After the research program was completed, decommissioning plans including single layer covers have been established for some Swedish deposits. Numerical simulations of the effects for some deposits show better results than the previously calculated. For the tailings pond at Aitik, e.g., the reduction is calculated at 99 % using a 1.0 m single till layer (Lindvall et al 1997).

A variety of the single layer concept is currently used for reclamation of the tailings deposit in Adak, Sweden. The cover is constructed with 0.5 m compacted till (barrier) overlaid with 1.5 m of uncompacted till. The permeability of the barrier layer is relatively low, about $5 \cdot 10^{-8}$ m/s, but not sufficient to act as a barrier against infiltration of precipitation. The annual oxygen flux for this construction is estimated at about 0.2 mol/m^2.

In Canada and US single layer covers not aiming primarily at reduction of oxygen supply, but to facilitate re-vegetation, often are applied. Instead are other methods, such as leachate treatment, used to prevent the effects of AMD.

Multi-layer covers

Theoretically, the so-called capillary barrier effect can be used in order to maintain a water saturated layer in the cover protected from losses due to evaporation or drainage. A capillary barrier can be generated when a fine-graded material with a high capacity to bind water by capillary forces is placed between two coarse-grained materials with no ability to transfer capillary suction (Rasmuson and Eriksson, 1987).

Modelling of covers including a capillary barrier suggests that it seems quite feasible to keep the fine grained layer at a high water content and thereby significantly reduce diffusional transport of oxygen (Collin and Rasmuson 1986). It has also been suggested that a capillary barrier layer can act as an infiltration barrier by lateral transport of water under unsaturated conditions, thereby avoiding positive pore water pressures on the barrier, which is theoretically needed to achieve vertical flow through the barrier. However, for normal Swedish conditions, this seems possible to achieve only occasionally. To act as an effective infiltration barrier, the capillary layer instead must have a sufficient low permeability (Rasmuson and Eriksson 1987).

The possibilities to utilise a capillary barrier have been investigated in laboratory columns by Aachib et al. (1994). The barrier tested consisted of (from top to bottom) 0.2 m sand, 0.5 m non reactive tailings and 0.3 m of sand placed on top of a tailings sand. The capillary layer (non reactive tailings) could be characterised as a silt with a high air entry value indicating that suction pressures up to about 25 kPa could be applied before the material should start to drain. The column was instrumented with tensiometers for measurement of suction pressure and TDR (Time Domain Reflectrometry) probes for measurement of volumetric water content. The results indicated that the water content in the capillary layer should be maintained at a high degree of saturation and consequently that tailings can be of interest to use as a water retention layer in capillary barriers.

In field scale, the capillary barrier concept has been investigated by e.g. Bell et al (1994). The tests were carried out as a reclamation of a waste rock pile with an area of 2 500 m^2 at Heath Steele Mine, New Brunswick, Canada. A compacted fine-grained till was used as the 0.6 m thick capillary layer and was placed in-between two layers of 0.3 m sand. The barrier was covered with a 0.1 m superficial layer as protection against erosion.

The cover was established in September 1991. Field hydraulic conductivity tests by single-ring infiltrometers indicated a hydraulic conductivity in the capillary layer of $1 \cdot 10^{-8}$ m/s. Following up showed no significant change in the water content of the till during the reported period (until May 1993), i.e. during the first 1.5 years after placement. The oxygen concentration in the pile decreased successively from 18-21 % in May 1991 over 0.8-1.1 % in May 1992 to 0.1-0.2 % in May 1993. The temperature decreased in a similar way and the pH in the leachate increased.

A similar construction was investigated at Waite Amulet, Quebec, Canada (Yanful et al, 1994). Test plots were constructed with a capillary layer of 0.6 m compacted clay placed between two layers of 0.3 m sand and protected against erosion by a gravel crust. The water content in the clay remained close to the construction content at a degree of saturation ≥93 % during the following-up period (1990-1993). Modelling of oxygen transport based on field measurements indicated a barrier effect of >99.9 % regarding oxygen transport. Furthermore, the compacted clay had a significant effect as a barrier against infiltration due to it’s fairly low permeability (estimated at about $1 \cdot 10^{-9}$ m/s).

One problem involved in the establishment of covers is the supply of appropriately fine-grained cover materials. Bussiére et al (1997) have investigated the possibilities to use desulphurized tailings as the capillary layer in a capillary barrier. Column experiments, including reference columns as well as columns with capillary barriers constructed of tailings with different (but low) sulphur contents, were carried out. Results were measured as oxygen-consumption values (Eberling et al 1994). The column with a capillary barrier constructed of desulphurized tailings with the lowest concentration of sulphur (0.14 %) showed a reduction of the oxidation rate by a factor 20 (95 %) compared to the reference column.

A similar capillary barrier, with non-acid generating tailings in the barrier, have been constructed in full scale at Les Terrains Auriferes in Quebec, Canada (Ricard et al 1997). AMD modelling showed that a multi-layered cover with an effective oxygen diffusivity of $1{\cdot}10^{-8}$ m^2/s with a thickness of 0.8 m would be adequate to inhibit acid production from the underlying tailings. To achieve this, the tailings in the barrier must be saturated at 85 % or more according to the characterisation of the material. For design of the construction, the HELP and SEEP/W models were used for two-dimensional evaluation of the water budget. The cover was constructed during the winter of 1995/96, which called for a strict QA/QC program to ensure that minimum standards were met. Winter construction was chosen to avoid trafficability problems since the performance of the cover would have been more jeopardised by these problems than by those encountered during winter construction. The first six months of monitoring of the cover's performance showed an average saturation of 86.0 % on top of the dump and 84.4 % in the slopes. Implementation of oxygen consumption measurements indicates a reduction of the rate of oxygen flux to the underlying tailings by a factor of 75.

Showed to be an economic feasible alternative, covers with a capillary barrier have been the subject for several studies and have also been established in full scale. However, the importance of adequate characterisation of the construction materials and modelling of the actual site and water budget prior to design and construction of a multi-layer barrier should be stressed (Aubertin et al 1997, Ricard et al 1997). For example, 2D numerical finite element investigations of a large scale deposit show that the capillary barrier effect is dependent not only on the properties of the used capillary materials but also on the geometry of the deposit, and illustrate the risk of desaturation of a portion of the cover if the slope is too steep or too long.

2.3 Oxygen consuming barriers

Oxygen transport to the waste can also be reduced by covering the deposit with an oxygen consuming material. Tremblay (1994) reports from the reclamation of the East Sullivan Mine in Quebec, where dumping of wood wastes on tailings began in 1984. In 1990, management of wood waste in order to reclaim the tailings deposit by covering started. The cover consists of 2 m of organic waste (85 % bark, 10 % pulpwood and 5 % sawdust). Following up during 1991 showed that the oxygen decreased with depth in the cover. At a depth of 0.7 m the oxygen content was 1.5 %. Monitoring also showed increasing pH and decreasing metal release in leachate from the covered areas. Some organic contamination by small concentrations of phenol and tannin occurred. Further monitoring and tests with different cover design and effect on oxidised and fresh tailings respectively are planned.

Three different organic materials (peat, lime stabilised sewage sludge and municipal solid waste compost) were investigated in a combination of bench and pilot scale laboratory test in order to evaluate their effectiveness as oxygen consuming covers (Elliot et al 1997). To provide comparative data also a cover with desulphurised tailings were tested. From these studies it was concluded that lime stabilised sewage sludge appeared to offer the greatest potential for reducing metal loading to the environment. This is achieved by changing the underlying tailings environment by reversing the AMD processes with an increase in pH, decreased metal concentrations and formation of a reducing environment at the tailings - cover interface. For the compost cover a significant decrease in oxygen concentration with depth was observed. The oxygen content at the tailings interface at 0.9 m depth was close to zero, indicating that the material worked as an oxygen barrier. However, the effect on metal release from the underlying tailings was moderate, probably because these already were partly oxidised and no change in pH was achieved. The peat layer showed a minor decrease in oxygen concentration with depth indicating that the oxygen consumption in the material was too low to function as an oxygen transport barrier. This was probably due to the character of the peat as an older, already largely decomposed material.

At Galgbergsmagasinet, a tailings pond in Falun, Sweden, a cover with a high content of organic material was constructed from paper mill sludge, fly ash and wood waste. On the top of the tailings pond a totally 1 m thick layer of fly ash mixed with paper mill sludge was laid out and compacted in two layers and thereafter covered with a 0.5 m layer of wood waste and coarse till. This cover is believed to form an effective barrier against oxygen transport partly due to consumption of oxygen in the cover and partly due to a physical barrier effect in the compacted low permeable mixture of fly ash and paper mill sludge. The hydraulic conductivity of the mixture was measured in the laboratory at $\leq 5 \cdot 10^{-9}$ m/s and the water retention capacity was measured and considered satisfactory to maintain a high degree of saturation in the barrier (Lundqvist). Other possible positive effects are inhibiting of the acidophilic leaching bacteria due to the high content of calcium hydroxide in the fly ash that will raise the pH in the percolating water, and the formation of a sustainable environment for sulphate-reducing bacteria producing hydrogen sulphide that precipitates metals. However, there is also a risk that the combination of organic compounds and iron hydroxides in the upper (oxidised) part of the deposit could produce bacterial iron reduction that would dissolve co-precipitated heavy metals. The ongoing following up indicates so far that the oxidation of sulphides has decreased and that the pH at the site is higher than at the reference site. No evidence of any significant bacterial sulphate reduction has yet been noticed (Granhagen et al 1996).

2.4 Low permeability barriers

Low permeability barriers are commonly used to limit the infiltration of precipitation into landfills. In addition, low permeability soil generally shows favourable water retention properties in order to maintain a high degree of saturation in the barrier. Limiting of percolating water through the barrier also lead to water saturation in a portion of the layer above the barrier for long periods, unless this layer is drained. Consequently, low permeability barriers also act as water saturated layers preventing oxygen transport provided they are covered by a protective layer.

Being a barrier against both the diffusion of oxygen and percolating water, covers including low permeability barriers often show high potentials for limiting of AMD and metal release from mine waste dumps. Estimations based on modelling within the former Swedish research program ”Deposits of waste from mining industry” indicate a possible reduction of the oxidation rate from 95 % up to more than 99 % and a probable reduction of the percolation rate from 80 % up to more than 95 % depending on the barrier soil (Naturvårdsverket 1993). Hence the pollution transport is reduced in proportion to the reduced infiltration.

Low permeability barriers can be constructed of fine-grained soils, mainly clay and clayey moraines, geosynthetic clay liners (geotextile/bentonite liners), geomembranes (plastic liners), cement-stabilised products and some fine-grained residues from industrial processes (mainly sludge) (MEND 1994, Lundgren 1995).

At a waste rock dump in British Colombia, Canada, a cover including a low permeability barrier was constructed with a fine-grained till classified as SC-CL using the Unified Soils Classification System (clayey Siltmoraine to silty Claymoraine according to the Swedish classification system) (Wilson et al 1995). On the top of the deposit, a barrier of 0.5 m compacted till with a saturated hydraulic conductivity estimated at $2 \cdot 10^{-9}$ m/s was established and covered with 0.3 m loose till as a protective layer. Modelling based on short term following up indicated that the water flux percolating into the dump was reduced by approximately 99 % and that the oxygen flux would be reduced up to 99.7 % compared to the uncovered waste rock.

The earlier described capillary barrier construction established at the Waite Amulet site in Quebec is also shown to act as a low permeability barrier (Yanful et al, 1994). The barrier consists of 0.6 m compacted clay between two 0.3 m sand layers. The hydraulic conductivity of the clay barrier was measured in field at about $1 \cdot 10^{-9}$ m/s. Monitoring of the infiltration through the cover by means of a lysimeter below the barrier yielded 37 mm of leachate during a year. Simulation of 20 years varying climate data and calculations of the resulting infiltration through the barrier with the quasi-two-dimensional deterministic HELP (Hydrologic Evaluation of Leachate Performance) model indicated an average infiltration rate of 39 mm/year. Finite element modelling simplified to steady state conditions supports the results indicating an infiltration rate of 34 mm/year. As previously noticed the reduction of oxygen transport was estimated at >99.9 %.

During 1987-1989, the deposits of mining waste and metal contaminated ground areas in Bersbo were remedied (Lundgren 1997). The waste - around 750.000 m^3 - was produced during a long period of mining activities reaching back to the 13th century. The mine was closed in 1901. The remediation works became a pilot project for the decommissioning projects for waste deposits from mining of sulphidic ores, which followed and still are under way in Sweden. Since 1989, a series of investigations has been conducted in Bersbo with the objective to show the results of the remediation measures. The follow up studies can be divided in four main programs:

1. The efficiency of the dry covers on the two waste rock deposits
2. The efficiency of other counter measures in the mining area
3. The quality of the surface waters affected by AMD from the mining area
4. The effect of the measures on the ecology of the area that was (and still is) affected by AMD.

The follow up studies on the efficiency of the covers shows that the cover on one of the two deposits - the Steffenburg deposit - works very well and the water quality monitoring verifies that the transport of metals from this deposit has been reduced to low levels. The cover on the other deposit - the Storgruve deposit - has not functioned equally satisfactory, the reason being air leakage from old galleries in the mine. Supplementary sealing operations have been carried out and the result has been followed up during 1.5 years. The supplementary sealing measures have had some effects which are too early to specify.

The drainage from the Storgruve deposit is also charged with an acid drainage from the mine. This AMD is generated by a quite large mass of waste rock (250 000 m^3) that was dumped under the groundwater level in the mine shafts in order to be wet covered. It has been shown that the drainage from the mine increased significantly due to secondary oxidation and leaching of this mass of waste rock and that the drainage from the waste rock deposit was significantly reduced. Even if there are indications on improvements, the resulting transport of AMD in the common outlet from the mine and the deposit to lake Gruvsjön is still about the same as before remediation.

On the Storgruve deposit, the cover consists of a 0.25 m thick sealing layer of cement stabilised coal fly ash (”Cefyll”) which was grouted as slurry in a crushed rock aggregate. This layer was covered by a protective layer of 2.0 m of glacial clay. On the Steffenburg deposit, the sealing layer consists of glacial clay, which was compacted in three lifts and protected by 2.0 m of glacial till. The saturated, hydraulic conductivity of the Cefyll-layer was registered on small samples to around $1 \cdot 10^{-9}$ m/s which was the specification set up for the sealing layers. The corresponding conductivity of the compacted clay was registered at $1 \cdot 10^{-10}$ m/s, i.e. 10 times lower than the specifications. The vertical, full scale percolation of water through the sealing layers registered as the leachate production rate corresponds to 3-5 times the saturated conductivity of the clay liner but only 0.5 times the saturated conductivity of the Cefyll liner.

The latter result is believed to be an effect of not fully water saturated conditions in the sealing layer which is caused by the capillary conditions in the Cefyll liner on the long slope of this deposit.

The transport of oxygen through the clay liner is very slow and the oxygen concentration is normally below 0.5 % with a seasonal fluctuation between less than 0.1 % and 1 %. The transport of oxygen through the Cefyll liner is higher, normally between 1 and 3 % but with seasonal fluctuations between 0.1 and 5 %. The peaks in oxygen concentration are caused by the seasonal low groundwater levels in the cover, which take place in August-September each year. Due to the low oxygen diffusivity of the sealing layers, the peaks are not registered until 6 months later when the groundwater levels are the highest, normally.

In conclusion, the clay liner acts as a good water and oxygen barrier, while the Cefyll liner acts as a comparatively good water barrier but a less good oxygen barrier. As previously mentioned the Storgruve deposit has also suffered from leakage of air with advective transport of oxygen to the waste. Chemical analysis shows that the higher oxygen transport rates in the Storgruve deposit has a less effect on the metal concentrations of the discharges of the deposits than the very low oxygen transport to the waste in the Steffenburg deposit.

Beside the wet and dry cover of deposits of mining wastes, the counter measures in Bersbo comprise rinsing of wastes and concentrating the waste to the two main deposits. This lead to a 50 % reduction of the size of the ground areas that were exposed to AMD. The follow up studies show that the surface and ground water quality rather soon and significantly improved at certain subareas while high concentrations of sulphate and metals still can be found after more than 8 years on other subareas.

The Saxberg mine was in operation from 1892 to 1988. Waste products from the activities were deposited in the Saxdalen Village where the first beneficiation plant was situated, at the Boliden Mineral AB mine site (from 1957) and in two large impoundments containing mill tailings. The decommissioning which is a joint project between the Swedish Environmental Protection Agency and Boliden Mineral AB started in 1991 with cleaning up the old plant site in the village and disposing of some 100 000 m^3 of mining wastes and contaminated soils in one of the tailings ponds. The project will probably end in 1997 by demolishing the rest of industrial buildings and disposing these residues in mine shafts, which will be secured (Lindvall et al 1997).

The most costly and time-consuming part of the project was the coverage of the two large tailings ponds which was carried out between 1993 and 1995. The cover consists of a sealing layer and a protective layer. The sealing layer, 0.3 m thick, consists of a glacial, clayey till that was compacted in 2-3 lifts. The protective layer consists of a 1.5 m thick layer of sandy, glacial till. Both till types were found close to the ponds and were used without other treatment than the removal of blocks bigger than 150 mm.

The original specification of the sealing layer material was that it should be compacted to 95 % modified Proctor and give a hydraulic conductivity that is less than $5 \cdot 10^{-9}$ m/s. This turned out to be hard to achieve with the selected materials at site which resulted in that a minor part of the sealing layer may have a slightly higher conductivity. The specified conductivity of the sealing layer is too high to result in efficient water barrier properties. However, according to numerical modelling it should be low enough to create saturated conditions in the bottom part of the cover during a large period of the season. Oxygen transport should only be possible during one or two months in the dry season.

Follow up studies have shown that the oxygen concentration under the cover is very low, only occasionally above 0.5 %. Hence, the cover must act as a good oxygen barrier. However, the monitoring also shows that the air transport processes are very sensitive to leakage through discontinuities in the cover. The reason for this is probably, like in Bersbo; the gas pressure changes that takes place in the covered deposit. In this case both in the short term and long term perspectives. Changes in the short-term perspective (diurnal) are due to tidal effects (gravitational) and possibly atmospheric air pressure variations. The long-term effects are due to variations in groundwater levels.

The monitoring also indicates that not only the seasonal but also the spatial variations in the water balance of the cover are significant.

So far, it is too early to specify what effect the cover has on the transport of metals from the tailings ponds to surface and ground waters.

At Ranstad, uranium mill tailings have been reclaimed using a cover including a low permeability till layer, similar to the construction in Saxberget.

In the case of shortage of low permeability soil at the site, low permeability barriers often involve high costs for transportation or processing of materials. In certain cases it is possible to reduce the cost by the use of alternative materials supplied by nearby industries. The earlier described rehabilitation of the tailings pond at Galgberget in Falun is such an example. At Eustis mine site in Quebec a similar low permeability barrier constructed with deinking sludge from recycling of paper was investigated (Cabral et al, 1997). An experimental cell was built with partly oxidised tailings covered with a barrier of 1.2 m of compacted deinking sludge and a vegetative cover of 0.3 m compost-deinking sludge mix. Prior to the covering, the upper 0.3 m of oxidised tailings was mixed with alkaline sludge in order to increase the pH.

The deinking sludge had a very high water content, 160-185 %. The sludge consisted of kaolin clay, 38-50 %, and organics. The permeability according to laboratory measurements on compacted samples varied between $5 \cdot 10^{-10}$ - $2 \cdot 10^{-9}$ m/s. The sludge also showed favourable water retention characteristics with an air entry value of 35-45 kPa.

Experiences showed that the sludge, unlike fine-grained soils, could be compacted in a wide range of water contents and still show low permeability calculated from lysimeter readings (comparable to the values measured in laboratory). Monitoring also showed that the oxygen content in the cover generally was below detection limits at a depth of 0.5 m, indicating an excellent barrier against oxygen transport. The temperature profile in the cover showed heat generation, which could be associated with degradation of the organics. It is likely that the oxygen diffusion barrier effect was strengthened by oxygen consumption in the barrier.

2.5 Long term performance

The long-term performance of the cover is a key issue as most mine waste dumps are potential sources for spreading of metals during a very long time. Despite this, only few investigations on physical stability and long term durability have been reported.

The short-term efficiency of a dry cover can decrease in the long term as a consequence of different destructive processes that may cause cracks or other discontinuities in barrier layers. Such processes are erosion, frost action, drying, differential settlements, root penetration, digging animals and manmade intrusion.

Owuputi et al (1995) carried out an erosion study for the cover system constructed on a waste rock pile at Equity Silver Mine, British Columbia. The cover consisted of 0.3 m loose till above 0.5 m compacted till. It was concluded, that topographic control and establishment of stable vegetation are the most important issues regarding the long-term stability with respect to erosion of the soil cover system.

Regarding frost resistance, several studies have been focused on how clay soils are influenced by repeated cycles of freezing and thawing. Viklander (1997) has put together results that show increase in hydraulic conductivity in clays due to freezing and thawing, often by as much as about two orders of magnitude (Benson and Othman 1993, Benson et al 1995, Eigenbrod 1996).

Yanful et al (1997) carried out laboratory measurements of hydraulic conductivity of a compacted low-permeability clay as a function of freeze-thaw. They found that the permeability increased by one to two orders of magnitude after the first two freeze-thaw cycles and then remained steady. However, this was not consistent with field measurements. According to these, the permeability of a barrier at a test plot, using the same clay as barrier material, remained constant despite freezing during winter. In general, impairment of the barrier function due to freezing has not been subject for studies in the field investigations cited above. However, following up measurements have mostly covered a period of at least one, in some cases up to three or four years. Based on the data collected during these short periods no references reporting any trend against decreasing barrier effect have been found.

Only a few investigation of the influence by freezing and thawing on coarse-grained soils like till have been found. Viklander (1997) refers to a study performed by Johnston and Haug that indicated an increase in hydraulic conductivity by two to three orders of magnitude for a glacial till. On the other hand, in laboratory studies he found no significant increase in hydraulic conductivity for a fine-grained till with a low content of clayey particles. Tests on silts show generally much smaller changes in hydraulic conductivity than tests on clays. Viklander also refers to White and Williams that noticed changes in the range 2-5 times while Konrad (1989) measured the increase to be 2 to 10 times in a clayey silt. The influence by freezing and thawing thus seems to be dependent on the clay content. In Sweden, fine-grained till is often used as a barrier material on mine waste deposits.

The knowledge of frost susceptibility in these materials, especially in field situations, is limited and the basis for design of covers with respect to frost need to be improved. Swedish practises for cover dimensioning with respect to frost today are based on empirical data for snow-cleaned surfaces, using correction factors.

Field experiments at a temperate site in Wisconsin showed a considerable effect on a clay barrier due to drying and root intrusion (Montgomery and Parsons 1989). Two test plots with a 1.2 m clay barrier was covered with 0.15 m and 0.45 m topsoil respectively. After three years, the conditions at both plots were about the same with weathered and blocky clay in the upper 0.20-0.25 m of the barrier and cracks extending up to 1 m in the barrier, roots penetrating up to 0.25 m into the clay in a continuous mat, and some roots extending up to 0.75 m into the clay.

Similar studies showed that also a soil protection of 0.6 m was inadequate to protect a clay barrier against significant drying, while a geomembrane overlaid by 0.45 m of topsoil showed to be a sufficient protection (Corser and Cranston 1991). This study however, was conducted in a relatively arid part of California and is probably of little relevance for Swedish conditions.

In the U.S. special stone layers are used as barriers against root intrusion (Lundgren, 1995). In the literature such barrier effects have been reported for natural analogues (Köstler, 1968). Another possibility is the construction of heavily compacted and firm layers. Results from field measurements indicate that roots don't penetrate layers that show a penetration resistance higher than 5 MPa measured by a steel point penetrometer (Eriksson, 1982).

Deformations in a deposit may cause a threat to the integrity of overlaying barriers. Clay barriers are able to withstand differential settlement up to 5-10 % without cracking. At larger differential displacement, cracks are developed and the flow rate through the barrier increases dramatically (Murphy and Gilbert 1985, Jessberger and Stone 1991). Similar studies of deformation effects on barriers constructed of till, which is usual in Sweden, have not been found. However, the function of such barriers are probably more sensitive to deformations than clay layers. On the other hand, deformations usually can be predicted by adequate geotechnical investigations prior to capping. Furthermore, the development of differential settlements is a function of the localisation and construction of a deposit.

2.6 New barrier concepts

At the East Mine tailings impoundment a field test of a ”self-sealing” barrier has been carried out by McGregor et al (1997). The self-sealing layer is formed by precipitates at the interface between two ”parent” materials, a 0.3 m thick ”reactive material” overlaid by a 0.2 m thick layer of mill washings rich in Fe and S. Above these layers a superficial cover of 0.25 m oxidised tailings were laid.

Following-up measurements showed that a thin barrier was formed at the interface of the mill tailings and the underlying reactive material (the thickness of the barrier was not specified). Moisture measurements indicated that the tailings immediately above the barrier were saturated and oxygen analyses of pore gas showed an oxygen content of less than 3.4 % below the barrier compared to 18.5 % above the barrier. Laboratory measurements of hydraulic conductivity on cores from the site indicated a hydraulic conductivity of $3.6 \cdot 10^{-10}$ m/s in the barrier. Costs for the construction was estimated at \$3 - \$5/m^2.

3 Flooding

Pioneering work on underwater deposition methods for tailings was carried out in Norway during the 1980:s. At the mines Løkken and Hjerkinn, sulphide rich tailings were distributed in ponds using floating pipelines, never allowing access of air to the tailings. The record proved that the pyrite oxidation rate under water was virtually interrupted, and metal release was orders of magnitude lower than in the drained deposit case.

Important scientific work was undertaken by among others NIVA (Norsk Institutt for Vannforskning), who developed methods for quantification and prediction of the metal release from sediments. Based on this work, the underwater deposition method was developed to be applied also on previously drained and oxidised material.

In Sweden, the Stekenjokk mine was closed 1988, and the operator Boliden worked out a decommissioning plan for the area focusing on the tailings pond. At that time, the predominantly discussed method was soil covers, however realised only on small deposits and unsatisfactory proven. Due to the circumstances - mainly lack of material - Boliden went for a post closure flooding method with elevation of the dykes and some redistribution of the material to reduce the need for dam height. The results are very satisfactory, and Stekenjokk has since become the reference object for tailings ponds.

Important references on post closure flooding are at hand also in Canada. In the uranium mining district Elliot Lake, a number of pyrite containing tailings ponds have been subject to flooding, now constituting a system of artificial lakes. The project has been carefully followed up, and a research centre administrated by CanMet, has been founded in Elliot Lake.

Within the research program MEND/NEDEM in Canada, important investigations of underwater deposition practises has been undertaken. A number of lakes in Canada being used for continuous underwater deposition of sulphidic tailings were studied, and the signs of disturbances due to slimes etc, except for the close proximity of the discharge points, were limited. The positive experience regarding the chemical stability of the material together with the favourable visual impression of the deposit even when it is in operation has caused Canadian mining companies to strongly consider the method for future projects. One of the most recently realised projects, the Louvicourt mine in Quebec, is operating a continuous underwater deposition pond.

3.1 Underwater deposition

To secure the deposit, some design parameters need to be established. The most important one is the design water depth, ruled by the water balance and the wind conditions. Depending on the exchange rate of the water, the design must accommodate for the losses due to evaporation and dyke leakage during dry periods. The minimum water depth is ruled by the wave action combined with the particle size, which yields a shear force or an erosion velocity at the sediment surface, which must not be exceeded.

Continuous underwater deposition

In its most downright form, underwater deposition means the discharge of tailings at or below the water surface. No beach is allowed to develop. Fluctuations of the water level must not take place to such an extent, that the tailings surface is exposed. As the angle of distribution is steeper below surface than above, the depth of the lake or pond is important. An early example was the Grängesberg mine in Sweden, which until closing 1986 practised underwater deposition mainly aiming at reduction of fugitive dust.

In some cases ocean deposits have been operated. Two such examples may be mentioned, Titania in Norway and Greenex in Greenland. Titania, producing inert titanium ore tailings, was forced by the regulators to move the deposit from a fjord to an area on land. Greenex, who closed operation 1990, utilised the fjord with only limited impact on the aquatic life recorded. After closure, the metal content in the species is declining towards the background value.

The management of the deposit involves a strict planning. The tailings are discharged using a floating pipeline, and to minimise the need for changing the set-up, batymetric measurements need to be carried out. Based on these measurements, the pipeline is directed along an intended line, anchored and in sequences shortened to create an underwater ridge of tailings. Depending on the fineness of the tailings and the depth of the pond, more or less frequent relocation of the discharge point is required. In the best case, the method does not call for more frequent changes than conventional discharge onto a beach.

Disadvantages with this method are the relatively high initial costs due to preparatory dyke building to final height and expensive pipeline arrangement.

Apart from the benefits of the water cover-inhibiting oxidation, the costs for decommissioning of an underwater deposit is low. In the most favourable form, only removal of the pipeline and final cleanup of the area is required. This is under the circumstances, that the dyke stability is sufficient, and that the dykes are built using appropriate material or already covered when the material is sulphidic.

Post closure flooding of waste

Also conventionally operated ponds, with discharge along a beach, may be permanently protected by a water cover after mine closure. To accomplish this, either the water level needs to be raised or the surface of the material needs to be flattened out to reduce the need for dyke height. As one parameter, Boliden uses breakwaters and waste rock to reduce the wave forces and stabilise the bottom in the shallow parts of the tailings ponds being flooded.

A problem that needs to be addressed is the content of weathering products in the material, originating from oxidisation during the dry periods. When the watershed is raised, often following major excavation work in the pond, substantial amounts of metals and salts are mobilised, complicating the evaluation of the oxidation rate of the bottom sediments. To clarify the processes, the development of the water quality in the Stekenjokk pond is being studied both by Luleå University of Technology and Boliden.

Apart from the high decommissioning costs, the post closure flooding option is connected with a few drawbacks during operation. One is the behaviour of the beach, which during unfavourable weather conditions may generate severe spread of fugitive dust. Additionally, the exposed tailings beach cause weathering of the material with increased metals and sulphate content in the pond water. References on this are available from all mines operating conventionally managed tailings ponds.

Flooded deposits interaction with the surrounding watercourses

A mechanism recently focused on in e.g. Kristineberg is the influence from natural sediments on top of the sand. Simulations carried out by NIVA suggested that even a shallow inert sediment layer would reduce the transport of oxygen and metals to fractions of the base case. Hitherto realised decommissioning projects have been aiming at isolating the deposits from the surrounding environment, without taking into account any benefit from natural sediments. In Kristineberg, the regional watercourse will be directed through the tailings pond, aiming at a rapid restoration of the pond to a lake with normal aquatic life.

Today's knowledge of the rehabilitation process is limited. As one example, the establishment of fish in the ponds is investigated only to a limited extent. There is a need to follow the immigration of various species in to the restored environment.

3.2 Water saturation

An intermediate solution for water saturation without creating open ponds is the saturation of the actual deposit by raising the phreatic level. This method is being practised in two ponds in Kristineberg, both containing strongly weathered material. As the material is entirely water saturated, further oxidation is inhibited. This is accomplished without the complex of problems connected to flooding. The basis for such a measure is a careful groundwater modelling, taking into account the influence from surface water management and groundwater raising dykes.

3.3 Combined wetland and flooding

One possible component of water covers is wetlands, used for reducing the need for water depth. An active wetland provides protection from wave forces. With time, a layer of organic material will develop on the bottom, acting as an oxygen-consuming barrier during dry seasons. It is underlined, that the main object with such a solution is not biological treatment of water, only the building up of a live barrier.

4 Selective management of waste

For various reasons, selective management of tailings has been practised in many cases, however only recently as a result of environmental considerations. In a typical case, waste rock with intermediate content of metals has been deposited separately, aiming at a future recovery of the metals. This practice, in many cases, also has full relevance for environment protection efforts, as elevated content of metals normally indicates more reactive rock.

4.1 Mill tailings

Selective management of mill tailings aiming at improved environmental performance either aims at isolating the pyrite content into a waste product of lesser volume than the main tailings product, or to produce a material suitable for covering a deposit containing reactive tailings.

If pyrite separation is desired, the object can quite easily be accomplished by flotation. The problem is normally the costs, as an additional separation step is required. An obvious consequence is furthermore that facilities for separate deposition of the reactive product needs to be arranged.

A cover material with favourable water retention properties, which can be used in oxygen transport barriers, may be produced by recovering the fines from the mill tailings. A useful separation mechanism is gravity classification, in many cases integrated in the ordinary milling process to recover the coarse fraction, which in many cases is, used for mine backfill.

To date, such practice is not common. Falconbridge, as one example, is developing a selective management system for low sulphur mill tailings in the Moose Lake system in Sudbury. The object is to produce an inert cover for a number of tailings ponds, while the final pyrrhotite rich product will be deposited under water.

4.2 Waste rock

Selective management of waste rock to control acidic drainage has been subject to studies for about a decade. The methods for determining the weathering properties of the rock have been developed mainly in Canada, within the MEND programme and by British Columbia Acid Mine Drainage Task Force. BC Task Force 1989 issued standards for testing waste rock and tailings, which still are used, however being subject to development. There are numerous references from around the world, where waste rock is sampled either using diamond drill cores or later in the process, and then classified depending on the composition.

What limit the successful and economic use of such practise are the methods for classifying the material. The predominantly used method, the Acid-Base Accounting, provides a conservative estimation of the acid generating properties. The method involves grinding of the samples and treatment with acid at elevated temperature. In field, the conditions differ significantly, in particular with respect to particle size and hence exposed reactive surfaces.

Future development is required to improve the prediction methods, to provide better tools for classifying the waste rock according to the need for decommissioning measures.

5 Treatment of drainage

5.1 General

The treatment of acid mine drainage aims not surprisingly at a reduction of the acidity of the water. This acidity includes not only the proton concentration of the water but also mineral acidity caused by hydrolysis of metals such as Fe and Al. Estimations of acidity of mine drainage in eastern U.S. are e.g. calculated from concentrations of Fe(II), Fe(III), pH, Al and Mn. This is often expressed as mg/l $CaCO_3$ equivalent. Supply of alkalinity is often done in active treatments by additions of limestone and subsequent handling of the precipitated sludge. Also in passive systems the addition or production of alkalinity is in focus. This can be achieved by anoxic limestone drains into which the acid mine drainage is directed. The inactivation or clogging of the drains that can be caused by ferric hydroxides is avoided if the water is kept isolated from air during the contact with the limestone. Other means to produce alkalinity are by microbial sulphate reduction. This has the additional benefit that sparingly soluble sulphides of several metals can be formed.

5.2 *Active treatment systems*

The prevailing method today is based on lime treatment, in some cases with strict demands on the effluent quality, combined with sulphide precipitation. Under normal circumstances, this method provides a reliable and cost-efficient water treatment method with excellent performance. When the drainage water - or mine water - is heavily contaminated, generating large quantities of sludge. This problem is normally solved by using a process in which the neutralisation process is controlled, and sludge is recycled, which reduces the water content in the sludge and hence the amount of sludge. The process is recognised as the HDS- process, of which a variety of alternative solutions are used

The possibility to recover metals has been investigated. Some methods that can include this step in the treatment were mentioned by Mattsson, (1990) in a review of methods for acid mine drainage. The aspect of recovery was possible to meet for precipitation of metals with fatty acids, selective precipitation of zincsulphide, or selective ion exchange. None of these methods were at that time thoroughly studied. The possible use of recovery methods is motivated because the handling and disposal of voluminous and metal contaminated sludge in conventional lime treatment can be limited or avoided. Selective ion exchangers were tested by Riveros (1995) who found that the presence of iron and the low pH decreased the efficiency of the exchangers. A laboratory test of consecutive hydroxide and sulphide precipitation conducted by Diaz et al. (1995) showed that precipitation of Fe(III) was possible without any substantial loss of Cu, Zn and Mn to the precipitate, however, As did co-precipitate.

5.3 Passive treatment of leachate

High costs for conventional treatment and the need for long term maintenance of the treatment systems has raised an interest in passive systems especially for abandoned mines. A comprehensive review on passive treatment of acid mine drainage is found in (Skousen and Ziemkiewicz, 1995) where results from both natural and constructed wetlands are found as well as discussions of other passive systems such as anoxic or open limestone drains or other alkalinity producing systems.

Natural wetlands

In a review by Sobolewski (1997) mostly Canadian experiences of wetland treatment are studied. Interest in wetland treatments in the Eastern United States initiated four wetland studies sponsored by MEND (MEND 1990; Kalin, 1993; Gormely et al. 1994 and Davé 1993). Several wetland tests in Canada as well as in the United States have shown ambiguous results according to Sobolewski. For instance several Spagnum-based wetlands were short-lived due to the limited metal-retaining capacity of peat. Few examples of constructed wetlands applied to metal mining are available. Sobolewski discusses examples from minerotrophic wetlands containing high concentrations of uranium, copper and pyrite, sometimes used as indicators of ore-bodies, as a principal argument that wetlands retain metals. In an examination of 11 mines where drainage passes natural wetlands, Sobolewski finds that in all drainage streams where water quality is satisfactory, the water is non-acid. He also points out the difficulties of making adequate mass-balance studies because of the influence of groundwater and complicated drainage patterns. The existence of a detritial layer may also contribute to an increased efficiency in metal retention, in contrast to immature constructed wetlands. It is also noted that water drainage may cease due to closed operations and thus the wetland vegetation may die out. Sobolewski concludes that constructed wetlands are an option for passive treatment of mine drainage.

A natural bog exposed to acid mine drainage was studied with the objective to test if a natural wetland can tolerate and ameliorate AMD (Wheeler et al., 1991). A pristine bog was exposed to a re-routed stream of acid mine drainage. The re-routing appeared to kill all vegetation. Work to provide alkalinity-generating bacteria started after the re-routing. The study included both laboratory and field studies. Alkalinity generating bacteria such as sulphate-reducing bacteria, iron-reducing bacteria amonifiers and denitrifiers were present in the bog in large numbers. These alkalinity-generating bacteria are favoured by reducing conditions. Based on laboratory experiments an increase in the microbial activity was expected with the application of hay as an easily degradable source of carbon. The reappearance of vegetation, cattail (Typha latifolia) was found during 1990 in areas of the bog with reduced flow. A decrease in conductivity but no clear trends in pH could be seen during 1990. (Timescale: summer 1988 re-routing of AMD; 1989 "very few plants alive"; summer 1989, application of hay; summer 1990, reappearance of cattail.)

Metal retention has also been studied in a larger scale than wetlands in Montana, USA (Williamson and Evans 1994). In that study ponds with similar characteristics as lakes were studied. The inlet to the pond was similar to limed drainage with a pH of about 9.5. The purpose of the ponds was to provide a place for suspended precipitates to settle. It was found that at some instances metal retention of the ponds were decreased. This was attributed to algal blooms. The vegetation increased pH, which resulted in the precipitation of calcium carbonate, but the presence of algae also increased the complexing capacity of the pond water. Overall this meant that the decrease in total hardness could affect the ability to meet the hardness based discharge criteria for copper documented increased sensitivity for biota at low total hardness and also that an increase in copper concentrations would be the case during these events (mainly spring).

Constructed wetlands

Constructed wetlands are reviewed by Kleinman et al. (1991). They report that several processes take place in constructed wetlands namely: adsorption and ion exchange, bioaccumulation, biotic and abiotic oxidation, sedimentation, bacterial sulphate reduction, neutralisation and dissolution of calcium carbonate. Generally they conclude that wetlands are cost efficient in the neutralisation of acidic drainage but supplemental chemical treatment is necessary.

The treatment of coalmine drainage is typically designed to optimise oxidative processes (e.g. the rate of oxidation of Fe(II) increases by the activity of iron-oxidising bacteria. In anaerobic parts of the wetland, bacterial sulphate reduction takes place producing alkalinity and hydrogen sulphide. This process will however be less important for the removal of iron compared to the oxidative process. The authors state that the most important effect of anaerobic processes on coalmine drainage is the production of alkalinity. Adsorption and ion exchange have limited effects after saturation of the adsorptive sites is completed. Bioaccumulation can be important for some species but the efficiency of accumulation can be contrary to the tolerance of mine water e.g. Spagnum accumulates iron effectively but may petrify, while Typha (cattails) is tolerant to mine water but has a poor ability to accumulate metals. Stimulation of microbial processes may however take place caused by plants. Organic material can induce oxidation by iron-oxidising bacteria or act as a substrate for sulphate reducing bacteria.

In order to stimulate sulphate reduction and passively increase the production of alkalinity it is important to make sure that a maximum contact with the organic substances and the drainage water is maintained. This could be obtained by the use of compost-based wetlands or by pre-treatment with anoxic limestone drains (ALD). ALD has been tested throughout the Appalachia by (Turner and McCoy 1990; Britt et al. 1990; Nairn et al. 1991). A combination of laboratory and field tests (Halifax International Airport) for the treatment of acidic drainage have been studied by Béchard et al. (1991). They used systems with straw together with wood shavings and added nutrients. Straw containing sulphate-reducing bacteria was also added to the field tests. Massive unintended introduction of oxygen as well as high flow events has affected the results of the field tests. The systems have been run for about 2.5 y with additions of urea, fresh straw, alfalfa etc. at several occasions. It was concluded in a follow-up presented 1995 (Béchard et al. 1995) that small mine water streams could be handled with this removal mechanism (sulphate reduction). In order to make the process a low maintenance alternative to liming, constant water levels, good flow control (minimising channelling) and a pre-precipitation pond for iron is needed.

In an attempt to elucidate changes in spatial patterns over time in wetlands systems with different types of organic materials were constructed (Wieder, 1991). No clear patterns were found but measured as inflow/outflow concentrations for the three types of studied wetlands (peat, straw/manure, mushroom compost) it was found that mushroom compost was most efficient in increasing pH and reducing Fe, Al and Mn. Straw manure had similar effect regarding Fe and Al but not so much effect on pH and Mn. The duration of the tests was 10 months.

Taddeo and Wieder, (1991) described five different wetlands that were constructed with Spagnum peat, Sphagnum peat with lime and fertiliser, sawdust, straw/manure and mushroom compost. Accumulation of iron was 1.7, 8.7, 11.1, 30.8 and 37.6 % of the total non-water soluble iron sulphide-Fe. A wet-chemical method for the determination of sulphide was used. The differences between the wetlands may be caused by differences in sulphate reduction rates or the stability of Fe-sulphides or both. Measurements took place during 16 months.

In a similar study by Calabrese et al. (1991) the effect of cattail (Typha latifolia) on iron removal was studied. Iron sulphide formation was believed to account for between 20 -28 % of the total iron removed during passage from the wetlands. Sulphate-reducing bacteria was counted and Eh and pH was measured, a speciation procedure for the iron was compared to the results from the wet-chemical extraction. The system was studied during one year.

The removal of copper in a neutral and low-strength mine drainage was investigated in British Columbia (Sobolewski et al. 1995). Two ponds were fertilised and planted with floating peat. pH was slowly decreasing in the outlet water. Sulphate reduction was indicated by the smell and by enumeration of sulphate reducing bacteria. However, the sequential leaching of copper instead indicated that copper was present in organic and exchangeable forms and also, surprisingly, in the oxidised iron phase. The experiments lasted for two years.

In Pennsylvania, USA constructed wetlands have been identified as the best available technology for the treatment of groundwater seepage from surface coal mining operations (Hellier et al., 1994). In another study from Pennsylvania in which six wetland treatment systems were constructed (Dietz et al., 1994) it was concluded that total acidity was the most appropriate parameter for the design of treatment systems and that acidity removal correlated with the size of the wetland. Total acidity was defined as the sum of hydrolyzable metals, mineral acidity, pH and carbonate acidity. The duration of the study was two years.

The removal of Ni from neutral mine drainage was measured (Eger et al., 1994) from concentration data in the inflow and outflow water from a small test cell, as well as from changes in concentrations in the vegetation and the substrate, peat. Limestone berms as a filter construction were introduced to increase alkalinity and prevent short-circuiting of the water. It was found that over 99% of the solid forms of Ni were in the peat. Based on a sequential extraction procedure and consistent with the constant concentrations of SO_4^{2-} it was concluded that the removal was associated with the organic fraction of the peat rather than precipitation of sulphides. A follow-up of this system was made by Eger et al., (1997). Assuming Ni retention of 10 g Ni /kg the system would loose its removal capacity in a few years. The area of the system was 930 m^2, water flow 0.3 - 0.6 l/s, Ni concentrations ca 20 mg/l and Cu concentrations ca 1 mg/l. The system has been monitored 2.5 y before treatment and 4 y after treatment.

Constructed wetlands have been applied for a neutral mine waste drainage in Ohio, USA (Stark et al., 1994). The study encompasses 8 years during which water quality has been studied. Treatment efficiency has been measured as removal of iron. The relative removal of iron measured as comparisons between outlet and inlet have increased during the time span of the study, but the removal measured as area-adjusted retention was constant. During the time of the study, the drainage water was treated with chemicals on several occasions, and the duration of "no-treatment" was 3.75 yr.

Other methods

Simple methods to reduce the amount of water that has to be treated may also be considered. Stuparyk et al. (1997) have designed a system where (surface) runoff is diverted from seepage. The runoff is then directed to a stream and thus reduces the volumes of water needed to treat. The seepage is collected anoxic in order to prevent precipitation of e.g. jarosite or iron-hydroxides. Clogging may otherwise effect the hydraulic properties of the drainage material.

Anoxic limestone drains can be used in order to increase the pH of the mine water. They can also be used as an initial step before the treatment in a wetland or separately. In the latter case it is preferred that a settling pond is available, since precipitation in the inside the ALD will decrease the water flow. Water flow is especially sensitive to the precipitation of iron, which can be avoided if the water is kept anoxic, however precipitation of aluminium will occur in the drain. In that case, water velocity should be sufficiently fast to rinse the ALD. Microbial activity may also decrease the efficiency of the ALD (Skousen and Ziemkiewicz, 1995). While it is easy to calculate the expected longevity of ALD based on water quality data and the dissolution of limestone in practice no long-term experiments have been done. Skousen and Ziemkiewicz reported that 8 of eleven investigated ALDs in West Virginia were functioning properly after one year. An investigation of the applicability of anoxic limestone drains (ALD) based on criteria set by the U.S. Bureau of Mines found that Colorado drains were not suitable for that treatment due to high concentrations of O_2 (Wildeman et al., 1997). The limitations to the technique, is according to the US Bureau of Mines, is set by the concentrations of Fe, Al, pH and dissolved oxygen.

A similar system for treatment of mine drainage water has been started by Blowes et al. (1995). A reactive wall consisting of composted leaf mulch, pine bark and wood chips agricultural limestone and creek sediments was constructed, in order to provide; an easily degradable source of carbon, a pH-buffering matrix and a culture of sulphate reduction bacteria. After 7 months of study, an increase in alkalinity as well as a reduction of Fe(II) and sulphate was found.

Miedicke et al. (1997) identifies three strategies available to minimise the long-term environmental impact: 1) Load reduction by control of the acid generating process or the acid drainage migration. 2) Metal recovery. By speeding up the oxidation rate, only small amounts of pyrite will be left after mine closure. 3) Acid drainage treatment. The treatment should continue until the load drops to acceptable levels without treatment. At the studied site Mount Lyell, Tasmania, neither treatment of the source diversion of water nor onsite containment was considered viable for different reasons. The options considered were treatment of the AD with SX/EW (Solvent extraction/Electrowinning. If this active system is able to reduce the loadings the Successive Alkalinity Producing Systems (SAPS) are suggested as a passive method to treat point sources. These systems consist of a combination of tanks filled with compost and limestone and settling tanks. It is considered to be a medium term solution for twenty years approximately. According to Miedicke et al. this systems have proven to be efficient in the eastern United States but performance on metal mine systems has yet to be proven.

5.4 Conclusions

There is a lack of long term evaluation of passive treatment systems. Most of the studies concerning passive systems have been investigated for only one or two years. In constructed systems where utilisation of precipitation of sulphides is intended, it seems to be a need for intermittent supply of easily degradable carbon. Conclusive evidence of precipitation of metal sulphides has not been found. The retention of metals must then at least initially be accomplished by other processes. One important process is the production or addition of alkalinity. If sulphate reduction is a slow process, additions of alkalinity are needed. A sustainable production of alkalinity in anoxic limestone drains is yet to be found. Several restrictions are put on the use of anoxic limestone drains such as flow rate, dissolved oxygen, concentrations of aluminium etc. All this together indicates that a totally passive system is not easily achievable and that surveillance to some extent is needed.

However, the capability of wetlands to retain metals, is verified by direct analysis of the material found in wetlands and is not questioned in principle. This can be caused by interactions between metals and solid forms of the organic material, but other mechanisms are also highly probable. Given sufficiently long retention times and low loadings and low acidity in the leachate, concentrations of metals in the water leaving the passive system may also become acceptable.

Comparisons of costs between passive systems and active treatment have not been done here. The costs for the land areas needed for passive systems might be an important factor in the choice of the most cost-efficient treatment method.

6 References

Aachib M., Aubertin M., Chapuis R.P., 1994: Column test investigation of milling wastes properties used to build cover systems. International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh.

Aubertin M., Chapuis R.P., Bouchentouf A., Bussière B., 1997: Unsaturated flow modelling of inclined layers for the analysis of covers. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Béchard, G., Goudey, P., Rajan, S. and McCready, R.G.L. 1991. Microbiological process for the treatment of acidic drainage at the Halifax International Airport. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept.18, 1991, Tome 3: pp 171 - 184, Montreal, Canada.

Béchard, G., McCready, R.G.L., Koren, D. W. and Rajan, S. 1995. Proceedings of the Sudbury 95, Conference on Mining and the Environment, May 28th - June 1st, 1995 Volume 2: pp 545 - 554.

Bell A.V., Riley M.D., Yanful E.K., 1994: Evaluation of a composite soil cover to control acid waste rock pile drainage. International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh.

Benson C. H. And Othman, M. A., 1993. Hydraulöic conductivity of compacted clay frozen and thawed in situ, Journal of Geotechnical Engineering, 119(2): pp 276-294.

Benson C. H., Abichou T. H., Olson M. A., and Bosscher P. J., 1995. Winter effects on hydraulic conductivity of compacted clay. Journal of Geotechnical Engineering 121(1): pp 69-79.

Bussière B., Nicholson R.V., Aubertin M., Servant S., 1997: Effectiveness of covers built with desulpharized tailings: Column test investigation. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Cabral A., Lefebvre G., Burnotte F., Proulx M-F., Audet C., Labbé M., Michaud C., 1997: Use of deinking residues as cover material in the prevention of AMD generation at an abandoned mine site. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Calabrese, J., Sexstone, A., Bhumbla, D., Bissonnette, G. and Sencindiver, J. 1991. Application of a constructed cattail wetland for the removal of iron from acid mine drainage. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept. 18, 1991, Tome 3: pp 559 - 576, Montreal, Canada

Collin M., Rasmuson A., 1986: Efficiency and design of layered soil covers for mine tailing dumps containing pyrite. Report to the Swedish Environmental Protection Agency.

Corser P., and Cranston M., 1991: Observations of long-time performance of composite clay liners and covers. Proc., Geosynthetics Design and Performance, Vancouver Geotech. Soc., Vancouver, British Columbia.

Day, Hope, Kuit (Cominco): Waste rock management planning for Khudz Ze Kayah Project, Yukon Territory, 1: Predictive static and kinetic tests work and 2: Waste management strategy (Fourth International Conference on Aciod Rock Drainage, Vancouver 1997)

Destouni G., Malmström M., Lindgren M., 1997: Predictive Modelling. MiMi State-of-the-art-report.

Diaz, M. A., Monhemius, A. J. and Narayanan, A. 1995. Consecutive hydroxide-sulphide precipitation treatment of acid rock drainage. Proceedings of the Sudbury 95 Conference on Mining and the Environment, May 28th - June 1st, 1995 Volume 2: pp 1181 - 1193.

Dietz, J., Watts, R., Stidinger, D. 1994. Evaluation of acidic mine drainage treatment in constructed wetland systems. Proceedings of the Int. Land Reclamation and Mine Drainage Conf. and the Third Int. Conf. on the Abatement of Acidic Drainage, April 24 - April 29, 1994, Vol 1: pp 70 - 79, Pittsburgh, USA.

Eberling B., Nicholson R.V., Reardon E.J., Tibble P., 1994: Evaluation of sulphide oxidation rates: laboratory study comparing oxygen fluxes and rates of oxidation product release. Canadian Geotechnical Journal, 31:375-383.

Eger, P., Wagner, J. R. and Melchert, G. 1997. The use of a peat/limestone system to treat acid rock drainage. Proceedings of the Fourth Int. Conf. on Acid Mine Drainage May 31 - June 6, 1997, Volume III: pp 1195- 1209, Vancouver, Canada.

Eger, P., Wagner, J., Kassa, Z. and Melchert, G. 1994. Metal removal in wetland treatment systems.Proceedings of the Int. Land Reclamation and Mine Drainage Conf. and the Third Int. Conf. on the Abatement of Acidic Drainage, April 24 - April 29, 1994, Vol 1: pp 80 - 88, Pittsburgh, USA.

Eigenbrod, K.D., 1996. Effects of cyclic freezing and thawing on volume changes and permeabilities of soft finr-grained soils. Canadian Geotechnical Journal, 33: pp 529-537.

Eriksson, J., 1982. Markpackning och rotmiljö/Ground compaction and root environment. Sveriges lantbruksuniversitet, Institutionen för markvetenskap, Report 126, Uppsala (in Swedish).

Granhagen J., Hallberg R., 1996: In situ investigation of acid mine drainage from a reclamation site at Galgberget, Falun. Annual report to the Swedish Waste Research Council. Department of Geology and Geochemistry, Stockholm University, Sweden.

Hellier, W., Giovanotti, E. and Slack, P. 1994. Best professional judgment analysis for constructed wetlands as a best available technology for the treatment of post-mining groundwater seeps. Proceedings of the Int. Land Reclamation and Mine Drainage Conf. and the Third Int. Conf. on the Abatement of Acidic Drainage, April 24 - April 29, 1994, Vol 1: pp 60 - 69, Pittsburgh, USA.

Höglund L-O., Lindgren M., Brandberg F., 1993: Metallfrigörelse från sandmagasinet i Adak vid alternativa täckskiktsutformningar /Metal release from the tailings pond in Adak depending of the cover design/. Report to the Swedish National Board for Industrial and Technical Development. Kemakta Konsult AB AR-92 (in Swedish).

Jessberger H.L., Stone K.J.L., 1991: Subsidence effects on clay barriers. Geotechnique, 41(2) pp 185-194.

Kalin M.,1993. Treatment of acidic seepage using microbiology. Final report MEND Project 3.11.1.

Kleinmann R.L.P., Hedin R.S. and Edenborn, H.M. 1991. Biological treatment of mine water - An overview. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept. 18, 1991, Tome 4: pp 27 - 42, Montreal, Canada.

Konrad J-M., 1989. Physical processes during freeze-thaw cycles in clayey silts. Cold Regions Science and Technology, 16: pp 291-303.

Köstler, J. N., 1968: Die Wurzeln der Waldbäume - Untersuchungen zur Morphologie der Waldbäume in Mitteleuropa. Verlag Paul Parey, Hamburg & Berlin (in German).

Lewis, Susetyo, Miller and Jeffrey, 1997: Waste Rock Management Planning and implementation at Freeport Indonesia Company´s Mining Operation in Irian Jaya. Fourth International Conference on Acid Rock Drainage, Vancouver 1997.

Lindvall M., Lindahl L-Å., Eriksson N., 1997: The reclamation project at the Saxberget mine, Sweden. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Lindvall M., Eriksson N., Jönsson H., 1997: Design of decommissioning plans at Boliden Mineral´s Aitik Mine. Fourth International Conference on Acid Rock Drainage, Vancouver 1997.

Lindvall M., Göransson T., Broman P.G., 1995: Efterbehandlingsplan Kristineberg, magasin 1 och 2 *(Decommissioning plan Kristineberg, tailings ponds #1 and 2)*. Boliden Mineral AB, Sweden. In Swedish.

Lundgren T., 1997: Bersbo pilot project - physical behaviour seven years after covering the waste rock piles. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Lundgren T., 1995: Sluttäckning av avfallsupplag /Closure of waste deposits/. Swedish Environmental Protection Agency, Report 4474.

Lundqvist G. (1997): Personal communication. Stora AB, Falun, Sweden.

Magnusson M., Rasmuson A., 1983: Transportberäkningar på vittringsförloppet i gruvavfall. Swedisn Environment Protection Agency, SNV PM 1689 (in Swedish)

MEND (Mine Environment Neutral Drainage Program) 1990. Assessment of existing natural wetlands affected by low pH, metal contaminated seepages (acid mine drainage) MEND Project 3.12.1.

MEND, 1994: Evaluation of alternate dry covers for the inhibition of acid mine drainage from tailings. MEND (Mine Environment Neutral Drainage Programme) Project 2.20.1.

Miedicke, J., Miller, S., Gowen, M., Ritchie, I., Johnston, J., and McBride, P. 1997: Remediation options (including copper recovery by SX/EW to reduce acid mine drainage from historical mining operations at Mount Lyell, Western Tasmania, Australia. Proceedings of the Fourth Int. Conf. on Acid Mine Drainage May 31 - June 6, 1997, Volume IV: pp 1451- 1468, Vancouver, Canada.

Miller, Jeffrey, Wong, Goldstone (Cyprus Gold New Zealand Ltd): In Pit Identification and Management of Acid Forming Waste Rock at the Golden Cross Gold Mine, New Zealand (Second International Conference on the Abatement of Acidic Drainage, Montreal 1991)

Millington R.J. and Shearer R.C., 1971: Diffusion in aggregated porous media. Soil Science, vol. 111, p. 372-378.

Montgomery R.J., Parsons L.J., 1989: The Omega Hills final cover test plot study: three year data summary. The 1989 annual meeting of the National Solid Waste Management Association, Washington D.C.

Murphy W.L., Gilbert P.A., 1985: Settlement and cover subsidence of hazardous waste landfills, US Environmental Protection Agency, EPA/600/2-85/035, Cincinnati, Ohio.

Naturvårdsverket, 1993: Gruvavfall från sulfidmalmsbrytning /Mine waste from sulphide ores/. Swedish Environmental Protection Agency Report No 4202 (in Swedish).

Nicholson R.V., Akindunni F.F., Sydor R.C., Gilham R.W. 1991: saturated tailings covers above the water table: The physics and criteria for design. Proceedings, Second International Conference on the Abatement of Acidic Drainage, Montreal.

Rasmuson A., Eriksson J-C., 1987: Capillary barriers in covers for mine tailing dumps. Swedish Environmental Protection Agency Report No 3307.

Ricard J.F., Aubertin M., Firlotte F.W., Knapp R., McMullen J., Julien M., 1997: Design and construction of a dry cover made of tailings for the closure of Les Terrains Aurifères Site, Malartic, Quebec. Fourth International Conference on Acid Rock Drainage, Vancouver, B.C. Canada, May 31-June 6, 1997.

Riveros, P. A. 1995. Applications of ion exchangers to the treatment of acid mine drainage. Proceedings of the Sudbury 95 Conference on Mining and the Environment, May 28th - June 1st, 1995 Volume 2: pp 441 - 449.

Mining Journal, Oct 1994: Setting a New Environmental Standard

Skousen, J.G. and Ziemkiewicz P.F. Ed. 1995. Acid mine drainage control and treatment. West Virginia University and the National Mine Land Reclamation Center, Morgantown, West Virginia, USA

Sobolewski, A. 1997. The capacity of natural wetlands to ameliorate water quality: A review of case studies. Proceedings of the Fourth Int. Conf. on Acid Mine Drainage May 31 - June 6, 1997, Volume IV: pp 1549 - 1566, Vancouver, Canada.

Sobolewski, A., Gormely, L. and Kistritz, R. U. 1995. Copper removal from mine drainage by an experimental wetland at Bell copper mine, Smithers, B.C. Proceedings of the Sudbury 95 Conference on Mining and the Environment, May 28th - June 1st, 1995 Volume 2: pp 683 - 692.

Stark, L.R., Williams, F. M. and Stevens, S. E., and Eddy, D. P. 1994. Iron retention and vegetative cover at the SIMCO constructed wetland: an appraisal through year eight of operationulated treatment wetlands improves mine water quality. Proceedings of the Int. Land Reclamation and Mine Drainage Conf. and the Third Int. Conf. on the Abatement of Acidic Drainage, April 24 - April 29, 1994, Vol 1: pp 89 - 98, Pittsburgh, USA.

Stuparyk, R.A., Dippong, J.P., Kerr, A.N., McDonald, T.J. and Wong, M. 1997. Methods of diverting surface runoff water from acid mine drainage at INCO´s copper cliff tailings areas. Proceedings of the Fourth Int. Conf. on Acid Mine Drainage May 31 - June 6, 1997, Volume IV: pp. 1579 - 1593 Vancouver, Canada.

Taddeo, F. and Wieder, R. 1991. The accumulation of iron sulphides in wetlands constructed for acid coal mine drainage (AMD) treatment. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept. 18, 1991, Tome 3: pp 529 - 548, Montreal, Canada.

Tremblay M., 1995: Modelling of groundwater conditions in silts and fine sands. a study of induced groundwater changes based on laboratory and full-scale field tests. Dissertation, Chalmers University of Technology, Sweden. Swedish Geotechnical Institute Report No 50.

Wheeler, W.N., Kalin, M. and Cairns, J.E. 1991. The ecological response of a bog to acidic coal mine drainage - deterioration and subsequent initiation of recovery. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept. 18, 1991, Tome 2: pp 449- 464, Montreal, Canada

Wieder, R. 1991. Spatial and temporal patterns in surface and subsurface water chemistry in wetlands constructed for acid coal mine drainage (AMD) treatment. Proceedings of the Second Int. Conf. on the Abatement of Acidic Drainage, Sept. 16 - Sept. 18, 1991, Tome 3: pp 507- 528, Montreal, Canada.

Wildeman, T.R., Dinkel, J.W., Smith, R.M., McCallister, M.L. 1997. Field assessment of Fe(III) , Al and dissolved O_2 for passive treatment options. Proceedings of the Fourth Int. Conf. on Acid Mine Drainage May 31 - June 6, 1997, Volume IV: pp. 1659 - 1672 Vancouver, Canada.

Viklander, P., 1997: Compaction and thaw deformation of frozen soil - permeability and structural effects due to freezing and thawing. Division of Soil Mechanics and Foundation Engineering, Department of Civil and Mining Engineering, Luleå University of technology (Diss.).

Williamson, R.L. and Evans, A. 1994. The effects of limnological parameters on lime slurry treated water in large detention ponds. Proceedings of the First Int. Conf. on Tailings and Mine Waste, Jan. 19 - Jan. 21, 1994, 1: pp 137 - 146, Fort Collins, USA.

Wilson G.W., Barbour S.L., Swanson D., O´Kane M., 1995: Instrumentation and modelling for saturated/unsaturated performance of soil covers for acid generating waste rock. Hydrogéologie, n° 4, 1995, pp. 99-108.

Yanful E.K., Aubé B.C., Woyshner M., Arnaud L.C., 1994: Field and laboratory performance of engineered soil covers on the Waite Amulet tailings. International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh.

Öhlander B., Ingri J., Widerlund A., Holmström H. 1997: Field studies and characterisation. MiMi State-of-the-art-report.

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.

MiMi
Mitigation of the environmental impact from mining waste

Visit us at:
http://www.mimi.kiruna.se

ISSN 1403-9478
ISBN 91-89350-02-2