Environment and Security Initiative in South Eastern Europe:
Improving regional cooperation for risk management from pollution hotspots as well as the transboundary management of shared natural resources

Mining for Closure: Innovations for contaminated mine waters assessment, management and remediation

Innovative Techniques and Technologies for Contaminated Mine Waters Assessment, Management and Remediation

Technical Workshop Report

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Prepared by
Christina Stuhlberger, UNEP/GRID Arendal
Philip Peck, UNEP/GRID Arendal
Gilles Tremblay, Natural Resources Canada
Nand Davé, Natural Resources Canada

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Summary

The Environment & Security Initiative seeks to promote the reduction of environmental and human health risks resulting from substandard mining operations within its project “Environment and Security in South Eastern Europe: Improving regional cooperation for risk management from pollution hotspots as well as the transboundary management of shared natural resources”. In March 2007, it organised and delivered the “Technical workshop on innovative techniques and technologies for contaminated mine waters assessment, management and remediation” in Brestovacka Spa, Bor, Serbia to which this technical report is an outcome. Innovative techniques for mine water management have been identified, delineated and analysed against the background of their general suitability and applicability in the SEE region.

Metal contamination of soils and waters has severe impacts on the environment and on human health around the world. Contaminated water from mines is one of the largest sources of heavy metals that pollute the environment. Common characteristics are low pH and dissolved toxic metals, which result from natural weathering processes of sulphide minerals. The responsible processes are considerably accelerated by mining activity which create the issue of contaminated mine water.

Mine water contamination issues are further aggravated in regions such as South Eastern Europe (SEE) where a lack of economic and technical capacity obstructs a sustainable approach to solving the problem. Environmental pollution causes damage not only in the country where it is generated, but also in neighbouring countries as mine water pollution migrates via transboundary waterways. The adverse impacts on human health, biodiversity, and local economies have the potential to cause political instability in the region. On the other hand, countries with pending EU accession, such as the countries in the Western Balkans, are motivated to address their environmental issues to comply with the requirements of EU legislation. For mine water in particular, complying with the Water Framework Directive will be a major challenge.

During the last 25 years, innovative, cost-efficient and robust mine water treatment techniques have been developed that rely on naturally occurring processes, such as bacterial activity and oxidation. These techniques are most commonly used in North America, though they have begun to be used in Western Europe as well. Due to their advantages compared to conventional treatment, these techniques are a viable alternative for decontaminating mine water in SEE and other regions.

The main advantages of innovative mine water management measures are their lower overall cost, their more attractive cost-structure and their higher robustness compared to conventional techniques. Furthermore, most of the techniques presented in this thesis add to the amenity and habitat value of the region.

It could be shown that a range of innovative management measures is available...
for Acid Mine Drainage (AMD) abatement whose applicability varies strongly with the particular site specific conditions. Innovative mine water treatment techniques are restricted by a number of factors and cannot be seen as a panacea for the whole issue. Compared to conventional treatment, innovative techniques are less reliable, less flexible and may be incapable of complying with legal requirements in some circumstances. However, they may offer an opportunity to reduce mining related risks where they are applicable and where flexible solutions for legal requirements can be found. In particular these techniques should be taken into consideration at abandoned and orphaned mining sites where conventional techniques are not feasible due to financial and technical constraints.

Climatic and geological conditions in SEE generally favour the application of innovative techniques. Exceptions are those areas where extreme conditions with regard to temperature, topography and precipitation prevail.

Knowledge about innovative mine water management should be further disseminated to decision-makers in SEE to promote implementation of the techniques. At the same time, assessment of mining sites and installation of innovative pilot mine water management measures in the region should be encouraged. It is expected that with further research, greater application frequency and more long-term experience, innovative mine water management will become more reliable, effective and eventually more applicable.
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<td>ADA</td>
<td>Austrian Development Agency</td>
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<tr>
<td>AMD/ARD</td>
<td>Acid mine drainage/Acid Rock Drainage</td>
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<td>ALD</td>
<td>Anoxic Limestone Drain</td>
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<tr>
<td>CIDA</td>
<td>Canadian International Development Agency</td>
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<tr>
<td>CSM</td>
<td>Conceptional Site Model</td>
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<tr>
<td>CSR</td>
<td>Corporate Social Responsibility</td>
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<tr>
<td>CW</td>
<td>Constructed Wetland</td>
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<td>ENVSEC</td>
<td>Environment and Security Initiative</td>
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<tr>
<td>EU</td>
<td>European Union</td>
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<tr>
<td>OLC</td>
<td>Open Limestone Channel</td>
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<tr>
<td>OLD</td>
<td>Open Limestone Drain</td>
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<tr>
<td>OSCE</td>
<td>Organization for Security and Cooperation in Europe</td>
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<tr>
<td>MNA</td>
<td>Monitored Natural Attenuation</td>
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<tr>
<td>NATO</td>
<td>Northern Atlantic Treaty Organisation</td>
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<td>NMD</td>
<td>Neutral Mine Drainage</td>
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<td>PRB</td>
<td>Permeable Reactive Barrier</td>
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<td>RAPS</td>
<td>Reducing Alkalinity Producing System</td>
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<tr>
<td>REC</td>
<td>Regional Environmental Centre for CEE</td>
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<td>SEE</td>
<td>South Eastern Europe</td>
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<tr>
<td>SCOOFI</td>
<td>Surface-Catalysed Oxidation Of Ferrous Iron</td>
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<tr>
<td>UNDP</td>
<td>United Nations Development Programme</td>
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<tr>
<td>UNEP</td>
<td>United Nations Environment Programme</td>
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<tr>
<td>VFR</td>
<td>Vertical Flow Reactor</td>
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<td>VFW</td>
<td>Vertical Flow Wetland</td>
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<td>WFD</td>
<td>Water Framework Directive</td>
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1 Introduction

In March 2007, the Environment & Security Initiative organised and delivered the “Technical workshop on innovative techniques and technologies for contaminated mine waters assessment, management and remediation” in Brestovacka Spa, Bor, Serbia. This report seeks to provide a comprehensive presentation of the topics discussed at the workshop and reflects the experiences gained by international and local experts during the four days in Bor.

The aim of the process of which this workshop and report contribute to, is the desire to stimulate broad and effective uptake of innovative techniques and technologies for cost effective, robust, and socially responsible remediation of contaminated mine water throughout South East Europe. The workshop included a range of stakeholders from across the South Eastern European region. Among them were key decision-makers from ministries, academia and mining industry. The central theme, examined within the workshop, is the abatement of ongoing generation of acidic, metals bearing effluents from sub-standard mining sites or operations that affect surface waters and groundwaters. Therefore, possibilities of addressing these challenges with innovative and cost-efficient prevention and treatment techniques were introduced and it was delineated where they may be feasible. With regard to this issue, workshop topics related to mining site management, such as governance principles and risk assessment, as well as case studies and design exercises are presented to provide a concise report on issues relevant to mine water management to the reader. As such, the workshop and its outcomes constitute an integral part of the UNEP project “Environment and Security in South Eastern Europe: Improving regional cooperation for risk management from pollution hotspots as well as the transboundary management of shared natural resources”. In essence, the Initiative is structured upon three distinct but interlinked pillars, dealing with:

- vulnerability assessment and monitoring
- capacity building and institutional development
- policy development and implementation.
By implementing the types of approaches and techniques discussed at the workshop, risk mitigation is supported and a general improvement of environmental quality and living conditions in the region can be attained. Moreover, the workshop sought to substantially increase the knowledge of important decision makers and practitioners of:

- best practice or innovative and robust mine water treatment techniques and technologies, with a focus on passive and semi passive techniques;
- technical needs and requirements for cost effective site assessment techniques – particularly focused on assessment of mine waters and effluents;
- the current status of European legislation as it relates to mine waters and effluents;
- design procedures and examples that can be utilized as a basis for experience sharing among practitioners in their own home countries.
# Background

## Environment & Security Initiative work in SEE

The Environment and Security (ENVSEC) initiative was formed in 2003 by three organizations – the United Nations Environment Programme (UNEP), the United Nations Development Programme (UNDP), and the Organization for Security and Co-operation in Europe (OSCE). In 2004, it was joined by NATO and recently by the REC and the UNECE. The Initiative aims to provide a framework for dealing with environmental issues across borders and promoting peace and stability through environmental co-operation and sustainable development. After the launching of the Initiative at the Kiev “Environment for Europe” Ministerial Conference in May 2003, the ENVSEC Initiative has initially focused on the three regions: Central Asia, the Caucasus and South Eastern Europe (SEE).

The ENVSEC SEE Consultations in Kiev and in Skopje, September 2004, identified hazardous activities, including mining operations and associated processing activities, as posing potential risks of trans-boundary character in the region. This was confirmed as a main priority for the region at a ministerial-level meeting in Cluj-Napoca, Romania in May 2005. Within this field, the UNEP project “Environment and Security in South Eastern Europe: Improving regional cooperation for risk management from pollution hotspots as well as the transboundary management of shared natural resources” contributes to the management of such trans-boundary risks by: providing deeper risk assessments for decision-makers; clearer identification and/or verification of potential “hot spots”, and recommendations for priority measures of mitigation and risk management at the regional level. Moreover, the project will carry out capacity building in the field of tools for early warning and conflict resolution. ENVSEC work in South Eastern Europe has thus far been supported by ADA and CIDA.
2.2 Mine Water Issues

Inadequate pollution control during operations, unacceptable waste disposal and/or storage practices and a general absence of mine rehabilitation and closure activities are typical outcomes of sub-standard mining operations in the SEE region. This has resulted in – and continues to cause – significant adverse environmental and health and safety impacts and related liabilities.

Of major concern is the ongoing generation of contaminated mine waters that affect both surface waters and groundwaters. Contaminated mine waters are formed by the oxidation of sulphide minerals and they are generally referred to as Acid Mine Drainage (AMD) or Acid Rock Drainage (ARD). They can originate at various stages of the mine cycle (Kroll, Amezaga et al. 2002):

- during dewatering for mining purposes
- by seepage of contaminated leachate from waste rock piles and tailings dams
- during flooding of workings after extraction has ceased
- by discharge of untreated waters after inundation.

Contaminated mine waters typically have low pH, elevated concentrations of heavy metals, and/or increased salinity loads that can cause substantial damage in freshwater ecosystems and in public water sources due to their bio-chemical and eco-toxicological effects. As such pollution generation can continue for decades and even centuries after the cessation of industrial activity, there is an urgent need for cheap, largely self-sustaining remediation methods.

The ENVSEC project on mining related hazards has focused on water pollution and the abatement of waterborne contaminant distribution. Mining has many potential environmental impacts – and some inevitable, but in the sphere of water pollution, impacts of mining on the environment and consequently on human health as well as political issues, such as transboundary risks are of special concern.

Waterways have been identified as the single greatest pathway and receptor for mining related contaminants. Moreover, in clustered regions, waterways often flow across jurisdictional boundaries. Pollution of those waterbodies often results in transboundary pollution. This is the case for many waterways in South Eastern Europe. While one must bear in mind that mining can seriously impact other key compartments of the natural environment, the significant background work within the Environment
and Security initiative1 has indicated that a focused on mine water pollution prevention and treatment must be a (if not “the”) priority area.

Major changes in mine water treatment approaches have resulted from the development of innovative and low cost techniques in this field of engineering over last two decades. While this development has mainly taken place in developed “western” economies, the fact remains that these countries DO share similar mining related environmental problems to those experienced by the countries in South Eastern Europe.

2.3 Technical Workshop in Brestovacka Spa, Bor, Serbia

A major goal in the project is to facilitate the transfer of this knowledge gained over the decades between those countries who have already successfully implemented remediation projects and those who still require support to handle their challenges.

In order to encourage this knowledge transfer, decision-makers and other relevant stakeholders are given the opportunity to incorporate themselves into already existing networks that are either dealing with the relevant issues or learning to do, and from which support can be obtained. Eventually, the aim is to increase implementation of remediation measures at problematic sites throughout SEE. However, it is not the ENVSEC Initiative who is in the position to execute these implementations. Its role is rather the one of a facilitator who provides stakeholders with relevant information on the issues and a forum where further, specific support can located.

With these points in mind, ENVSEC organised the “Technical workshop on innovative techniques and technologies for contaminated mine waters assessment, management and remediation” from 26th to 29th of March 2007 in Bor, Serbia. The key contributors to the workshop consisted of five international experts, from Canada, the UK and Germany who have extensive experience in the field of mine water treatment and management, and around thirty regional participants/contributors who are dealing mining issues and/or the environmental relevance of mining activities. Participants came from institutions across SEE and included representatives of Ministries of Environment, research institutes, mining companies operating in the region, donors and

other contributors. The countries represented at the workshop were Serbia, Bosnia and Herzegovina, Montenegro, Kosovo (Territory currently under interim UN administration) and Macedonia.

Moreover, Canadian experts extended the duration of their visit to the Balkans and performed visits to some of the mining sites in Albania (Gjegjan, Repsi, Kurbnesh, Reshen, Fushe Arres, Rubik) and Bosnia and Herzegovina (Srebrenica and Vares), that were identified as high priority during ENVSEC missions in 2006, in the auspices of the ENVSEC programme. The purpose of the visits to the mining sites was to verify the actual status quo of these sites on the ground and what, according to the experience, might be a good, simple and affordable technical option for remediation.
2.4 Technical Workshop Report

This report is a deliverable of the workshop held in Brestovacka Spa March 26 – 29, 2007, by the ENVSEC initiative. It seeks to summarise and present the core portions of the information communicated and experiences gained during the workshop interactions and presentations and as such it is intended to concretise the workshop outcomes. That which it cannot communicate are the many conversations, exchanges or information and experiences and networking that took place among the participants – however, it is stressed that these items were a central deliverable of the event.

In summary for this introductory section, it is hoped that the workshop and this report will foster future cooperation between local and international experts and stimulate the further dissemination of expertise on mine water issues to an audience that embraces many more than the workshop attendees.

Photo courtesy of Edin Delic
3 Mine Water Generation

This section aims to introduce the reader to the basic principles of mine water generation and to outline the key parameters which are later required to select and design effective treatment schemes. Moreover, the main impacts of contaminated mine water are summarized to underpin the relevance of mine water treatment to human health and environment.

Mechanisms will be explained that underlie the generation of contaminated mine water as a result of sulphide mineral weathering, which occurs when sulphide minerals are exposed to water and oxygen. Other sources of water pollution associated to mining such as chemicals used for leaching processes (e.g. cyanide) or accidental spilling will not be regarded at this point but are nonetheless relevant.

3.1 What is Contaminated Mine Water and where does it come from?

Mine water generation is a problem that has occurred ever since the first mining activities started thousands of years ago. The famous Rio Tinto (engl. “Red River”) in south-western Spain, for example, owes his name to the mining of massive sulphide deposits extending back to the Copper and Bronze Ages (Davis, Welty et al. 2000). This example also displays one of the biggest concerns regarding mine water; its longevity once its generation has started.

Contaminated mine water is often referred to as acid mine drainage (AMD) or acid rock drainage (ARD). In this context, it shall be mentioned that not all mine waters have to be necessarily acidic in order to be contaminated. In some cases, dissolution of minerals such as calcite in the host rock can neutralize mine water before emerging to the surface. Alkaline water commonly displays a lower contaminant concentration, but even when water is of neutral pH, it can still contain considerable amounts of toxic elements.
Commonly, neutral mine waters are less problematic and easier to treat than those who are highly acidic and therefore this quality has influence on the choice of a water treatment scheme. The crucial parameters are net-alkalinity and net-acidity. Basically, acidity comprises all components present in a solution which contribute or will under certain conditions contribute to proton activity (in mine water these compounds are commonly metals and free protons). The same principle applies to alkalinity with the difference that it accounts for all actual and possibly alkalinity generating components in a solution (commonly bicarbonate, carbonate, and hydroxide) (Kirby and Cravotta 2005a). As potentially proton/hydroxide releasing elements are also into account in addition to actual proton concentration, pH is not clearly dependant on a certain acidity/alkalinity concentration. Depending whether alkalinity or acidity concentration is higher, the water is either net-acidic or net-alkaline.

According to Brown, Barley et al (2003) sources for contaminated mine water from mining operations include:

- drainage from underground workings,
- runoff from open pit workings,
- waste rock dumps piles usually due to infiltration of rain water or groundwater,
- spent ore piles from heap leach operations,
- upon abandonment of mines, which is typically followed by flooding,
- active and abandoned tailings lagoons.

![Golf course on a reclaimed colliery spoil heap in Bowden Close, UK.](image)
At mining site, water can discharge into the environment from various sites and in various manners. The schematic design of a tailings\(^2\) dam in Figure 3.2 shows the water flux of a common mining facility. Tailings impoundments are at many sites the origin for mine water discharging into the environment and they pose risks to the environment when dam stability is not guaranteed. This example is aimed to demonstrate the various flows that contribute to mine water generation and have to be regarded when planning a water management system for such a facility. It also introduces the reader to the principle of tailings dams, as they are often a major concern when it comes to mining associated risks (BRGM 2001).

![Water gains and losses at a terrestrial impoundment](image)

Figure 3.2 Water flows entering and leaving a tailings impoundment (UNEP).

## 3.2 Impacts of Contaminated Mine Water

The prevalent contaminating characteristics associated to mine water are low pH, high metals and sulphate load and discoloration of water. Risk to the environment is posed by the direct toxic effects of the contaminants, their physical properties (i.e.

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\(^2\) Tailings are materials left over after the process of separating the valuable fraction from the worthless fraction of an ore.
sedimentation of precipitates in watercourses) and their adverse effects on amenity in a region that can strongly influence tourism or the marketing of agricultural products. See Table 3.1 for a summary on environmental impacts of common mine water contaminants.

Indirectly, mine water can cause serious damage when the water table rises after mine closure and cessation of pumping. Firstly, the rising water may displace gasses like methane present in the mine voids upward and cause emission to the surface and related health risks. Secondly, changes in the water saturation of the underground may cause structural problems, such as land subsidence, in the area.

A thorough examination of impacts of mine water on aquatic ecosystems, social and cultural heritage issues is presented by (Kroll, Amezaga et al. 2002).

<table>
<thead>
<tr>
<th>Property</th>
<th>Chemical species</th>
<th>Concentration range in solution</th>
<th>Environmental impact</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acidity</td>
<td>H⁺</td>
<td>pH &lt; 4.5</td>
<td>Loss of bicarbonate to photosynthetic organisms, degradation and death to animals and plants; reduction in drinking water quality, mobilisation of metal ions; corrosion of man-made structures</td>
</tr>
<tr>
<td>Iron precipitates</td>
<td>Fe³⁺, Fe²⁺, Fe(OH)₃ (s)</td>
<td>100 to 1-9x10³ mg/l</td>
<td>Discoloration and turbidity in receiving water as pH increases and ferric salts precipitate; smothering of benthic organisms and clogging up of fish gills; reduction in light penetrating the water column; encrustation of man-made structures</td>
</tr>
<tr>
<td>Trace metals</td>
<td>Cu, Pb, Zn, Cd, Co, Ni, Hg, As, Sb</td>
<td>0.01 to 1-9x10³ mg/l</td>
<td>Degradation and death to animals and plants; bioaccumulation; reduction in drinking water quality: soil and sediment contamination</td>
</tr>
<tr>
<td>Total dissolved solids</td>
<td>Ca, Mg, K, Na, Fe, Al, Mn, Si, sulphate</td>
<td>100 to more than 1-9x10⁴ mg/l</td>
<td>Reduction in drinking water quality, reduction in stockwater quality, encrustation in of man-made structures when solids precipitate; soil and sediment contamination</td>
</tr>
</tbody>
</table>

Table 3.1 Main characteristics of contaminated mine waters and their potential environmental impact (Lottermoser 2003)
3.3 Mine water principles

3.3.1 Generation processes

In order to understand the formation and problems arising from contaminated mine water, one has to take a closer look at the generation process.

Basically, mine water generation is a weathering process that takes place in varying extent in everywhere where sulfide bearing minerals are exposed to water and oxygen. The reason why mining activities are having adverse effects on the water environment is the intensification of this process due to drastically increased exposure of relevant minerals to water and oxygen by excavation and milling.

Excavation removes sulfide minerals as in Table 3.2 from their non-reactive environment and brings them into contact with water and oxygen. Crushing and milling for ore extraction enhance the mineral surface area which further increases reaction rates (Lottermoser 2003).

<table>
<thead>
<tr>
<th>Mineral</th>
<th>Composition</th>
<th>Aqueous end-products of complete oxidation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amorphous FeS</td>
<td>Fe$^{3+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Arsenopyrite FeAsS</td>
<td>Fe$^{3+}$, AsO$_4^{3-}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Chalcocite Cu$_2$S</td>
<td>Cu$^{2+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Chalcopyrite CuFeS$_2$</td>
<td>Cu$^{2+}$, Fe$^{3+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Cinnabar HgS</td>
<td>Hg$^{2+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Galena PbS</td>
<td>Pb$^{2+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Opiment As$_2$S$_3$</td>
<td>AsO$_4^{2-}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Pyrite FeS$_2$</td>
<td>Fe$^{2+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
<tr>
<td>Sphalerite ZnS</td>
<td>Zn$^{2+}$, SO$_4^{2-}$, H$^+$</td>
<td></td>
</tr>
</tbody>
</table>

Table 3.2 Common sulphide minerals implicated in the production of contaminated mine water (adapted from Brown, Barely et al 2003).

A commonly abundant sulphide mineral present at ore and coal mining sites is pyrite that contributes significantly to mine water generation. Apart from its relevance to mine water generation, pyrite oxidation is also the most studied mechanism in mine water generation. Therefore the process is described in the following.
Reaction 3.1 shows that the oxidation of one part of pyrite releases two parts of protons. Hence, the pH of a solution decreases when sufficient pyrite is turned into ferric iron, sulfate and protons

$$2 \text{FeS}_2 + 7 \text{O}_2 + 2 \text{H}_2\text{O} \rightarrow 2 \text{Fe}^{2+} + 4 \text{SO}_4^{2-} + 4\text{H}^+ \quad (3.1)$$

Mine water generation is a self-sustaining cycle, which means that once mine water generation has taken off, it does not stop until one of the necessary agents, commonly sulphide minerals, becomes depleted.

Apart from oxygen, ferric iron (Fe$^{3+}$) is the relevant element in oxidising pyrite (see Reaction 3.4) but it is only provided after ferrous iron (Fe$^{2+}$), produced in Reactions 3.1 and 3.4, is oxidised in Reaction 3.2. The oxidation step is relatively slow on a purely abiotic basis but significantly accelerated by microbiological activity, in particular at low pH (Evangelou 1995).

Once ferric iron has formed, it can either hydrolyse to ferric hydroxide and precipitate, Reaction 3.3, or oxidise pyrite as in Reaction 3.4. Both reactions release protons which will consequently lead to pH decrease unless the water is sufficiently buffered. Metal hydrolysis is a crucial process in passive mine water treatment and should therefore be remembered. Products of this reaction are protons and metal hydroxide flocs that accumulate as low density sludge of around 1 – 5 % solid content (Evangelou 1995).
Whether ferric iron precipitates or involves into pyrite oxidation depends on the pH of the solution. Where the pH exceeds a value of approximately 3, Reaction 3.5 will occur and consume ferric iron (Lottermoser 2003), when pH conditions preclude iron precipitation (pH < 3), ferric iron contributes to pyrite oxidation.

\[
\begin{align*}
4 \text{Fe}^{2+} + \text{O}_2 + 4\text{H}^+ & \rightarrow 4 \text{Fe}^{3+} + 2\text{H}_2\text{O} \quad (3.2) \\
\text{Fe}^{3+} + 3\text{H}_2\text{O} & \rightarrow \text{Fe(OH)}_3(\text{s}) + 3\text{H}^+ \quad (3.3) \\
\text{FeS}_2 + 14 \text{Fe}^{3+} + 8\text{H}_2\text{O} & \rightarrow 15 \text{Fe}^{2+} + 2\text{SO}_4^{2-} + 16\text{H}^+ \quad (3.4)
\end{align*}
\]

The perpetual process of mine water generation, resulting in contaminant release, constitutes the most difficult aspect in achieving sustainable mine water treatment. To overcome the problem, self-sustainable and low-cost techniques are required that allow economical application in the same timescales in which mine water generates. Passive prevention and treatment options which will be presented in the course of the report are addressing this issue by various means and their development and performance so far has given reason to raise expectations that the solution can be found in such applications.

### 3.3.2 pH as a master variable

When sulfide minerals oxidise, protons are released into the water. Once the natural buffering capacity of the water is depleted, the pH will drop according to the amount of protons generated by sulfide weathering. The direct effect of a low pH may be the loss of the aquifer or stream as drinking water supply. Also, a decreasing pH correlates with the degradation of life conditions for most organisms so that loss of biodiversity for an AMD affected stream will be observed. Except for these direct effects, there are significant indirect effects that come along with low pH in waters.

Most ore and coal deposits exist in association with other elements apart from the desired commodity, such as arsenic, cadmium, copper, iron, mercury, lead, antimony, zinc and other metals/metalloids. When sulfide minerals are oxidised, their associated metals are liberated. Most of them increase solubility when pH decreases. Elements in solution are more toxic to the environment because of their much higher bioavailability compared to metals in solid compounds. Moreover, they can distribute further as their
downstream transportation is facilitated. In Figure 3.4 the correlation between the solubility of some metals and pH is demonstrated.

![Figure 3.4 Solubility of various metal ions relation to pH (Wolkersdorfer)](image)

From these relations it becomes clear that for successful AMD treatment pH adjustment is crucial, if not the most important goal that has to be achieved. Once one has managed to raise pH to a level where the target metals leave solution, subsequent deposition of the precipitates by sedimentation processes can take place.

### 3.3.3 Precipitation and sedimentation

In natural environments, a major cause for metal removal is hydrolysis and subsequent precipitation and sedimentation. The pH in a contaminated stream rises when mixing with unaffected, neutral water, e.g. from a stream or from atmospheric precipitation. Resulting hydroxide flocs settle in the streambed and form a low-density sludge.

Because of this processes, metal hydroxide sludge is regularly encountered on the bottom of AMD affected streams. Except for the unsightliness which such sediments cause in a river, there are further implications concerning the ecological value of a stream. By precipitating and settling along the riverbed, the metals are smothering the substrate interstices. These little spaces in the riverbed are habitat to many species,
vertebrates and invertebrates. The sedimentation of the metal flocs renders the interstices inaccessible to organisms whereby the biodiversity of the area becomes significantly affected. Figures 3.5 and 3.6 show how metal precipitation can manifest itself and have readily visible impact on a stream.

Figure 3.5  Aluminium hydroxide precipitates in a streambed in Bowden Close, UK.
3.3.4 Temporal changes in mine water quality

With regard to the long-term persistence of water pollution after a mine is flooded, it has been observed that contaminant concentration usually reaches its climax shortly after the mine has been completely flooded. After reaching the climax, contaminant concentrations decrease steadily afterwards. The gradient is related to sulphide mineral oxidation and dissolution of hydrogen releasing minerals. These processes slow down once the sulfide minerals are submerged beneath water and oxygen ingress becomes restricted. This characteristics of mine water concentration from flooded deep mines has been termed “first flush” by Younger (2000) and its characteristics are shown in Figure 3.7. It can be expected that the period of time for the “first flush” is about four times longer than it took the mine to flood (Younger 2000). This characteristic is important when it comes to the selection of the treatment system for a particular mining site and will be further discussed in the following section.

Although it shows that improvement of water quality takes place over time, it cannot be assumed that water treatment will necessarily become dispensable. Ultimately, pollution will go on until such time as pollutant source minerals are finally exhausted, this may extend over hundreds of years (ERMITE-Consortium 2004).
Figure 3.7 Temporal change in mine water acidity (ERMITE-Consortium 2004)
4 Coming to Terms with Contaminated Mine Waters

Discharge of contaminated mine water into the environment has shown to be responsible for many problems, including severe impact on human health, environment and commercial values. Additional to the severity of direct mine water impacts, the perpetuity of mine water generation further aggravates the whole issue and makes ignoring the problem impossible or at least very unreasonable. There is a need for action where mine water causes adverse effects and steps have to be taken to reduce these impacts. In principle there are three ways how mine water remediation can be approached:

- Monitored Natural Attenuation
- Mine Water Prevention/Minimisation
- Mine Water Treatment.

Mine water treatment can be divided into active treatment measures and passive treatment measures. Within the ENVSEC project, it was focused on passive treatment techniques. Where passive treatment is applicable, its characteristics suit in many cases the needs of mining sites in South Eastern Europe which call for cost-effective, robust and socially responsible solutions.

It should be pointed out that passive treatment is not a panacea to the mine water problem. On the contrary, in situations where high contaminant concentrations and/or high flow rates occur and consequently high risk is posed to recipients, active treatment options are more reliable and therefore the better option. Also, passive treatment techniques are in many applications not used as a stand alone treatment system, combinations of active and passive components within a system are common, just like the combination of prevention measures and treatment measures. This combination is often deployed either to minimise the discharge values to meet regulatory compliance or to make passive treatment feasible in the first place as many passive treatment techniques require relatively low to moderate contamination rates and low to moderate
flow rates. The following section will present the commonly applied techniques for mine water abatement and explain the conditions under which they can be implemented.

4.1 Monitored Natural Attenuation (MNA)

The term monitored natural attenuation (MNA) basically refers to an approach where natural processes are regarded to be sufficient to handle the contamination and has been defined by the US EPA as:

“...the reliance on natural attenuation processes (within the context of a carefully controlled and monitored site cleanup approach) to achieve site specific remediation objectives within a timeframe that is reasonable compared to that offered by other more active methods. The natural attenuation processes that are at work in such a remediation approach include a variety of physical, chemical, or biological processes that, under favourable conditions, act without human intervention to reduce the mass, toxicity, mobility volume, or concentration of contaminants in soil or groundwater. These in-situ processes include biodegradation, dispersion, dilution, sorption, volatilisation, radioactive decay and chemical or biological stabilisation, transformation or destruction of contaminants”.

(U.S. EPA 1999)

Where MNA is possible, it will be a cost-efficient and sustainable solution to deal with mine water and conditions should be assessed if MNA application is feasible. Factors determining the rate of natural attenuation and consequently its applicability as viable mine water treatment option are dilution and the degree of natural “self cleansing” occurring in the flooded workings (ERMITE-Consortium 2004). In order to assess availability and degree of dilution at a site, hydrometric and water quality monitoring installations are required, ideally supported by mathematical modelling (ERMITE-Consortium 2004).

Geochemical factors that have influence on the contaminant release of mining site are the relative abundances of sulfides, carbonates, and silicate minerals within the mined systems and receiving catchment. Calcite, for example, consumes acidity upon
dissolution which reduces metal solubility and hence pollutant release. The suitability of MNA as only abatement measure requires the evaluation of acidity and alkalinity generating compounds present in the workings, pollutant release rates and the rate of mixing processes as well as the rates of geochemical attenuation reactions (ERMITE- Consortium 2004).

4.2 Mine Water Prevention

It is widely known that “prevention is better than cure” so avoiding or at least reducing contaminated mine water generation in the first place is desirable. Prevention measures therefore seek to reduce the amount of contaminants released into the water and to reduce the total amount of water emanating from a mining site. Unfortunately, prevention is not always possible due to technical restrictions and local conditions (Brown, Barley et al. 2002).

The aim of mine water prevention is to minimize contaminant release and can be achieved by excluding at least one or more of the factors that are relevant for mine water generation. The essential components for mine water generation are sulphide minerals, water and oxygen. Other factors that influence the process are bacterial activity, temperature, pH and iron content (Lottermoser 2003).

Like mine water treatment, mine water prevention can also be executed under passive conditions. The PIRAMID project defines passive prevention as follows:

“Passive prevention of pollutant release is achieved by the surface or subsurface installation of physical barriers (requiring little or no long-term maintenance) which inhibit pollution-generating chemical reactions (for instance, by permanently altering redox and / or moisture dynamics), and / or directly prevent the migration of existing polluted waters.”

(PIRAMID Consortium 2003)

Possible techniques for mine water prevention

- dry covers,
- alkaline addition,
- water covers,
- alkaline injection,
- selective diversion of surface waters,
- inundation,
- underground mine sealing,
- coating/encapsulation,
- biocides,
- separation of sulphides.

This report we have a closer look at dry covers and wet covers and water diversion as they can provide a successful track record compared to others which still lack experiences from full-scale implantation.

The MEND Manual, Report 5.4.2d Volume 4 - Prevention and Control, March 2001 provides an excellent source of information on mine water reduction.

4.2.1 Covers

Covers seek to stabilise acid-forming mine waste through control of oxygen and water ingress/flux. In addition, they help to control dust formation and erosion and provide a growth medium for sustainable vegetation.

Because cover system design is highly site specific, it is necessary to assess the local conditions that are given at a site and have influence on the conception of the cover. Such factors are (O’Kane and Wels 2003):

- climate, in particular evaporation and precipitation
- topography
- type and volume of mine waste
- local availability of cover materials
- size and geometry of the waste storage facility.

The two cover types presented in the following are water covers and dry covers. These cover types belong to the most commonly applied systems due to their practicability and successful implementation.

The Canadian MEND project has extensively investigated multi-layer earth covers and water covers for tailings and waste rock. It provides a valuable source for more detailed information on the conception of covers.

A tabular overview on covers and functional cover elements can be found in Appendix A.
4.2.1.1 Water Covers

The submergence of wastes by flooding or relocation in suitable basins (e.g. open pit mines) creates an effective way to reduce mine water generation. By placing the mine wastes under water, oxygen ingress minimised and sulfide oxidation is hindered. This effect is due to the low oxygen solubility and diffusivity of water compared to air. For example, at 25°C, oxygen concentration in water is 8.6mg/L, which is approximately 25000 times lower than in the air (Government of Western Australia 2006).

For installing a water cover, an impoundment must be provided which allows sufficient water depth (> 1m) to be maintained above the waste material at all times (Younger, Banwart et al. 2002). Where water levels are lower, oxygen availability is not minimized because of mixing processes which cause the oxygen concentration gradient to remain constant rather than to decrease with depth. The consequence is a relatively high oxygen concentration at the sediment/water interface where reactions take place (Younger, Banwart et al. 2002).

When the water level is too low in an impoundment, under windy conditions waves can cause re-suspension and erosion of mine wastes. The elevated concentration of suspended materials may eventually cause problems complying with effluent requirements (Younger, Banwart et al. 2002),

![Oxygen concentration gradient in a water cover](Davé 2007)
Limitations to the application of water covers occur where topographic or hydrological conditions do not allow the construction of impoundments and flooding/inundation. Also, the submergence of tailings is not always possible for the fact that not all tailings dams were designed to be suitable for flooding.

Water covers are considered to be natural, economical and less costly than dry covers and generally the preferable cover option in humid regions where feasible. However, it must be regarded that such structures leave a large footprint in the landscape in the case of failure and there is a medium to high risk associated with the long-term maintenance of water retention dams and flow structures (Davé 2007).

The application of wet covers requires constant maintenance, in particular with regard to the water level. In case the water table falls so low that it exposes mine wastes to the atmosphere, intensive sulfide oxidation begins and subsequent contaminant release occurs when the water table rises again. Factors such as climate change and human interference with water regimes can make long-term predictions about the water availability for a particular site difficult in timescales relevant to these structures.

After water cover installation, the remaining oxygen in the material and already formed sulfide oxidation products will continue to result in contaminated mine water until they become depleted. Because of that, immediate effects of waters covers cannot be expected but in the long run, water quality will gradually improve (Davé 2007).

4.2.1.2 Dry Covers

The objectives of a dry cover system are to minimise the influx of water and to provide an oxygen barrier (Brown, Barley et al. 2002). Apart from reducing the amount of mine water, dry covers also form the basis for a sustainable vegetation cover; they reduce erosion and subsequent dust generation. Compared to water covers, dry cover are considered to be more costly (Davé 2007).

Dry covers can be constructed in several ways which differ from each other in their complexity. Most covers used to isolate wastes comprise a layer of compacted, locally available material (such as clay or till) or industrial materials (e.g. geosynthetics). This layer displays low hydraulic conductivity and serves as infiltration-limiting barrier. For the final top layer, non-compacted soil is the most suitable material as its purpose is to
support plant growth for cover stabilisation and addition of amenity value (Government of Western Australia 2006). The vegetated top layer also protects the low permeability layer from deterioration due to desiccation, frost action, erosion, animal burrowing and plant rooting (O’Kane and Wels 2003).

To enhance oxygen and water exclusion, further functional layers can be incorporated to increase the performance of a dry cover system. Among them are:

- reactive organic matter for oxygen consumption (Younger, Banwart et al. 2002)
- capillary breaks to reduce water ingress (Tremblay 1999)
- reaction-inhibiting materials (e.g. limestone) (Government of Western Australia 2006)

The primary effect of dry covers is reduction of water in the mine waste body by low permeability layers but, as mentioned above, decreasing the oxygen level is also desired. For this purpose, just like for water covers, the low oxygen solubility and diffusivity of water can be harnessed to achieve best results. In order to attain this, a layer of finely grained, porous material, designed to maintain high saturation levels, can be incorporated. Ideally, the diffusion coefficient of water applies to the layer and oxygen ingress into the sulphidic material is significantly reduced.

The oxygen diffusion coefficient for these layers is consequently reciprocal to the degree of saturation of the porous material. If high a saturation level ( > 90%) can be maintained in the cover through capillary barrier effects, then a layer of fine material (e.g. non-reactive tailings) between two sand layers will effectively reduce the oxygen flux to the reactive tailings materials by a factor of 1000 or more. Theoretically, the efficiency of a dry cover then becomes comparable to that of a water cover of the same thickness (Tremblay 1999).

Dry covers have a relatively small footprint compared to water covers and are less prone to catastrophic failure. On the other hand there are no long-term performance records available which prove the reliability of the system. Problems that may occur in the long run are penetration of the low-permeability layer with plant roots and burrow animals or longevity of applied materials (Davé 2007).

Climatic conditions are restricting the applicability of cover systems that require permanently high saturation as this is difficult maintain and usually not economically feasible in arid or semi-arid areas (O’Kane and Wels 2003). Reactive wastes will
usually require higher quality cover system to ensure cessation of AMD generating reactions.

Successful long-term cover performance requires detection and resolution of problems prior to significant environmental impacts. This involves monitoring, maintenance, repair and contingency plans. When considering covers for AMD management, best environmental practice requires site-specific adaptation of local resources and environment conditions (Government of Western Australia 2006).

4.3 Mine water treatment

In many cases where contaminated mine water occurs, subsequent water treatment cannot be avoided. This may be due to contaminant concentrations that are so high that MNA has to be excluded as a responsible solution or due to missing or insufficient source control measures.

As water treatment is not tackling the contamination source, but “only” preventing the distribution of the contaminants into the environment, it is considered as an “End-of-pipe” technology and therefore treatment applications are not a truly sustainable solution to the problem. On the other hand, it is often the only remaining way out at sites where mine water has significant adverse impacts on the surroundings.

In this context, it is important to acknowledge, that in the long run, the goal has to be that mine water abatement becomes an integral part of mine planning. Postponing the examination of the issue to the end of the mine life is likely to result in long-lasting mine water pollution that requires constant treatment. For the fact that this is costly and there are no revenues after mining activities have ceased, the implementation of the required treatment measures often turns out be very difficult to achieve. This may due to the absence of a legal owner of a site or a lack of regulatory frameworks addressing liability issues.

Numerous abandoned and orphaned mining sites sharing this problem can be found all over South Eastern Europe as well as in other regions. Mine water treatment in these cases is a vital to avoid and/or reduce hazards to human health, environment and political stability.
More information on integral mine planning and sustainable mining practices is provided in the ENVSEC publication “Mining for Closure” available from www.envsec.org.

4.3.1 Conventional mine water treatment

Conventional or active mine water treatment techniques will only be covered in an overview as the emphasis of this report is on passive treatment techniques. Nevertheless, active treatment is crucial and due to its long history, a vast amount of literature on active technology is available for those who are interested in more details.

The definition of active treatment is summarised by Younger (2002) as:

“Improvement of water quality by methods which require ongoing inputs of artificial energy and/or (bio)-chemical reagents”

There are several reasons why active treatment is still the most common system for AMD treatment and very likely to remain so. One is that active treatment techniques rely on conventional, well-recognized technology and is considered as “proven technology”. They have been implemented for decades all over the world and the experience gained over time has lead to reliable techniques. Compared to conventional treatment, passive systems have a background of less than twenty-five years to look on, which indicates that there is still a significant need for more experience and long-term performance data (Gazea, Adam et al. 1996).

Apart from the current state-of-the-art of the two treatment approaches, they also differ from each other in terms of application areas. The most striking advantage of active treatment plants is the high contaminant load they can handle and the reliability to comply with regulations for effluent qualities. This is possible because the variables can flexibly adjust to changing mine water quality and quantity. In passive systems, intervening with the system properties is hardly possible once the treatment system is constructed. A passive system is hence incapable of responding to changing treatment requirements. Excess flow and contaminant load will be discharged untreated into the receiving watercourse.

A further reason for the better applicability of active systems is the lower land demand. Compared to passive systems, they can handle higher contaminant
concentrations and flow rates whereby they only require a fraction of the amount of land.

The crucial drawback of an active system is that it is very expensive. The major parts of the costs are generated during the operational phase of the plant. Active treatment systems need constant energy and/or chemicals input, and monitoring and maintenance that has to be undertaken permanently by staff on the ground. Moreover, a relevant cost factor in an active treatment system is the disposal of the resulting metal laden sludge (Gazea, Adam et al. 1996) – the amounts of which can accumulate to very significant volumes over long periods of time. It is not uncommon for water treatment costs to exceed $200,000 per year at AMD sites using active treatment (Demchak 2005). The costs associated to the operation of an active mine water treatment plant are ongoing for the lifetime of the plant, or better, for the time mine water generation continues. And this, as presented before, can be very long-term.

Table 4.1 presents commonly used or proposed active mine water treatment options. Most applied conventional mine drainage treatment systems involve neutralization by addition of alkaline chemicals such as limestone, lime, sodium hydroxide, sodium carbonate or magnesia. These chemicals cause pH rise and subsequent precipitation of metals. Active systems on the basis of alkalinity addition generally require the installation of a plant with agitated reactors, precipitators, clarifiers and thickeners. Compared to other active treatment systems the ODAS system (Oxidation, Dosing with Alkali, accelerated Sedimentation) is a relatively uncomplicated and low tech operation scheme with high efficiency.

Currently, chemical precipitation is the most widely used technique metal removal from mine waters. Although it is an attractive process, there are also several disadvantages, such as: generation of large sludge volumes, the need for further treatment of sludge to meet the disposal criteria, and loss of the valuable metals (Jordanov, Maletii et al. 2006a).

The mine water issues that arises in SEE could technically be solved by the application of active mine water treatment plants. However, that which restricts their applicability are the high costs associated with their installation, and even more the high and ongoing costs for their operation. So far, mine water treatment plants have not yet found wide application in the region and with the economic situation only very gradually improving; this is likely to remain so the case. This is where passive, cost-
effective and robust techniques could fill a gap and are therefore discussed in this report. Because of the emphasis on these techniques in this work, passive treatment will be presented separately in the following chapter although systematically they would belong to this section.

<table>
<thead>
<tr>
<th>Name</th>
<th>Overall aims</th>
<th>Brief description</th>
<th>Advantages</th>
<th>Limitations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oxidation, Dosing with Alkali, accelerated</td>
<td>pH rise and precipitation of</td>
<td>Oxidation by active air introduction is followed by addition of an alkalinity</td>
<td>Technically no very demanding, proven technique</td>
<td>Salinity can not be treated, high sludge production, no application for</td>
</tr>
<tr>
<td>Sedimentation (ODAS)</td>
<td>metal hydroxides</td>
<td>generating agent such as lime or caustic soda. Resulting sluge is removed by</td>
<td>with long track record</td>
<td>products</td>
</tr>
<tr>
<td></td>
<td></td>
<td>sedimentation accelerating processes like thickener, addition of flocculants or</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>recirculation (HDS-p)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ionexchange and Sorption</td>
<td>Metal attenuation and recovery</td>
<td>Metal ions sorb to an exchange media such as resins or zeolithes and after</td>
<td>High efficiency and metal recovery possible</td>
<td>High costs for sorbent purchase and recovery</td>
</tr>
<tr>
<td></td>
<td></td>
<td>depletion of the sorbent the metals can be obtained in a purified and</td>
<td></td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>concentrated form</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sulphatereduction and Biodesalination</td>
<td>Metal attenuation and sulphate removal</td>
<td>Co-treatment of mine water and … waste water (contains short-chained carbon</td>
<td>Sulphate removal is achieved and co-</td>
<td>Sulphide sludges are difficult to handle, some techniques are not</td>
</tr>
<tr>
<td></td>
<td></td>
<td>sources) under anaerobic conditions favours the activity of Sulphate Reducing</td>
<td>treatment of second waste stream, low sludge</td>
<td>yet proven</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Bacteria (SRB). Thereby effective metal and sulphate removal is achieved</td>
<td>volume</td>
<td></td>
</tr>
<tr>
<td>Membrane processes</td>
<td>Metal attenuation</td>
<td>Pressure driven filtration of mine water through a membrane. Techniques</td>
<td>High water purity can be achieved</td>
<td>Expensive to operate due to high energy demand and high costs for</td>
</tr>
<tr>
<td></td>
<td></td>
<td>covered are inter alia Reverse Osmosis, Ultrafiltration and Microfiltration</td>
<td></td>
<td>membranes</td>
</tr>
</tbody>
</table>

Table 4.1  Brief typology of active treatment units.
5 Passive Treatment of Contaminated Mine Water

This chapter will discuss passive mine water treatment from several aspects. First, the term will be defined and general characteristics will be presented, in particular compared to characteristics of active treatment. Limits to the applicability are shown but also opportunities where innovative techniques can outpace conventional treatment. The discussion will be followed by the presentation of basic principles underlying passive treatment techniques and the delineation of the actual techniques as they can be applied on a mining site.

5.1 What is passive treatment?

Over the last twenty-five years, the possibility of harnessing natural ameliorative processes for mine water treatment has been investigated and the practice developed out of these investigations has been termed passive treatment (Younger, Banwart et al. 2002). These techniques are mainly applied in North America and more recently also in Western Europe.

One output of research and dissemination of these techniques in Europe has been the PIRAMID project (PIRAMID Consortium 2003), this defines passive treatment solutions as:

'\textit{the improvement of water quality using only naturally-available energy sources in gravity-flow treatment systems which are designed to require only infrequent, albeit regular, maintenance to operate successfully over their design lives.}'

The principle of passive treatment schemes lies in the use of naturally occurring processes in order to improve the quality of the influent waters with minimal operation and maintenance requirements. These processes are of chemical, biological and physical nature:
- Chemical removal processes comprise oxidation, reduction, coagulation, adsorption, absorption, hydrolysis, precipitation.
- Physical processes harnessed within passive treatment are gravity, aeration, dilution.
- Biological removal processes refer to biosorption, biomineralization, bioreduction, alkalinity generation.

(after MEND, 1996)

Basically, passive treatment can be described as an example for “enhanced natural attenuation” (Younger, Banwart et al. 2002) where naturally occurring processes are accelerated by manipulating environmental conditions in a treatment system. The aim is to provide such conditions where the highest removal rate for a particular contaminant can be achieved (MEND 2000).

Truly passive systems hence function without any regular input of cost-intensive resources, such as manpower, energy and chemicals. In reality, a completely passive system is hard to achieve as many sites often require active components such as pumping or aeration. As mentioned by a leading UK practitioner and researcher during a recent workshop presentation on UK passive treatment experiences (Jarvis 2007), out of 50 installed treatment systems described as passive, only three meet the definition of a truly passive system. However, even when systems are not entirely passive according to the definition, overall operational life cost profile is still reduced compared to adequate, fully active systems.

Another characteristic of passive applications that distinguishes them from active treatment is their economic structure (PIRAMID Consortium 2003). With active treatment, the costs are distributed over time, i.e. operating costs are high and exceed by far the cost of designing, building and commissioning of a plant. These operating costs are caused by needs such as constant energy and/or chemicals input, staff and high maintenance costs.

With passive systems, on the other side, the main financial input is required at the construction period of the treatment scheme. Indications from experts (PIRAMID Consortium 2003) are that the upfront installation costs for a passive system are, depending on the size of the application, similar or at times marginally higher than an active system. As the nature of passive systems is to be self sustaining, at least to a certain degree, the cost following successful commissioning of the plant will be low
compared to an active solution. To put these facts into figures, Cohen (2006) states that compared to conventional treatment, passive systems are about half of the capital costs and less than 1/20 of the maintenance costs. Other calculations are less positive and indicate that the cost advantage is only small in the early years of operation but starts to develop afterwards (Kunze, Kiessig et al. 2007).

5.1.1 *What are the advantages of passive systems?*

The reasons why the implementation of a passive treatment system should be considered can be summarised as follows:

- Overall treatment costs are much less than compared to an active system with comparable treatment efficiency. With small to medium sized passive operations not only the operating costs are lower but also the implementation cost.

- Passive systems are very robust and hard to destroy physically, whereas active systems are prone to be subject to vandalism or even equipment/metal theft or “salvaging” (particularly relevant in economies where people are poor).

- Maintenance of a passive system is easy and requires no professional personnel, which creates possibilities for community involvement at the actual site.

- The economic structure of passive systems facilitates external funding. Donors can easier be motivated to support a project with relatively finite costs (system implementation) rather than to support a project where financial input has to be distributed over 20 years or more (system operation).

- Many passive systems, (e.g. in particular wetland type systems), can add significant amenity value to an area and contribute to the tourist or community usage value of an area as well as to the ecological/biodiversity value by providing a wildlife habitat.

- When the “infrequent but regular” maintenance of passive is system is kept up, long system lifetimes can be attained and the requirement for long-term mine
water treatment can be met.

- Passive treatment operations usually do not involve any hazardous materials.

5.1.2 *What are the pitfalls of passive systems?*

Application is restricted to low to medium flow rates and low to moderate contaminant concentrations as passive systems generally require large land areas for operation. In most cases, land scarcity is the factor restricting the treatment capacity of a passive system.

- Passive mine water treatment is still a young technology and therefore there is a lack in detailed knowledge on the relevant processes taking place in the system (e.g. sulphate reduction) and experience in long term application (e.g. sludge handling).

- Precise adjustment to change of influent quality and flow rates is not possible in a passive system. Additionally, performance of passive systems is subject to seasonal and other variations.

- Restrictions in removal ability of certain elements (e.g. manganese) or suitability (i.e. highly toxic elements).

(Johnson and Hallberg 2005)

5.1.3 *When are passive systems a viable option for mine water treatment?*

The question whether an active or a passive approach is more suitable and which technique will yield the best results for a particular site can only be answered after a detailed and scientific examination of the conditions given at the site.

There are numerous aspects that have to be seriously taken into consideration. Further, it must be recognised each site will be unique for its combination of characteristics. For the design of a treatment scheme, this means there are no “off-the-shelf” technologies available but each site has to be addressed individually and the solutions are always site specific.
Among the criteria which have significant impact on the conception of a treatment system are (summarised from PIRAMID Consortium (2003)):

- flow rate of the mine water
- hydrochemistry of the mine water
- topography of the site
- geology of the site
- hydrology of the site
- land availability
- “Contaminated Land” issues

A key parameter for the choice of a treatment approach is the contamination load that has to be removed from the water (Younger, Banwart et al 2003). The necessary treatment intensity varies with it and therewith the treatment techniques that can handle particular contamination levels. Although it is crucial to bear in mind that there are a variety of factors that play a role in treatment design, Figures 5.1 and 5.2 give an orientation where the line between active and passive treatment, in terms of flow rate and contaminant concentration, can be drawn. The diagram axes are deliberately not mentioning any concrete figures as the actual treatment capacity eventually depends on the local conditions (Willscher 2003).

![Figure 5.1](image1.png)

**Figure 5.1** Suitability of mine water treatment approaches (ERMITE 2003).

Temporal implications of mine water pollution are on the one hand its longevity but also the change of mine water quality over time as has been indicated in Section 3.3.4. When planning remedial works, it will frequently be necessary to use active treatment techniques in the beginning of mine water generation to cope with the high contaminant
loads. However, after a period of time, the application of passive treatment may become feasible after the “first flush” period when the contamination load has decreased to a level which allows passive treatment application. Figure 5.2 illustrates the principle of successive treatment application (Willscher 2003).

5.2 Passive Treatment Techniques

This section will explain the basic principles of the most commonly applied passive treatment techniques with the aim to promote a deeper understanding for the requirements and applications of these techniques. This text has also been developed with non-technical readers in mind in recognition of the many stakeholders affected by mine water pollution that may have interest in this field.

5.2.1 Constructed Wetlands

The possibility of using constructed wetlands to treat AMD was first indicated by observations made on the treatment of mine drainage by naturally-existing wetlands. The flow of AMD through Sphagnum moss bogs illustrated that iron and acidity concentrations could be reduced (Gazea, Adam et al. 1996).

When the first wetlands were constructed with the purpose to treat mine water it was observed that Spagnum moss is not the most suitable vegetation for a treatment wetland due to its sensitivity to water level fluctuations and low metal retention capacity (U.S. EPA 2000). Instead of using Sphagnum, it was found that some macrophytes,
notably *Typha* and *Phragmites spp.*, displayed far better adaptation to the relatively harsh conditions that exist in these environments (Johnson and Hallberg 2002.). Today, constructed wetlands are the most commonly applied passive treatment system (Younger, Banwart et al. 2002). This can be ascribed to the excellent track record for aerobic wetlands, eliminating iron in alkaline mine waters and the attractiveness with regard to their habitat/biodiversity value.

Large wetlands can also cope with the fluctuation of water levels as it may occur during periods of low precipitation periods thanks to flexible storage volumes (Younger, Banwart et al. 2002).

Under certain circumstances the application of constructed wetlands for treatment purposes may not be the adequate solution even if good removal rates can be achieved. This for example the case when highly toxic metals are present in the mine water and retained by the wetland. These metals remain accessible to organisms because of the open structure of a wetland. For waters containing considerable amounts of such high risk elements (e.g. cadmium or mercury) one has to make sure that they are permanently removed from the cycle.

The basic types of constructed wetlands are aerobic wetlands, compost wetlands and RAPS-systems which differ significantly from each other with regard to form, function and applicability.

Usually, a constructed wetland will always incorporate both aerobic and anaerobic environments but with varying design and substrate input either aerobic or anaerobic conditions are promoted in order to achieve desired reactions (Younger, Banwart et al. 2002).

Sedimentation ponds and aeration units are presented in this section as they are mostly applied in combination with constructed wetlands and removal processes occurring in these installations are similar to those responsible for contaminant removal in wetlands.

**5.2.1.1 Sedimentation ponds and aeration units**

Where mine water is alkaline, aeration may result in extensive metal precipitation so that simple ponds can be sufficient for achieving acceptable water qualities. The
requirements for treating contaminated mine waters with aeration and sedimentation as the only treatment are according to Younger (2002).

- pH is above 6.5 and net-alkaline,
- iron is the only contaminant of interest,
- for each 50 mg/l of iron in the water one aeration cascade followed by a sedimentation pond is provided.

In general, aeration and sedimentation units form part of treatment systems which seek to remove contaminants via precipitation of metal hydroxides. As explained above, oxidation of iron and manganese enhances their hydrolysis and although oxygen is introduced in a wetland by plant roots and the water/atmosphere interface, pre-aeration of the inflowing mine water significantly facilitates the removal processes (MEND 2000).

Sedimentation ponds between the aeration unit and a wetland allow metal precipitates and other suspended materials to settle. So instead of entering the wetland, they are removed in the sedimentation pond. This is important as the sedimentation pond is far easier to clean once it requires excavation than a wetland with all its vegetation and organic material mixed with the precipitates. As such, the wetland only receives those metals and resulting precipitates that could not be removed by the aeration/sedimentation unit (Younger, Banwart et al. 2002).
For passive water oxygenation, (i.e. the process of introducing oxygen into water) aeration cascades are commonly applied, as shown in Figure 5.4. Usually, one cascade consists of a series of 4 to 6 wide steps over which water can spread to a thin film so oxygen contact is maximised. Further developments of aeration cascades incorporate so called “plunge pools” after each step so bubbles can fully diseminate in the water, see Figure 5.6 (PIRAMID Consortium 2003).
Theoretically, one cascade followed by a sedimentation pond can oxidise and consequently remove 50 mg/L of iron from mine water, but to ensure system reliability it is recommended to calculate with 30 mg/L (PIRAMID Consortium 2003). To execute passive aeration via cascades, sufficient topographic gradient must be provided at the treatment site to drive the water through the cascade. Another option to introduce oxygen to mine waters is the application of an in-line venturi aeration device. The functional principle can be seen in Figure 5.5. Although venturi aerators are very attractive because of their high oxidation rate for ferric iron (~ 900 mg/l per aeration step), their applicability is quite restricted as they require a minimum operating head of about 15 m (PIRAMID Consortium 2003).

For some strongly alkaline mine waters, aeration often results in CO₂ degassing upon which mine pH level rises and thus increases the rate of metal precipitation (Brown, Barley et al. 2002).

The design of sedimentation ponds is based on the settlement velocity of ferric hydroxide flocs which is relatively low due to small particle size and low density (Younger, Banwart et al. 2002). When retention times of > 24 h can be achieved,
settlement alone has proven to be able to reduce iron concentration below 1 mg/l in alkaline mine waters (Younger, Banwart et al. 2002). The difficulty or drawback is that such high residence times require large areas for the sedimentation ponds. As has been mentioned – quite commonly land availability is the limiting factor in passive treatment application

5.2.1.2 Aerobic wetlands

An aerobic wetland seeks to favor, as its name already indicates, oxidizing conditions. Its goal is to remove contaminants via oxidation, hydrolysis, precipitation and subsequent sedimentation. Their application is the one of the most common among passive treatment techniques, partly for the long experience of both natural and man-made aerobic wetlands in mine water treatment. Moreover, they are also relatively simple to design and to build. Figure 5.6 shows an aerobic wetland with reed vegetation designed for alkaline mine drainage treatment at a former coal mining site in the UK (Nairn and Mercer 2000).

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Figure 5.6 Aerobic wetland in Whittle, UK.

Design scheme
Aerobic wetlands are shallow ponds, vegetated with emergent plant species and water depths around 0.15 m to 0.25 m. See Figure 5.7 for the conceptual design scheme. The area of the wetland depends on the contaminant load entering the wetland and is calculated through area-adjusted removal rates. Iron removal rate in an aerobic wetland is assumed between 10 to 20 g/m²/d (Younger, Banwart et al. 2002; PIRAMID Consortium 2003).

As has been indicated, in order to achieve optimal removal rates within an aerobic wetland, it is recommendable to install aeration cascades and settlement lagoons prior to the wetland. The introduction of oxygen via aeration cascade ensures oxidation of ferrous iron to ferric iron in order to facilitate precipitation (Brown, Barley et al. 2002).

It is recommended to design a wetland in a way that prevents preferential flows and short-cutting of the mine water. This can be achieved by adding barriers or little islands of sufficient height in the basin so the flow is diverted over the width of the wetland (PIRAMID Consortium 2003). Plants help to baffle the surface flow with their stalks and significantly influence the velocity profile of the flow (Sheoran 2006). Apart from their physical properties plants are mainly responsible for oxygen influx and accumulation of organic matter in the wetland (Batty and Younger 2004).

Typical wetland vegetation includes Typha latifolia (Cattail), Phragmites australis (Common reed), Juncus effusus (Soft rush), and Iris pseudacorus (Yellow flag iris). Organic matter is the habitat for alkalinity producing bacteria and can help the aerobic wetland to cope with elevated acidity levels. The processes responsible for this will be described in the following section on compost wetlands. Sorption processes are also enhanced by elevated levels of organic matter in the wetland (Batty and Younger 2002).
The aimed processes in an aerobic wetland are oxidation and hydrolysis of metals. After precipitation, metals are removed from the water by several processes among them are sedimentation of suspended flocs, filtration of flocs by stems of plants, adsorption of aqueous metal species, precipitation of hydroxides on plant stems and the wetland sediment surface, and direct plant uptake of iron and other metals (which are retained primarily in the plant roots). Sulphate removal and pH rise are not foreseen to take place in an aerobic wetland but do still occur to some extent as anaerobic zones exist and organic matter is provided by plant die-back and other sources (Cohen 1996).

**Applications and limitations**

Aerobic wetlands are applied for net-alkaline mine waters with moderate contaminant load and where land availability is not a problem. The application of aerobic wetland is limited when the mine water is net-acidic as hydrolysis of metal ions releases hydrogen ions into solution (Sheoran and Sheoran 2006).

When the mine water is net-acidic, its buffering capacity is already depleted and proton release will decrease the pH value. Consequently, metal removal rates are lowered or even remobilisation of already precipitated metal may occur. Because aerobic wetlands have only very little capacity to generate alkalinity in the few anaerobic zones, net-acidic water would be discharged with low pH which is undesirable for the environment. For this reason, the alkalinity - acidity balance has to be carefully calculated when an aerobic wetland is considered for mine water treatment in order to prevent disappointing system failures as they have occurred before (Hedin 2006).

The range of contaminants that can be efficiently removed in an aerobic wetland are quite restricted because the lack of capacity to raise pH. For example, abiotic manganese oxidation does not set in until pH values of around 8 are achieved – and further, it requires the complete removal of iron as it interferes with the manganese removal process. As such, the main application of this technique is consequently the removal of iron and aluminium which readily form precipitates at relatively low pH values.

Aerobic wetlands can be used as stand-alone treatment system for net-alkaline mine waters with moderate contaminant concentrations but also they are often applied as
“polishing step” after other passive or active treatment steps which have taken care of those contaminants an aerobic wetland cannot cope with.

5.2.1.3 Compost wetlands

In contrast to aerobic wetlands, compost based wetlands, Figure 5.8, try to avoid oxidising conditions in the waterbody in order to support the activity of Sulphate Reducing Bacteria (SRB) that require anoxic conditions to thrive. Therefore, compost wetlands incorporate an organic substrate layer (“compost”) which serves as habitat and organic carbon source for the SRB. The major advantage of these systems is their ability to raise the pH they work for net-acidic mine waters which are generally more problematic than net-alkaline mine waters.

![Compost wetland at Quaking Houses, UK.](image)

**Design scheme**

Water that is treated with a compost wetland should enter the wetland as soon as possible after emerging from the underground, so a low oxygen level can be maintained. Oxidising conditions are to be avoided as the SRB are obligate anaerobes. Hence, compost wetlands do not require any preliminary treatment steps such as aeration cascades or settlement ponds. An exception to this rule may be when the water has a high inert solid content so the wetland capacity is not reduced by sedimentation of these solids. The conceptual design of a compost wetland can understood from Figure 5.9.
The PIRAMID project (2003) recommends a thickness for the compost layer of ~0.5 m with a water level on top of the compost of 0.1 m – 0.2 m. Material that is used as compost has to support several conditions to ensure successful treatment:

- contains sulphate reducing bacteria and promotes growth of associated microbial flora,
- contains short-term and long-term available carbon sources for SRB activity,
- displays adequate hydraulic conductivity,
- is not hazardous.

Many substrates have been used for these purposes and the choice depends most of the time if they are readily available on the site. Among them are:

- spent mushroom compost (Chang, Shin et al. 2000),
- paper recycling sluge (Chang, Shin et al. 2000),
- cow/horse manure with straw (Younger, Banwart et al. 2002),
- wood chips (Chang, Shin et al. 2000),
- composted municipal waste (Younger, Banwart et al. 2002),
- softwood sawdust and hay (Johnson and Hallberg 2005).

The predominant removal processes in a compost wetland are alkalinity generation and precipitation of metal sulfides by SRB activity. These bacteria use short-chained hydrocarbons like alcohols and organic acid and transform them to hydrogen sulfide gas (H₂S) and bicarbonate (HCO₃⁻), Reaction 5.1. The produced hydrogen sulfide gas reacts with metal cations to solid metal sulfides, Reaction 5.2. Bicarbonate alkalinity
eventually leads pH rise via proton consumption, Reaction 5.3. These processes can be seen as the reverse of pyrite oxidation – the very cause of much AMD.

\[
\begin{align*}
2 \text{CH}_2\text{O} + \text{SO}_4^{2-} & \rightarrow \text{H}_2\text{S} + 2 \text{HCO}_3^- \quad (5.1) \\
\text{M}^{2+} + \text{H}_2\text{S} + 2 \text{HCO}_3^- & \rightarrow \text{MS} + 2 \text{H}_2\text{O} + 2 \text{CO}_2 \quad (5.2) \\
2 \text{HCO}_3^- + 2 \text{H}^+ & \rightarrow 2 \text{H}_2\text{O} + 2 \text{CO}_2 \quad (5.3)
\end{align*}
\]

Sulphate reducing bacteria tolerate high heavy metal concentrations and show their best performance at a pH > 4 (Gazea, Adam et al. 1996).

Metals that cannot form sulfides under natural conditions, like aluminium (Al\(^{3+}\)) and ferric iron (Fe\(^{3+}\)), leave solution as hydroxides upon pH rise. Due to the high content of organic material in a compost wetland, sorption processes are also contributing to contaminant removal, especially shortly after commissioning the treatment system ("honeymoon period").

Alkalinity generating processes and metal sulphide precipitation take place in anaerobic areas, i.e. in the organic substrate. Compost wetlands are commonly designed as surface flow systems. However, the alkalinity generating processes are based in the compost layer so that exchange of relevant agents takes place via diffusion and is therefore relatively slow. As the system is open to the atmosphere, ingress of oxygen cannot be avoided but is also not necessary. It has been shown that anoxic conditions are can be continuously maintained even when plant roots penetrate the organic layer. Analysis of spent compost material showed that already a few millimetres distance from the root anoxic conditions were prevailing again. This also indicates it is not necessary to preclude plant growth in SRB based systems to maintain anoxic conditions. It is rather that vegetation becomes an organic carbon source once the plant decay begins and may be very useful when the organic carbon initially introduced to the system becomes depleted. For amenity reasons and habitat value, vegetation is always preferable (Jarvis, personal communication, 23.01.2007).

After water leaves the compost wetland, an aerobic treatment step may follow such as a settlement pond or an aerobic wetland to remove remaining metals by oxidation and hydrolysis. At this point, proton release from hydrolysis is not a problem anymore because of the alkalinity added in the compost wetland.
**Applications and limitations**

When compared to aerobic wetlands, the most striking advantage of compost wetlands is their ability to treat acidic mine waters and higher contamination levels. Apart from metals and acidity, a compost wetland is also eliminates sulphate contamination, although sulphate levels usually decrease less than 20% in a SRB based wetland (Jarvis 2000).

Because of the organic material to which contaminants greatly sorb until sorption capacity becomes depleted, the treatment performance of compost based systems is very impressive in the first weeks after commissioning. This effect has been termed “honeymoon period” by Younger et al. (2002) and one should be aware that the results obtained during this period are very likely to be unique throughout the life-time of the system.

Metal sulphide sludge formed under reducing conditions is less bulky than sludge resulting from metal hydroxide precipitation. This increases the longevity of a treatment system and reduces maintenance works. It should be remembered that, although the techniques presented in this thesis are termed passive, maintenance still has to take place on a “infrequent but regular” basis. This includes sludge removal and maintenance of pipe works.

On the other hand, metal sulphides are likely to remobilise metals when they are not kept under anoxic conditions, e.g. when sludge needs to be dug out and disposed of. That is also why it is important to permanently maintain sufficient water in the wetland to preclude exposure of metal sulphides to atmospheric oxygen.

However, researchers and practitioners have not yet gathered enough experience with disposal and remobilisation behaviour of metal sulphide sludge generated in compost wetlands in order to make precise statements on these issues. Results from the excavation of the compost wetland in Quaking House, UK, indicate that remobilisation of metal is not necessarily the case when metal sulphide sludge is exposed to the atmosphere (Jarvis, personal communication, 23.01.2007).

Regarding land demand, compost wetlands do not display advantages compared to aerobic wetlands. Anaerobic reactions are slow and consequently retention times for compost wetlands are relatively high. Problems that may occur are safety risks (e.g. children may fall into a compost wetland) and unpleasant odour because of hydrogen sulphide gas bubbling out of the compost wetland (PIRAMID Consortium 2003).
5.2.1.4 Reducing and Alkalinity Producing System (RAPS)

RAPS are a more recent development compared to previously presented wetland designs and were developed as response to experiences gained with the application of anoxic limestone drains (ALD) (see Section 5.3.1). ALDs require incoming mine water with low dissolved oxygen content (< 1 mg/l) and low levels of other elements that readily precipitate upon pH rise such as ferric iron and aluminium (< 2mg/l). For a RAPS however, these requirements are much loosened, so a wider range of mine waters can be treated (Watzlaf, Schröder et al. 2000).

The aim of a RAPS is to significantly increase the pH of the mine water and subsequently precipitate and remove metal hydroxides in an aerobic treatment step. Initially Kepler and McCleary (1994) named this system SAPS (Successive Alkalinity Producing System) but it was then renamed to RAPS by Watzlaf, Schröder et al. (2000) so both names can be found in respective publications. Particularly in North American sources, RAPS are also often referred to as vertical flow systems/vertical flow wetlands (VFS/VFW). In Figure 5.10 the one lagoon of a two staged RAPS in the UK is pictured.

![RAPS lagoon in Bowden Close, UK.](image)

**Design scheme**
In a RAPS, pH rise is achieved by calcite dissolution from limestone gravel and bicarbonate generation by SRB. Normally, armouring\(^3\) of the limestone and consequent system failure occurs when water with elevated oxygen concentrations and/or ferric iron/aluminium concentration come into contact with the limestone. In order to prevent limestone armouring, a 0.15 m to 0.6 m layer of organic substrate is placed on top of a 0.6 to 1.2 m limestone gravel layer (Watzlaf, Schröder et al. 2000). The water is forced to penetrate this organic layer whereby a vertical flow through the wetland system develops and greatly increases the interaction of water with organic matter and limestone (Watzlaf, Schröder et al. 2000).

Because of oxygen consuming microbial activity, anoxic conditions are prevailing in the organic substrate and ferric iron entering this environment is reduced to ferrous iron. Just as in compost wetlands, SRB are active in the organic material and create bicarbonate alkalinity. Usually, before the water gets in contact with the limestone, a pH above 4.5 is created by SRB which forces aluminium to precipitate as Al(OH)\(_3\). Thereby, it can be retained within the organic layer. When the water reaches the limestone, most of the problematic compounds have been removed and calcite dissolution can take place without undesired precipitation of hydroxides. Downstream, the RAPS is commonly followed by an aerobic pond where precipitation and sedimentation processes can take place before discharging the water to the receiving watercourse (MEND 2000).

Vegetation is not required for a RAPS to operate successfully but as mentioned with compost wetlands, amenity and habitat values of the site increase when vegetation present. By adjusting the water level above the compost layer, plant growth can be prevented (>1 m) or permitted (~0.2 m). Material for the organic layer in a RAPS needs to fulfill the same requirements as the substrate used for compost wetlands. Hence the materials listed in the preceding section can also be applied for a RAPS (Younger, Banwart et al. 2002).

There are several possible configurations for a RAPS. In Figure 5.11, the initial design recommendation of a RAPS is presented with layered limestone and compost compartments. A drawback of this configuration is the restricted hydraulic conductivity

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\(^3\) «Armouring» of the limestone occurs when iron and aluminium precipitate in the proximity to limestone surface due to the highly alkaline conditions given there. The resulting metal hydroxides coat the limestone surface whereby limestone dissolution and consequently alkalinity generation is hindered.
for the system due to low permeability of the compost layer, whereas the limestone layer displays high permeability.

When limestone and compost are mixed together instead of being layered, shown in Figure 5.12, the overall permeability rises significantly (Fabian, Jarvis et al. 2006). This reduces the required retention times and less land is needed for the treatment system.

The key processes taking place in RAPS are:

- pH rise via limestone dissolution and SRB activity (bicarbonate release from sulphate reduction);
- reduction of ferrous iron and oxygen removal in the organic layer to prevent limestone armouring;
- hydrolysis of aluminium and other compounds in the organic layer.
Due to the presence of SRB, sulphate removal can be expected, but as with compost wetlands, removal rates are low to moderate.

**Applications and limitations**

As for compost wetlands, RAPS are applied to treat net-acidic mine waters. Compared with compost wetlands, RAPS can treat higher concentration of metal contaminants and require less land for their application. A key prerequisite for RAPS application is the provision of sufficient topographic relief on a site to make sure the water is forced through the system. The PIRAMID Guidelines suggest at least 2.5 m site relief after the point of water emergence and about 1 m of freeboard above the compost should be provided at a treatment site to guarantee sufficient driving head. Practically, this is the only significant drawback of RAPS when compared to compost wetlands. RAPS require about 15 - 20% of land area a compost wetlands would cover. With regard to land scarcity being a major constraint to passive treatment application and the treatment efficiency of RAPS, this technique is becoming more and more the preferred solution where applicable (Younger, personal communication, 23.01.2007).

The major part of the sludge obtained from a RAPS consists of hydroxides and accumulates mainly in the aerobic pond following the RAPS but also the system itself. As for compost wetlands, the water level should not fall below the level of the organic material because it would allow oxygen ingress to the limestone gravel and consequently lead to system failure. This constitutes a problem where the incoming flow strongly varies, for example in arid and semi-arid areas.

5.2.2 *Chemical Passive Treatment*

Chemical passive treatment seeks to neutralize acidity, raise pH and precipitate metals as hydroxides by dissolving alkalinity generating minerals such as calcite or dolomite (Lottermoser 2003). This is achieved by allowing acidic mine water to flow through a bed of calcite or dolomite rich limestone. Calcitic limestone is commonly the preferred mineral as it is more reactive and hence more effective (Lottermoser 2003). After leaving the limestone bed, the mine water flows into an aerobic treatment unit so oxidation and precipitation can take place. Compared to treatment systems which rely on biological activity for treatment, non-biological treatment options are less subject to
seasonal changes and other uncertainties associated to biological processes (Champagne, Geel et al. 2005).

Limestone is a cost-effective alkaline material that is often easy to source locally, but it also has major drawbacks when used for mine water treatment. When ferric iron and aluminium are present at elevated concentrations they cause the limestone system to fail due to so-called limestone “armouring”.

Ferric iron precipitates readily when pH is raised above a certain level which is quickly exceeded in the proximity of limestone. The resulting iron hydroxides form an ochre coating on the limestone surface whereby further calcite dissolution is hindered. Another prerequisite for mine waters treated with limestone is therefore low dissolved oxygen content so iron remains as ferrous iron (Fe$^{2+}$). Aluminium also forms hydroxide flocs that clog the limestone pores and prevent the carbonates from dissolution. Under natural conditions aluminium can only be found as Al$^{3+}$ so aluminium levels should always be low for these treatment schemes (Johnson and Hallberg 2002.).

Eventually, the limestone will dissolve and needs to be renewed. Although this way of alkalinity generation is much cheaper and less demanding than active lime dosing, techniques based on limestone dissolution are not truly self-sustaining (Barley, Hutton et al. 2005)

According to Watzlaf, Schröder et al (2000) are factors that affect the level of alkalinity generation in chemical treatment systems:

- contact time,
- partial pressure of CO$_2$
- calcite content of the limestone,
- initial pH,
- particle size of the crushed limestone

5.2.2.1 Anoxic Limestone Drains (ALD)

ALDs have been developed in the early nineties by the Tennessee Valley Authority and have found wide acceptance throughout the US where water conditions allow ALD treatment (Costello 2003; Ziemkiewicz, Skousen et al. 2003; Skousen and Ziemkiewicz 2005 ). ALDs overcome the problem of the limestone armouring by exclusion of oxygen so that oxidation of ferrous iron within the limestone aggregate is minimized
and alkalinity generation can continue. ALDs have strict requirements for influent water quality which must be taken into account for successful ALD application.

**Design scheme**

Basically, an ALD is a trench backfilled with limestone aggregate which intercepts mine water very soon after its point of emergence so anoxic conditions are maintained until the flow enters the limestone bed, see Figure 5.13. The conceptual design of an ALD is provided in Figure 5.14.

![ALD under construction in Eastern USA](from Skousen (2005))

Dimensions are typically in the order of 1 m deep, 1-7 m wide, and 25-100 m long. (Watzlaf, Schröder et al. 2000; Lottermoser 2003). The trench is covered with an impermeable layer, commonly clay or synthetic materials which prevent oxygen ingress and to create “closed-system” conditions. When such conditions are given, bicarbonate alkalinity and consequently pH increase because of the higher CO₂ partial pressure in the ALD than in the atmosphere. This process is explained in more detail by Lottermoser (2003).
Field tests show that relatively high rates of limestone dissolution occur within the initial 14 hours of contact with mine water (Hedin and Watzlaf 1994). After this contact period, the rate of limestone dissolution is much slower so longer retention times are not beneficial (Watzlaf, Schröder et al. 2000).

An ALD is not a stand alone treatment system and has to be followed by an aerobic treatment unit, such as a sedimentation pond or an aerobic wetland for final metal removal. Once water leaves the ALD, oxidation of the particular metal ions occurs in the subsequent aerobic wetland or sedimentation pond. Like in other aerobic systems, hydrolysis and precipitation are the processes for improving the water quality.

**Applications and limitations**

As the aim of ALD is to achieve pH rise, net-acidic mine waters are the target for ALD treatment. Commonly, ALDs are incorporated as first treatment stage into a passive system so the neutral conditions created by the ALD to enable further treatment, e.g. in an aerobic wetland. In some cases, it may be useful to have an ALD as final treatment step, especially where initial water conditions restrict ALD application. By the time mine water leaves other treatment steps, its quality may have reached the conditions suitable for treatment in an ALD. In this case an ALD serves as final pH adjustment and polishing unit before discharge. Compared to a RAPS, the installation of an ALD is more flexible because it requires less topographic gradient to function and less material input.

Limitations to incoming mine water are dissolved oxygen concentration (< 1 mg/L) and low levels of ferric iron and aluminium (< 2mg/L) (PIRAMID Consortium 2003). High overall iron levels increase the probability of ferric iron being present. In any case, the actual site specific distribution between ferrous and ferric iron has to be determined.
A significant indicator if a particular mine water is suitable for ALD treatment is the pH. The lower the pH of a mine water, the more ferric iron is likely to be present as iron precipitation before entering the ALD is precluded. In mine water with pH below 4.5, the ferric iron and aluminium content can be expected above 2 mg/L so successful application of ALDs for such mine waters becomes very unlikely (Younger, personal communication, 23.01.2007)

5.2.2.2 Oxic Limestone Drains and Open Limestone Channels

In practice, often it showed that AMD does not meet the requirements to be treated optimally in an ALD. Regardless this circumstance, it may still be necessary to create alkalinity for achieving treatment and where installation of other measures for pH adjustment are not feasible, Oxic Limestone Drains (OLD) and Open Limestone Channels (OLC) offer alternative treatment options.

**Design scheme**

OLDs receiving oxic mine water are physically identical to ALDs, so Figures 5.15 and 5.16 are valid for describing their conception. In contrast to ALDs, precipitation of hydroxides within the system is expected. In order to reduce the risk of system failure, high interstitial flow velocities (above 0.1 m/s) have to be maintained to keep hydroxide flocs in suspension and to clean precipitates from limestone surfaces (Younger, Banwart et al. 2002). Although water entering an OLD has already elevated oxygen concentrations, it is designed to avoid the ingress of atmospheric oxygen. The aim is to avoid further oxygenation but also to achieve closed-system conditions for increasing alkalinity generation.

In contrast to afore mentioned chemical systems, Oxic Limestone Channels (OLCs) surrender the air-tight sealing of the limestone channel, see Figure 5.17. Practically, OLCs are ditches lined with limestone and are therefore very easy to construct. The consequence of the unhindered oxygen ingress is even more limestone armouring and clogging of the system. Despite these objections, it has shown that dissolution of alkaline minerals still occurs at significant rates so that OLCs can be applied successfully for AMD treatment (Skousen, Rose et al. 1998). Just as with OLDs, high flow velocities are required to reduce clogging and armouring. These can be ensured by sufficient slope gradient, Skousen and Rose (1998) suggest 20% or more for optimal
results. In order to counter low dissolution rates, more limestone is required and longer trenches should be constructed compared to ALDs and OLDs to achieve good treatment results.

![OLC in Eastern USA (from Skousen (2005 ))](image)

Both OLDs and OLCs should be followed by an aerobic treatment step where metal precipitates formed in the limestone drain/channel are removed and oxidation and precipitation are continued.

### Applications and limitations

Where mine water fails to be treatable with ALDs and the installation of a RAPS or a compost wetland is not possible, an OLD can be viable treatment choice. If the system manages to work long-term also it constitutes a low-cost treatment option. Although OLDs can bear higher levels of “risky” contaminants, it is recommended by Cravotta and Trahan (1999) to limit the concentration of iron and aluminium for water being treated in an OLD to 10 and 20 mg/l, respectively. Acidity should not exceed 90 mg/l (as CaCO₃).

An OLC is a treatment option that is justified when other alternatives are not feasible due to technical or financial restrictions. In remote, mountainous areas or where hard bedrock makes excavation works very expensive, the installation of a simple structured OLC is often the only treatment option that can be implemented (Younger, Banwart et al. 2002). OLCs are also applied as emergency measures where immediate water treatment is needed until a more elaborate treatment system can be installed.
5.2.3 **Permeable Reactive Barrier**

The techniques that have been described so far are all designed to treat surface water flows of point source character. However, this does not represent the reality of mine water occurrence. In many cases, mine water discharges as diffuse subsurface flow into the groundwater, posing risks to drinking water supply and vegetation. Even where no water supply is threatened, it is likely that the mine water will eventually discharge in a surface water body in a diffuse manner where it causes adverse environmental impacts (PIRAMID Consortium 2003).

The usual active treatment approach to such a problem is “pump & treat” which, as the name implies, involves pumping of the groundwater at a well and subsequent active water treatment. Pumping can also be combined with a passive treatment step but this contravenes with the definition of a passive system which precludes external energy input. Truly passive options for this kind of mine water are restricted to a Permeable Reactive Barrier (PRB) which is pictured in Figure 5.16.

![Image of mine water emanating from the downgradient face of the PRB in Shilbottle, UK.](image)

**Figure 5.16** Mine water emanating from the downgradient face of the PRB in Shilbottle, UK.

### Design scheme

PRBs seek to achieve in-situ remediation of contaminated mine water by taking advantage of the natural hydrogeology of a site. For implementing a PRB, a trench in the flow path of a contaminant plume is backfilled with reactive material. The water is
intercepted and lead through the permeable, reactive zone where processes for contaminant removal take place. Figure 5.17 should be consulted for the basic design concept of a PRB. Water leaving the PRB at the downgradient face is significantly less contaminated than water entering the installation.

![Figure 5.17 Basic design concept of a PRB.](image)

There are two generic options available for the configuration of a PRB (Fabian, Jarvis et al. 2006):

- A “continuous wall” arrangement, in Figure 5.18, where the full length of the barrier consists of reactive material.

- A “funnel-and-gate” system, in Figure 5.19, where the water is directed to a reactive section by impermeable limbs so all water has to pass through the restricted reactive zone.
The reactive media responsible for water treatment is usually compost-based with the aim to promote SRB activity. Substrates described in the section on compost wetlands and RAPS are in general suitable for PRB, but special attention must be paid to the hydraulic conductivity of the medium in order to prevent water by-passing. By adding coarse, high structure materials, such as pea gravel and limestone, further alkalinity generation can be attained and insufficient permeability of the compost is counterbalanced. Younger et al. (2002) recommend adjusting the hydraulic conductivity of the reactive medium to a value around 5 to 10 times higher than the conductivity of the surrounding aquifer.

It is also recommended to key the reactive barrier into the aquitard in order to prevent treatment failure by contaminated mine water passing underneath the barrier. Eventually, the reactive media needs to be replenished or completely removed and renewed. This may be due to depletion of necessary agents in the media (e.g. carbon sources for SRB activity) and/or loss of permeability resulting from clogged pores.
(Younger, Banwart et al. 2002). As this creates significant costs, the design should aim to minimize the frequency with that such maintenance is required.

The removal processes taking place in a PRB are basically the same as occurring in compost wetlands or RAPS (when limestone is part of the reactive media). Key functions of the PRB are neutralisation by sulphate reduction, precipitation of metal sulphides (reaction with H₂S-gas), metal hydrolysis (caused by pH rise) and adsorption on organic matter.

Applications and limitations

PRBs are designed for the treatment of diffuse source, subsurface, acidic mine water. Currently, no other viable passive alternatives for subsurface flow treatment exists so if a passive treatment for this type of mine water is required, a PRB has to be installed (Fabian, Jarvis et al. 2006).

Apart from the criteria mentioned above, the choice of the reactive media depends on available material that can be sourced locally at reasonable costs. This may at some places mean that there is no choice due to the scarcity of organic material (Younger, Banwart et al. 2002). The problem can be addressed by adjusting the configuration of the PRB to this circumstance. In funnel-and-gate PRBs, less reactive media is required and therefore recommendable where cheap and sufficient reactive media is hard to obtain. On the other hand, this option requires a wider construction as higher residence times must be achieved in the gate compared to a “continuous wall” design. In cases where space is very limited this may be an issue (Jarvis and Younger 2006).

Another factor which significantly influences the cost of a PRB installation is the depth of the aquifer. Due to stabilization measures that become necessary at certain depths and special equipment which is required to dig deep trenches, PRBs are economically more feasible at shallow aquifers.

5.2.4 Other Passive Treatment options

Surface-Catalysed Oxidation Of Ferrous Iron (SCOOFI) reactors

The principle of SCOOFI-reactors is based on the property of ochre to accrete to high surface area media when oxygenated mine water is passed over it. The resulting ochre layer absorbs ferrous iron and catalyses the oxidation of ferrous iron in-situ.
Compared to oxidation of ferrous iron in open water, the process is more rapid when it occurs in association with ochre. Resulting ferric iron will precipitate as ochre and the layer on the high surface media will grow. Consequently, clogging of the porous media will occur if no regular ochre removal takes place. The application of SCOOFI-reactors is restricted to net-alkaline waters as they have not alkalinity generating capacity and ochre formation (hydrolysis) releases protons into solution (Jarvis and Younger 2001).

It has been mentioned before but it is deemed worth mentioning again, in most cases the restrictive factor for installing a passive system is insufficient land availability. An option to overcome these limitations is the application of SCOOFI-reactors at sites where iron is the contaminant of concern. They require far less space than other alternative passive treatment systems (e.g. aerobic wetland) because of their high area-adjusted removal rates. Depending on the flow conditions in the SCOOFI, between 36 g/m²/d and 4000 g/m²/d of iron can be removed. For comparison, the area adjusted removal rate for an aerobic wetland lies between 10 g/m²/d and 20 g/m²/d. Although this sounds very promising, SCOOFI-reactors are a newer technology and therefore experiences in their application are very restricted. Reliable design guidance is not yet available and installation should only be considered, when land availability excludes any other treatment option (PIRAMID Consortium 2003).

**SRB reactors**

Usually, mine water displays a huge molar excess of sulphate over dissolved metals but it is not very often a contaminant of concern whose removal it targeted. As mentioned before in Section 3.3.4, this may be different in arid and semi-arid areas where the concentration of total dissolved solids has a significant impact on water usability. The commonly achieved amount of sulphate reduction in anaerobic systems results in enough hydrogen sulphide gas to successfully precipitate metals present in the mine water. However, when the aim is to address sulphate in particular by SRB, a highly reducing environment must be provided (Eh < -150 mV). This is best achieved in a reactor where oxygen ingress is precluded. Like SCOOFI-reactors, no extensive track record of full-scale applications exists for anaerobic SRB reactors and their application is only recommended where other solutions are not possible (Younger, Banwart et al.2002).

**Pyrolysite process**
Manganese removal is relatively difficult in mine waters and often not accomplished in passive systems (see Section 5.2.2.3). For manganese to precipitate it is necessary to obtain a relatively high pH (consists of a shallow bed of limestone aggregate inundated with mine water. After laboratory testing determines the proper combinations, the microorganisms are introduced into the limestone beds via inoculation ports located throughout the bed. The use of limestone as structure for the micro-organisms also increases the alkalinity and raises the pH of the water (Berghorn 2001).

**Algae and microbial mats**

The characteristic of algae to bioaccumulate considerable amounts of contaminants has lead to the development of algae based passive treatment systems for mine water (Brown, Barley et al. 2002). Applications that involve algae for water quality improvement require a porous media, such a rock filters that encourage algae growth. Water percolating through this media passes the algae which take up metals from the mine water and the provide nutrients required for other biological treatment processes taking place in the system. In natural and constructed wetlands, algae contribute to metal removal, especially in open and deeper water where other plants cannot settle (Knight 1992). In particular, the application of algae has shown to be successful for manganese removal from mine waters as practiced at the Wheal Jane site in Cornwall, England (Whitehead 2005) and for uranium removal at the Pöhla site in Germany (Künze, Kiessig et al. 2007).

Similar to algae, mixed populations of heterotrophic bacteria, blue-green algae and filamentous green algae forming microbial mats have been applied for treatment purposes. Among the metals that have been successfully removed from mine waters with microbial mats are iron and manganese. The responsible mechanisms involved in metal immobilisation comprise:

- surface binding to the mat or mat exudates present in the porous media,
- precipitation with anions present in the mat,
- mat-mediation of the water conditions in favour of metal-oxide precipitation, and
- active transport of the metals into the cell.

There are various means that have been investigated to harness microbial mats for water quality improvement. These include free-floating mats, mats immobilised on
glass-wool in baffle tanks, mats immobilised on floating glass-wool balls and mats established in field ponds by application of silage (Brown, Barley et al. 2002).
6 Environmental Mining Challenges in South Eastern Europe

6.1 Introduction

During the “Mining for Closure” Workshop presentations were held that showed mining site management cases from the region to visualise the issues concerning mining and mine water management in the Western Balkans. It was also a forum that offered local experts to present the challenges they are facing and to initiate future cooperation between regional and international professionals.

6.2 Trepca Mining Complex, Kosovo (Territory under interim UN – administration)

6.2.1 Background

The territory of Kosovo (Territory under UN interim administration) is of particular interest in the trans-boundary context within the Balkans. It is both a centre of considerable minerals sector activity and a source of waters for each of the three distinct watersheds in the region. As such, all fluvial flows from Kosovo are trans-boundary. Therefore, water-born pollutants originating the mining operations present a permanent risk to the surrounding communities and to communities throughout the region, affecting the health of surrounding populations, flora and fauna both in Kosovo and downstream in Serbia proper. Information on this section has largely been drawn from Blacklock (2007).

Among the many problems the Kosovo region has to cope with low level of economic development, very slow development of privatization process of socially owned enterprises and lack of vision for economic development. One of the factors hindering economic development is insufficient electricity supply in the region which is, so far, mainly provided by monopolistically owned coal fired power stations. Kosovo has limited water reserves, which in the future will be a limiting factor for the economic
and social development of the country. The population suffers from a high level of poverty of which about 50% are considered to live in general poverty and 12-16% in extreme poverty. The difficult social situation of the region reflects itself in a deficit of good qualified professional cadres both in central and local institutions.

In 1988, energy, colour metals and their processing made up 63% of Kosovo’s industrial production. These activities have caused great damage to the environment and their negative impacts have yet to be rehabilitated. Considerable process in mitigating problems resulting from sub-standard mining practices has been achieved by the ENVSEC initiative under the coordination of the UNDP office in Kosovo. In 2006, the Environmental Assessment and Remediation Project Action Plan (EARAP) Project was launched to implement a feasibility study for the remediation of two Trepa mines, i.e. Stan Trg/Stari Trg Mine and Artana/Novo Brdo Mine.

6.2.2 Trepa Mining Complex

The name “Trepa” refers to an industrial conglomerate which comprises mines and processing installations all through Kosovo. Regular production of the Trepa Mine Ltd. started 1930 at the Stan Trg/Stari Trg Mine. At the height of its operations, Trepa operated nine mines primarily in the North and Eastern parts of the territory of which five have outstanding extraction potential. Since the early 1980s Trepa suffered from the lack of reinvestment, repair and maintenance. Trepa’s processing infrastructure was in poor condition, some of which have declined even further since 1999, as many facilities have remained stagnant or operating at a fraction of their capacities. Today, the mines are for the major part not functioning for technical as well as legal reasons linked to the ownership of Trepa.
6.2.3 The Stan Trg/Stari Trg Mine

The Stan Trg/Stari Trg Mine is located approximately 8 km north east of Mitrovica town and it is considered by some to be one of the richest lead, zinc and silver mines in Europe. Since 2000, rehabilitation works have been ongoing with support from several international donors. According to Trepca’s estimates in 2006, the mine is expected to gradually build up to an annual production rate of 400,000 tonnes per year. Most serious environmental problems associated to this site are contaminated mine waters which are continuing to pollute the surrounding areas via contaminants leeching into the soil and...
ground water as well dust generation of tailings material from uncovered tailings dumps. It is estimated that on average 6.5 tonnes of Zn and 30 kg of Cd are ejected via the mine waters into the Barska River (and subsequently into the Ibar River) annually.

![Photo courtesy of Denika Blacklock](image)

**Figure 6.2 Mine water pollution at the Stan Trg/Stari Trg Mine.**

For abating the environmental issues related to this site, implementation of a mine water treatment system is an important step that has to be taken. Remediation measures as recommended by the feasibility study, included chemical mine water treatment to reduce heavy metal concentrations in the water discharging into the Barska River, dust prevention as well as possible tailings stabilization.

### 6.2.4 The Artana/Novo Brdo Mine

Located 51km south of Pristina, the lead-zinc mine remained operational until the 1999 conflict. The site comprises the mine itself and two tailings dumps adjacent to the former Artana concentrator along the Marec/Kriva riverbank. Acidic mine waters (pH 2.7-3) are discharged from the mine workings and flow to a short drainage, then directly into the Marec/Kriva river. Two large tailings impoundments are located adjacent to the Marec/Kriva river at and downstream from the former concentrator. The upper tailings are severely eroded and significant amounts of waste have been transported into the river.
At the Artana/Novo Brdo Mine AMD generation is of major concern and must be addressed by either active or passive treatment. It is also crucial to implement measures to halt ongoing degradation of tailings material into the Marec/Kriva River. When tailings are discharged in the riverbed they may cause clogging benthic structures, change flow characteristics of the river (e.g. threatening stability of bridges) and continue to release heavy metals and acidity. In shallow streams they can also render the river unnavigable.

Challenges to Remediation in Kosovo (Territory currently under UN interim administration):

- serious financial resource constraints;
- capacity building for effective public administration, mine management and operations;
- marginalization of the environment sector due to status, economy and security issues;
- delays in the privatization process of Trepca.
6.3 Bucim Mine Site, Macedonia

6.3.1 Background

The Republic of Macedonia (also referred to as the Former Yugoslavian Republic of Macedonia or FYROM) is situated in the central southern part of the Balkan Peninsula.

Macedonia has hosts deposits containing economic grades of copper, iron, lead, precious metals such as silver and gold, and zinc. In the second half of the 20th century, an extensive processing and fabricating infrastructure was also established that allowed the production of not only these metals and their alloys, but also such ferroalloys as ferrochromium, ferromanganese, and ferronickel, and aluminium. In 2003, Macedonia’s lead-zinc mining and processing industry faced several stoppages that stemmed from pollution-related closures, raw material shortages, and financial difficulties (USGS).

In 2003 and 2004, the Government of Macedonia included its lead-zinc mines in the national privatization program whereby it was indicated that past environmental liabilities would be the responsibility of the state (USGS).

Most trans-boundary pollution issues associated with sites within Macedonia are of a moderate nature for a number of sites have largely been based upon the considerable distance to the Greek Border. Relevant operations lie in the catchment of the Vardar River, where Greece is the receiving nation of any cross-boundary pollution for this river.

6.3.2 Bucim mine

The Bucim mine, near Radovis in eastern Macedonia, has started operations again on May 6, 2005, under Russian ownership. It is the country’s only major copper mine and largest metal mine. The Bucim mine is consistently listed as a major environmental problem for the country with pollution related risks encompassing: heavy metals contamination in water and soil, particulate emissions to air, and (the possibility of) stability concerns in tailings impoundments.
The mines total output is 4,000,000 tons per year of ore and 4,000,000 tons per year of waste rock. Ultimately, 99.7% of the ore processed is disposed as tailings.

Water pollution arising from AMD generation is of major concern at the site. Main sources for water pollution include water pumped from the open pit, discharge from the flotation plant and drainage water from a large uncontained waste rock dump with flow rates between 2 to 15 l/s. Additional concerns arise from a large tailings dam that affects local communities with airborne particulate pollution. In addition to effluents from the dam that mix with the acidic and copper rich effluents from other mine sources, dust from the tails surface can blow toward the very nearby village of Polnica (located directly below the tailings dam). In efforts to abate the dust, trees have been planted and a polymer has been applied to a 4-hectare area.

The waste rock dump is very large but appears unlikely to have risk of a significant physical failure. As this waste rock dump today appears to constitute a significant economic resource suitable for processing, the new owners are looking for investors for a process plant to recover copper from the waste rock. In this case, the environmental issues with this dump could focus onto the acidic/metallic effluents in the short to
medium term. This reservation is, of course, conditional upon investor support for processing of the waste rock (secondary quality ore) dump being secured.

Soils and waters around the site are significantly affected by pollution from the site. AMD from the waste rock dump has a pH of 3 and contains 200-400 ppm Cu (clearly visible blue/green tint). Surface water and sediment samples indicate heavy metals and Cu concentrations and mine waters flow onto agricultural lands.

![Photo courtesy of Dejan Mirakowski](image)

**Figure 6.5**  Copper contamination in a stream at Bucim

Mining related hazards at the Bucim mining site involve:

- toxic/acidic effluents,
- uncontained waste rock, dust emissions and unsecured workings,
- poorly contained and/unstable tailings wastes

The mine facilities are in a generally good condition and the new operators expressed clear interest in improving the environmental problems, in particular, the discharge of open pit, rock pile and tailing waters.

The Bucim site was chosen to serve as theoretical case study for the workshop exercise. During the workgroup session, lead by two internationals expert, possible mine water treatment solutions or abatement measures were to be identified with regard to site specific conditions. Therefore in the following section more detailed information
will be given on site characteristics and the outcome of the working group will be presented.

6.4 Bor Mine Site, Serbia

6.4.1 Background

Only a century ago, the town of Bor was a small village and has changed with the discovery of copper ore and its exploitation since 1903. The village has transformed into an industrial and urban centre of Serbia with a population rising to around 60000 inhabitants in the early nineties. The Rudarsko Topionicki Bazen’s (RTB) Bor mining, beneficiation, and smelting complex comprises several mines and processing facilities in north-eastern Serbia. At the height of its productivity, around 1990, 14000 people were employed at the Mining and Smelting Company Bor. From that time on, decrease and economic decline were dominating the scene. As a consequence, infrastructures degraded as well as unemployment and poverty rapidly increased.

In the course of the privatisation process of RTB Bor an Environmental Impact Assessment was conducted. It identified contamination pathways affected by the different operation units of the complex, see Table 6.1.
<table>
<thead>
<tr>
<th>Ser. No.</th>
<th>Description</th>
<th>Degree of effect to the environmental pollution</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>RTB Bor Plants</td>
<td>Water</td>
</tr>
<tr>
<td></td>
<td>RBB Bor Plants</td>
<td></td>
</tr>
<tr>
<td>1</td>
<td>Veliki Krivelj open pit mining</td>
<td>****</td>
</tr>
<tr>
<td>2</td>
<td>Cerovo open pit mining</td>
<td>***</td>
</tr>
<tr>
<td>3</td>
<td>Veliki Krivelj flotation</td>
<td>****</td>
</tr>
<tr>
<td>4</td>
<td>Bor flotation</td>
<td>***</td>
</tr>
<tr>
<td>5</td>
<td>Pit (Hydrometallurgy)</td>
<td>*****</td>
</tr>
<tr>
<td>6</td>
<td>Zagradje lime-burning factory</td>
<td>*</td>
</tr>
<tr>
<td>7</td>
<td>Bela reka sandstone</td>
<td>**</td>
</tr>
<tr>
<td></td>
<td>TIR Bor Plants</td>
<td></td>
</tr>
<tr>
<td>8</td>
<td>Smelting plant</td>
<td></td>
</tr>
<tr>
<td>9</td>
<td>Electrolytic refinement</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>Sulphur acid plant</td>
<td>*****</td>
</tr>
<tr>
<td>11</td>
<td>Copper and copper alloy foundry</td>
<td>**</td>
</tr>
<tr>
<td>12</td>
<td>Thermal power station</td>
<td>***</td>
</tr>
<tr>
<td>13</td>
<td>Transportation</td>
<td>*</td>
</tr>
<tr>
<td>14</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* neglige or small
** and *** important
**** and ***** considerable

Table 6.1 Degree of environment pollution in RTB Bor complex plants (FIDECO 2006)

The process of copper ore treatment has produced large amounts of ore waste and flotation tailing heaps, located in the vicinity of the towns of Bor and Majdanpek. The waste and tailing heaps pollute the environment in a number of ways, sometimes to the extent of ecological catastrophes. Reportedly, hundred years of mining have left over 11000 tons of waste per citizen of Bor. Other problems people in the region have to face are air pollution, lifeless rivers and damaged and destroyed agricultural soil.

In Serbia, trans-boundary risks may arise from mining sites located on tributaries of the Timok River and the Pek River, which in turn are tributaries to the Danube. Other mining operations are located at the near the Drina River and the BiH border which is also rises concerns regarding possible transboundary pollution.

Mining related hazards at the Bor mining site involve:

- toxic/acidic effluents,
- uncontained waste rock,
- dust emissions & unsecured workings,
- toxic solid waste,
- airborne toxics & SO2,
- poorly contained and/unstable tailings wastes, and
- poorly contained smelter residues and chemicals.

Three mining sites were presented at “Mining for Closure” Workshop; all are located in the Bor region and display different features.

![Figure 6.6 Mining sites in the Bor area](image)

6.4.1.1 Bor mine

The active copper mine Bor is the oldest copper ore mine, with exploitation in modern times dating back to the beginning of 20th century. Mine waters from underground
mining flow directly into the Bor river. This river later joins the Kriveljska river, where also mine waters of Veliki Krivelj and Cerovo mines are discharged into. This results in excessive surface water pollution in the mentioned streams, as well as in groundwater pollution in alluvial and karst aquifers in the surroundings of the mentioned mines.

About 9 km to the east from the mine, after joining of the Borska and Kriveljska streams, following water characteristics have been identified in long-term studies: pH 3.7-6.5 \( \text{SO}_4^{2-}: 880-3235 \text{ mg/l} \) Fe: 0.8-223mg/l Cu: 0.5-315 mg/l

Additional sources for contamination at the site are discharges from copper melting and electrolysis processes into the water basin of the Borska river. After the the Borska and Kriveljska rivers conjoin, the originating river is called Crna reka (Black river). It is a tributary to the Danube and considered to be the most polluted stream in Serbia, if not in Europe.

### 6.4.1.2 Gornja Lipa

The abandoned copper mine Gornja Lipa is situated on the northern slopes of the mountain Crni Vrh, close to the river Lipa representing the spring part of the Veliki Pek. The mine was closed in the middle of the sixties, when 500000 tons of the ore were left unworked. Table 6.2 shows the metal concentrations downstream the Gornja Lipa mine in various distances to the site.

<table>
<thead>
<tr>
<th>Locality</th>
<th>Metal content (mg/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td>River Lipa (100 m downstream of the mine)</td>
<td>Fe=2.9; Cu=1.10; As=0.95; Zn=0.46</td>
</tr>
<tr>
<td>River Lipa (200 m downstream of the mine)</td>
<td>Fe=3.0; Cu=0.95; As=1.90; Zn=0.38</td>
</tr>
<tr>
<td>River Lipa (500 m downstream of the mine)</td>
<td>Fe=9.7; Cu=1.65; As=0.94; Zn=0.46</td>
</tr>
</tbody>
</table>

Table 6.2 Concentrations of relevant metals downstream the Gornja Lipa mine, Serbia.

### 6.4.1.3 Majdanpek

In the past 30 years, the area surrounding the town of Majdanpek has witnessed two ecological disasters caused by uncontrolled discharge of large amounts of flotation tailings and waste waters into the surface streams and the surrounding karst and alluvial aquifers.
The Majdanpek copper deposit is situated in the vicinity of the town of Majdanpek, near the Yugoslav-Romanian border. Long exploitation and processing of low-grade copper ore took place in the vicinity of the town. As a result of these activities, two tailings impoundments were created, the "Valja Fundata" and the "Saski potok".

In 1974, an uncontrolled discharge of several million cubic metres of tailings and waste waters occurred via underground streams from the "Valja Fundata" tailings impoundment which was installed on karstified underground. Other sources indicate that the entire River Pek was wiped out by 7 Mm$^3$ solids and 4.5 Megalitres of cyanide waters. This resulted in significant pollution of the karst aquifers, destruction of speleological structures, contamination of surface waters of the Veliki Pek river and of the neighbouring aquifers.
7 Related considerations for Mining & Environment

When talking about mine water treatment and other remediation measures, one has to keep in mind that at sites, where due to insufficient regulatory frameworks liabilities are not clearly defined or where irresponsible mine planning results in non-compliance, getting to the point where the implementation of a treatment system can be seriously considered is a long way go. To make environmentally sound and socially responsible mining practices feasible, financially as well as technically, environmental protection, human health risks and social aspects have to be taken more into consideration in mining regulation and practice. Another reason why issues like environmentally and socially responsible mine operation and closure have become increasingly important over the past years, and continue to do so, are growing expectations for environmental protection, desire for reduced human health risks and increased value of land as recreational space.

Sustainable mining practices take into account that in many regions the mining sector is a very important contributor to local and national economy and therefore the viability of a mining operation has to be an integral part of the solution.

The Regional Environmental Center for Central and Eastern Europe (REC) as well as ENVSEC have each developed tools and guidelines for promoting sustainable mining practices with key focus upon economies in transition. At the “Mining for Closure” Workshop, the “Governance Principles for Foreign Direct Investment in Hazardous Activities” developed by the REC were presented. The Governance Principles are referred to in the the “Mining for Closure – Guidelines for sustainable mining and mine closure”, therefore both will be introduced in this report.

7.1 “Mining for Closure”

In 2005, the Environment & Security Initiative has published the “Mining for Closure – Guidelines for sustainable mining and mine closure” to promote “best
practices” related to mining. The report provides principles and guidelines for government, mining industry, NGOs and other stakeholders with the goal to stimulate incorporation of sustainable corporate practice, adequate regulatory frameworks and incorporation of governance principles. Furthermore, it emphasises the importance of ongoing assessment of trans-boundary, environmental and human safety risks posed by sub-standard mining operations and the importance of implementing risk reduction measures.

Tangible benefits at specific sites in the region are expected from pilot projects which address such risk reduction measures like mine water treatment facilities. Thereby, know-how is disseminated and experiences can be gained in the regional context to improve future applications.

Eventually, it is hoped that promoting risk reduction tools, providing the guidelines and principles will help to reduce environmental and security risks from mining operations.

Although the “Mining for Closure” report has not in particular been subject to the Workshop, the issues it discusses are highly relevant to mine water treatment as ensuring a sustainable approach in mine management is a prerequisite for successful risk reduction and eventually remediation.

“Mining for Closure – Guidelines for sustainable mining and mine closure” is provided on the ENVSEC website: http://www.envsec.org/see/

The application of governance principles such as developed by REC are considered as important component to achieve “best practice” in risk reduction for mining operations and are referred to in the “Mining for Closure” guidelines.

### 7.2 Governance Principles for Foreign Direct Investment in Hazardous Activities

The Governance Principles are intended to apply to foreign direct investment in industrial, mining and other activities and have been developed to complement voluntary codes of conduct, compacts and other instruments.

Since the Baia Mare accident, the REC has led a process of dialogue on improved governance over foreign direct investment in hazardous activities, taking into account
the complex root causes of mining and other disasters, and acknowledging the lack of appropriate controls. The result is a tool for risk reduction in the investment process that has been widely praised and appreciated in all contexts in which it has come up.

The “Governance Principles for Foreign Direct Investment in Hazardous Activities” were launched at the World Summit on Sustainable Development in Johannesburg in 2002. Since 2005 they have been included in the initiative “Mining for Closure” in connection with the Environment and Security initiative.

With following the conduct of Good Governance, a formal or informal institution has the chance to establish a system of responsibility and accountability in decision-making to build trust and capacity to cooperate. While it is clear that there are costs involved implementing best social and environmental practice, savings might only be short-term because when taking the mine-life into account, an improperly managed investment process can lead to an increase rather than a decrease of expenses due to risks of or harm to human health and the environment. Investors that apply the PGs make use of a potent risk reduction tool, and authorities and stakeholders can also benefit from their application.

The Governance Principles take into account the state of the art of initiatives in the field of corporate social responsibility, and extend these principles towards the specific characteristics of investment processes, in recognition of the fact that the methods of engaging in high-risk activities are profoundly affected by major investments. Themes covered by the principles include corporate good citizenship, company environmental policy, public participation and stakeholder relations, environmentally responsible corporate values, and accident prevention and management.

General principles on corporate responsibilities include the Polluter Pays principle, and public accessibility and precautionary principles. On hazardous investment, principles such as the utilisation of Sustainability Impact Assessments, continuous EIA, and financial assurance play a key role.

The “Governance Principles for Foreign Direct Investment in Hazardous Activities” are available at the REC webpage:

http://www.rec.org/REC/Programs/EnvironmentalLaw/PDF/Governance_Principles.pdf
7.3 Risk Assessment and Investigation of Contaminated Land

7.3.1 Introduction

The adoption of sustainable and socially responsible mining practices is necessary and an efficient way to abate risks posed to the environment and human health by present and future mining operations. Although these are issues which are definitely highly relevant, in many regions major concern arise from mineral activity related sites where mining operations have ceased and no efforts are made to prevent or abate ongoing contaminant release and associated risks. In fact, there are very significant environmental and trans-boundary security risks associated to non-operational, abandoned or orphaned mine sites, often arising from uncontained, AMD generating mine wastes.

Frequently, the responsible party no longer exists or cannot be located. The costs associated to a systematic rehabilitation programme are enormous and governments alone rarely are capable to take on financial and technical responsibility for it. An opportunity to reduce financial constraints is to mobilise public and private funds, but even if this is successful, money will in most cases remain the limiting factor. To ensure the best use of available financial resources, a phased approach to remediation projects must be developed. This involves prioritisation of sites where the most significant adverse impacts are identified and where significant tangible benefits can be expected.

Prioritization itself is preceded and based on building of detail inventory of relevant mining related sites to eventually determine the degree of risk posed by a site to the receptors. The inventory requires the collection of qualitative and quantitative data of the given conditions such as the morphology of sites and their geochemistry. Moreover, status of ownership and activity status are crucial information to be obtained. It becomes clear that the process of prioritisation depends largely on scientific assessment of key physical and geochemical parameters and broad risk assessments.

7.3.2 Key issues for Risk Assessment and Contaminated Land

In order to approach the issue of contaminated land, a common approach to take into account the specific site in question is required. The mixture of current use and
environmental setting are the two main considerations for the Risk Matrix based approach employed in the UK. The term “suitable for use” provides an outlook to determine the extent of risk that is present on a specific site.

To assess this, a risk assessment matrix is employed as a consistent basis across varying types of sites with the ability to transfer the information across a broad spectrum of stakeholders and interested parties. This risk based approach introduces the idea of a Conceptual Site Model (CSM).

The CSM aims to establish whether a pollutant linkage between the sources of contamination and the receptors exists in a site and to what extent. This is achieved through a phased approach and although it can be applied to any site, it is always site specific.

The CSM consists of a Source, Receptor and Pathway. For a pollutant linkage to exist, all three must be present. The phased approach involves further determination and improvement of the CSM at each phase of the work. This enables both a qualitative and quantitative risk assessment to be performed on each site, enabling transfer of information on both a technical and non-technical form.

The phased approach has three main phases, each improving the CSM further at each stage. The initial Phase 1 stage is a desk top study of the site, gathering all available data on the three parts of the CSM. An initial overview at the site with respect to whether a pollutant linkage exists at this stage is attained. If established, it may well be necessary to further determine the extent of a pollutant linkage and thus the risk to sensitive receptors.

This leads to the Phase II where by further information mainly regarding the soil and water conditions relating to the site are desired. This usually involves intrusive investigation work with collection of soil and water samples for laboratory analysis. Comparative guideline values are employed in order to assess whether a significant degree of pollution may exist on the site. Exceeding soil and water guidelines would lead to a Phase III in the risk assessment of the site.

The Phase III helps quantify to what extent a site poses a risk to sensitive receptors. This involves a quantitative risk assessment employing the data and information obtained in the first two phases. Several models are available to quantify the risk to receptors and can be both probabilistic and deterministic types of models. Generation
of Site Specific Target Levels are produced which can be used to determine whether or not remediation is required on the site, in agreement with the regulators.

Risk assessment matrix and CSM is easily transferred between a variety of stakeholders and parties. It enables a degree of use and reuse options to be established on land that may be contaminated to some extent, establishing “suitable for use”. It embraces sustainability by providing information valuable to improve the social, economic and environmental setting specific to the site in question.

7.4 Legal frameworks in Europe pertaining to mining

A key driver for progress in environmental legislation in the Western Balkans is the potential EU membership according to the “Stabilisation and Association process”. The process was launched in 1999 with the eventual prospect of EU membership for countries of the region. The Thessaloniki Summit of 2003 confirmed that all countries and territories were "potential candidates". The process covers the following countries:

- Albania
- Bosnia and Herzegovina (BiH)
- Serbia
- Montenegro
- Kosovo issues

For the accession process, regional co-operation is a highly important field of action. In this context, the adaptation of environment relevant EU directives plays a significant role due to their regional nature and for environmental issues being covered by a number of EU laws.

With this background, EU directives pertaining to mining issues are of highly relevant to countries in the SEE region as they will form the basis of future requirements that will have to be met by all mining operations. Eventually, the adapted legislation will lead more sustainable mining practices in the accession countries and therewith to a growing demand for efficient mine water abatement measures.

Like other industrial activities, mining has to comply with environment protection laws, regulation and standards. Mining operation and environment protection
requirements are most commonly implemented through a variety of different legal tools, such as (BRGM 2001):

- Mining legislation,
- Environmental planning and assessment legislation,
- Environment protection legislation,
- Other legislation and standards, including occupational health and safety.

Government roles in environment protection are gradually evolving in response to changing perceptions in mining operations. Developments in the ownership and control of mines and metal production facilities have greatly influenced both the locations of mining and investment in new mines in Europe and around the world (BRGM 2001).

Recent major mining accidents in Baia Mare/Romania in 2000 and in Aznalcóllar/Spain in 1998 increased the awareness of environmental and safety hazards of mining activities. As a consequence of these and other accidents which visualised the risks associated to mining activities, the European Commission initiated a re-evaluation of European policies in relation to mining accidents in general and mine water pollution in particular. Resulting from this process, EU environmental legislation relevant to mining activities has been revised and following legal instruments were introduced to improve the safety of mines:

- Amendment of Seveso II Directive
- Best Available Techniques Document on waste-rock and tailings under the IPPC Directive
- Mining Waste Directive

With regard to water protection, the Water Framework Directive is the key European legislation and pertains significantly to mine water management. The principal objectives of the WFD are to prevent future deterioration and to promote sustainable use of water. Because of its relevance to the issues discussed in this report, the WFD will be briefly presented following.

7.4.1 Overview on European legislation pertaining to mining activities

This section provides an overview of legislation relevant to mining activities in the European Union. The wording of the particular directives can be obtained via the
website of the European Union at http://europa.eu/. For summaries and analysis of the legal texts, their implications on mining activities and background information on the development of the new legislation, following sources are recommended: Kroll, Amezaga et al. (2002), BRGM (2001) and Wolkersdorfer (2005, )

**Water**
- Water Framework Directive 2000/60/EC and Daughter Directive 80/68 on Groundwater protection:
- Directive 75/440/EEC on potentially drinkable water, which introduce the notion of protection of raw water resources and define target values
- other Directives such 82/176/EEc or 84/156/EEc on mercury, 85/513/EEc on cadmium,

**Waste**
- Mining Waste Directive 2006/21/EC
- The Landfill of Waste Directive 99/31/EC

**Industry**
- The IPPC Directive 96/61/EEC concerning integrated pollution prevention and control
- Environmental Liabilities Directive 2004/35/EC

**Other**
- Habitat Protection Directive 92/43/EEC
- United Nation Convention on Access to Information, Public Participation in Decision Making and Access to Justice in Environmental Matters (Aarhus
## 7.4.2 Mine Water and the Water Framework Directive

The Water Framework Directive provides a unified framework for management of surface and groundwater in Europe and it is the most relevant piece of legislation for coordinated regulation of water impacts of contaminated land, including mine water pollution (ERMITE-Consortium 2004).

The WFD is intended as a general framework for protection of all waters including rivers, lakes, coastal waters and groundwaters. It has set itself the objective to achieve “good ecological and chemical status” for all surface water bodies and “good chemical and quantitative status” for all groundwater bodies by 2015. In order to meet the goal, key objectives are reduction and control of water pollution from all sources, such as agriculture, urban areas and industrial activities. A difference to previous legislations is to include ecological standards so that any activity leading to biological changes, e.g. morphological impacts, changes in flow rate or introduction of alien species will have influence on compliance. Furthermore, diffuse pollution is increasingly recognised as a major source of metals to freshwaters.

An approach introduced by the WFD is the river basin based management that recognises that water systems do not stop at political borders. This requires close cross border co-operation between countries and all involved parties to succeed in sustainable water management. This stipulates the parties to organise a common River Basin Management Plan which all EU member countries have to set up until the year 2009. As the WFD is meant to be a framework, not every issue is regulated into detail but when necessary it can be amended, such as for the Groundwater Directive. (Brown, Barley et al. 2002; Wolkersdorfer 2005,)

Although mining and consequently mine water, is not directly mentioned in the WFD, water management has to be in accordance with the targets set out by the directive as polluted mine water affects surface and ground water. The WFD consists of different steps and monitoring procedures as shown in Table (). Sources of water pollution, such as mine water, affecting water quality within catchments have to be identified and all possible means for water quality improvement must be investigated (Wolkersdorfer 2005,).
Furthermore, economic principles such as “polluter pays” are incorporated in the WFD. This principle ensures that the mine operator has to guarantee that the quality standards of the discharged water are met at all times and have to be continued after mine closure. Water treatment has to go on until no more adverse impacts from the operation on the receiving watercourse can be expected (Wolkersdorfer 2005, ). For this regulation, water treatment during active mining operations is usually not of major concern as operators have to be in accordance with the “polluter pays” principle.

However, the situation is different for the large amount of uncontrolled discharges of abandoned mine sites which lack a (potent) legal owner. To address the problem of water pollution arising from abandoned sites, an economically optimal management plan is required to get the most out of the scarce resources available for water quality protection. Under such circumstances, the application of passive treatment options offers an alternative to expensive conventional treatment measures to remediate affected water environments and to meet compliance with the WFD.

The WFD opens up further possibilities to implement cost-effective, innovative mine water treatment techniques where they were not applicable due to regulatory restrictions. Mitigation of the environmental impacts from mining before the WFD has commonly focused on source regulation, i.e. on pollutant release processes and their resultant emissions. Remedial measures therefore were mostly installed at the mine site itself. As described by the ERMITE-Consortium (ERMITE-Consortium 2004)) the WFD, however, draws its attention to the water quality standards in the various water environments, on the welfare of aquatic ecosystems and to sustainable development and cost-effective management of water bodies. This allows the consideration of installing mine water treatment measures downstream of mine sites near to compliance boundaries thereby widening the range of different possible measure allocations within a catchment.

On the other hand, there is a possibility the implementation of the WFD into national law will lead to much stricter limits on maximum concentrations of metals allowed to be discharged to freshwater environments. The final figures are still in discussion, and will then need to be transposed to national law (Jarvis presentation). Lower discharge limits and stricter compliance requirements would clearly reduce the applicability of passive systems.
<table>
<thead>
<tr>
<th>Year</th>
<th>Issue</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>Directive entered into force</td>
<td>Art. 25</td>
</tr>
<tr>
<td>2003</td>
<td>Transposition in national legislation</td>
<td>Art. 23</td>
</tr>
<tr>
<td></td>
<td>Identification of River Basin Districts and Authorities</td>
<td>Art. 3</td>
</tr>
<tr>
<td>2004</td>
<td>Characterisation of river basin: pressures, impacts and</td>
<td>Art. 5</td>
</tr>
<tr>
<td>2006</td>
<td>Establishment of monitoring network</td>
<td>Art. 8</td>
</tr>
<tr>
<td></td>
<td>Start public consultation (at the latest)</td>
<td>Art. 14</td>
</tr>
<tr>
<td>2008</td>
<td>Present draft river basin management plan</td>
<td>Art. 13</td>
</tr>
<tr>
<td>2009</td>
<td>Finalise river basin management plan including programme of</td>
<td>Art. 13 &amp; 11</td>
</tr>
<tr>
<td>2010</td>
<td>Introduce pricing policies</td>
<td>Art. 9</td>
</tr>
<tr>
<td>2012</td>
<td>Make operational programmes of measures</td>
<td>Art. 11</td>
</tr>
<tr>
<td>2015</td>
<td>Meet environmental objectives</td>
<td>Art. 4</td>
</tr>
<tr>
<td>2021</td>
<td>First management cycle ends</td>
<td>Art. 4 &amp; 13</td>
</tr>
</tbody>
</table>

Table 7.1 Timeline of the European Water Framework Directive (European Commission)
8 Partnerships in Mining Site Management

Existing partnership programmes all over the world addressing the issue of mining site management and mine water treatment provide access to information gathered by governments, academia and mining industry. They constitute a valuable source of information often based on a vast amount of experience gathered throughout the last few decades in those areas where mining has been the in the focus of research and public concern. Especially for stakeholders from regions where the local conditions do not favour intensive research and thorough analysis of the mining related issues on their own, such knowledge transfer and collaboration with partnership programmes are outstanding opportunities to take advantage of and participate in networks already existing to identify solutions and find support for implementing successful mining site management.

Examples for successful partnerships, mainly Canadian and European but also international, were presented at the workshop. They showed how the partnerships are organised, what their main objectives are and how information can be obtained through them.

8.1 North American Partnerships and Projects

Canada has a long history in mining and consequently also a long history of mine water pollution. To address these issues, in the 1980s Canada developed a collective approach to address technical issues of national importance. The results were multi-stakeholder initiatives that have been models for cooperation among industry, various levels of government, NGOs (non-governmental organizations) and First Nations (i.e. aboriginal Canadians).

Domestically, Canada partners with industry, the provincial/territorial governments, non-governmental organizations and aboriginal Canadians to cooperate in technology development. Internationally, Canada partners with other governments to build capacity and transfer knowledge to improve the environmental management of their mining sectors.
In Canada, mining becomes more and more recognised as a sustainable industry where environmental problems can be prevented by sustainable practices and no public funding is necessary due to responsible management of mining operations. This achievement is partly the consequence of cooperative problem-solving that has proven to be efficient, very sustainable and successful in achieving synergetic effects.

The consortia are formed by Federal Government, Provincial/Territorial Governments and Industry partnerships and with participation of non-governmental organizations and Aboriginal Canadians. They also create a forum for disseminating information gathered and for technology transfer with parties outside Canada. According to Gilles Tremblay (2007), reasons for the success of the consortium approach are science and technology based programs with focused plans where results are shared with full disclosure. Volunteer participation of the members and government-industry-NGOs partnership are further characteristics of the consortium approach which contribute to its efficiency. Relevant consortia and projects in Canada:

- Mine Environment Neutral Drainage (MEND)  
  http://www.nrcan.gc.ca/mms/canmet-mtb/mmsm-mmst/mend/default_e.htm

- Aquatic Effects Technology Evaluation Program (AETE)  

- Toxicological Investigations of Mining Effluents (TIME)  
  http://www.nrcan.gc.ca/ms/canmet-mtb/mmsm-mmst/enviro/time/time-e.htm

- National Orphaned/Abandoned Mines Initiative (NOAMI)  
  http://www.abandoned-mines.org/intro_e.htm

- The Mining Association of Canada’s Toward Sustainable Mining  
  http://www.mining.ca/www/

- The Global Alliance convened by the International Network for Acid Prevention (INAP)  

- International Conference on Acid Rock Drainage (ICARD)  
MEND and NAOMI are partnerships of high relevance to mine management and mine water abatement on a regional and international level as well as the Global Alliance which is currently preparing a guide to Best Practice for ARD management. Therefore their aims, objectives and means of communication will be introduced in this report to encourage future cooperation with these partnerships.

8.1.1 Mine Environment Neutral Drainage – MEND

The Mine Environment Neutral Drainage (MEND) program was founded in 1998 in response to the projected high liabilities resulting from long term mine water pollution. MEND is a cooperative research organization sponsored, financed and administered by a voluntary consortium consisting of the mining industry, the Government of Canada and eight provincial governments.

The Mine Environment Neutral Drainage (MEND) initiative was the first international multistakeholder program to develop scientifically based technologies to reduce the effect of acidic drainage. It was implemented to develop and apply new technologies to prevent and control acidic drainage.

MEND has since provided a knowledge base that has made it possible to manage the complexity of acidic drainage. Of particular importance was the development of a common understanding among participants. It has allowed partners to take actions with greater confidence and to gain multi-stakeholder acceptance more quickly. MEND has essentially developed a toolbox of technologies that is available to all stakeholders, including operators, regulators and consulting engineers.

MEND results are available through reports, newsletters, seminars and conferences, workshops and the website: http://www.nrcan.gc.ca/mms/canmet-mtb/mmsl-lmsm/mend/mendpubs_e.htm.

A key document is the MEND manual which summarizes in six volumes the work completed by MEND in a format that provides practitioners in industry and government with a manageable single reference document. Although it cannot be seen as a detailed guide for mine management and remediation, it offers comprehensive working references on relevant techniques and practices. Furthermore, there are over 200 technical documents available and 3 CD’s with over 160 reports.
Technologies reviewed and discussed by MEND:

- prediction techniques
- water covers for unoxidized and partially oxidized wastes
- dry covers using soils/tailings, organic materials and geomembranes
- mine waste management options for tailings, waste rock and treatment sludges
- active and passive treatment
- cold climate issues (permafrost).

8.1.2 National Orphaned and Abandoned Mines Initiative – NOAMI

The National Orphaned/Abandoned Mines Initiative (NOAMI) is a co-operative Canadian program that is guided by an Advisory Committee consisting of the mining industry, federal/provincial/territorial governments, environmental non-government organizations and First Nations. NOAMI adopted the MEND framework to develop a policy-based program for remediation of orphaned and abandoned mine sites in Canada. It was established in 2002 to address challenges arising from over 10000 orphaned or abandoned sites across Canada. Various issues and initiatives concerning the implementation of remediation programs are studied by the project. Five task groups are overseeing the key programme areas of NOAMI:

- information gathering (for inventory),
- community involvement,
- barriers to legislative collaboration,
- funding approaches,
- jurisdictional legislative review.

The aims of NOAMI are to ensure that approaches across jurisdictions are consistent, clear, transparent, coordinated and efficient for orphaned and abandoned sites in Canada. Their work comprises several studies, among them a review of funding models and community involvement. Moreover, a national inventory of non-active mining sites has been created to provide a database upon which sound decision making, cost-effective planning and sustainable remediation are enabled. The compiled information on abandoned and orphaned sites in Canada will be available to the public on the internet on to ensure transparency of decision-making and access to information by governments, civil society, industry and other stakeholders.

All reports generated by NOAMI are available from www.abandoned-mines.org
8.1.3 Global Alliance – Global ARD Guide (GARD Guide)

In 2003, several initiatives devoted to acid mine drainage have formed the Global Alliance under the coordination of INAP to foster collaboration between the regional groups around the world. Currently, the Global Alliance is developing a Global Acid Rock Drainage (GARD) Guide, a worldwide guide that captures and summarizes best science and a risk based approach to ARD management. The GARD Guide aims to be a reference for acid prevention and to identify Best Practice in the field of ARD.

The Guide is supposed to cover all phases of an operation from initial discovery through to final closure. In that way it is meant to assist industry in providing high levels of environmental protection, assist governments in the assessment and regulation of affairs under their jurisdiction and enable the public to have a higher degree of confidence in and understanding of acid prevention proposals and practices. A so-called preliminary “beta version” of the guide is scheduled for review and testing in 2008.

8.2 European partnerships and projects

In response to a number of dramatic pollution events associated with abandoned mines (e.g. Baia Mare, Aznalcollar, Wheal Jane), the European Union launched a policy process which among others, put forth the Mine Waste Directive (YOUNGER PADRE). During the development of this policy process several EU – funded research projects were launched to address topics like passive remediation and management of catchments affected by mining activities such as the PIRAMID and the EREMITE project. In the course of the projects, a network of acidic drainage researchers within Europe developed, who are in turn linked to many other researchers and practitioners throughout Europe.

- Passive In situ Remediation of Acidic Mine / Industrial Drainage (PIRAMID)
  http://www.ncl.ac.uk/piramid/

- Environmental Regulation of Mine waters in the EU (ERMITE)
  http://www.ncl.ac.uk/environment/research/Ermite.htm
8.2.1 Passive In situ Remediation of Acidic Mine / Industrial Drainage – PIRAMID

The aim of the PIRAMID project was to harmonise research and practice efforts in Europe to create passive in situ remediation (PIR) methods for acidic drainage treatment. Starting in the year 2000, a key objective of this three year project has been the development of a handbook presenting engineering guidelines for the design and installation of passive treatment systems. These guidelines are intended to provide practitioners in the field of environmental engineering with sufficient information to enable them to confidently undertake feasibility studies and develop conceptual design statements for passive mine water remediation systems.

The PIRAMID guidelines provide the reader with fundamental information on development and implementation of robust engineering designs for the passive treatment and / or passive prevention of mine water pollution. They address operational aspects of water management in the particular context of the design and construction of passive remediation schemes. Each chapter considers a particular aspect of passive technology and describes the key principles involved. The selection of the appropriate form of passive treatment for specific types of effluent is discussed and appropriate methods of design are recommended.

The PIRAMID Guidelines are available from http://www.ncl.ac.uk/piramid/

8.2.2 Environmental Regulation of Mine waters in the EU - ERMITE

The ERMITE research and development project ran from 2001 until 2004 and was conducted by several European institutions under the coordination of the University of Oviedo and the University of Newcastle upon Tyne. Its objective was to evaluate existing European mine water guidelines, including open legal questions. The goal of the ERMITE project was to provide integrated policy guidelines for developing
European legislation and practice in relation to water management in the mining sector. These guidelines took into account the catchment management approach defined by the Water Framework Directive and the sustainability principles enshrined in the Treaty governing the EU (ERMITE). ERMITE’s interaction with the European Commission and the European Parliament has contributed to the development Mining Waste Directive.

The project has produced the following deliverables available at http://www.minewater.net.ermite:

- National Case Studies
- Overview of the EU and Eastern Europe
- Institutional Research
- European Policies and Mine Waters
- Economic Analysis of Mine Water Pollution Abatement
- Mining Impacts on the Freshwater Environment: Technical and managerial guidelines for catchment scale management
- National Recommendations and Workshop Reports
- Policy Guidelines
- Policy Briefs

8.2.3 Partnership for Acid Drainage Remediation in Europe - PADRE

PADRE has been established in 2003 as a permanent commission of the International Mine Water Association (IMWA). It is a non-governmental, non-profit, scientific and technical organization with the aim of fostering best practice, based on the latest research, in the remediation of acidic drainage from active and abandoned mine sites throughout Europe. With acidic drainage research and remediation practices maturing throughout Europe, there is also a need for a central ‘watching post’ in order to avoid unnecessary duplication of effort. Therefore PADRE has been established to meet these needs. Another task of the partnership is to act as the European hub for the Global Alliance convened by INAP, and to participate with in the development of a proposal for the GARD guidelines.

Objectives of PADRE are:

- Promote international best practice in the stewardship of waters and soils on European sites subject to the generation and migration of acidic drainage.
- Foster collaborative, international research and development into techniques for
characterization and abatement of acidic drainage in Europe.

- Promote dissemination of knowledge of current best-practice and innovations relating to acidic drainage prevention and remediation, with particular reference to European conditions, including the evolving framework of relevant EU legislation.
- Advance the training of present and future generations of European professionals who will engage in the art and science of acidic drainage prevention and remediation.
- Actively collaborate with a Global Alliance of organizations based in other continents which share similar objectives, which has been convened by INAP.

PADRE activities include:

- maintaining best practice guidelines on passive remediation (the PIRAMID Guidelines)
- maintaining best practice guidelines catchment-scale mine water management (the ERMITE Guidelines)
- developing further sources of guidance
- implementing training and professional development activities for European scientists and engineers
- European branch of the Global Alliance.
9 Mine water management cases

A very tangible topic at the “Mining for Closure” Workshop was the demonstration of mine water management cases in Canada, Germany and the UK by the international experts. They showed how mining sites with significant AMD generation were successfully remediated with innovative mine water prevention and treatment techniques. Most of the techniques applied in the presented cases should be familiar to the reader as they have been described in the previous section on mine water treatment in this report.

9.1 Mine water management cases from the UK

There are approximately 50 full-scale mine water treatment systems across the UK, treating in excess of 100,000 m³/day. Over the last decade alone, about 45 full-scale mine water treatment systems have been installation by the UK Coal Authority. The Coal Authority (CA) was formed in 1994, and one of its remits is to address the legacies of the formerly privatised coal mining industry in the UK.

Mine water treated in the UK, with only one exception, emanates from coal mines. Of them, the majority are waters from abandoned deep mines. Most of the treatment systems are passive, but often rely on pumping to deliver mine water to the system, to avoid uncontrolled discharges, or to protect overlying aquifers. Of the about 50 full-scale treatment systems in the UK currently operational:

- One treats water from a metal mine (Wheal Jane tin mine, Cornwall).
- Two are fully automated chemical treatment plants (both High Density Sludge).
- Approximately 6 have preliminary chemical dosing (H₂O₂, lime, caustic), followed by passive units
- Three are fully passive, according to the definition above.
- The remainder treat water from coal mines and rely on pumping from workings, followed by passive treatment (no chemical addition)
Special attention should be paid in this context to the CoSTaR project (Coal Mine Sites Targeted for Remediation Research) which comprises six full-scale operating passive treatment systems open for research purposes in North East England. The applied techniques are all treating coal mine drainage and include bioreactors (including a PRB), a compost wetland, a RAPS system and four varieties of aerobic wetlands. The project has two basic objectives on its agenda. Firstly, the applications are intended to ameliorate mine water pollution arising from the underground mine workings and colliery spoil heaps. The second objective is to disseminate expertise in passive mine water treatment across the EU by giving researchers from across Europe the outstanding opportunity to visit the CoSTaR sites and to carry out their studies alongside with the researchers from the University of Newcastle upon Tyne/HERO research group.

Figure 9.1 Passive mine water treatment sites of the CoSTaR project in England

For more information on the CoSTaR sites and research opportunities please visit:
http://www.ncl.ac.uk/environment/research/HEROCOSTAR.htm

At the Workshop, two sites which currently undergo mine water treatment were presented. The one first is located at Shilbottle, where a PRB treats diffuse mine water emanating underground from a colliery spoil heap. The second example showed the application of a conventional lime dosing active treatment plant with a HDS (high
density sludge) process. As mentioned before, active treatment is inevitable to achieve treatment at various sites. However, the Horden site will not be further discussed as this report focuses on passive prevention and treatment applications.

Information on the Horden active treatment site can be obtained from the Coal Authority on:
http://www.coal.gov.uk/resources/environment/hydrolog/horden/hordenhydrolog.cfm

### 9.1.1 Shilbottle PRB system

At the site, severe contamination of the adjacent stream occurred from a colliery spoil heap producing acidic mine drainage. The mine water is draining subsurface from the spoil heap and finally discharges into the Tylaw Burn. To address ongoing contamination at the site, mine water is intercepted by a PRB which is followed by aerobic treatment. The characteristics of the Shilbottle are presented in the following as summarised from (Jarvis and Younger 2006).

The Shilbottle spoil heap is located in Northumberland, North East England where it spreads over 15 hectares. It is the remainder of the Shilbottle Grange Colliery, which ceased its operation in 1982. In the late 1990s, investigation of the site by staff and students from Newcastle University began and lead to the development of a treatment system that is now successfully treating the contaminated mine water originating from the spoil heap. The treatment system is one of the six passive treatment sites that constitute the CoSTaR project.

#### 9.1.1.1 Site conditions

The spoil heap at Shilbottle contains a large amount of highly pyrite-rich material. Due to a lack of sufficient neutralising material, the resulting leachate is highly acidic. The diffuse nature of the drainage made flow rate measurement quite difficult but estimates suggested a value in the order of <10 l/s.

With regard to its chemical characteristics, the mine water displayed concerning amounts of several contaminants. The leachate was highly acidic with a pH value around 3 – 4. Concentrations for other contaminants, such as iron, manganese and
aluminium were among the highest ever recorded in the UK for mine drainage (see Table 9.1).

<table>
<thead>
<tr>
<th></th>
<th>GW9</th>
<th>GW10</th>
<th>GW11</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>4.17</td>
<td>3.55</td>
<td>3.29</td>
</tr>
<tr>
<td>Acidity (mg/l as CaCO₃)</td>
<td>3322.00</td>
<td>2534.00</td>
<td>1360.00</td>
</tr>
<tr>
<td>Fe (mg/l)</td>
<td>688.00</td>
<td>452.00</td>
<td>278.00</td>
</tr>
<tr>
<td>Mn (mg/l)</td>
<td>238.00</td>
<td>181.00</td>
<td>165.00</td>
</tr>
<tr>
<td>Al (mg/l)</td>
<td>298.00</td>
<td>249.00</td>
<td>97.00</td>
</tr>
<tr>
<td>SO₄²⁻ (mg/l)</td>
<td>11176.00</td>
<td>9288.00</td>
<td>6334.00</td>
</tr>
</tbody>
</table>

Table 9.1 Chemical characteristics from Shilbottle spoil heap at three sampling points

The stream impacted by the mine water is a tributary to the River Croquet. Pollution of this river is particularly undesirable because of its significance to Salmonid fisheries in the UK and its use of drinking water abstraction after the inflow of the Tyelaw Burn, carrying the contaminated mine water.

9.1.1.2 Treatment solution

A key characteristic at the Shilbottle site is the diffuse subsurface mine water flow draining from the spoil heap and discharges along the toe of the spoil heap, often only just before the Tyelaw Burn. Previous efforts in achieving passive mine water treatment at this site failed, partly for not taking into account the discharge structure of the leachate. About 60% of the leachate bypassed a wetland series installed in an earlier attempt to achieve passive treatment at the site.

A second reason for poor performance of the system was the nature of the installed treatment units which were aerobic wetlands. For the kind of mine water occurring at this site, it would have been necessary to achieve a significant rise in alkalinity for successful treatment. Aerobic wetlands on the other side have no capacity to generate alkalinity, so hydrolysis, which occurs upon oxidation in the wetlands, causes further proton release. The low pH conditions resulting from the process did not allow extensive metal removal via precipitation (see section 3.31).

To achieve better results with passive treatment, it was necessary to find more suitable approaches which regard the specific features of the site. As the water emerges
to the surface, very close or even in the stream, the physical conditions precluded land intensive treatment methods, as long as there was not any pumping unit to be involved which could have directed the water suitable area.

The solution was found in the application of a PRB followed by settling pond and aerobic wetland. The PRB contained a mixture of limestone gravel, cow manure and green waste that could all be sourced locally.

To allow metal to precipitate after the PRB, the mine water is forced to emerge to the surface from where is directed to a sedimentation pond (see). When the water emerges to the surface metals can oxidise and together with pH rise from interaction with the PRB, metals are precipitated in the settlement pond. Because of the high acidity concentrations entering PRB, it is important to ensure sufficient alkalinity generation. Otherwise, proton release in following aerobic applications would negatively impact the treatment performance.

One of the previously installed aerobic wetlands previously was incorporated into the new treatment system as polishing step before discharge. Compared to the first attempt, water entering the wetland was then net-alkaline so treatment efficiency of the wetland could be significantly improved.

9.1.1.3 Implementation process

After the suitable treatment scheme for given conditions had been identified, it was important to undertake preparatory steps before moving of the full-scale application. At Shilbottle, enhancing the system design was supported by laboratory and field work and involved:

- Identification of optimal location and adequate dimensions of the PRB;
- Determination of permeability of the spoil heap material to adjust PRB media;
- Analysis of potential PRB media to find the most suitable substrate to obtain optimal results.

Regarding the design options of a PRB, a continuous wall arrangement was chosen, because reactive media was at very low cost and land was scarce between the spoil heap and the stream, a “funnel-and-gate” seemed less attractive.

Construction works required three months and started in July 2002. The first activity was the diversion of the Tyelaw Burn as its proximity to the soil heap precluded
construction works. For the PRB a 140 m long trench of and varying depth was excavated. Special attention was paid to key the PRB into the underlying boulder clay so the excavation cut about 0.5 m into the aquitard. At the downgradient face of the PRB a permeable brick rubble berm was installed to create hydraulic conditions that facilitate water flow into the settlement lagoons.

Commonly, PRBs are covered by a clay layer to ensure anoxic conditions as well as to reduce water ingress. In Shilbottle, the installation does without one, as oxygen entering with surface run-off and rainwater is expected to be sufficiently stripped by the substrate in the PRB.

After a few years, a grass sward has developed on the PRB so there are hardly any signs that indicate the presence of the treatment unit.

![Figure 9.2 System design of the PRB at the Shilbottle site.](image)

### 9.1.1.4 Performance

The treatment system proved to substantially reduce the amount of contaminants between the spoil heap and discharge point. Since commissioning the system water quality has consistently improved, with approximate removal efficiencies of 96% for Fe, 78% for Zn, 71% for Ni, 52% for Mn, and 59% for sulphate.

Within the system, the greatest level of removal is achieved in the settlement ponds receiving the water from the PRB. As mentioned before, the following aerobic wetland acts as polishing step to optimise treatment performance. Table 9.2 shows how the total percentage of metal removal only increases moderately after the water leaves the wetland compared to the effluent of the second sedimentation lagoon.
<table>
<thead>
<tr>
<th></th>
<th>Lagoon 2</th>
<th>Wetland Discharge</th>
</tr>
</thead>
<tbody>
<tr>
<td>Iron</td>
<td>82</td>
<td>96</td>
</tr>
<tr>
<td>Manganese</td>
<td>41</td>
<td>52</td>
</tr>
<tr>
<td>Sulphate</td>
<td>48</td>
<td>59</td>
</tr>
<tr>
<td>Aluminium</td>
<td>85</td>
<td>88</td>
</tr>
<tr>
<td>Nickel</td>
<td>60</td>
<td>71</td>
</tr>
<tr>
<td>Zinc</td>
<td>71</td>
<td>78</td>
</tr>
</tbody>
</table>

Table 9.2 Percentage removal rates after sedimentation ponds and aerobic wetland

9.1.1.5 Costs

Jarvis and Younger (2000) calculated the costs for the Shilbottle system in Table 9.3 and compared the cost to an equivalent active treatment solution suitable for the site. Costs for the active treatment are estimated and based on a containerised HDS plant. Calculation includes the necessary components such as polymer dosing equipment, reactor mixers, clarifier rake, monitoring equipment and so on but not planning and enabling works.

<table>
<thead>
<tr>
<th></th>
<th>Passive system</th>
<th>Active system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Capital expenditure - PRB and lagoons</td>
<td>78,000 GBP</td>
<td>200,000 GBP</td>
</tr>
<tr>
<td>Capital expenditure - wetland</td>
<td>200,000 GBP</td>
<td>-</td>
</tr>
<tr>
<td>Operational expenditure</td>
<td>7,500 GBP/year</td>
<td>44,500 GBP/year</td>
</tr>
</tbody>
</table>

Table 9.3 Actual costs of the Shilbottle PRB system and estimated costs of a potential active system

With AMD generation at the site expected to be ongoing for centuries, the life-cycle costs are an important figure for calculating the overall costs for the application. For the PRB at Shilbottle a life cycle of about 40 years was estimated which will ultimately governed by the availability of organic carbon. For comparison with possible active treatment, the calculation in Table 9.4 is based on a 10-year lifetime.
### 9.2 Mine Water Management Cases from Germany

In Germany, one of the most impressive mining remediation projects of the world has been ongoing over the last 16 years. The project is financed by the German government with a total volume of more than 6.5 billion Euros and is executed by the state-run WISMUT Project. The purpose of the project is the large-scale remediation at sites of former uranium mining and milling in East Germany. The information on mine water management in Germany presented here is largely drawn from Kiessig (2007) and will be indicated if other.

During the period of the cold war, uranium mining was a key industry in Eastern Germany. The Soviet and later Soviet-German company WISMUT became the world’s
third largest producer of uranium. When Germany was reunified in 1989, the WISMUT became part of the German Ministry of Economic. At that time, mining operations were no longer viable and had to cease. In 1991, WISMUT was transformed from a mining company to a remediation project addressing the legacies of 45 years of intensive mining.

Finalisation of the project is scheduled for 2015. By the time, 1400 km of open mine workings, 311 million m³ of mine wastes and 160 million m³ of radioactive tailings (10 tailings ponds) will have been subject to remediation processes. The mine water discharge WISMUT is dealing with mounts up to 31000 m³ per year.

With respect to the finite character of the project, it became necessary to address those problems that will not be solved within the remaining period of time. So far, the project was utilizing modern, state-of-art conventional mine water treatment. Because of the high costs associated to them, the decreasing contaminant concentration over time (see section ) and the end of the project, the WISMUT started to investigate on passive treatment as viable alternative.

As a result from these investigations, a pilot wetland was designed and operated at the Pöhla mining site. The pilot plant was followed by a full-scale application in 2003 and was presented at the “Mining for Closure” Workshop.

9.2.1 Constructed Wetland system at the Pöhla Mine

The former uranium mine Pöhla is located in the Ore Mountains in South-Eastern Germany where mining activities finally ceased in 1990. It was the first mine within the project to be flooded which was followed by the installation of a mine water treatment plant in 1995. The initial treatment plant was developed by WISMUT and tailored to the specific properties of mine water occurring at the site. It was based on a selective precipitation/-flocculation process, comprising components such as uranium adsorption on a polymeric flocculant, co-precipitation of radium, arsenic removal via FeCl₃, iron and manganese oxidation, sludge dewatering and disposal in dry parts of the mine.

Soon after the commissioning of the plant, it became clear that for long-term treatment, a passive or semi-passive alternative is desirable. After the installation of a pilot treatment plant, the concept for a full-scale application was developed and implemented.
9.2.1.1 Site conditions

At the Pöhla site, mine water from the flooded underground reaches the level of natural overflow to the surface mine workings with a flow-rate around 4.7 l/s. The water is net-alkaline so pH adjustment is not a primary goal like in the preceding example. At Pöhla, elements of problematic concentrations were U, Ra, As, Fe, and Mn. Since flooding was completed in 1995, the concentrations constantly decreased, so today elements significantly exceeding the discharge limits are iron, arsenic and radium. From Table 9.5 it can be seen how contaminant concentration contaminant concentration decreased over time.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Fe</td>
<td>mg/L</td>
<td>2</td>
<td>5</td>
<td>4</td>
<td>4</td>
<td>4</td>
<td>4</td>
</tr>
<tr>
<td>Mn</td>
<td>mg/L</td>
<td></td>
<td>3.7</td>
<td>1.6</td>
<td>1.1</td>
<td>0.5</td>
<td>0.4</td>
</tr>
<tr>
<td>As</td>
<td>mg/L</td>
<td>0.1</td>
<td>0.5</td>
<td>2</td>
<td>2.2</td>
<td>2.2</td>
<td>2.9</td>
</tr>
<tr>
<td>U</td>
<td>mg/L</td>
<td>0.2</td>
<td>1.8</td>
<td>0.2</td>
<td>0.2</td>
<td>&lt; 0.1</td>
<td>&lt; 0.1</td>
</tr>
<tr>
<td>Ra-226</td>
<td>Bq/l</td>
<td>0.3</td>
<td>1.1</td>
<td>3.9</td>
<td>4.5</td>
<td>4.3</td>
<td>4.3</td>
</tr>
</tbody>
</table>

Table 9.5 Main components of the Pöhla mine effluent (Kunze 2007)

Although the number of critical components had decreased, geochemical modelling predicted that water treatment for the remaining elements would have to continue over approximately 15 years. With unit treatment costs for the conventional treatment plant of about 4 EUR/m3 at flow rates around 17 m3/h, the overall costs for mine water treatment at the site would be very high if no other solution to the problem had been found.

9.2.1.2 Treatment solution

A possible treatment system at the site had to remove iron, radium and arsenic from the mine water. Iron, due to its relatively low concentrations and the neutral conditions could be removed straightforward by oxidative processes. The priority task was to intensify both radium and arsenic removal using biological effects (KUNZE IMWA). To accomplish these goals, a constructed wetland system preceded by an aeration cascade was identified to be the most suitable approach.
The wetlands are intended to promote populations of various algae which function as accumulators for radium and arsenic. To provide sufficient colonisation area for microorganisms, floating mats and similar installations were used in Pöhla. These installations further support sedimentation processes for iron precipitates so the required wetland size is reduced. Arsenic strongly binds to iron precipitates so they are jointly removed from the mine water by sedimentation. Investigations on suitable macrophytes for contaminant removal found that Characeae are capable of accumulating considerably high radium activities and were therefore populated in the treatment system.

In general, when considering the application of biologically based processes, one has to bear in mind that it will take several growing seasons for purely biological treatment stages involving areas established with algae or plants to reach their full performance.

Apart from technical issues, constraints to apply passive treatment arise from legal requirements. To comply with them, a system must ensure treatment reliability. For a systems that strongly rely on biological processes, it is very difficult to guarantee stable discharge values as wetland performance varies with temperature, flow-rate and chemistry of the inflow. Peaks of arsenic concentration were observed to correlate with high-flow rates, frequently occurring at the Pöhla site. That is why there was need to incorporate counter-measures to achieve permissible discharge standards. At the full-scale constructed wetland in Pöhla, the last treatment step comprises adsorption filters consisting of granulated barium sulphate to intercept radium and granulated ferric hydroxide for reducing arsenic concentration in the discharge. Under normal conditions theses filters are not necessary to meet the required discharge values as the wetland capacity is sufficient to cope with the contaminant load. But when fluctuations occur that overwork the system capacity, the filters stand in to abate the exceeding concentration.

The structure of the treatment system can be summarized as follows:

1. Aeration cascade
2. Settlement pond
3. Pond with floating mats
4. Pond with Characeae algae
5. Polishing pond
6. Adsorption filters
Starting from the pond which comprises floating mats, the system is laid out in two parallel lines that work independent from each other. The aim of having a double line design is to achieve adequate redundancy even when one line is non-operational due to maintenance work or other implications.

Figure 9.3  Design scheme for the Pöhla constructed wetland

Figure 9.4  Arial view of the full-scale wetland at the Pöhla site (Kunze imwa)
9.2.1.3 Implementation process

Prior to the implementation of the full-scale treatment plant which replaced the conventional treatment facility, a pilot-plant was built to identify optimal the design and operating parameters. For the pilot wetland, an already existing concrete pond from former mining operation was used to host the system.

Parallel to the implementation of the constructed wetland a R&D project (“BioRobust”) investigated the treatment performance of the system under the various conditions that apply to the site. As a result, it was shown that wetlands can cope to a some extent with changing external conditions but beyond a certain level, system compliance cannot be guaranteed.

Because of the novel status of the wetland, the WISMUT agreed with the regulators on a relatively intensive monitoring programme, taking into account the possibility of non-compliance. It is relatively cost-intensive and requires substantial manpower which is to some extent offsetting the advantages of the passive approach. In 2007, the monitoring programme is anticipated to required the collection and analysis of 3000 samples per year. This amount will reduce over time when the system proves its reliability and less surveillance is necessary.

9.2.1.4 Performance

The Pöhla wetland system successfully reduces contaminant concentration of inflowing mine water to meet legal requirements as shown in Table 9.6.

<table>
<thead>
<tr>
<th></th>
<th>Discharge limit</th>
<th>Inflow</th>
<th>Outflow</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arsenic</td>
<td>100 μg/l</td>
<td>2623 μg/l</td>
<td>65 μg/l</td>
</tr>
<tr>
<td>Radium</td>
<td>300 mBq/l</td>
<td>4430 mBq/l</td>
<td>30 mBq/l</td>
</tr>
<tr>
<td>Iron</td>
<td>2 mg/L</td>
<td>6.6 mg/l</td>
<td>0.04 mg/l</td>
</tr>
</tbody>
</table>

Table 9.6 Average contents of relevant contaminants in inflow and discharge.

On a yearly basis, the system manages to remove 1400 kg of iron, 300 kg of arsenic and 600 MBq of radium.
9.2.1.5 Costs

The construction cost for the constructed wetland amounted about 700 000 Euro. In the initial phase of the wetland operation biological processes were not yet contributing to contaminant removal. To achieve acceptable discharge values, a major part of the contaminant removal was carried out by the adsorption filters. Costs for the frequent replacement of the adsorbent and the adsorbent itself resulted in higher unit treatment costs in the initial phase. Further costs arising in the initial phase were associated to stricter monitoring requirements.

Once, biological processes have established and less personnel is required to execute site monitoring, operational costs decrease significantly:

- Operating cost of conventional plant: 4 EUR/m3
- Operating cost for the first years of passive operation: approximately 2 EUR/m3
- Operating cost for long-term, steady passive operation: approximately 1 - 1.50 EUR/m3

9.2.1.6 Conclusion

The treatment mine water originating from flooded mine workings and spoil heap drainage at the Pöhla mining site via passive treatment proved to be sucessful and economical. Although the operating costs are significantly lower than the ones for conventional treatment, an important lesson learned at the Pöhla site was that passive systems are not maintenance-free and “zero-cost”, as often misleadingly acclaimed.
10 Technical Exercises

In the following, technical solutions for mine water abatement are worked out for two mine sites. The solutions are, as in every mine water treatment case, highly site specific but the aim of the exercises is to demonstrate how a case can be approached and to give examples how to identify adequate treatment options.

10.1 Bucim Mine, Macedonia

The situation given in at the Bucim mining site has been described in Section 6.3 and its main characteristics are briefly summarized in Table 6.1. A table with more detailed Bucim mine water data can be found in Annex A
Figure 10.1  Sources for mine water drainage at the Bucim site

<table>
<thead>
<tr>
<th></th>
<th><strong>Stream 1</strong> (Open pit)</th>
<th><strong>Stream 2</strong> (Waste rock)</th>
<th><strong>Stream 3</strong> (Tailings dam)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copper concentration</td>
<td>moderate (25 mg/L)</td>
<td>very high (430 mg/L)</td>
<td>low (0.06 mg/L)</td>
</tr>
<tr>
<td>Flow rate</td>
<td>high (21 L/sec)</td>
<td>low (3 L/sec)</td>
<td>low (3 L/sec)</td>
</tr>
<tr>
<td>pH</td>
<td>moderate (4.68)</td>
<td>low (3.81)</td>
<td>elevated (7.56)</td>
</tr>
</tbody>
</table>

Table 10.1  Main flow characteristics at the Bucim site.
Further facts relevant for a possible mine water treatment scheme:

- Water from the tailings dam is recycled and used for the flotation processes.
- The value of copper in Stream 1 discharged per year has been estimated to about 2 million USD.$^4$
- Precipitation rate in Radovis lies around 500 mm/yr (Miladinovic, Blinkov et al. 2006).

10.1.1 General considerations

A general question which opens the analysis of a potential treatment scheme for this site is whether the contaminated streams should be treated separately or together. Higher concentrations lead to a more efficient treatment process, i.e. higher removal rates can be achieved if the Stream 2 is treated separately. At the same time, a second treatment unit would have to be installed to treat the water from Stream 1.

By treating the two streams jointly, contaminant concentration is lowered in the water entering the treatment facility, which under often enables the application of certain treatment options in the first place. This option also requires only one treatment solution for the two streams and would therefore be less costly.

To answer to this question, it is recommendable to identify potential treatment systems first and then to evaluate the different configurations.

10.1.2 Passive treatment

A possible passive system would need to remove the two main contaminants of concern at this site which are acidity and copper. Copper precipitates in passive systems mainly as sulphide which suggests the application of compost based anaerobic systems that favor SRB activity. Possible options for referred to in this thesis are compost wetlands, RAPS, SRB reactors.

Apart from increased solubility of metals at low pH, SRB activity is hindered as its optimum lies at circum-neutral pH. Due to the high acidity in the solution, alkalinity

$^4$ Copper price assumed for 7000 USD/tonne at the workshop excersise.
generation in addition to SRB induced alkalinity would be required. Possible passive treatment options for increased alkalinity generation include ALD/OLD/OLC and RAPS. With regard to these applications, further parameters have to be taken into consideration such as atmospheric precipitation rate and topography.

The application of a RAPS and/or OLD/OLC is very likely to be precluded due to the lack of hydraulic gradient at the site. Although no specific data is provided, it can be gathered from Figure 2.3 that conditions are not suitable. Furthermore, RAPS require the water level to be maintained above the compost layer, otherwise the treatment capacity is significantly reduced and even remobilisation of retained contaminants can occur. Given the low precipitation/high evapotranspiration levels at the Bucim site, it can be assumed that flow rates are strongly fluctuating throughout the year. However, for both topography and climate, more detailed data are necessary to evaluate the applicability of a RAPS.

Chemical passive treatment has been described as an effective measure for alkalinity generation with limitations in its use for certain water qualities. ALDs do not require as much hydraulic gradient as OLCs and OLDs, but are very restricted with regard to the incoming water quality (Fe $^{3+}$ and Al $^{3+}$ concentration $>$ 2 mg/l). As mentioned in Section 5.6.2.1, mine waters with a pH below 4.5 are very likely to exceed these requirements. At the Bucim site, the iron values would be acceptable for ALDs but aluminium values are by far exceeding the thresholds, see Table 2.2. An ALD would be prone to fail short after installation and is therefore not an option at this site.

The probably most striking argument against any passive treatment option as single approach at this site is the very high copper content in the effluent. Firstly, as presented and explained in Section 4.3.2.4 and shown in Figure 4.1, passive treatment is most suitable at low to medium contaminant concentrations. Secondly, the high value of the copper in the effluent calls for recovery which is not possible by passive treatment. The last two arguments make the further consideration of passive treatment options at this site practically redundant.

Although a considerable data set was provided for the Bucim site and some important design decisions could be made on this basis, it became clear that for the conception of a passive treatment system, even if only a basic one, a very wide range of information is required. This reflects the fact that apart from main contaminants, passive mine water treatment is largely depending on further site specific conditions.
10.1.3 Active treatment

Because of the high copper concentration, metal recovery is one of the first options that should be examined when considering active treatment at the Bucim site. Common technologies that are able to recover marketable metals from wastewater are:

- electrolysis,
- solvent extraction,
- adsorption/ion exchange,
- membrane separation.

Among the listed technologies, ion-exchange considered as reliable, simple, and cost-effective (Hancke and Wilhelm 2003). For removing acidity from the discharges at the Bucim site, a simple chemical treatment option such as alkali-dosing is assumed to be suitable due to its robustness and reliability.

Factors that influence the applicability of these techniques are far less compared to those influencing passive treatment techniques. A main issue with chemical treatment is the generation of voluminous metal hydroxide sludge that needs to be disposed of. In most cases, the resulting sludge is taken to landfill which creates an additional and considerable cost factor. In Bucim, it may be worth assessing if a there is a possibility for the disposal of treatment sludge within the tailings dam. This would save costs for transport and disposal in a landfill.

The higher costs associated to an active treatment plant could be offset by the revenue from metal recovery.

Apart from the approaches described so far, there is also another possibility given at the Bucim site to reduce the contaminant release from the mine workings. The recirculation of flotation water reduces significantly the contaminant concentration of the tailings dam discharges. Other streams, such as the flow from the open-pit could also be captured by the circulation and be added to the flotation water. This may also be of particular advantage for the flotation processes as water scarcity has been reported to be a problem in the area (Mirakovski 2007). On the other hand, the salinity level would
increase in the flotation water, due to the high sulphate concentrations. This could interfere with the flotation processes and preclude this option.

Another problem that arises if Stream 1 enters the flotation water cycle instead of discharging into the receiving stream would be a lack of water in the receiving stream as the discharge from the open-pit constitutes the major part of it.

10.1.4 Tentative technical solution

In order to make a clear statement which would be the most effective and advisable solution, the conditions and possible consequences need to be assessed more closely. For now, three options as they have been identified at the “Mining for Closure” Workshop will be presented.

**Option 1:**

The contaminated streams are treated together in a treatment facility incorporating an ion exchange step for metal recovery and a chemical active treatment step to raise pH and remove other contaminants such as manganese and aluminium.

**Advantages:** Only one installation is required (no pumping), no restrictions regarding salinity problems for flotation processes and water level in the river can be maintained.

**Disadvantages:** Efficiency of the treatment plant is lowered due to high flow and low concentration, larger treatment capacity is required to treat higher flow-rate.

**Option 2:**

The high-concentration/low-flow stream discharging from the waste rock dump is treated with an active treatment system as described in the preceding option and the low-concentration/high-flow stream discharging from the open-pit is pumped up (250m) to the flotation plant where it joins the recirculation system.

**Advantages:** Metal recovery can focus on the high concentration stream and therefore increase efficiency, lower treatment capacity required and less sludge production due to lower treatment volume.

**Disadvantages:** Energy and equipment for pumping is required, oversalination of
the flotation water may occur and the water level of the receiving stream is significantly lowered.

**Option 3:**

No treatment plant will be installed and both contaminated streams are pumped to the flotation plant to enter the water cycle.

**Advantage:** No costs for treatment plant installation and maintenance.

**Disadvantages:** No metal recovery possible, salinity level will probably be too high for flotation purposes, tailings dam stability is not ensured for additional amounts of water, water level in the receiving stream is significantly lowered and pumping of considerable amount of water is required.

The option to apply passive treatment at the Bucim site is by no means generally excluded; especially not with regard to future mine water conditions after mine closure when contaminant levels will gradually decrease. This will also be the case for the copper content in the water and eventually render a potential ion exchange unit unprofitable, whereby the source for financial support would also run dry. Mine water drainage, however, will continue for a long time and needs to be treated, that is when the application of passive techniques should be reconsidered at the Bucim site.

**10.2 Tentative technical solution for Rosia Montana, Romania**

According to the chemical analysis conducted for the Environmental Impact Assessment at the Rosia Montana site, a significant contributor to the contamination of waterways in the region is the flow discharging from Adit 714. In Figure 10.2, the adit is indicated as number 4. More detailed mine water data for Rosia Montana are provided in Appendix A.

The main characteristics of the flow are:

- high iron concentration which has been fluctuating over the past years, see Figure 2.9, with an average concentration of ~380 mg/l for dissolved iron,
- low pH (~3),
- high but also fluctuating arsenic concentration (50 – 1700 μg/L).

Further site considerations are:

- Precipitation rates, including heavy rainfalls and snow, in the region where Rosia Montana is located are among the highest in the country with around 1200 mm per year (Murzin-Bencovski 2007).
- Being located in the Transylvanian Alps, winters are cold and freezing with temperatures below -20°C occurs.
- The terrain is generally mountainous.
- The site is located in a karst area (Peter, Racataianu et al. 2006).
10.2.1 Passive treatment

Of major concern at the site are acidity and iron at this site. Arsenic tends to co-precipitate with the most abundant element, so once a high iron precipitation rate can be achieved in a system, arsenic is jointly removed from the water.

With a look on the low pH value of the flow emerging from Adit 714, it becomes clear that an alkalinity generating technique would be required if the mine water is to be treated with a passive system. Similar to the example from Bucim, the techniques to increase alkalinity and to remove metals in mine water as discussed in this thesis are ALD/OLD/OLC, compost wetland, RAPS and SRB reactor.

Given that the site is mountainous, techniques requiring a certain slope should be applicable. However, the topography of the site needs to be assessed more thoroughly as too steep terrain is also unfavorable to passive treatment applications which have a certain aerial extent, such as wetlands. Chemical treatment such as ALD, OLD and OLC can cope with and even improve (OLD and OLC) their performance with higher hydraulic gradient due to less armouring with higher flow velocity. If a technique should be applied that requires anoxic conditions (ALD, compost wetland, RAPS, SRB bioreactor), the selected site for the water treatment should be as close as possible to the point of emergence as oxygen concentration rises from there on.

Because of the high iron concentration in the discharge, sludge accumulation and resulting maintenance work may be a notable cost factor at the site and should be taken into consideration when designing a treatment system. Metal sludge containing predominantly metal sulphides resulting from sulphate reducing processes is far less voluminous than metal hydroxides. On the other hand, if precipitation of ferric oxides is induced mainly in a sedimentation pond and if excavation is not too troublesome (e.g. if appropriate access and excavation equipment is available) then precipitation as oxides is acceptable.

Due to the low pH, a compost wetland as the only alkalinity generating unit would not be expected to constitute an optimal solution. SRB require near neutral conditions to execute optimal rates for sulphate reduction. With such low pH at the inflow, alkalinity generation would be significantly reduced. For acceptable discharge characteristics, a
compost wetland would have to be relatively large so that pH can gradually rise as water flows through it.

The high iron concentration precludes the application of chemical passive treatment measures as main alkalinity generating step unless frequent renewal of the limestone would be acceptable. Even though the hydrochemical analysis does not distinguish between ferric and ferrous iron, at such high levels it can be assumed that too much ferric iron will be in solution. The system would be likely to cease effective alkalinity generation soon after commissioning due to armouring processes. Application of an ALD at this site can therefore only be considered as a polishing step after a process step that has largely eliminated iron from the water. Usually ALDs are more efficient than OLDs but if the discharge of the preceding system is high in oxygen, the installation of an OLD is logical.

Bioreactors have the advantage that they require less land area, so their application is recommended where site conditions are not favoring space consuming installations such as wetlands. On the other hand, the technique is less documented and remains subject to further research until a certain reliability can be expected. At present, the application of reactors is only recommended where other possibilities of passive treatment have to be precluded due to restricted land availability.

As mentioned in the text before, in many cases where site conditions are suitable, RAPS are often a preferred option due to their ability to handle high loads of contaminants and cope with low pH levels. Another advantage of a RAPS application at this site is that it seems likely that they can handle the cold climate better than other techniques, see Section 5.6.1.5. Topography at the site needs to be confirmed but in this case, it can be assumed to provide sufficient hydraulic gradient. It may be that the terrain even proves to be too steep for a wetland installation. A good opportunity for a treatment area at this site would be the installation of the system on top of the Gura Rosiei tailings dam located about 3 km southwest from the Adit 714. This no longer utilised tailings dam is indicated on the map in Figure 6.1 and its surface is shown on the photo in Figure10.3.
The precipitation level in the region also promises a constant water supply so that drying out of anoxic parts in the RAPS can be avoided. Cost-wise, it should be assessed if required materials for the construction can be sourced locally. This is also important from a social development aspect on the implementation. The regional market profits from application of local products and maintenance work can be easily undertaken by locals as materials are readily available. The most important materials for the construction of a RAPS are limestone, clay and compost material of which all can be found in the area. Limestone is abundant as Rosia Montana is located in one of the numerous karst areas of Romania, organic material for the compost is assumed to be available as it is an agricultural region, and as clay tends to accumulate in river areas so it is expected to find clay deposits downstream the Rosia Valley.

**10.2.2 Active treatment**

Conventional treatment at the site could be executed by a regular lime dosing facility. Main contaminants would easily be removed and sludge should be disposed of at the tailings dam if possible. The application of active treatment is therefore rather opposed by the cost and maintenance managerial issues than by technical issues.
For the numerous reasons described in Chapter 2 and 5, the technical solution for Rosia Montana will be a passive approach to the mine water problem.

10.2.3 Tentative technical solution

A chosen treatment technique for the Rosia Montana site, more precisely for the flow emerging from Adit 714, is a RAPS treatment system and will be exercised in the following manner. The reader should again note, that is one potential solution among many that would then require diligent review and even comparison against other configurations. This potential configuration is provided as a continuation of the workshop exercise and is an example only.

10.2.3.1 Preliminary steps

The base of operations is an adequate set of information upon which decisions can be made. Local conditions do not only decide which is the most suitable technique to be applied is but also define the parameters of the system. A thorough examination of the local conditions in terms of mine water chemistry, flow rate, land availability etc. is key for successful treatment implementation.

The design information required for a passive treatment system comprises various aspects. The PIRAMID Consortium (2003) considers the following issues important to be addresses with regard to treatment design:

- Flow measurement
- Hydrochemical sampling and analysis
- Treatment site selection
- Site topography
- Site appraisal
- Ground investigation
- Contaminated land assessment

Other factors that may have influence on design values are ownership of targeted land, neighbor activities and accessibility of the area.
Although it is important to address all of these issues to successfully implement a full-scale passive treatment project, some data issues are indispensable and have to be collected very thoroughly (e.g. water quality and flow-rate data), while others (e.g. contaminated land issues) may require only cursory investigation, if only to discount as possible constraints on site-specific designs (PIRAMID Consortium 2003).

In Appendix D, a table on relevant hydrochemical parameters for different stages of the analytical process is provided.

It is also recommended to calculate the cation-anion balance to ensure all major ions have been determined and no significant element present in the mine water has been overlooked. This typically includes sulphate, chloride, bicarbonate (anions) and calcium, magnesium, natrium, potasium (cations), and usually at least one of the main metal contaminants if the concentration is high (e.g. Fe\(^{2+}\) / Fe\(^{3+}\) and Al\(^{3+}\)). The basic idea is that, when converted to units of milliequivalents per litre (which accounts for ion charge), the sum of the cations (positively charged) and anions (negatively charged) should be equivalent. If not, it can be assumed that one or more constituents are missing in the analysis.

The most critical value for a passive treatment system is net-acidity/net-alkalinity for selection and sizing of the treatment system. In this case study, the low pH already indicates that the mine water is net-acidic but in case a non-alkalinity generating treatment option is considered where pH is less of a problem, the technical reader is referred to Hedin (2007) in order to obtain detailed information on acidity/alkalinity measurement.

If the acidity has not been measured onsite therefore it must be calculated from the major acidity generating constituents in the water. Because pH is below 4, no bicarbonate is present and acidity equals net-acidity. In Equation 6.1, the formula for calculating net-acidity is provided.

The data set from the EIA on Rosia Montana that was available for the “Mining for Closure” workshop exercise and for the continuation of this design exercise does not contain all data necessary for calculating acidity. The analysis indicates an average value for dissolved iron (no specification between ferric and ferrous) of 380 mg/l. In the absence of real data, assumptions were required to allow a speculative design to continue – in a real situation, this data would need to be collected from the field. Assuming that under given pH conditions ferric and ferrous iron are present in equal
proportions, and even if no aluminium were present, acidity would be around 900 mg/L (as CaCO₃). However, it is very likely that other elements contribute to acidity, so for this exercise, 1200 mg/L for acidity has been assumed

\[
\text{Acid}^{\text{calc}} = 50*(2*\text{Fe}^{2+}/56 + 3*\text{Fe}^{3+}/56 + 3*\text{Al}/27 + 2*\text{Mn}/55 + 1000*10^{-\text{pH}}) - \text{alkalinity}
\]

(10.1)

10.2.3.2 Mine water prevention

As the Adit 714 is the lowest point of the historic workings, groundwater emerging from tunnels in the ore body – and from porous rock structures flow out at this point. As such it is highly likely that a large proportion of the water emerging from the Adit 714 is atmospheric precipitation that has infiltrated into the ground via the surrounding mine surface area (mine workings are indicated on the map in Figure 6.1).

As mentioned earlier, it is desirable to reduce water and contamination quantity as a first step in a mine water management system. Possible solutions that have been discussed at the “Mining for Closure” workshop for this site included the application of a clay cap on the surface area, water diversion (i.e. channels that divert surface runoff that would infiltrate to the adit) and interception of the mine water at an earlier point in the adit. This latter strategy would reduce the contact of water with reactive material and lower its contaminant concentration. However, the strategies to reduce flows and reduce pollution intensity are “projects in themselves” and are not further addressed here. Nor could they be addressed in detail in the workshop.

During the workshop, the necessity to take these approaches was clearly recognised – but in order to move forward with a demonstration water treatment design example, it was assumed by the experts present at the workshop that a nominal 30% of the mine water flow and contaminant concentration can be reduced by afore mention measures at the site. As such, the flow rate utilised for this hypothetical exercise be calculated 10 L/s (36 m³/h) with an acidity of 800 mg/l.
10.2.3.3 Design

There are two sizing approaches that can be applied in order to identify the required land and amount of material for a RAPS. The first one is to calculate the size based on the required retention time.

The limestone gravel layer beneath the compost is typically made 0.5 – 1.0 m thick, using single size gravel of 25 – 50 mm diameter (Watzlaf et al., 2003; Younger et al., 2002). It is usual practice to size the RAPS such that this limestone gravel layer has a nominal retention time (= total pore volume / design flow rate) in excess of 14 hours, and to then fit adjust the area and thickness of the layer to achieve this total pore volume. It is recommendable to calculate with a retention time of 24 h in order ensure sufficient contact time between the reactive material and the mine water throughout the life time of the system.

Commonly, endtipped, single-size limestone gravel displays a porosity of 50%, so that the excavation required to providing a 24 hour retention time will be double the calculated volume of limestone. Thus, for a mine water with a flow-rate of 36 m$^3$/h as at the Rosia Montana site the volume of limestone required would be 1728 m$^3$ (36m$^3$ per hour * 24hours/0.5 = 1728 m$^3$). The surface area of the RAPS would be about 2160 m$^2$ (calculated with a 0.8 m gravel layer), which results in 1080 m$^3$ compost that would have to be incorporated into the system (calculated with a 0.5 m compost layer).

An alternative to calculating the dimension of a RAPS based upon retention time, is to base the design on the “area adjusted removal rate”. This value is indicated by the PIRAMID Consortium (2003) to be calculated with 25 – 30 g/m$^2$/d which is a conservative guideline figure. Fabian et al. (2006) and Skousen et al. (2005 ) showed that real life applications often achieve higher removal rates. The latter reviewed of the performance 16 RAPS throughout the U.S that had an average area-adjusted removal rate of 87.5 g/m$^2$/d. Nevertheless this exercise will follow the recommendations of the PIRAMID Guidelines to maintain conservative practice. The result looks slightly different from the one achieved via retention time. At a flow rate of 10 L/s and acidity of 800 mg/l as CaCO$_3$, the area required to eliminate all acidity in a RAPS would be 23040 m$^2$ (2.3 hectares).

The two calculations yield results that are about the order of a magnitude apart from each other. This suggests that the contaminant concentration is too high to be treated with a wetland system, where reasonable retention times should be maintained. On the
other hand, it may be worth asking if it is necessary to remove all acidity from the mine water in order to reduce negative impacts on the environment. In the review of Skousen et al. (2005) the largest RAPS had a surface area of no more than 4200 m². It may be advisable remain in this size category of RAPS where experience is provided. As such, and in order to overcome the problem with high acidity, an additional treatment step may be necessary to achieve desired water quality.

One possibility would be to incorporate aeration cascades between the RAPS wetland and the sedimentation pond. Commonly, a sedimentation pond follows directly the RAPS as it is assumed the generated alkalinity in the RAPS is sufficient to raise pH to a level where extensive precipitation of metals occurs. This process can be promoted by oxidation as most of the iron will be present in as ferrous iron due to the reducing conditions in the RAPS. Instead of having one big RAPS it is may be advisable to have two or three smaller ones in a row, thereby short-circuiting is reduced and the system is more adjustable. This is also a potential solution seen as being potentially viable for this exercise. Because of the high acidity, it can be assumed that pH continues to be low after the sedimentation pond also for the hydrolysis which releases protons into solution. By entering into a second RAPS, the water becomes finally neutralized. If necessary, the second sedimentation pond may also be preceded by an aeration cascade.

The last step in the treatment system can be an aerobic wetland which is at this stage able to remove still remaining pollutants as the water should be net-alkaline at this stage and acidification is not a danger anymore. As there is an already existing shallow lake one the surface of the Gura Rosiei tailings dam which can be used as aerobic wetland, the installation of the treatment system on top of it seem to be most favorable option.

The system should be laid out in two parallel lines that work independent from each other as shown in the design scheme in Figure 10.4. The aim of having a double line design is to achieve adequate treatment performance even when one line is non-operational due to maintenance work or other implications. Hence, the system as described above will incorporate four RAPS, four sedimentation ponds, two aeration cascades and one aerobic wetland (as it is already preexisting). The size of aerobic wetland is calculated with an area-adjusted-removal rate for which 10 g/m²/d is recommended. Sedimentation ponds are calculated with 48 h retention time and a depth of about 3m. For each cascade the mine water passes, it can be assumed that 30 mg/L of iron can be precipitated.
As such, the size of the RAPS would be approximately 4000 m² for this exercise. This value remains between the two values calculated for the given conditions and within the range of commonly applied RAPS. Each RAPS compartment therefore measures 1000 m² followed by an aeration cascade and a sedimentation pond of which each has a size of at least 288 m² according to afore mentioned design criteria. The size of the wetland depends on the incoming iron concentration and can at this stage only be estimated as treatment efficiency of the preceding steps may vary. For this exercise, it is assumed that 100 mg/L of iron is still dissolved in the mine water. In this case a wetland to completely remove remaining iron would measure 8640 m².

In this case, the already existing lake on the tailings dam should serve as aerobic wetland to reduce remaining pollutants. As this is more a polishing step, it is not crucial to divide it into two separate units as a stoppage of the aerobic wetland would have no major consequences on the mine water quality. The size of the lake is not provided. In case it is not sufficient for the treatment purpose, additional area could be excavated to enlarge the wetland.
Figure 10.4 Flow scheme of potential mine water treatment system at Rosia Montana.

Aerobic wetland (in the existing tailings dam lake)
Apart from the surface water area, extra surface area is required for the slopes of the wetland installations as they are not recommended have an angle of at least 2:1. For the RAPS, with a gravel/compost layer of 1.5 m and a recommended freeboard above 1.5 m, this would add extra 1.5 m at every side of the basin, see Figure 6.4. Between and at the sides of the basins, a distance of at least 6 m should be provided so that vehicles can access the site for construction and maintenance purposes. Finally, the external slopes must be considered. Where the retaining walls are 2 m high, additional 4 m width are recommended on each side of the RAPS and the other basin for wetland and sedimentation pond.

![Diagram of a RAPS to treat mine water at Rosia Montana.](image)

**Figure 10.5** Dimensions (length) of a RAPS to treat mine water at Rosia Montana.

The net-surface area required for this treatment scheme would be about 13 800 m² only for the treatment surface area if measures are as follows: RAPS (20 m x 50 m), sedimentation pond (12 m x 24 m) and wetland (60 m x 140 m). Including the necessary installations (internal and external slope gradient, access allowance) the minimal total area to accommodate the treatment scheme would be 26100 m².

For this calculation, a rectangular design was chosen for the RAPS and sedimentation pond and wetland in order to not unnecessarily complicate the task. In reality, sharp engineered structures should rather be avoided for all passive systems presented in this thesis where possible. Natural, soft shapes fit better into the landscape and minimise the visual impact of the treatment scheme.

The land demand of the treatment scheme is quite large and even in not very populated areas, a suitable area is not easy to find. At Rosia Montana, it is recommended to install the treatment system on top Gura Rosiei tailings dam. The
surface dimensions are about 500 m x 200 m (~10 ha) with a gradient of about 6.5 %. These conditions fit very much the needs of the presented treatment concept.

The tailings dam is located in a distance of 3 km from the discharge point. The water could be conducted through a pipe to the dam which is situated downstream in the neighbouring valley of to the discharge point at 690 m elevation. Event though the tailings dam and the Adit 714 are located in two different valleys, it seems possible to conduct the water from the discharge point to the tailings dam using a siphon construction in order to make use of the hydraulic gradient and avoid active pumping. It can also be assumed that pipe work from the tailings dam operations exists and may be used for the mine water transport.

Next steps to get closer to the actual implementation of the treatment system would include an estimation of costs. For a rough calculation, the cost for successful RAPS applications in other regions, such as documented by Fabian (2006), can be applied in order get an idea of the magnitude associated to the implementation. It must be kept in mind that the cost may be varying with the local conditions given at a site and the economic situation in the region where a passive system should be implemented. If local materials and manpower are available, their cost can be expected to be significantly lower in SEE than in Western Europe. As these are the main cost factors for constructing and maintaining a passive system, it can be assumed the cheaper these goods are in a place, the lower the overall cost for implementation and maintenance of a passive system will be. However, there may be issues that can increase the cost in SEE, for example if the site is located in a remote area and materials that to be transported or even shipped to the treatment site. For example if clay is not available or applicable for the sealing of a wetland and a HDPE liner has to be used instead.

As for the preceding case study it must be mentioned that the data on which this exercise is based on, were incomplete. In particular the element concentrations for calculating acidity would have been crucial for the accuracy of this exercise. But still, for aim of this exercise (i.e. demonstrating the applicability of passive mine water treatment techniques) they served the purpose. It was shown that at the Rosia Montana site, passive mine water treatment could be a viable solution which is worth further assessment.
The two case studies showed how mine water issues can be addressed and what is required to identify potential solutions. Although innovative techniques seem to be promising for the particular situation in SEE, it became clear that there is no such thing as a panacea which will work at every site. A thorough assessment and close consideration of all site specific factors are the most important steps in implementing the optimal mine water management measures with respect to human health, environment, security and economy.
List of References


## Overview cover technologies

<table>
<thead>
<tr>
<th>Cover type</th>
<th>Primary function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil cover as oxygen diffusion barrier</td>
<td>To limit oxidation by acting as a barrier against the diffusion of atmospheric oxygen into the mine waste. The soil pore water effectively slows down the oxygen diffusion rate, compared to the diffusion rate in air. Establishment of vegetation on soil cover</td>
</tr>
<tr>
<td>Oxygen consuming barriers within a soil cover</td>
<td>To limit the transport of atmospheric oxygen through the soil cover, by promoting reactions that consume the penetrating oxygen.</td>
</tr>
<tr>
<td>Low permeability barriers within a soil cover</td>
<td>To limit the formation of leakage, by acting as a barrier against water infiltration from precipitation into the soil cover, and limit the oxygen diffusion rate by maintaining a high water content</td>
</tr>
<tr>
<td>Geochemical barriers within a soil cover</td>
<td>To provide a (bio)geochemical environment that limits metal release rates and metal mobility.</td>
</tr>
<tr>
<td>Underwater deposition, or creation of a surface water cover over the mine waste by hydrologichydraulic engineering methods</td>
<td>The water cover above the deposited mine waste reduces the amount of atmospheric oxygen that is available for oxidation reactions, by slowing down the oxygen diffusion rate considerably, compared to the diffusion rate in air.</td>
</tr>
<tr>
<td>Groundwater saturation of mine wastes</td>
<td>The groundwater table is raised until it covers the mine waste, thereby creating an oxygen diffusion barrier by the same principle as for a surface water cover, or underwater deposition.</td>
</tr>
<tr>
<td>Wetland construction upon mine wastes</td>
<td>To form an oxygen diffusion barrier, by the same principle as surface/groundwater covering and underwater deposition, with wetland plants providing further protection and/or possibly requiring less water covering, by stabilising sediments and thereby hind</td>
</tr>
</tbody>
</table>

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### Bucim mine water data

<table>
<thead>
<tr>
<th></th>
<th>Pit waters</th>
<th>Landfill drainage</th>
<th>Tailings drainage</th>
<th>Topolnica river after junction of stream 2 and 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow (l/sec)</td>
<td>21</td>
<td>3</td>
<td>3</td>
<td>3</td>
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<tr>
<td>pH</td>
<td>4.68</td>
<td>3.81</td>
<td>7.56</td>
<td>6.13</td>
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<tr>
<td>TSS</td>
<td>0.037</td>
<td>0.104</td>
<td>0.104</td>
<td>0.048</td>
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<tr>
<td>As</td>
<td>0.108</td>
<td>0.572</td>
<td>0.169</td>
<td>0.367</td>
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<tr>
<td>Ag</td>
<td>0.003</td>
<td>0.026</td>
<td>0.000</td>
<td>0.013</td>
</tr>
<tr>
<td>Ti</td>
<td></td>
<td>0.003</td>
<td>0.025</td>
<td>0.026</td>
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<tr>
<td>Al</td>
<td>25.454</td>
<td>215.730</td>
<td>0.150</td>
<td>92.079</td>
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<tr>
<td>Sr</td>
<td>1.303</td>
<td>0.890</td>
<td></td>
<td>0.624</td>
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<tr>
<td>Ca</td>
<td>439.102</td>
<td>290.154</td>
<td></td>
<td>214.138</td>
</tr>
<tr>
<td>Ba</td>
<td>0.009</td>
<td>0.019</td>
<td>0.056</td>
<td>0.037</td>
</tr>
<tr>
<td>Ni</td>
<td>0.673</td>
<td>2.247</td>
<td>0.002</td>
<td>1.146</td>
</tr>
<tr>
<td>Mn</td>
<td>47.425</td>
<td>149.874</td>
<td>0.057</td>
<td>79.363</td>
</tr>
<tr>
<td>Fe</td>
<td>0.028</td>
<td>1.495</td>
<td>0.044</td>
<td>0.324</td>
</tr>
<tr>
<td>Cr</td>
<td>0.034</td>
<td>0.075</td>
<td>0.001</td>
<td>0.034</td>
</tr>
<tr>
<td>Mg</td>
<td>233.826</td>
<td>574.796</td>
<td></td>
<td>360.866</td>
</tr>
<tr>
<td>Na</td>
<td>61.511</td>
<td>114.105</td>
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<td>131.257</td>
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<tr>
<td>V</td>
<td>0.000</td>
<td>0.053</td>
<td>0.002</td>
<td>0.031</td>
</tr>
<tr>
<td>P</td>
<td>&lt; 0.005</td>
<td>0.177</td>
<td>0.047</td>
<td>0.154</td>
</tr>
<tr>
<td>Zn</td>
<td>2.906</td>
<td>3.915</td>
<td>0.003</td>
<td>1.988</td>
</tr>
<tr>
<td><strong>Cu</strong></td>
<td><strong>25.030</strong></td>
<td><strong>434.488</strong></td>
<td><strong>0.058</strong></td>
<td><strong>205.455</strong></td>
</tr>
<tr>
<td><strong>Pb</strong></td>
<td>0.013</td>
<td>0.009</td>
<td>0.011</td>
<td>0.014</td>
</tr>
<tr>
<td><strong>Cd</strong></td>
<td>0.024</td>
<td>0.021</td>
<td>0.000</td>
<td>0.011</td>
</tr>
<tr>
<td><strong>Co</strong></td>
<td>1.118</td>
<td>4.302</td>
<td>0.004</td>
<td>2.049</td>
</tr>
<tr>
<td><strong>COD</strong></td>
<td>0.013</td>
<td>0.019</td>
<td>0.009</td>
<td>0.008</td>
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</table>
Rosia Montana Adit 714 mine water data

<table>
<thead>
<tr>
<th>Quality parameter</th>
<th>Maximum Admissible Concentration (MAC)</th>
<th>Characteristic values</th>
<th>Total number of samples exceeding MAC</th>
<th>Maximum Exceeding MAC by</th>
</tr>
</thead>
<tbody>
<tr>
<td>Max</td>
<td>Min</td>
<td>Total number</td>
<td>Percentage</td>
<td></td>
</tr>
<tr>
<td>pH units</td>
<td>6.5-9.5</td>
<td>3.03</td>
<td>2.68</td>
<td>7</td>
</tr>
<tr>
<td>As total</td>
<td>μg/L</td>
<td>1852</td>
<td>65</td>
<td></td>
</tr>
<tr>
<td>As diss.</td>
<td>μg/L</td>
<td>100</td>
<td>1738</td>
<td>49.4</td>
</tr>
<tr>
<td>Cd total</td>
<td>μg/L</td>
<td>875</td>
<td>116</td>
<td></td>
</tr>
<tr>
<td>Cd diss.</td>
<td>μg/L</td>
<td>200</td>
<td>814</td>
<td>97.4</td>
</tr>
<tr>
<td>Fe diss</td>
<td>mg/L</td>
<td>720,151</td>
<td>105.9</td>
<td></td>
</tr>
<tr>
<td>Fe total</td>
<td>mg/L</td>
<td>5</td>
<td>1324</td>
<td>234.25</td>
</tr>
<tr>
<td>Ni total</td>
<td>μg/L</td>
<td>1132</td>
<td>507</td>
<td></td>
</tr>
<tr>
<td>Ni diss.</td>
<td>μg/L</td>
<td>500</td>
<td>732</td>
<td>483</td>
</tr>
<tr>
<td>Pb total</td>
<td>μg/L</td>
<td>266</td>
<td>52</td>
<td></td>
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<tr>
<td>Pb diss.</td>
<td>μg/L</td>
<td>200</td>
<td>246</td>
<td>3.21</td>
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<tr>
<td>Hg</td>
<td>μg/L</td>
<td>50</td>
<td>0.15</td>
<td>0</td>
</tr>
<tr>
<td>Cr total</td>
<td>μg/L</td>
<td>1000</td>
<td>2710</td>
<td>52</td>
</tr>
<tr>
<td>Se</td>
<td>μg/L</td>
<td>100</td>
<td>217</td>
<td>98.8</td>
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<tr>
<td>SO4</td>
<td>mg/L</td>
<td>600</td>
<td>2637.9</td>
<td>1736.42</td>
</tr>
<tr>
<td>HCO3</td>
<td>mg/L</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

*As in the Normative NTPA - 001/2002, which is the Romanian legal act related to the respective EU Directive and specifies the maximum limits of pollutants into the wastewaters evacuated into the water resources.
APPENDIX B WORKSHOP INFORMATION

Agenda
Mining for Closure:
Innovations for contaminated mine waters assessment, management and remediation

Technical workshop on innovative techniques and technologies for contaminated mine waters assessment, management and remediation

March 26 – 29, 2007
Brestovacka Spa, Bor, Serbia

Draft Agenda

DAY 0, Sunday, March 25

18:30 – 19:00 Registration in Belgrade’s Hotel “Metropol”

19:00 Dinner with the Canadian Ambassador

DAY 1, Monday, March 26

08:00 – 11:00 Travel from Belgrade to Bor - Brestovacka Spa

12:00 – 13:00 Welcome Buffet Lunch

13:00 – 13:30 Welcome and opening

- Representative of the Ministry of Science and Environment protection of the Republic of Serbia
- Ms. Sandra Wibmer, the Austrian Development Agency (ADA), Vienna
13:30 – 15:00 Introduction to Contaminated Mine Water and Treatment in SEE

- **Key Issues on Mining and Environment in SEE**
  Dr. Philip Peck - Extractive Industries Specialist, UNEP GRID Arendal; Associate Professor, International Institute for Industrial Environmental Economics (IIIEE) at Lund University, Sweden

- **Mine Water and Innovative Treatment (1)**
  Dr. Gilles Tremblay - Program Manager, Natural Resources Canada (NRCan), Canada
  Dr. Nand Dave - Senior Scientist, Natural Resources Canada (NRCan), Canada

- **Mine Water and Innovative Treatment (2)**
  Dr. Adam Jarvis - Environment Agency Research Fellow, Hydrogeochemical Engineering Research & Outreach (HERO), Institute for Research on Environment & Sustainability at University of Newcastle upon Tyne, UK

15:30 – 17:30 Mines and Environmental Problems

- General Mine & environment challenges: Regional Examples
  Serbia, Bor - Dr. Dejan Milenic (University of Belgrade, Faculty for Mining and Dept. of Hydrology and Geology)
  Macedonia, Buchim – Dr. Dejan Mirakovski (Faculty of Mining and Geology, University “St Cyril and Methodius”, Macedonia)
  Montenegro, Mojkovac - Goran Sekulic (Ministry of Environment and Physical Planning of Montenegro) – tbc
  Kosovo, Trepca – Denika Blacklock (UNDP Kosovo)
  Bosnia and Herzegovina – Zvjezdan Karadzin (Mining Institute Tuzla)

**DAY 2, Tuesday, March 27**

09:00 – 09:45 Delineation and management of Mining Sites

- Risk Assessment and Investigation of Contaminated Land- Chris McCormick (Freelance Environmental Geochemist Consultant)
09:45 – 12:30 International policy works and legal frameworks on Mining

- Policy works and partnerships from Canada/North America – Dr. Gilles Tremblay/Dr. Nand Dave (NRCan)
- Policy works and partnerships from Europe; introduction to Mine Water Legislation in Europe – Dr. Adam Jarvis (HERO)
- Governance principles for Foreign Direct Investments in Hazardous Activities; Prevention of transboundary environmental conflicts: the case of Timok river – Mr. Stephen Stec, The Regional Environmental Center for CEE, Hungary

13:30 - 17:30 Site visits to Bor

DAY 3, Wednesday, March 28

09:00 – 10:30 Innovative Mine Water management and Remediation

- Mine water management cases from the UK - Dr. Adam Jarvis (HERO)
- Mine water management cases from North America – Dr. Gilles Tremblay/Dr. Nand Dave (NRCan)
- Mine water management cases from Western Europe and application of Passive Mine Water Treatment Systems – Dr. Gunther Pieplow (Head of Water Treatment Department, WISUTEC Environmental Technologies, Germany)

11:00 – 12:30 Data Management and Regional Case Studies

- Presentation of Regional Case Studies
  - Introduction of Buchim – Dr. Dejan Mirakovski (Faculty of Mining and Geology, University “St Cyril and Methodius”, Macedonia)
  - Introduction of ”Site X at Bor” – Dr. Dejan Milenic (University of Belgrade)
  - Introduction of Rosia Montana – Philip Peck (UNEP/IIIEE)

14:00 – 15:30 Breakout Sessions

- Sketch solutions for Buchim
- Sketch solutions for one polluted stream in Romania
16:00 – 17:00 Presentation and Discussion of the Sketch Solutions
17:30 Summary of the workshop, conclusions and follow up

DAY 4, Thursday, March 29

09:00 – 12:30 Site Visits to Majdanpek

14:00 Return to Belgrade