

Water Management at Abandoned Flooded Underground Mines

Fundamentals – Tracer Tests – Modelling – Water Treatment

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We all seek an ideal in life.

Charlotte Brontë

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1 Preface

Before I started to write this publication, I was thinking which areas of the mine water field might be interesting for you, the reader, and which areas have not already been covered in detail. One of those subjects was “mine abandonment and mine water management”. Many articles have been written, and excellent books covering mine water issues are available, so this publication focuses on some of the following gaps in the existing literature:

- why and how to close a mine
- the hydrodynamics of a flooded mine
- tracer techniques for flooded mines
- stratification patterns in the mine water body

Since my first contact with mine water issues in 1991, I realised that scientists and consultants from many different backgrounds are working in this field of science: e.g. mining engineers, geologists, biologists, chemists, civil engineers, environmentalists, lawyers, and regulators. For some of them, mine water issues are their daily job; others work on a project for a short time period and then never come back to mining in their career or work. Most of the books I read about mine water issues – and back when I first started to think about this publication, there were not many books available – are intended for either audience.

Therefore, there are very detailed parts of this book, which address issues that are not well covered in the mine water literature, and other parts that simply provide an overview for the novice in this field. So, some facts are covered in detail (e.g. tracer tests or mine water stratification), others are discussed briefly (e.g. treatment techniques, surface mining) and others are just missing because covering them in detail would have been beyond the scope of this publication (e.g. salt mining, all types of solution mining, small-scale mining). Most of the book assumes that the reader is a practitioner with a basic knowledge of natural science.

But why should you care about mine water stratification and mine water tracer tests?

In the 1960s Stuart and Simpson (1961) showed that water in a flooded underground mine might stratify (Fig. 98). Many researchers since then have mistakenly thought that stratification (which is the layering of water with different physico-chemical properties) could be used to extract unpolluted mine water from a flooded mine; although many flooded and stratified mines exist and have been made public, no comprehensive synopsis of the density stratification problem has been published so far. This publication will fill this gap by describing the main physical laws involved, categorising the types of stratification, and describing several case studies. Furthermore, large parts of this book are based on my research about mine water tracing and mine water stratification and are embedded in the general issues needed to understand why tracer tests are necessary and how stratification can be explained.

When looking through the references you might notice that many of them have been written in a language with which you may not be very familiar: German. This might be unusual for an English publication, but it arose from two simple but decisive factors:

- In recent years, the Technische Universität Bergakademie Freiberg has organised several mine water related symposia and conferences. During the organization of such conferences, it became clear that most German-speaking colleagues from industry and public authorities mainly depend on publications written in their own language. This is probably why many aspects of passive treatment were reinvented in Germany without detailed studies of all the English publications available (an vice versa). Therefore, the list of references contains German as well as English publications, giving the German speaking colleagues several references to publications in their own language.
- On the other hand, if you are a reader with English as your mother tongue, you will have the opportunity to learn about research done in Germany that you might find useful.

2 Acknowledgements

Many colleagues have helped and assisted me in my research. Some accompanied my way for just a short time; others have been with me since I started to work in the mine water field, and again others were (and are) my teachers and friends. Yet, the list must start with four colleagues and friends who, mainly but not exclusively from a scientific point of view, made me what I am today:

Miran Veselič from Ljubljana/Slovenia was the first person to help me understand what happens when a mine is flooded. I owe him many ideas in many areas of science. Bob Kleinmann from Pittsburgh/USA (United States of America), then, was my mentor in what has to be known when one works in the mine water field. His knowledge helped to solve many of the problems I had. Next comes Paul Younger from Newcastle upon Tyne/United Kingdom. It is hard to copy him, nearly impossible, but it is worth trying because the way he thinks and works stimulates me to try harder all the time. Finally, Broder Merkel from Freiberg/Germany gave me the freedom to conduct research in the areas I deemed to be necessary for solving the difficulties related to mine water tracers and stratification. His “Why-do-we-need-that?”-questions were more than once the key to developing better procedures.

Well, this first part was easy while the second part is virtually impossible. So many people have helped me and there is no way of ranking them in an appropriate way. It would bother me very much if you looked for your name here and found it missing when you expected it to be here. Therefore, I thank all of you who contributed to my research and my growth as a scientist: thanks for whatever you did for me. Your ideas were important to me! Without your help – and even if it was only a reference list or a copy of a hard to find article – this work would not have been possible! Furthermore, I appreciate very much the assistance of all my colleagues and students at the Technische Universität Clausthal, the Ingenieurbüro für Geotechnik/Bautzen, and the Technische Universität Bergakademie Freiberg – which is a chronological ranking!

Yet, there are two person to whom I owe much of the fun, strength, knowledge and freedom that I had during the years since I started this work: my wife Ulrike and my friend Jana. They never lost her love and belief in me and my work!

3 Introduction

3.1 Mining Issues

“Everything we use is either grown or mined”, as miners say, and therefore, abandoned, closed, or orphaned mine sites cover approximately 240,000 km² of the earth’s surface and can be a hazard to both humans and the environment (Fig. 1). Countless papers about such sites and their remediation have appeared, most of them focused on a single mine site, and on acid mine/rock drainage/metalliferous leachates. This book attempts to provide a more comprehensive presentation of the water management that is necessary at such sites. It focuses on general aspects of flooded underground mines and aims to improve the understanding of the processes involved in underground mine flooding and mine water treatment. Furthermore, it explains some of the hydrodynamic aspects of flooded underground mines, which could then be used to plan new *in-situ* and on-site treatment options. To reach that goal, the following chapters will:

- summarise the necessary measures for the management of abandoned and flooded underground mines
- explain the fundamental hydrodynamics within flooded underground mines, and
- describing the reactive transport mechanisms within the flooded voids

While the first part is mainly a compilation of previous investigations of mine water research the second part consists of observations in flooded underground mines and my own, innovative tracer techniques for flooded underground mines. Since the 1990s the number of mine water related articles has burgeoned and a complete outline of all techniques and investigations is hardly possible. A database retrieval in the four major databases relevant for mine water (GeoRef, Geobase, FIZ-GEO, SCI Expanded) listed more than 10,000 entries with the key words ‘(mine or mining) and water’ (Wolkersdorfer 2004). If you think, your work or somebody else’s work is missing in this publication, please just let me know and it will be considered for inclusion in future editions.

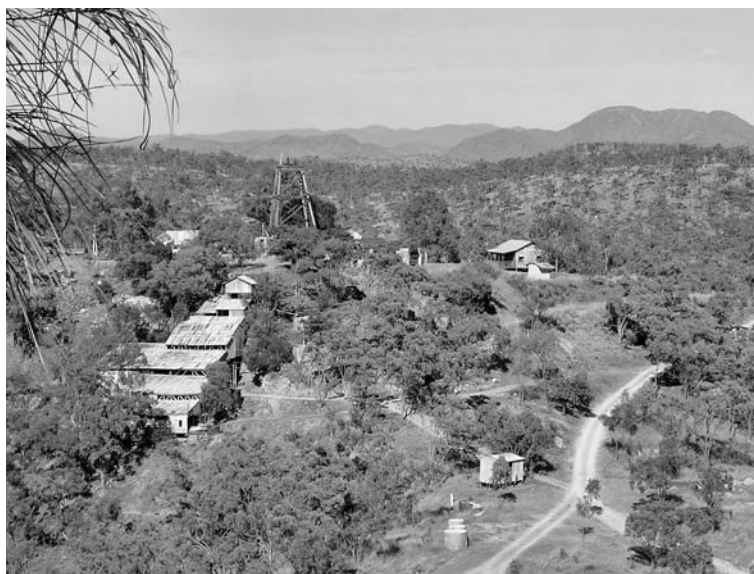


Fig. 1: Abandoned Tyrconel/Australia gold mine site. The headframe stands about the flooded main shaft.

To illustrate the techniques and the behaviour of water in flooded mines several case studies will be presented and all the details that are necessary to transfer the outcomes of those investigations to other mines sites will be described. Although several thousand mines were closed in recent decades, few were described in the literature; therefore I have included results of both published and unpublished mine closures that are relevant to water management.

Hámar et al. (2002) concluded that “illegal and orphan mines pose the highest potential risk to the environment over the long term”, whereas “orphaned” according to their definition means a closed, not remediated mine which has no owner. Similar, if not more pessimistic – but challenging – views have been given by Balkau (1999), who stated that “among the outstanding environmental problems confronting the mining industry, that of abandoned mine sites has been particularly slow to be tackled.” Such publications categorise “unused” mines as either ‘closed’, ‘abandoned’, or ‘orphaned’. Though such a nomenclature might be helpful on a legal basis, all such sites have one thing in common: mining is no longer taking place and somebody has to take care of the legacy. The geochemical and hydrodynamic processes do not care about such definitions and therefore no special focus will be given in this book to those terms. Wherever they appear in this book they can be considered as synonyms, as there is no generally accepted worldwide definition of these words, as was observed during a European Commission Workshop about “Mine and Quarry Waste – The Burden from the Past” (Puura et al. 2003).

In the USA, several programmes for abandoned mine land and for polluted mining-related superfund sites were installed on national and regional levels. The U.S. Department of the Interior’s “Office of Surface Mining” also hosts the “Abandoned Mine Land Reclamation” programme, which is financed through an “Abandoned Mine Land Fund” (\$1.44 Billion U.S. in 2003). Another programme is hosted by the “Bureau of Land Management” within the same department. Most of those activities are concentrated in the “National Association of Abandoned Mine Land Programs” and in the “National Coalition for Abandoned Mine Reclamation”. Although those programmes are collecting data about abandoned mine land and potential hazards from such sites, they do not carry out own research on mine flooding or acid mine drainage (Office of Surface Mining 2002). A different programme is the U.S. Geological Survey’s “Abandoned Mine Lands Initiative”, where also research is carried out (US Geological Survey 2000).

Canada runs the “Orphaned/Abandoned Mines Initiative” with the “National Orphaned/Abandoned Mines Advisory Committee” coordinating various initiatives (Castrilli 2002); and in Europe PADRE (Partnership for Acid Drainage Remediation in Europe) focuses on acid mine remediation works in Europe (Wolkersdorfer et al. 2004).

To date, no reliable statistics about the total number of abandoned mine sites exists, though their total worldwide number might go into the millions (e.g. International Institute for Environment and Development 2002). In the USA the Abandoned Mine Lands Inventory System (AMLIS) lists more than 10,000 abandoned mine sites that have been investigated, of which only 5% might pose a significant threat to watersheds or the environment. Copa et al. (1995) estimated that only 30,000 or 6% of the estimated 500,000 mining sites in the USA might cause potential risks to the public or the environment. Nevertheless, the States of Pennsylvania and West Virginia identified acid mine drainage as the № 1 threat to water quality (Diz 1998). Wireman et al. (2001) reported on tens of thousands of abandoned mines in the Rocky Mountains, and Benner et al. (1997) summarise several investigations in their introduction and come to similar conclusions. Furthermore, ten U.S. States reported that between 8,000 and 16,000 km of streams were polluted by mine drainage (U.S. Environmental Protection Agency et al. 2001). During the European Union PECOMINES project, the mine inventory of the candidate countries at that time (Bulgaria, Czech Republic, Estonia, Hungary, Latvia,

Lithuania, Poland, Romania, Slovakia, Slovenia) identified several thousand potentially dangerous mine sites – in just ten European countries (Jordan et al. 2004)! Also the “Abandoned Mines Working Paper” (International Institute for Environment and Development 2002) refers to only 5 countries (Australia, Canada, United Kingdom, United States, South Africa) though the numbers of four “other countries” (Ireland, Sweden, Japan, Chile) are listed. In addition, what is missing in many of these statistics is the scope of pollution emanating from small-scale mining, though some reports try to evaluate the problem (Wrobel et al. 2000; Hentschel et al. 2002). Zamora (1999) summarised some of the problems of small-scale mining and concluded that

it cannot be assumed that small-scale miners are knowledgeable about environmental technologies or that they can adopt international standards or codes of conduct. They do not generally have access to financing, so cannot invest in new technologies and systems. They cannot be treated like any other miners.

Though every mine site is special in itself and nearly every mine is – to borrow a mathematical term – a singularity, this work wants to classify abandoned underground mines concerning their hydrogeochemical behaviour. The question is whether, to a certain degree, the hydrogeochemical behaviour of an abandoned mine can be predicted and if that classification can be used to plan remediation strategies. Yet, as in weather forecasting, too many non-linear equations determine the hydrogeochemical and the hydrodynamic development of a flooded mine so that (as has often been shown) small changes in conditions can result in big differences at the end. There is a similar unpredictability on the atomic scale: the decay of a radioactive element. Though the lifetime of a single atom cannot be predicted, a huge quantity of them can be described with a characteristic constant: their half-life. The question, therefore, is whether flooded underground mines have similarities that can be used to estimate their behaviour after they have been flooded, and which can then be used to set up an environmentally sound mine water management plan.

When reading the following chapters, you must bear in mind that elevated metal concentrations may occur in both acid and neutral drainage. Although commonly referred to as “acid mine/rock drainage” or AMD/ARD, it is more accurate to refer to the phenomenon as “acid mine/rock drainage/metal leaching: AMD/ML or ARD/ML”. Therefore, whenever the term acid mine drainage is used in the following sections, it must be kept in mind that this includes elevated concentrations of metals as well. In this work, the term ‘metals’ is commonly used for both semi-metals and metals, whereas the term ‘heavy metals’ is avoided whenever possible, because several of the so-called heavy metals are neither heavy nor are they metals in our common understanding (Duffus 2002). Yet, if the term ‘metal’ is used for all elements that show cationic behaviour in aqueous solution or that have an acid soluble oxide (as defined by Parish 1977), then the term becomes clearer, because then also Al and Sb are included as metals. Only arsenic, being a metalloid, does not fall in the this category, so if one wants to be exact, one should write “metals and arsenic” instead of “heavy metals”.

Some readers might be bothered by the fact that standard hydrogeological methods are not described. A thorough hydrogeological and hydrological understanding is essential for all mine water researches. However, how can the already existing books about geological practises be beaten by a book focused on mine water hydrogeology for flooded underground mines? It would be beyond of this book to repeat what has already been written about good hydrogeological practices. Therefore, the interested reader – and obviously you belong to this group – is referred to the already existing literature (e.g. Freeze and Cherry 1979; Fetter 1988; Jordan and Weder 1995; Brassington 1999; Domenico and Schwartz 1998; Deming 2002; Langguth and Voigt 2004; Hiscock 2005; Hölting and Coldewey 2005).

Not all the literature cited in this publication is readily available in all libraries or can be bought through a bookshop. This publication covers many of the issues connected with mine water management and mine closure to give the interested researcher or consultant an insight into special cases as well as general rules. Publications concerning the latter can usually be found in most well equipped libraries. All the others should be accessible through the internet or through national or international book exchange. Also persistent digital object identifiers have been avoided since at least two competing systems are available; which of those will survive the next decade is still unclear. As web sites are changing quickly the reader will be referred to the excellent Search Engines or library OPACs available on the internet if he or she wants to find more information about a single issue.

3.2 How to read this book

Depending on your personal background, you might find some parts of this book more interesting than others. First, I will briefly introduce you to each of the chapter and then I will let you know which chapters might be of interest to you.

The overall idea was to describe the processes related to mine abandonment from a hydrogeologic perspective, following the path that a single drop of water might take through an abandoned mine. First, it flows through the fractures, fissures and bedding planes into the mine where it comes in contact with distinctive minerals. This process is described in chapter 4, “Hydrogeochemistry of Mine Water”, which also describes some attenuation and precipitation processes that might happen. Then we will look into the necessary measures that lead to the “enclosure” of that water drop in the abandoned mine; chapter 5: “Insights into Mine Closure” describes why a mine is closed and what is necessary for an environmentally safe mine abandonment. A short chapter, which attempts to sketch some important “Do’s and Don’ts of Mine Closure”, follows. This chapter mainly resulted from hearing statements like “the next time we close a mine, we should ...”. It is a summary of recommendations that did not fit into the other chapters but that I thought would be useful for people not directly involved in the closure process. In the following chapter, the water drop meets all the other water drops that are in the mine. They start to move around and therefore chapter 7, “Hydrodynamics”, describes the governing laws that force those water drops to flow through our abandoned mine workings. To understand how the water drops flow through the mine, how they interact with the rocks and each other, and how their behaviour could be predicted is the subject of chapter 8, “Mine Closure Flow Models”. Because all our models are simplifications of reality, we now have to prove if our water drop behaves as we predicted that it would. Chapter 9, “Monitoring and Sampling”, summarises the procedures that are necessary for sampling mine water. Because mine water is such a unique liquid, the pitfalls of mine water sampling and monitoring are given special consideration in this chapter. Because sampling and monitoring does not explain how, in which direction, and at which speed our water drop flows through the mine we have to trace its way by conducting tracer tests. A comprehensive description of the relevant procedures for conducting mine water tracer tests is given in chapter 10, “Tracer Techniques for mines”. Then, after minutes or decades in the dark and in contact with the rocks and minerals, our water drop leaves the mine. Perhaps it has been lucky and has a ‘clean record’; otherwise, chapter 11, “Mine Water Treatment and Ground Water Protection”, comes into play. Finally, in chapter 12, thirteen well-investigated case studies are described, and the interaction of the material discussed in the previous chapters is integrated. Because stratification could possibly be seen as a key instrument for *in-situ* treatment of mine water those case studies take up significant space in this publication.

After the references, you will find three appendices. I am not fond of such annotations, but as in an interesting letter, the post-script adds that extra something. Appendix I describes a simple procedure for checking the accuracy of water analyses. By comparing the calculated elec-

trical conductivity against the measured one (hopefully you calibrated or checked your conductivity meter!), you will get a better reliability than by the usual cation/anion balance (eqs. 62 and 63). Appendix II addresses flow measurements. While putting together equations for flow measurements I realised that the books I read used different equations, even for a simple mechanical structure like a V-notch weir. I went into many of the available books and publications about flow measurements and noticed that flow measurement is far more complicated than most of us think. Because flow measurement is so essential for our work as mine water researchers, this highly physical and mathematical appendix had to be added for you – culminating in simple and reliable equations that you can readily use. Finally, the last appendix summarises the Berlin I and II Guidelines that have been drawn up to guarantee environmentally sound mining and mine closure as well as sustainable mining and post-mining sites.

Well, after that short outline, where should you start to read or what should you read? Each chapter assumes that you are coming from a different background. I have balanced the chapters as I did based on the question that I received during 13 years of lectures and conference presentations. However, all of the chapters assume that you have at least a basic knowledge of chemistry, hydrogeology, mining issues, mathematics, and physics.

If you are new in the field of mine abandonment and would like to read about some general ideas, start with chapter 4 and proceed to chapters 5, 9 and 11. From the case studies, choose sections 12.3, 12.4, 12.6, and 12.10.

Should you wish to understand the hydrodynamics of an abandoned and flooded mine, or if you are planning to conduct a tracer test, start with chapter 7 and then read chapters 9 and 10. Out of the case studies you might find 12.4, 12.6, 12. 7, and 12.10 helpful.

If you are going to close a mine, you will want to read chapters 5, 6, 8, 9, and 11. Choose sections 12.4, 12.7, 12.11, and 12.12 if you want to get some ideas of the processes that can occur during mine flooding.

As a regulator or a lawyer, you might find parts of chapter 4 useful, as well as sections 5.4, 5.9, 6, 9.1, 9.3, 10.1, and 11. The case studies are probably more technical than you require but you might like to read appendix III.

4 Hydrogeochemistry of Mine Water

4.1 Introduction

Problems with mine drainage predominantly arise when ground and surface waters enter mine sites and come into contact with primary and secondary minerals under oxic conditions. During contact with those minerals, the water may dissolve components of the ore and rock-forming minerals. Most of the processes involved are controlled by the E_H -pH conditions at the water-mineral contact. If even relatively small amounts of disulphides, as pyrite or marcasite, oxidise and dissolve, the acidity formed can dissolve other water contaminants. Because metal or coal deposits are so diverse, the metals and the composition of mine water are unique at every mine site. Commonly, carbonates minerals neutralize (buffer) the acid leading to neutral or alkaline mine water, which might still be harmful to surface and ground waters in a catchment area. Barnes and Clarke (1964) nicely summarised the problem associated with disulphide oxidation: “if mine workings could have been isolated hydrologically, operators, who must pump as much as 23 t of water per ton of coal from the anthracite mines of Pennsylvania, could have saved a great amount of money”. That was in 1964 and even today, after nearly all of the questions discussed by Barnes and Clark are resolved, we are still far away from being able to definitively stop acid mine drainage and metal leaching. These problems were, in fact, described by Agricola (1556), and clearly existed long before that.

Though this chapter focuses on metal and semi-metal pollution of the mining environment, there are obviously other pollutants that might negatively affect watersheds. One of the most threatening is high salinity, mainly connected with salt mines but also with deep coal mines as in Germany and Poland and with mines cutting across mineral water aquifers. In addition, also ammonium, nitrate, organics, hydrocarbons (from machinery used underground), PCBs, radium, and barium are contaminants that can sometimes be found at mine sites (e.g. Bench 2000).

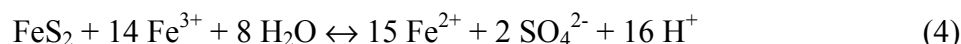
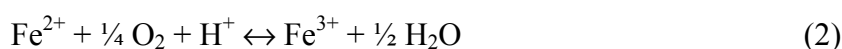
Besides the above-mentioned processes, microbial activity has to be considered. As the outstanding work of Baas Becking et al. (1960) shows, there is nearly no area in our environment which is not inhabited by some kind of bacteria (Fig. 5). Interestingly, when Baas Becking et al. (1960) wrote their work about the E_H -pH conditions of waters and the environment in which microorganisms live, they found “only two published readings on mine waters” but “fortunately, unpublished information increases the total number of readings to fifty-six”. Furthermore, by the time they wrote their paper, they realised that “oxidation of pyrite is a chemical system which has been neglected by geochemists” though “investigations by us have shown quite conclusively that the first stages of oxidation are entirely inorganic”. And even more interestingly, this work is hardly ever cited by mine water researchers, though it contains interesting data about the environments in which bacteria prefer to do their job! Luckily, the last 50 years saw a considerable change in that field of research, though the present fashion to describe the reactions of pyrite oxidation detailed in *every* small article about mine water issues should be resisted – a reference to Sigg and Stumm (1994), Stumm and Morgan (1996), or Younger et al. (2002) provides an adequate overview. Most people – in the meantime – know how the process is working. The following section has been written for those who are interested in the governing reactions. Others can directly skip ahead to section 4.3. If you do not like chemical formulae at all, you can go directly to chapter 5 – yet you will miss understanding all the processes that make up our interesting and often colourful work (Fig. 2)! A potential classification of mine waters is given in section 9.3.3.



Fig. 2: Iron- and manganese oxihydroxide as well as other metal precipitates as stalactites in the abandoned Holzappel/Germany mine. Width of picture: 50 cm.

4.2 The Process Starts: Abiotic Disulphide Weathering

It is commonly accepted that disulphide weathering and especially the oxidation of pyrite and marcasite are the initial reactions in the formation of acid mine drainage/metalliferous leachate (Fig. 3):



As can easily be seen, the overall reaction produces protons, and thus releases acid into the mine water (Singer and Stumm 1970). For that reason, pyrite weathering is the strongest acid-producing process of all oxidation processes known to occur in the natural environment (Stumm and Morgan 1996). If no buffering minerals are present, the pH of evaporating mine water can reach extremely low pH-values, sometimes as low as -3.6 (Nordstrom and Alpers 1995; Stumm and Morgan 1996; Nordstrom et al. 2000). As the reaction is exothermal, the mine air and mine water can reach unusually high temperatures (air temperatures up to 40°C were recorded in the Iron Mountain mine, D. Kirk Nordstrom, pers. comm.; Nordstrom 2004c). Consequently, owing to buoyancy effects, the warm oxygen depleted air will move into the upper parts of a mine, creating a ‘chimney effect’: fresh, cold and oxygen-rich air is drawn into the mine through the portals, fostering additional disulphide oxidation.

Due to its prominent role in many reactions that are important in the formation and attenuation of mine water, the pH is often called the “master variable” (Stumm and Morgan 1996):

$$\text{pH} = -\log_{10}\{\text{H}^+\} \quad (5)$$

For those not familiar with the conventions in chemistry, numbers in $[]$ are concentrations c and numbers in $\{\}$ are activities a . They are connected by the activity coefficient f according to the following relation:

$$\{a\} = f \times [c] \quad (6)$$

All chemical-thermodynamic codes use this relation for calculating the species distribution in the analysed water (Stumm and Morgan 1996; Nordstrom 2004c).

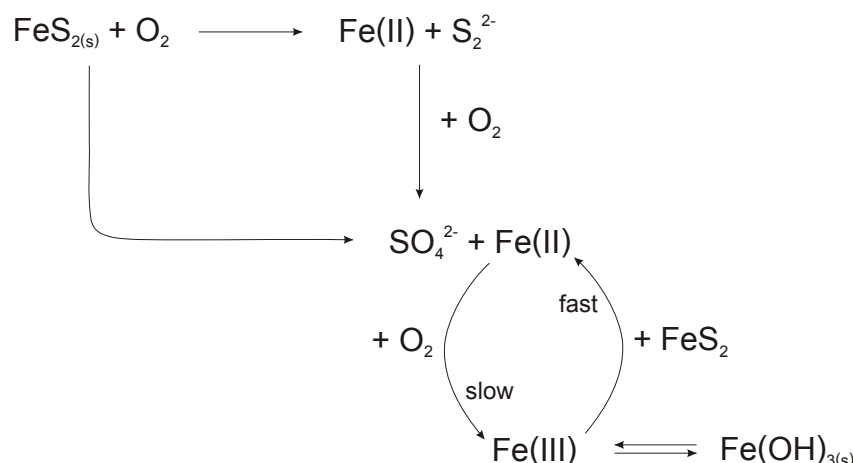


Fig. 3: Schematic representation of the pathways during abiotic pyrite or marcasite oxidation (modified after Kester et al. 1975 in Stumm and Morgan 1996; Singer and Stumm 1970).

Reactions 1 and 2 are catalysed by bacterial activity (see chapter 4.4). Reaction 3 is abiotic and often does not occur until the water has left the mine. The above reactions have a net proton production and therefore can cause the pH to decrease rapidly. As soon as the pH reaches about 4 ferric iron becomes the predominant oxidant and reaction 4 becomes the governing equation to oxidise pyrite (Singer and Stumm 1970). Though this might imply that pyrite oxidation can occur without oxygen involvement, oxygen is the initial oxidant either directly for pyrite or indirectly to generate ferric iron by reaction 2. Pyrite does not oxidise in anoxic environments unless previously generated ferric ion is present in the solution. The four reactions also prove that pyrite oxidation occurs as long as enough of the reactants are available as has nicely been shown by Nordstrom (2004b) for the Iron Mountain superfund site, where there is enough pyrite for another 3,200 years of acid generation!

The above disulphide weathering reactions require pyrite or marcasite, oxygen, and water. Because those three reactants can be found in nearly every surface and underground mine and because all it takes to keep the process going is 1–2% pyrite in the rocks exposed to the mine air, this process occurs worldwide. A large mineral surface area is preferable for the decomposition of the minerals: the smaller the crystals, the faster the process and the quicker the water gets acidified. Generally, if the pyrite crystals can be seen with the bare eye, the problem will be less intense than in cases where the same amount of pyrite is dispersed as sub-micron sized grains throughout the rock matrix. Of course, higher concentrations of pyrite make the problem worse. However, as long as the ground water is percolating through undisturbed rock, through open pore space, or natural fractures without human interference, and as long as the ground water is in a reducing state, no significant pyrite oxidation occurs. Therefore, acid rock drainage rarely occurs in undisturbed natural environments except in places such as the Austrian Rötzbach, where the first natural occurrence of schwertmannite was observed (section 4.6, Schwertmann et al. 1995; Rude 2004), or the Colorado/USA South Fork/Lake Creek near Red Mountain in the Grizzly Peak Caldera (Neubert 2000). The opposite is also true: if only one of the reaction components is missing, the reactions will stop and no acidity will be produced. That is the basis for most *in-situ* or on-site remediation procedures (section 11.6).

Even “dry” mines can contain a large amount of water that can oxidise pyrite. Such “dry” mines are dry because all the water that enters the mine through the rock matrix evaporates as

it contacts the mine air. Brewitz et al. (1985) measured the amount of this evaporating water in the Konrad/Germany mine and compared it with results from the Stripa/Sweden experiments. While the Konrad mine gained a volumetric evaporation rate of $0.016 \text{ m}^3 \text{ min}^{-1} \text{ m}^{-2}$, it was $0.076 \text{ m}^3 \text{ min}^{-1} \text{ m}^{-2}$ at Stripa (cited after Brewitz et al. 1985).

Mining significantly changes the natural ground water conditions and as the ground water table falls and large underground openings are created, oxygen enters a formerly reducing environment. While the pre-mining conditions may have supported the formation of pyrite from iron-enriched ground water, the process now is reversed and pyrite will be oxidised. This process occurs throughout the dewatered area but especially in the proximity of the mine workings, where the rocks are fragmented because of blasting and excavating. Disaggregation zones ('excavation disturbed zones') generally extend about 2 m around the mine workings, but can extend up to 8 m (Müller-Salzburg 1978; Jacobi and Everling 1981; Brewitz et al. 1985; Militzer et al. 1986; Stoll and Bauer 1991; Roschlau 1994; Wolkersdorfer 1996). Blasting, up to now, has generally not been seen as a factor that affects disulphide weathering. However, as destressing blasting protects the rock and thus minimises blasting induced fractures in the rock, it can reduce the formation of acid mine drainage/metalliferous leachate production in underground workings.

Kinetics and consequently mineral weathering rates are important in understanding the evolution of different mine waters. Different authors report different mineral weathering rates but a discussion of those important investigations is beyond the scope of this publication. A thorough discussion and summary of those investigations, which were conducted under various atmospheric conditions, has been given in Salmon and Malmström (2002), Younger et al. (2002), and Höglund et al. (2004).

4.3 The Process goes on: How the Metals come into Solution

Because pyrite is always found with other minerals (paragenesis), predominantly other sulphides, but also oxides, vanadates, arsenates, phosphates, and carbonates, mine water is commonly enriched in many elements. Their concentrations mainly depend on the mine water pH, with maximum concentrations typically found at lower pH values (Tab. 2). Those minerals may weather in a congruent, incongruent, or redox process (Stumm and Morgan 1996). Congruent weathering processes are reversible processes, such as the dissolution of gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$). Incongruent weathering processes, on the other hand, are irreversible because one of the products either vaporises, precipitates, or forms other, stable species in the mine water. Redox weathering reactions, similar to the incongruent weathering reactions, are irreversible under normal atmospheric conditions. When mine workings are eventually inundated with water (see section 5.4), several of these processes, including the dissolution of secondary mineral precipitates, again play an important role and eventually create the 'first flush' (section 5.4.1).

Tab. 1: Selection of weathering processes that occur in mines (modified after Stumm and Morgan 1996). $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}_s$: gypsum; $\text{K}[\text{AlSi}_3\text{O}_8]$: orthoclase; $\text{KAl}_2[(\text{OH})_2\text{AlSi}_3\text{O}_{10}]$: muscovite; FeS_{2s} : pyrite.

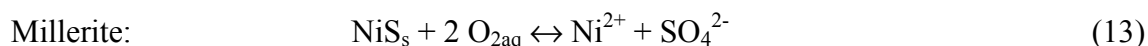
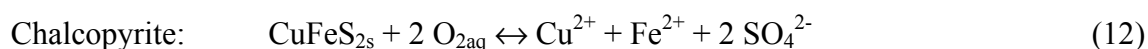
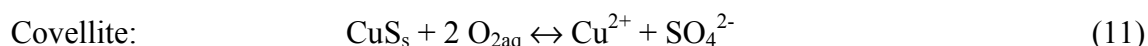
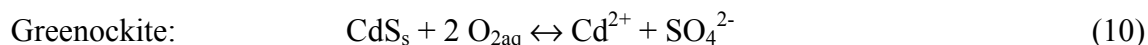
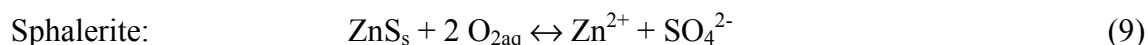
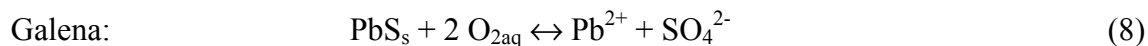
Type	Reaction
Congruent	$\text{CaSO}_4 \cdot 2\text{H}_2\text{O}_s + \text{H}_2\text{O} \leftrightarrow \text{Ca}^{2+} + \text{SO}_4^{2-} + 3 \text{H}_2\text{O}$
Incongruent	$3 \text{K}[\text{AlSi}_3\text{O}_8]_s + 2 \text{H}_2\text{CO}_3 + 12 \text{H}_2\text{O} \leftrightarrow 2 \text{K}^+ + 2 \text{HCO}_3^- + 6 \text{H}_4[\text{SiO}_4] + \text{KAl}_2[(\text{OH})_2\text{AlSi}_3\text{O}_{10}]_{(s)}$
Redox	$\text{FeS}_{2s} + \frac{7}{2} \text{O}_2 + \text{H}_2\text{O} \leftrightarrow \text{Fe}^{2+} + 2 \text{SO}_4^{2-} + 2 \text{H}^+$

Like pyrite, most of those minerals are stable at the E_H -pH conditions and the oxygen content of the ground water before mining. When oxygen enters the ore veins or the rock matrix the

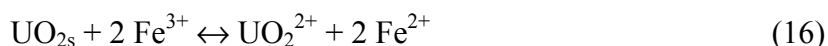
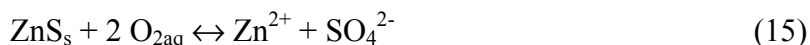
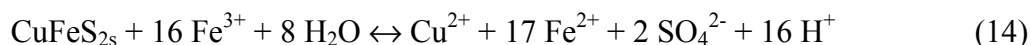
stable conditions change and those mineral phases start to oxidise to more mobile species, which are then transported with the water to the point of the lower hydraulic pressure, which is usually the open space of the underground mine or the mine sump (Me in equation 7 stands for any metal):



Because this looks rather theoretical, here are some examples of the oxidation of primary minerals commonly found in underground mines:



The second process is induced by disulphide oxidation and the lowering of the pH. Furthermore, the release of Fe^{3+} is a process induced by disulphide oxidation (Nordstrom 1977; Tuovinen and Bhatti 1999):



Gottschalk and Buehler (1912) were amongst the first authors to systematically investigate the interaction of sulphide minerals and pyrite or marcasite. They realised that non-iron metal concentrations in the solution increase in the presence of pyrite or marcasite. Thought they were able to satisfactorily explain their findings since, they were not aware that microorganisms were partially responsible for their findings, and that those microorganisms more easily ‘nibble’ on pyrite or marcasite than on other sulphides. Nevertheless, their work shows clearly that the release of metals and semi-metals from other sulphides increases if pyrite or marcasite are abundant in the system.

Viewed the other way: if mine water has elevated concentrations of Fe, Zn, Cu, Pb, and Cd, it can be assumed that pyrite, sphalerite (Cd always substitutes Zn in sphalerite), galena, and chalcopyrite are relevant minerals in the ore deposit from which it emanates. It is thus likely that it is flowing out of a lead-zinc-mine. Another possibility to find those mineral species is by chemical-thermodynamic codes. They all include the relevant stability constants for the formation and dissolution of the mineral species and are based on an iterative computation that calculates which mineral phases is, or was, in equilibrium with the water.

Arsenic, for example, is a very mobile metalloid commonly found in polluted mine water and has provoked considerable interest in recent years (e.g. Merkel and Sperling 1998; Williams 2001; Zhu et al. 2003). It is mainly released from arsenopyrite (FeAsS), in a reaction similar to equation 1 but also from other common As-minerals such as enargite or fahlores. Many uranium mines and nearly all hydrothermal gold deposits show elevated concentrations of As in their mine waters (e.g. Straskraba and Moran 1990; Wolkersdorfer 1996; Schneider et al. 2001; Howell 2001).

Tab. 2: Composition of different mine waters with the most predominant mine water constituents. Sources: Alpers et al. (1992); Yelpatyevsky (1995) [according to a pers. comm. by Vera Pavlovna Elpatyevskaya the values in Yelpatyevsky's publication are wrongly given in mg L^{-1} instead of $\mu\text{g L}^{-1}$]; Wolkersdorfer (1996); Banks et al. (1997); Nordstrom et al. (2000); Younger (2000b); Wolkersdorfer and Hasche (2001a); Banks (2004); unpublished data from author; concentrations in mg L^{-1} .

Locality	pH	SO_4^{2-}	Fe_{tot}	Al	Mn	Zn	Cu
Iron Mountain California (copper)	0.4	108,000	18,600	2,320	–	2,060	290
Iron Mountain California (copper)	1.1	41,000	7,820	1,410	11	1,860	360
Pyrite mine (anonymous)	2.5	5,110	1,460	84	3	1	0.2
Smirnovsky tin mine (Sichote Alin)	3.2	–	132	–	14	25	5
abandoned coal mine (anonymous)	3.6	1,044	101	17	4	0.2	0.007
Ynysarwed Wales (coal)	4.2	1,554	180	< 0.5	6	0.06	–
waste rock dump (coal)	5.5	146	287	1	5	0.05	< 0.007
Gernrode Harz Mountains (fluorite)	5.9	78	22	–	4	0.14	0.016
Straßberg Harz Mountains (fluorite)	6.3	359	31	–	6	0.9	0.08
Dunston Chesterfield (coal)	6.3	210	11	< 0.05	1.3	< 0.007	–
abandoned coal mine (anonymous)	6.3	83	5	0.08	0.4	0.05	0.005
metal mine (anonymous)	6.5	124	15	0.1	2	0.003	–
Niederschlema Erzgebirge (uranium)	7.1	1,138	3	0.4	3	0.1	0.03
Frazer's Grove Durham (fluorite)	7.7	260	11	–	–	0.4	0.04
Mine № 3 Svalbard (coal)	8.2	7	< 0.01	< 0.02	0.004	0.055	< 0.005
Schwaz Austria (dolomite, fahlore)	8.4	13	< 0.01	–	0.002	0.022	0.04

4.4 The Process is Speeded up: The Role of Microorganisms

4.4.1 Introduction and Metabolism

Microorganisms play an important role in the processes of the natural and man-made environments (Nordstrom 1977; Singh and Bhatnagar 1988; Smith et al. 1988; Benedix et al. 1991; Ledin and Pedersen 1996; Salmon and Malmström 2002; Ebena 2003). Most microorganisms in mine water are bacteria, but fungi, algae, archaeobacteria and diatoms can also be found (Perrin et al. 1992; Douglas et al. 1998; Ohimain 2003; Rohwerder et al. 2003). A good synopsis of microbial activities in mine water is given by Hallberg and Johnson (2005) and Mills (1999). For those who are interested in a more detailed approach, Edwards et al. (2000), Johnson (2003), or Nordstrom (2000) are recommended. Herbert Jr. (1999) and Rawlings (2005) give an excellent review of microbial activities in mine wastes. In the following paragraphs the main interactions, necessary to understand microbial activity in mine water is given, underlain by some basic references to published work.

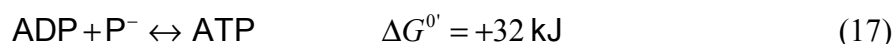
The role of microorganisms in mine water is twofold: they catalyse pyrite oxidation and help to remediate the polluted mine water. Because of such diversity, they cover nearly all ecosystems on our planet (Fig. 5), and if found in really extreme conditions, reports about such environments find their way into popular journals (e.g. Siefer 2000). Some microorganisms prefer extremely acidic conditions (Robbins et al. 2000) whereas others like to live under extremely alkaline environments (Takai et al. 2001), and are called extremophiles because they live in conditions which are considered to be extraordinarily unusual for normal organisms (Madigan et al. 2003; Hallberg and Johnson 2005). Due to their ability to catalyze the dissolution of metal sulphides, they are commonly used in bioleaching processes to extract metals from the host or mother rocks of low grade deposits or waste rock dumps (Ebner and Schwarz 1973; Bosecker 1980; Lundgren and Silver 1980; Beyer 1986; Duarte et al. 1990; Brunner et al. 1993; Ahonen and Tuovinen 1995; Gourdon and Funtowicz 1995; Modak et al. 1996; Cas-

sity and Pesic 1999; Vasan et al. 2001; Rawlings 2002; Rohwerder et al. 2003; Sand and Gerke 2006).

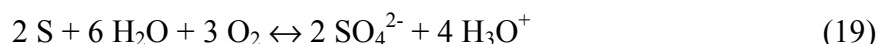
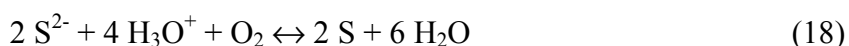
Until the year 2000, every mine water researcher was able to name the common bacteria most responsible for catalysing and accelerating the reactions that cause acid mine drainage: *Thiobacillus ferrooxidans* and *Thiobacillus thiooxidans*. Yet, in 2000 Kelly and Wood (2000) reclassified *Thiobacillus* into three new genera: *Acidithiobacillus*, *Halothiobacillus*, and *Thermithiobacillus*, based on 16S rRNA gene sequence. To distinguish the non-acidic “true” *Thiobacillus* from the acidophilic species, *Thiobacillus ferrooxidans* and *Thiobacillus thiooxidans*, which prefer pH-ranges from 0 to 5, were renamed *Acidithiobacillus ferrooxidans* and *Acidithiobacillus thiooxidans* (Hallberg and Johnson (2005), Tab. 5). The new names underline the acidic environment in which these bacteria prefer to live (all references to these bacteria in this publication were renamed using this convention, independent of the name in the original literature). In circum-neutral to alkaline conditions, *Acidithiobacilli* become less common and *Thiobacilli* take on their role, as can be seen in the Niederschlema/Alberoda uranium mine, where circum-neutral mine water dominates (Tab. 3).

Most of the bacteria responsible for the development of acid mine drainage are chemotrophic bacteria, which use organic or inorganic electron-donating sources as their energy supply. Chemolithotrophs use only inorganic chemicals (Fig. 4), and because CO₂ is the source of their cellular carbon, they can further be classified as autotrophic – as opposed to microorganisms that use organic carbon, which are called heterotrophs. Furthermore, lithoautotrophs use inorganic electron donors whereas phototrophs obtain energy from sunlight.

Whichever primary energy source the microorganisms use, they must phosphorylate ADP (adenosine diphosphate) to ATP (adenosine triphosphate), because all cell reactions depend on ATP as an energy source. The reaction process to produce ATP requires +32 kJ of energy and involves the enzymatic transfer of protons through the inner cell membrane:



For chemolithotrophs, this energy is obtained from the oxidation of sulphides, primarily pyrite and marcasite, but also from other sulphides (Tab. 4). Inside the bacteria cell, the following exothermal reactions (exemplified by *At. thiooxidans*, *Beggiatoa* and *Thiotrix*) take place:



The bacteria gain $\Delta G^0 = -498 \text{ kJ}$ of energy, which is used for producing ATP, in which the energy is stored until it is needed in endergonic metabolic processes, such as kinetic, chemical, osmotic, electrical, or thermal processes.

The biotic iron oxidation occurs at a rate which is 100 to $1 \cdot 10^6$ times faster than abiotic iron oxidation (Singer and Stumm 1970; Rackley 1976; Lundgren and Silver 1980). Though the bacteria accelerate the reactions, they cannot change the end points of reactions or thermodynamics, but certainly the natural endpoint.

Four microbiologically catalysed processes are important for mine water: two (sulphur and iron oxidation) are responsible for the formation of acid mine drainage/metalliferous leachate and two (sulphate and iron reduction) for the attenuation of the water (Mills 1999). Furthermore, the biotic activities of bacteria are responsible for many mechanisms that lead to metalliferous leachate, which is used for bioleaching (Lundgren and Silver 1980).

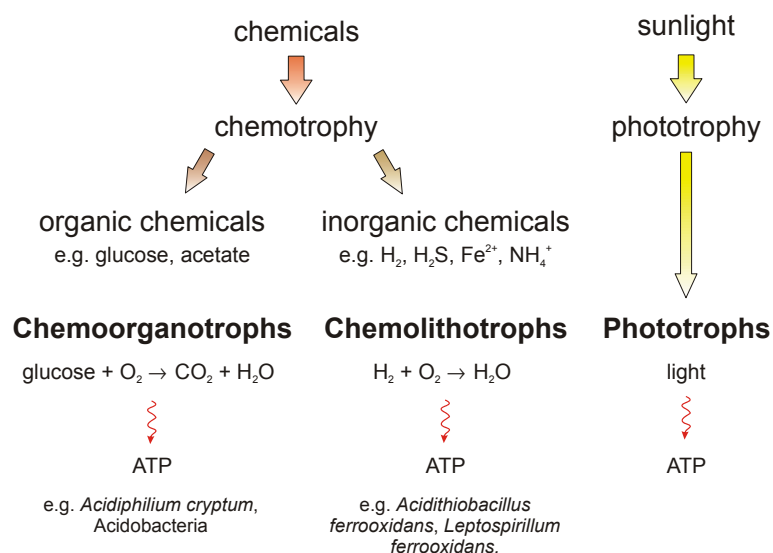


Fig. 4: Classification of some common bacteria occurring in mine water according to their metabolism. Chemical reactions just show one of the principle processes involved. Acidobacteria are a group of microbes that have yet to be named, but are closely related to *Acidobacterium capsulatum* (modified after Madigan et al. (2003) and adopted after Kevin Hallberg, pers. comm.).

Tab. 3: Results of a microbiological investigation in the Niederschlema/Alberoda/Germany uranium mine (Wolkersdorfer 1996); –: not measured, n.d.: not detectable, ^w: data measured by Wismut GmbH (microbiological analysis: Dr. M. Graff, Institut für Mikrobiologie der Technischen Universität Braunschweig 1996).

	Shaft 366 b, level -540,	Shaft 366 b, onsetting sta- tion level -720	Cross drift Q918, level -540	Surge tank, inlet, level -540
Air temperature, °C	–	13.6	22.8	16.0
Water temperature, °C	31.3 ^w	13.0	24.4	17.5
pH, –	9.45 ^w	8.10	7.33	7.73
Oxygen, mg L ⁻¹	2.5	10.0	7.2	9.5
Oxygen, % saturation	33	94	85	99
Redox potential, mV	315 ^w	345	335	365
neutr. thiobacteria (S ₀ -Ox.), L ⁻¹	4.5 · 10 ³	1.5 · 10 ³	4.5 · 10 ⁴	2.5 · 10 ⁴
acid. thiobacteria (S ₀ -Ox.), L ⁻¹	n.d.	n.d.	n.d.	n.d.
acid. iron oxidisers (Fe ²⁺ -Ox.), L ⁻¹	n.d.	0.4 · 10 ³	n.d.	0.4 · 10 ³

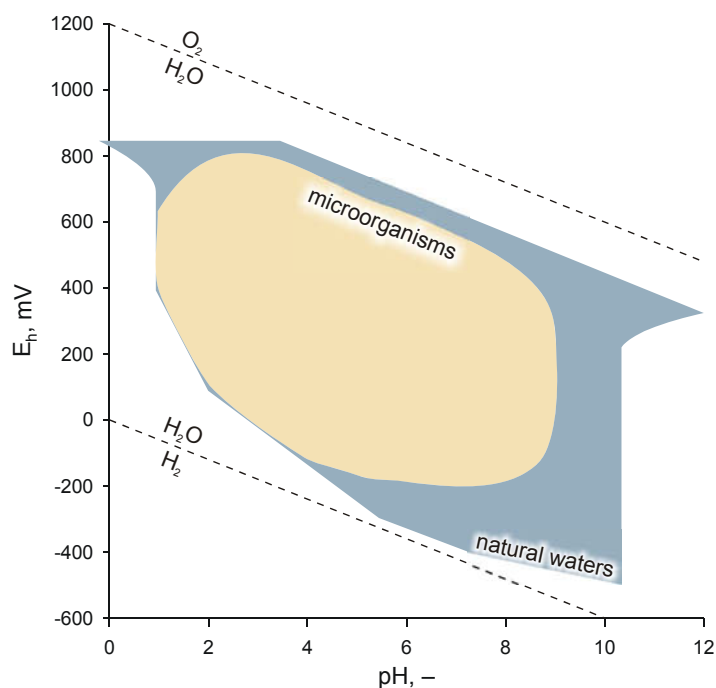


Fig. 5: Areas of natural E_h - and pH-conditions including mine waters and the preferred fields for commonly occurring microorganisms in mine waters (modified after Baas Becking et al. 1960).

Tab. 4: List of metal sulphides that were described to be oxidised by *At. ferrooxidans* or *At. thiooxidans* (Gottschalk and Buehler 1912; Nordstrom 1977; Lundgren and Silver 1980; Bosecker 1980; all chemical formulas according to Strunz and Nickel 2001).

Mineral	Chemical formula	Mineral	Chemical formula
antimonite	Sb_2S_3	galena	PbS
arsenopyrite	$FeAsS$	greenockite	CdS
orpiment	As_2S_3	marcasite	FeS_2
bismuthinite	Bi_2S_3	millerite	$\beta\text{-NiS}$
bornite	Cu_5FeS_4	molybdenite	MoS_2
chalcopyrite	$CuFeS_2$	pentlandite	$(Ni,Fe)_9S_8$
chalcocite	Cu_2S	pyrite	FeS_2
cinnabar	HgS	pyrrhotite	FeS
covellite	CuS	sphalerite	$\alpha\text{-ZnS}$
enargite	Cu_3AsS_4	tetrahedrite	$Cu_3SbS_{3.25}$

Tab. 5: Acidophilic prokaryotic microorganisms associated with mine water (from Hallberg and Johnson (2005), Kevin Hallberg, pers. comm.) Meso: mesophiles ($T_{\text{optimum}} < 40^{\circ}\text{C}$); Mod Thermo: moderate thermophiles ($T_{\text{optimum}} 40\text{--}60^{\circ}\text{C}$); Ext Thermo: extreme thermophiles ($T_{\text{optimum}} > 60^{\circ}\text{C}$); ¹: inferred ability to oxidise minerals (via production of sulphuric acid).

Mineral-degrading Acidophiles	Thermal Classification	Phylogenetic Affiliation
Iron-oxidisers		
<i>Leptospirillum ferrooxidans</i>	Meso	Nitrospira
<i>L. ferriphilum</i>	Meso	Nitrospira
<i>L. thermoferrooxidans</i>	Mod Thermo	Nitrospira
" <i>Thiobacillus ferrooxidans</i> " m-1	Meso	β -Proteobacteria
" <i>Ferrimicrobium acidiphilum</i> "	Meso	Actinobacteria
<i>Ferropasma acidiphilum</i>	Meso	Thermoplasmatales
" <i>Fp. acidarmanus</i> "	Meso	Thermoplasmatales
Sulphur-oxidisers		
<i>Acidithiobacillus thiooxidans</i>	Meso	β/γ -Proteobacteria
<i>At. caldus</i>	Mod Thermo	β/γ -Proteobacteria
<i>Thiomonas cuprina</i>	Meso	β -Proteobacteria
<i>Hydrogenobaculum acidophilum</i>	Mod Thermo	Aquificales ¹
Iron- and sulphur-oxidisers		
<i>Acidithiobacillus ferrooxidans</i>	Meso	β/γ -Proteobacteria
Iron-reducers		
<i>Acidiphilium spp.</i>	Meso	α -Proteobacteria
Iron-oxidisers/reducers		
<i>Acidimicrobium ferrooxidans</i>	Meso	Actinobacteria
Iron-oxidisers/reducers and sulphur-oxidisers		
<i>Sulfobacillus spp.</i>	Meso and Mod Thermo	Firmicutes
Heterotrophic acidophiles (non mineral-degrading)		
<i>Acidocella spp.</i>	Meso	α -Proteobacteria
<i>Acidisphaera rubrifiaciens</i>	Meso	α -Proteobacteria
<i>Acidobacterium capsulatum</i>	Meso	Acidobacterium
<i>Acidomonas methanolica</i>	Meso	α -Proteobacteria
<i>Alicyclobacillus spp.</i>	Meso	Firmicutes
<i>Picrophilus spp.</i>	Mod Thermo	Thermoplasmatales
<i>Thermoplasma spp.</i>	Mod Thermo	Thermoplasmatales

4.4.2 Mechanisms of Organism–Mineral Interaction

As discussed earlier, the bacteria require sulphur (*At. thiooxidans*, *At. ferrooxidans*) or iron (*At. ferrooxidans*, *Leptospirillum ferrooxidans*) as well as oxygen to produce energy and to maintain their metabolism. They oxidise the disulphides pyrite and marcasite (FeS_2) as well as other sulphide minerals (e.g. PbS , ZnS , CdS) under acidic conditions. Though the bacteria catalyse the chemical reactions, they are not able to initiate chemical reactions that would be thermodynamically impossible (Stumm and Morgan 1996). Other mechanisms catalyzed by bacteria are sulphate and iron reduction, especially in passive treatment systems. So far, not much is known about microorganisms in basic mine water environments (Hallberg and Johnson 2005). However, Takai et al. (2001) describe the bacteria *Alkaliphilus transvaalensis* from a borehole in an ultra-deep gold mine near Carletonville/South Africa. This bacteria growth over a temperature range of $20\text{--}50^{\circ}\text{C}$ and pH-range of $8.5\text{--}12.5$. Furthermore, Perry et al. (2001) described *Gallionella ferruginea* in neutral mine water discharges.

During the past decades there was an ongoing discussion how the microorganisms catalyze the metal-sulphide oxidation: by direct or indirect mechanisms (e.g. Bennett and Tributsch

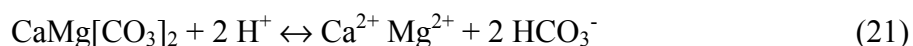
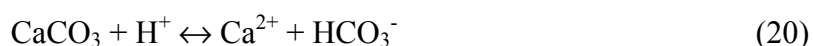
1978; Nordstrom and Alpers 1999). It was found that some bacteria are attached to the mineral surface and others are not and therefore, it was concluded that there must be a biotic and an abiotic process involved. In the meantime it seems to be clear that only abiotic processes start the reaction chain (Sand et al. 1995; Rawlings 2005). Depending on the type of mineral two mechanisms are relevant: iron disulphides release ferrous iron because of the oxidation through ferric iron or oxygen whereas the other metal sulphides are attacked by the acid in the mine water and release the metal and sulphur into solution.

Where direct contact between the mineral and the bacteria has been observed it seems as if the cell surface is important in this reaction process. This is indicated by the fact that bacteria with mineral contact have other cell surfaces than those without bacteria-solid contacts (Sand et al. 1995 Herbert Jr. 1999) After that process, *At. thiooxidans* or *At. ferrooxidans* oxidise the sulphur to sulphate and *At. ferrooxidans* ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}). Those two processes also occur without a bacteria-solid contact and inhibit a concentration gradient from the solid into the liquid which even releases more iron and sulphur from the pyrite or marcasite crystal. Finally, the bacteria use the ions in solution to maintain their metabolic processes according to equations 17 to 19 and, as *Leptospirillum ferrooxidans*, through the oxidation of ferrous iron to ferric iron.

4.5 The Process Slows Down: Buffering of the Acidity

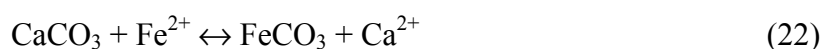
Besides the minerals and micro-organisms responsible for acid production, there are also minerals that have the ability to consume the protons produced during the above-described processes (eqs. 1—4). These minerals are usually present in the gangue or in the host rock of the raw material deposit. From a chemical point of view, these minerals are proton-acceptors; they buffer the acidity and prevent the mine water from becoming too acidic or acidic at all. Buffering reactions control the pH as well as the level of acidity in the mine water. Since the pH, in turn, controls the bioavailability and mobility of most metals, buffering reactions are fundamental for keeping the pH of mine water in an acceptable range.

Carbonate minerals, such as calcite, dolomite, or siderite, are well known to buffer mine water pH, but oxi-hydrate and silicate minerals also buffer pH when carbonates are not present or have been consumed (Tab. 6; Blowes et al. 1994; Jurjovec et al. 1995). The consumption of acidity by carbonate minerals can be seen in equations 20 and 21:



Other mechanisms that consume acidity are ion-exchange and surface protonation, but they are comparably slow and have limited capacity compared to the mineral dissolution reactions. It must be clear that the buffering minerals do not prevent pyrite oxidation (reactions 1—4). However, if the buffering occurs in the same environment as the pyrite oxidation, virtually all of the Fe^{3+} is precipitated as a hydroxide before it can dissolve more pyrite, which greatly decreases the rate of pyrite oxidation. Alternatively, the buffering can occur down-gradient of the pyrite; this can produce circum-neutral mine water that is enriched in sulphate in the same way as acid mine water (Tab. 2). Usually, the buffering ranges can be attributed to specific minerals and the mechanism of that process is referred to as sequential pH buffering. Those processes were first observed in mine tailings investigations (Blowes et al. 1994), but the pH-ranges of most acid mine waters prove that the same processes are occurring within most mines and in the receiving streams.

Calcite can also react with the Fe^{2+} produced during the disulphide oxidation forming secondary siderite (Blowes et al. 1994):



Tab. 6: Minerals that buffer protons produced during disulphide weathering (Blowes et al. 1994, Geller et al. 2000b, Goethite, K-Jarosite, and aluminosilicates from Jurjovec et al. 1995). Buffering ranges of K-Jarosite, and aluminosilicates deduced from column experiments. The values are possibly too low as reaction kinetics have not been considered (Paul Younger, pers. comm.).

Mineral	Formula	pH-Buffer Range
calcite	CaCO_3	6.5 — 7.5
dolomite	$\text{CaMg}[\text{CO}_3]_2$	6.5 — 7.5
siderite	FeCO_3	4.8 — 6.3
mixed carbonates	$(\text{Ca}, \text{Mg}, \text{Fe}, \text{Mn})\text{CO}_3$	4.8 — 6.3
gibbsite	$\text{Al}(\text{OH})_3$	4.0 — 4.3
ferrihydrate	$\text{Fe}(\text{OH})_3$	< 3.5
goethite	$\alpha\text{-FeOOH}$	1.3 — 1.8
K-jarosite	$\text{KFe}_3[(\text{OH})_6(\text{SO}_4)_2]$	1 — 2 (experimental)
aluminosilicates		1 — 2 (experimental)

Once calcite and dolomite are consumed, the pH rapidly drops to be buffered by the next occurring mineral phase in the system (Fig. 6). If all the buffering minerals are consumed, the pH drops below 1 and can reach the extreme pH-values reported in section 4.2. Another factor in buffering the acidity are the different kinetic processes involved. Carbonate weathering rates, compared to those of pyrite, biotite, feldspar, or chlorite, are comparatively high. Therefore, as long as carbonates are present, the acidity is quickly buffered. After the carbonates are consumed, the silicates buffer at a lower pH-range (Salmon and Malmström 2002).

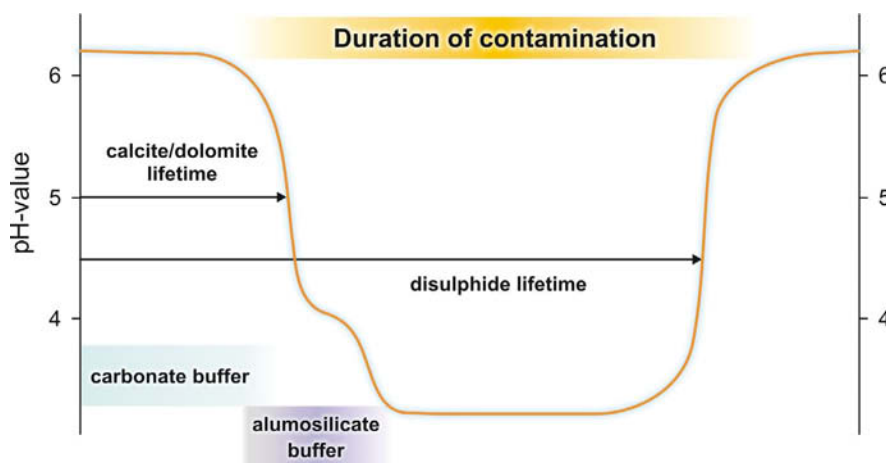


Fig. 6: Potential development of pH of a mine water in a mine with more pyrite than carbonates. The second rise in pH might be the result of different buffering processes and the fact that acidity is not formed any longer (modified after Younger et al. 2002).

In acidic mining lakes, the buffering mechanisms might be slightly different, as the ions in the lake water act as reaction partners for the protons. A low pH buffering range lies between pH 2.6 and 2.9, controlled by hydrogen ions and hydrogen sulphate. A second one ranges from pH 2.9 to 4.3 and is controlled by iron sulphate. Aluminium sulphate will buffer in the next step, from pH 4.3 to 5.5. Finally, surface exchange of sulphate and basic cations will buffer above pH 5.5 (Geller et al. 2000b; Uhlmann et al. 2004).

4.6 The Process Ends: Mineral Precipitates in the Mining Environment

One of the predominant features of mine water and one of the reasons why dealing with mine water problems also has an aesthetic aspect are the precipitates associated with acid mine water/metalliferous leachate (Fig. 2, Fig. 8). Precipitation of mineral species or amorphous substances occurs when the relevant solubility products of the species in solution are exceeded (Alpers et al. 1994). Such a situation can arise under several conditions:

- the water evaporates
- reduced species are oxidised
- the pH changes
- the redox potential changes
- flocculation or coagulation occurs
- two waters of different chemical compositions are mixed
- CO₂ is stripped off the water

Most of the secondary minerals in underground mines are iron minerals, as iron is usually the predominant cation in the water around a mine. This can be seen in the list of common secondary minerals in the mining environment (Tab. 7).

Two different groups of precipitates can be distinguished from each other: the minerals that are usually referred to as “ochre” and the water soluble minerals forming within the open space of a mine called secondary minerals (Fig. 7, Fig. 8). Most authors discuss them in separate chapters but chemically and physically, the processes forming them are similar and in both cases they are formed by mine water. Therefore, they are described together here. Also, when I write: “The process ends” this refers to a single ion’s perspective, as it is dissolved, transported and then precipitated. Because many of the secondary minerals can easily be dissolved when the water table in an abandoned mine rises, they could then restart their travel through geological space and time.



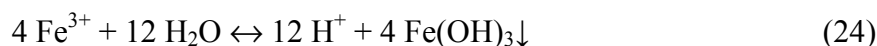
Fig. 7: Ochre precipitates in a mine water discharge pipe of the abandoned Straßberg/Harz fluorspar mine.

Secondary minerals can also be found on mine heaps and tailings where they can be used as a means of natural attenuation when they form low-permeable incrustations (e.g. Rammlmair and Grisseemann 2000; Jung 2002; Regenspurg et al. 2005). They will not be discussed further

here, because they are beyond the scope of this publication. Yet, as has nicely been shown by the Quaking Houses/County Durham incident, destroying that protective cover at mine heaps can lead to local mine water problems that needs much effort to be stopped (Kemp and Griffiths 1999). Therefore, old incrustations at mine heaps should better not be touched during remediation works as that might initiate mine water problems that did not exist before (Stefanie Hurst, pers. comm.).

The most common precipitates associated with mine water are called “ochre”, “yellow boy” or iron oxihydroxides, because they are a mixture of crystallised and amorphous substances with a yellowish to red colour. They can exist as soft gelatinous flocculants, as suspended colloids (as long as the water velocity is high enough), as a medium soft mud, as foam on the water’s surface, as coatings on rocks, and as hard, thick encrustations. Though, ironically, those minerals sometimes cause interesting and attractive features in abandoned underground mines or remote areas, the same minerals cause an ugly staining of surface streams or wetlands. Furthermore, the staining covers microorganisms, cause their die back and consequently macroorganisms leave those parts of the streams. Thus those streams or rivers are often biologically dead for several hundreds of meters and kilometres. It has to be kept in mind that “biologically dead” is only true for the usual macroorganisms that can be found in surface streams with a good ecological status. As discussed earlier, the biodiversity in mine water affected environments significantly decreases. Yet, other micro- and macroorganisms take over the niche and can be very efficient in such extreme environments, as investigations verified in heavily polluted river catchments (Ariza 1998; Batty 2005).

“Ochre” mainly consists of the following mineral species: goethite (yellowish brown), lepidocrocite (orange), ferrihydrite (reddish brown), schwertmannite (yellow), and jarosite (straw yellow) and, sometimes, also Mn minerals. All of those species are commonly found in acidic environments where iron is abundant as well. They form when Fe^{2+} in the solution is oxidised to Fe^{3+} and finally precipitates as a colloid or a mineral:



the sum of those two reactions is:



As can be seen from equation 25, this process releases a total of 2 protons per mole of iron (which is the reason, why acidic waters being “treated” in aerobic wetlands often decrease in pH). Those protons, if not buffered by proton accepting processes, lower the pH until ferric ion no longer precipitates as a hydroxide at a pH of around 2.2 (Stumm and Morgan 1996). Predominance diagrams (also called Speciation diagrams, E_{H} -pH-diagrams, Pourbaix-Diagrams), as shown in Stumm and Morgan (1996) or Appelo and Postma (1994) nicely show the relationship between the iron species and the E_{H} -pH conditions in natural environments. In Merkel and Planer-Friedrich (2002) the calculation of such diagrams with PHREEQC is explained and a small software tool for simple diagrammatic representations is provided. More sophisticated software tools for constructing predominance diagrams are FactSage (Thermfact/CRCT, Montreal/Canada; GTT-Technologies, Aachen/Germany) and ThermoCalc (Thermo-Calc Software, Stockholm/Sweden).

Rüde (2004) observed that during the course of the Rötzbach/Austria, while pH and sulphate concentrations of the water decreased, first jarosite, then schwertmannite, and finally goethite formed encrustations in the stream. Similar conditions can be found in many mining-impacted streams. It appears that the formation of these minerals is controlled by the pH and sulphate concentrations. As the ochre-minerals precipitate, other metals and semi-metals co-precipitate. Again, Rüde (2004) showed that in the Rötzbach, the concentrations of As, Ni,

and Pb in the precipitates increased while their concentrations in the water decreased to less than the detection limits after approximately 1,200 m. Exactly the same effects can be observed for mine waters, where co-precipitation of Al, Cr, Co, Cu, Pb, Mn, Ni, REE, Sc, U, Y, and Zn occurs (Bigham 1994; Fuge et al. 1994; Lottermoser 2003). Stumm and Morgan (1996) and Appelo and Postma (1994) give the following degree of affinities for co-precipitation with Fe- and Mn-oxides:

Fe-oxides: $\text{As(III)} - \text{Pb} \geq \text{Cu} > \text{Zn} > \text{Ni} \geq \text{Cd} > \text{Co} > \text{As(V)}$

Mn-oxides: $\text{As(III)} - \text{Pb} - \text{Co} > \text{Cu} > \text{Zn} > \text{Ni} > \text{Cd} > \text{As(V)}$

Co-precipitation is an important mechanism in the natural attenuation of post-mining landscapes and in mine water treatment. It results from pH and metal dependant surface complexation reactions (Dzombak and Morel 1990) and the zero point of charge (pH_{ZPC}). This occurs at pH 7.8 for FeOOH, and at 7.2 for $\beta\text{-MnO}_2$ (Stumm and Morgan 1996). Many metals (e.g. Pb, Cu, Mo, U, Sb, Mn, Se) and As co-precipitate with colloids formed by iron or manganese oxihydroxides as well as minerals of the alunite-jarosite-group. As, Sb, and Se tend to precipitate at acidic pH while the other elements precipitate at alkaline pH (Alpers et al. 1994; SENES Consultants Limited 1994; Skousen et al. 1998; Webster et al. 1998; Rothenhöfer et al. 2000; Ball and Brix 2000; Noubactep et al. 2002; Tutu et al. 2003; Merkel 2004).

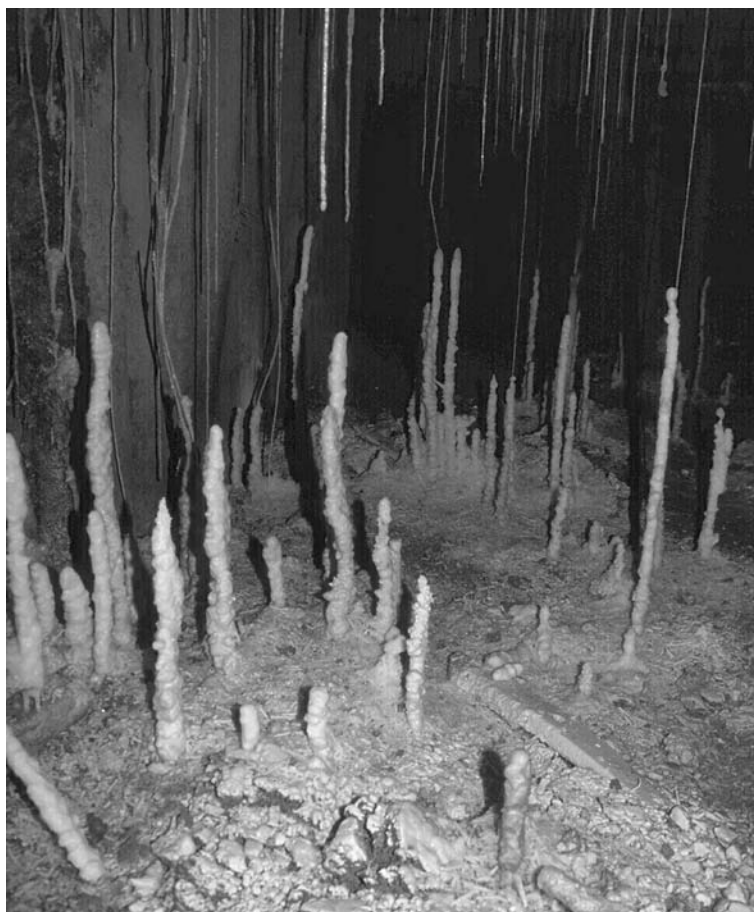


Fig. 8: Epsomite ($\text{Mg}[\text{SO}_4] \cdot 7\text{H}_2\text{O}$) stalactites and stalagmites in the Idrija/Slovenia mercury mine (the stalagmites are about 1 m high). Secondary minerals, such as epsomite, are dissolved during the flooding of the mine and – especially during the first flush – contribute to the acidity and metal content of the mine water.

Yelpatyevsky (1995) quantified this effect in an acidic “mining red stream leaking from [an] adit at [a] cassiterite-sulphide deposit” with a pH of 3.2 in south-east Russia (the Smirnovsky

mine in the Rudnaya river basin, Vera Pavlovna Elpatyevskaya, pers. comm.). He observed the following removal factors after a river length of 300 m: As 78%, Pb 45%, Fe 38%, Cd 27%, Zn 10%, Cu 10%, Mn 1.2% (initial element concentrations are given in Tab. 2). In the Italian Libiola Fe-Cu mining area, Dinelli and Tateo (2002) observed that after the precipitation of the Fe-mineral-associations, Al-, and finally Cu-phases formed in the stream.

Tab. 7: Some of the secondary minerals commonly found in the mine water environment and responsible for the first flush and their mineralogical classification; a more comprehensive review of such minerals is given in Alpers et al. (1994) and Nordstrom and Alpers (1999).

Mineral	Chemical formula	Colour
class V Carbonates		
malachite	$\text{Cu}_2[(\text{OH})_2 \text{CO}_3]$	green
cerussite	PbCO_3	white
hydrozincite	$\text{Zn}_5[(\text{OH})_3 \text{CO}_3]_2$	white
class VI Sulphates		
jurbanite	$\text{Al}[\text{OH} \text{SO}_4]\cdot 5\text{H}_2\text{O}$	white
basalumnite	$\text{Al}_4[(\text{OH})_{10} \text{SO}_4]\cdot 5\text{H}_2\text{O}$	white
melanterite	$\text{Fe}[\text{SO}_4]\cdot 7\text{H}_2\text{O}$	blue-green
szomolnokite (schmöllnitzite)	$\text{Fe}[\text{SO}_4]\cdot \text{H}_2\text{O}$	white, green
coquimbite	$\text{Fe}_2[\text{SO}_4]_3\cdot 9\text{H}_2\text{O}$	purple, white
alunite	$\text{KAl}_3[(\text{OH})_6 (\text{SO}_4)_2]$	white, orange
jarosite	$\text{KFe}_3[(\text{OH})_6 (\text{SO}_4)_2]$	orange, red
epsomite	$\text{Mg}[\text{SO}_4]\cdot 7\text{H}_2\text{O}$	white, orange
anglesite	$\text{Pb}[\text{SO}_4]$	white
goslarite	$\text{Zn}[\text{SO}_4]\cdot 7\text{H}_2\text{O}$	white

Besides the minerals that form ochre, many other mineral phases are formed when water enters a mine and oxidises primary minerals within the ore veins or the host rock. Most, but not all, of those minerals are soluble in water: cerussite, malachite, jarosite, goslarite, melanterite, szomolnokite, epsomite, alunite, jurbanite, coquimbite, basalumnite, and anglesite. Other minerals precipitating from mine water when being oxidised are gibbsite, and Zn-hydroxides. They can commonly be found on the adit and shaft surfaces because of the oxygen and the humidity within the mine workings or where the mine water drains into the receiving stream. Williams et al. (2002) showed that the pH of the water can be estimated using the colour and spectral reflectance of those minerals. The general process forming those minerals is that mining lowers the ground water table, allowing oxygen and water to contact the primary or ore minerals. Those minerals are oxidised or dissolved. Chemical-thermodynamic codes such as PHREEQC can be used to calculate in which form or complex the ions are transported. Then the water transports the ions in the direction of the lowest hydraulic pressure, usually towards the mine pumps or the open mine workings. There, the water collects in the sump, spillways, drainage gullies, or puddles within the adits. If the water is flowing slowly through the rock matrix or small fractures, the mine air is able to evaporate the mineralised water and carbonate, hydroxycarbonate, sulphate, hydroxisulphate, or oxihydroxides minerals form on the surfaces of the mine workings. Because the metal ions forming those minerals are proton acceptors, they store the protons and consequently the acidity. As long as those minerals are in contact to the mine air and the overall situation does not change significantly, this acidity will be stored in the minerals *ad infinitum*. Yet, after mining ceases, the mine is abandoned and inundated (if below the ground water level), and the water soluble carbonates, hydroxycarbonates, sulphates, and hydroxisulphates are dissolved by the mine water. Thus, the mine water, which commonly is already acidic, gets enriched in the ions constituting the minerals (section 4.3); this causes what is known as the first flush (section 5.4).

4.7 Predicting the Post-Mining Chemical Composition

Predicting the post-mining chemical composition of the mine water is an integral part of nearly all mine closure procedures. The procedures commonly used are geochemical modelling, empirical conclusions (see chapter 8), acid-base-accounting (ABA), calculation of net neutralisation potential (NNP), or static and kinetic leach tests. Since this book is about mine water management, the prediction of the chemical development of mine waters will not be covered in detail. Some good descriptions and references to other methods of how to predict post-mining water chemistry can be found in Henton (1976), Sobek et al. (1978), MEND [Mine Environment Neutral Drainage Program] (1991), Werner et al. (1999), Perry (2000), Prediction Workgroup of the Acid Drainage Technology Initiative (2000), Younger (2000d), Younger and Robins (2002), or Kruse and Younger (2005). Yet, Younger and Sapsford (2006) discusses several of the shortfalls of the before mentioned techniques and concludes that a conceptual model and site specific investigations are more important than the before mentioned tests (see section 4.5).

The differences in mineral weathering rates are a potential pitfall for researchers and consultants and can lead to incorrect predictions of future pH-values. Many mine water investigations are based on column or batch experiments and last several days, weeks, or months (the duration of a master's or diploma thesis). Seldom do such investigations last more than a year. In the initial phase of those experiments, the carbonates quickly dissolve, buffer the acidity produced by disulphide weathering in the column, so that the leachate has a pH between 6.5 and 7.5. Some researchers therefore tend to conclude that the final mine water discharge will be circum-neutral and negative environmental impacts should not be expected. If only those researchers had employed a PhD-student who would have had *two years* to conduct his or her experiments! They would then see the total depletion of the carbonates and watch the pH drop to acidic conditions. They would conclude that the final pH of the mine water would be acidic – which is true for the significant time period, though eventually the disulphides will be consumed and the pH will rise to circum-neutral conditions again. This example illustrates that all mine water investigations must include site investigations, geochemical modelling, acid-base accounting and, if possible, any other method mentioned in Prediction Workgroup of the Acid Drainage Technology Initiative (2000) and The Pennsylvanian Department of Environmental Protection (1998). Younger and Sapsford (2006) conclude after critically reviewing the relevant literature that “it emerged that there has been an excessive concentration on simple pollution potential assay tests (acid-base accounting, humidity cell tests) at the expense of rational assessments of contaminant transport pathways”. Furthermore, it must be kept in mind, that most column tests are not standardised (U.S. Environmental Protection Agency 2003) – applying the results from such tests to the real world is not always an easy task.

One key feature of many abandoned mines is the increase of pollutants shortly after the mine water discharges to the receiving water courses and their decrease thereafter. This typical scenario is called first flush (see section 5.4). Knowledge about the first flush, its peak concentrations and the pattern of how it decreases, are essential for the planning of remediation technologies, especially mine water treatment plants. Many active treatment plants work economically only with high concentrations of metals, while passive treatment plants require relatively low concentrations and loads. If, therefore, the duration of the first flush can be predicted to a reasonable degree, the time when an operator can change from active to passive treatment can be estimated (ERMITE Consortium et al. 2004).

Such an estimation using a black-box model was conducted at the German Straßberg/Harz mine (unpublished data). All available data, such as weekly flow, iron measurements, and time series analyses of up to 10 years were used (Fig. 9). The left graph shows the measured and predicted decrease in the iron concentration and the right one shows the expected loads

until the year 2015. For all calculations the following commercially available software was used: statistical analyses (SPSS), correlation analyses (TableCurve 2D), spreadsheet software, and scientific graphic software (Sigma Plot 2D). Burke and Younger (2000) predicted the first flush scenario for the South Yorkshire Coalfield by applying expert knowledge and the numerical code GRAM (Groundwater Rebound in Abandoned Mine Workings). Another method was described by Younger (2000d), who analysed more than 80 British mine water discharges and came up with a flow chart to predict the medium and long term mine water qualities with respect to iron concentrations. However, the latter flow chart is coal mine drainage biased and focuses on the United Kingdom, and has not been tested for many other areas of the world, except in French coalfields (Paul Younger, pers. comm.). Depending on the total sulphur content of the mined strata, which ranges between below 1 and 5 weight percent, Younger (2002d) estimates total peak iron concentrations of 0.01–1,500 mg L⁻¹ (Tab. 10). However, as only the disulphide content of a stratum is important for the release of acidity and consequently iron, it is more appropriate to use the total disulphide content instead of the total sulphur content of a stratum.

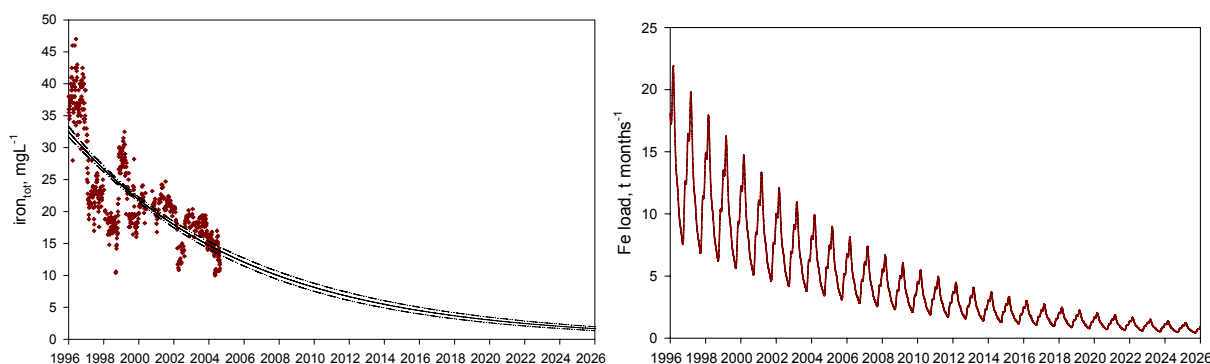


Fig. 9: Prediction of future Fe_{tot} concentrations and loads in the Straßberg/Germany Fluorspar mine until the year 2026 by using past data and a black-box model assumption (raw data kindly provided by BST Mansfeld GmbH and GVV).

4.8 Radionuclides in non-Uranium Mines

Enriched radionuclide concentrations in mine waters from uranium mines are a frequent occurrence (Merkel et al. (1997) and many others), and the radionuclides commonly found in mine waters are thorium, radon, radium, and uranium. In non-uranium mines those radionuclides are usually not considered by researchers or authorities, though the phenomenon of naturally occurring radioactive materials (NORM) has been known for a long time. Sources of that radioactivity are especially water of hot springs and effluent from water separators from gas and oil wells, (e.g. Sohrabi 1998; Chałupnik and Wysocka 2000; Gillmore et al. 2002; Dowdall et al. 2004; Skowronek et al. 2004). This must not imply that those mine waters are of no danger to human beings, it just means that there is very often a lack of awareness concerning the radioactivity of mine waters, which Paschoa (1998) can also see in other industrial areas. Yet, the problem should not be overemphasised as the concentrations of the radionuclides – from a radiotoxic and very often also from a chemotoxic point of view – are usually very low, though several sites are known with elevated radionuclide concentrations (Sohrabi 1998; Gillmore et al. 2002; Schmid and Wiegand 2003). Besides the radionuclides in the water phase there is the issue of radon in the mine air, in the vicinity of ventilation shafts, and mine adits ('mine gases'), sometimes referred to as elevated natural radiation areas (ENRA; Sohrabi 1998). Radon is known to cause lung cancer, originally called the *Schneeberger Lungenkrankheit* (Schneeberg lung disease; Enderle and Friedrich 1995; Boice et al. 2003), and

therefore the public must be protected against elevated concentrations of radon in their environment (Gillmore et al. 2002). However, there is also a positive effect known, as there are numerous radon spas throughout the world (section 11.7) and at which concentration radon becomes toxic is still a matter of discussion – even if many countries, for obvious reasons – have set up legal standards for radon concentrations in houses or at the working place (e.g. U.S. Environmental Protection Agency 2000; 2001/928/EURATOM; Austrian Drinking Water Guidelines). Finally, elevated radon concentrations might also be expected in zones with fissures in volcanic rocks (Davis 1983; King et al. 1996).

According to Sohrabi (1998), who gives an excellent summary of areas with elevated levels of natural radioactivity, NORM should be classified into 4 categories and depending on the mean annual potential public exposure, no remedial action, remedial action within 5 years or urgent remedial action should be taken (Tab. 8). However, this is not the only classification known, but one of the many which gives clear guidelines of potential actions that should be taken in areas with elevated natural radioactivity.

Tab. 8: Summary of classification of natural radiation areas according to Sohrabi (1998). LLNRA: low or normal level natural radiation area, MLNRA: medium level natural radiation area, HLNRA: high level natural radiation area, VHLNRA: very high level natural radiation area, UNSCERA: United Nations Scientific Committee on the effects of Atomic Radiation, ICRP: International Commission on Radiological Protection.

Classification	Criteria: potential public exposure	Remedial action
LLNRA	$\approx 5 \text{ mSv y}^{-1}$ (twice natural average global effective dose, UNSCEAR)	None
MLNRA	$\approx 5 - 20 \text{ mSv y}^{-1}$ (present ICRP limits)	Within 5 years
HLNRA	$\approx 20 - 50 \text{ mSv y}^{-1}$ (former ICRP limits)	Within 5 years
VHLNRA	$\geq 50 \text{ mSv y}^{-1}$	Urgent, possible evacuation

Recent publications focused on radionuclides in the Polish and German Ruhr coal mining districts (Lebecka et al. 1997; Chałupnik and Wysocka 2000; Chałupnik et al. 2001; Schmid and Wiegand 2003) and in elevated radon concentrations in British tin-copper-uranium mines (Gillmore et al. 2002). In the Silesian coal mining area, two types of mine waters exist: type A with a higher mineralization and type B with a comparatively lower mineralization (Lebecka et al. 1994, 1997). Both types contain elevated concentrations of ^{226}Ra and ^{228}Ra , with typical ^{226}Ra activities of $62.8 \cdot 10^3 \text{ Bq L}^{-1}$ for type A and $3.4 \cdot 10^3 \text{ Bq L}^{-1}$ for type B. Concurrently, type A mine water contains high Ba concentrations with typical values around $1,500 \text{ mg L}^{-1}$. To prevent the precipitation of $(\text{Ba}, \text{Ra})[\text{SO}_4]$ in pipelines or river banks (Pluta 2000), the mine water is chemically treated with different methods (Lebecka et al. 1994; Chałupnik and Wysocka 2000; the different publications are not consistent in which type of water is type A and which B). Elevated radioactivity is also known from the flooded Stockheim coal mine in Franconia/Germany (Ebner and Schwarz 1973; Ziehr 1983).

Though Gillmore et al. (2002) present an interesting study of extreme radon concentrations in abandoned mines in Devon and Cornwall/UK, they do not really fit in this chapter, as they were also mined for uranium. Yet, their conclusions concerning radon emanations from those abandoned mines are also relevant for non-uranium mines. For the South Terras mine in Cornwall, which was abandoned and flooded in 1928, they reported a radon concentration of $3.9 \cdot 10^6 \text{ Bq m}^{-3}$ in a distance of 70 m from the mine's entrance. Within the mine they measured doses for a duration of one hour which equalled the 18 year personal dose for a member of the public. Everybody working in even long abandoned mines where uranium was a by-product must be aware that extreme radon concentrations might be expected. Those extreme radon concentrations were used by the Soviet occupiers of Saxony to locate potential uranium

sources in the silver/lead mines of the Erzgebirge/Germany. They knew of the radiological investigations in the early 20th century by Schiffner and Genser (Schiffner (1908); Schiffner and Weidig (1909); Schiffner et al. (1911); Weidig (1912)) and used those data to concentrate their exploration at the localities with the highest radioactivity and temperature anomalies in the working and abandoned polymetallic mines (Tab. 9). It should be noted that the water of the Hindenburg and Bismarck springs was used in the Bad Schlema Spa which was the world's most radioactive spa water by this time (Genser (1933)).

Potential dangers of radionuclides can be seen for people living very close to places where radon emanates from flooded underground mines, for researchers working in abandoned mines, for ecologists taking care of bats, for private archaeologists, and for mineral collectors. Schmid and Wiegand (2003) calculated the doses for the public along the river Lippe between the Auguste Victoria mine and its confluent with the river Rhine for external exposure and ingestion of contaminated, Ra-barite rich soil. External exposure, depending on their scenario, ranged between 0.03 mSv a⁻¹ and 2.4 mSv a⁻¹, while ingestion of the sediments could result in an additional 0.01—15.5 mSv a⁻¹ (this number largely depends on several parameters, thereunder the isotopic composition of the sediment). They conclude that underground precipitation of the radionuclides would be a measure to prevent the drainage of radium into the anthroposphere.

Another very special case are the South African gold mines in the Witwatersrand Basin, where elevated concentrations of uranium are known to occur in the ore body as a by product within the carbon leaders (Pohl 2005). Pollution comes from the drained mine water as well as from the seepage water of the tailings dams and can locally reach high concentrations of uranium (Coetzee 1995; Tutu et al. 2003), which are sometimes remediated with bioremediation (Weiersbye and Witkowski 2003). Wendel (1998) describes the different sources of radioactive mine waters in South Africa and he recommends to develop cost effective prevention and remediation methods based on further investigations. Because more and more mines in the Rand will be closed during the next decades and their mine water will drain into the surrounding areas, there is a real need for the South African mining industry and the regulators to invest more research on NORM for those cases.

None of the authors gives a general recommendation of how to prevent elevated concentrations of radionuclides in the anthroposphere. Yet, the Polish experiences show that high radionuclide concentrations in non-uranium mines can be prevented by using underground treatment facilities. Furthermore, it is possible to diminish the mine water's radium concentration by active treatment with barium chloride (Skeaff and Campbell 1980; SENES Consultants Limited 1994). In most of the other cases a thorough monitoring of the radon exhalation in or at abandoned mines in which uranium or thorium minerals were found is possibly the only precautionary measure that can be taken. Apparently, U.S. Environmental Protection Agency (2000) clarifies that only the incremental radioactivity owing to the abandoned mine must be considered when speaking about legislative measures. Elevated background radioactivity due to geological reasons, as it can be found in many granitic settings, must be separated from that radioactivity. However, it is known from many localities that even if the radon concentrations outside the abandoned mine workings are under the national limits, closed rooms, especially cellars, can be contaminated with radon and therefore have to be treated in a special way. From a water managerial point of view, if uranium or thorium minerals are known to be found in the mine or its vicinity, uranium and radium measurements shall be conducted in the mine water and radon measurements in the mine air.

Tab. 9: Radioactivities of mine waters in the Markus-Semmler-Adit near Schlema, Erzgebirge/Germany before large scale uranium mining started. Localities ordered by their activities (Wolkersdorfer 1996 after Schiffner 1908; Schiffner and Weidig 1909; Schiffner et al. 1911; Weidig 1912). Institute Fresenius (H. Zerbe, pers. comm.) analysed an activity of 246,500 Bq L⁻¹ in the 1930s. The German names are typical old fashioned miner's names. Ort: face; Flügel: wing (of a main gallery); Spat: ore vein in easterly direction; Morgengang: ore vein in north-easterly direction; Flächen: ore vein in south-easterly direction; Stehend: ore vein in northerly direction.

Date	Location	Radioactivity, Bq L ⁻¹
1932	Radiumort, Hindenburg Spring	181,818 (246,500)
1932	Radiumort, Bismarck Spring	40,404
1932	Friedrichflügel	10,774
1932	Heinrichflügel, Drill Hole III	10,101
7.2.1911	Radiumort, 25 m inside Markus-Semmler-Adit	9,400 — 11,000
1932	Radiumort, Drill Hole I	8,350
19.10.1911	Radiumort, 25 m inside Markus-Semmler-Adit (<i>n</i> = 7)	7,886
19.9.1911	Radiumort, drainage into Markus-Semmler-Adit (<i>n</i> = 16)	5,375
23.11.1910	Radiumort, 27 m inside Markus-Semmler-Adit	4,539
1932	Radiumort, Radium Shaft	4,444
1932	Radiumort, Drill Hole II	4,108
1932	Radiumort, B-Flügel	4,040
1932	Heinrichflügel, Heinrich Shaft	3,771
19.10.1911	Radiumort, 35 m inside Markus-Semmler-Adit (<i>n</i> = 5)	2,979
1932	Wolfgangflügel	1,886
1932	Gleesbergflügel	1,886
1932	Johannesflügel	1,347
23.3.1909	Maximilian Spat	1,048
7.01.1909	Eva Spat	640
1932	Jung-König-David-Flügel	337
27.3.1909	Fleischer Morgengang	226
30.3.1909	Neuglück Flächen	218
30.3.1909	Quergestein close to Gottes Schickung	191
30.3.1909	Türk Flächen	121
30.3.1909	unnamed Spat at the crossing with Glück Flächen	110
27.3.1909	Gabe Gottes Working Place north-west	86
30.3.1909	Hohe Fichte Flächen	55
26.3.1909	Priester Spat	51
25.3.1909	between Neujahr Stehenden and Frischglück Morgengang	31
5.1.1909	Freudenschall Stehenden and Hoffnung Gottes Flächen	24
26.3.1909	unnamed Morgengang in the hanging wall of Rappold Flächen	22
27.3.1909	Himmelfahrter Flügel	11

5 Insights into Mine Closure

5.1 Reasons for Mine Closure

Mining is an economic activity that ensures the daily needs of a growing world population for raw materials such as metals, fuels, and building materials. Depending on the available technology and the mineral mix to be extracted, the lifetime of a mine might last for several years or millennia, as in the case of the Tyrolean Schwaz Silver-Dolomite mines, the Salzburg Mitterberg Silver-Uranium deposit, or the Spanish Rio Tinto mining area (Zschocke et al. 1932; Salkield 1987; Ammann 1990; Günther et al. [ca. 1993]; Leblanc et al. 2000). However, eventually, all mines must close, and nowadays closure procedures have to meet particular standards that have been established by the various governments and regulatory agencies. These environmental requirements are usually drawn up by the authorities based on expert reports or on environmental impact assessments, whereas already abandoned mines usually did not undergo such procedures. Nevertheless, both cases are very similar with the respect to the available remediation options and the time scales or financial efforts necessary to remediate these sites, which cover about 240,000 km² of our planet (Furrer et al. 2002).

The reasons for mine closure that Lässl listed in 1556 in his *Schwazer Bergbuch* (Lässl 1556; Winkelmann 1956), have not changed considerably. He described that “four things ruin a mine”:

- war (and accidents)
- diseases
- inflation (economy)
- listlessness

In addition two more reasons now exist, of which the first was known by the miners of that time (Agricola 1556), but never was a matter of real concern:

- environment
- deposit depletion

Anyway, the effects of mine closure are various and have to be solved on a site or catchment scale. None of these aspects is strictly site specific and therefore, the consequences of mine closure have to be resolved by involving all the relevant stakeholders, and keeping in close communication with all stakeholders involved:

- economics and financial
- social and community
- legal aspects
- political and psychosocial
- environmental
- technological

Responsible mine closure, as outlined by Sheldon et al. (2002), guarantees that plant and equipment (if not an industrial heritage) as well as hazardous substances are removed, the used land is reclaimed, and all reasonable measures are taken to exclude potential ground- or surface water pollution. This does not only apply to the mine site itself but for all installations that were used by the mine operator. Too many sites are known worldwide where the mine operator just switched off the lights and left the mining site without any after care. If the mining industry wants to be acknowledged again as the key industry for all industrial processes

small and medium scale as well as large scale mining operators have to ensure that they leave their mining sites as environmentally sound as possible. Some of those aspects will be discussed in the next section.

5.2 Post-Mining Sustainability of Mine Sites

Sustainability as a technical term was introduced by the Saxonian Oberberghauptmann (senior chief-inspector of mines) Hannß Carl von Carlowitz. Like Georgius Agricola he came from the German Erzgebirge and in 1713, he published *Sylvicultura Oeconomica*, where he first described sustainability in forestry (Carlowitz 1713). Since then, forests in Germany have generally been maintained in a sustainable manner, which means that no more trees are cut than are growing, or the land is reforested at the same time. In the 19th Century, the principles of sustainable forestry became the method of choice in nearly all developed countries of the world. Because sustainability is so important in today's mining and mine closure processes, von Carlowitz' first mentioning of the word shall be cited here. Interestingly, it stands in a direct context to mining (the words are written exactly as they appeared in the original publication):

Aber da der unterste Theil der Erden sich an Ertzen durch so viel Mühe und Unkosten hat offenbahr machen lassen, da will nun Mangel vorfallen an Holtz und Kohlen dieselbe gut zu machen; Wird derhalben die größte Kunst, Wissenschaft, Fleiß, und Einrichtung hiesiger Lande darinnen beruhen, wie eine sothane *Conservation* und Anbau des Holtzes anzustellen, daß es eine *continnuierliche* beständige und nachhaltige Nutzung gebe, weils es eine unentberliche Sache ist, ohne welche das Land in seinem Esse nicht bleiben mag. Denn gleich wie andre Länder und Königreiche, mit Getreide, Viehe, Fischereyen, Schiffarthen, und andern von GOtt gesegnet seyn, und dadurch erhalten werden; also ist es allhier das Holtz, mit welchem das edle Kleinod dieser Lande der Berg-Bau nemlich erhalten und Ertze zu gut gemacht, und auch zu anderer Nothdurft gebraucht wird. (Hannß Carl von Carlowitz, *Sylvicultura Oeconomica*, Capter VII. Von Nothwendigkeit und Nutzen des Holtz-Anbaues, p. 105—106; Leipzig 1713).

Translated, this means:

Now, that the ores in the lowermost part of the earth, through effort and expenses, were revealed, a shortage of wood and coal for their beneficiation occurs. Thus, the greatest skills of art, science, diligence, and institutions of the local states will examine how a conservation and cultivation of wood shall be done so that a continuous, lasting, and sustainable use is given, because it is an indispensable thing, without which this state will not continue to exist. Since other countries and kingdoms are blessed by God with crops, cattle, fishing, ships, and others, and survive because of that; so, here it is wood that is the precious jewel of these states: it allows mining to stay alive and ore to be beneficiated, and also used for other affairs.

Since the Berlin round table on Mining and the Environment (United Nations Department of Technical Co-operation for Development and Development Policy Forum of the German Foundation for International Development (1992), see appendix III) and the UNCED conference in Rio de Janeiro (1992) which resulted in the Agenda 21 (United Nations Conference on Environment and Development [UNCED] 1992), sustainability and environmental sound mining also became crucial for mine operators. Furthermore, these principles are a fundamental part of the UNDESA and UNEP guidelines for environmentally safe mining and post-mining procedures (United Nations Environment Programme [UNEP] 1994). A two year project called "Mining, Minerals and Sustainable Development (MMSD) Project" focused on many aspects of sustainable mining, with a critical view on the mining companies, but al-

though the project produced a lot of material on many different aspects of mining, mine water issues were not covered in great detail (Mining Minerals and Sustainable Development [MMSD] Project 2002).

Post-mining sustainability regarding mine water means that mine water quality after mining will reach a good ecological status and that the water quality can be sustained without further input of energy or financial subsidy. This does not only apply for the mine water *sensu strictu* but for the surface and ground waters that could be affected by the abandoned mine. Whereas this is the case for the chemical constituents emanating from the abandoned mine and should also be the case for suspended solids, which in some areas of the world account for a large part of the mining-impacted pollution, as in the Indonesian Grasberg-Ertsberg (Lottermoser 2003), the Papua New Guinea Ok Tedi (Miller et al. 1990; Hettler 1995; Lottermoser 2003), or Misima (Jones and Ellis 1995), the Tasmanian Mount Lyell mines (Harries 1990; Sustainable Development Advisory Council and Copper Mines of Tasmania 1995; Featherstone and O'Grady 1997), or the sludge from mine water treatment operations (Fig. 10). The question is always which is worse: precipitated solids or acid mine drainage.

MPC, the Mineral Policy Center, an NGO based in the USA, called for a complete ban on submarine tailing disposals in 1999. Indeed, submarine tailing disposal usually will not be permitted in the USA. One exception are the Alaskan A-J Kensington, and Quartz Hill mines where exceptional circumstances had to be considered (U.S. Environmental Protection Agency 2003). In Indonesia, submarine tailing disposal is commonly used due to limited land space and the potential for earth-quakes that might cause disastrous tailing dam failures (Natural Resources Management Unit Division „Protection of the Environment and Natural Resources“1999). Those cases show that submarine tailing disposal, compared to alternative options, might be comparatively harmless, and from a water managerial point of view is usually of no great concern (Ellis et al. 1995). Yet, submarine tailings disposal needs all the environmental precautions that are necessary in a up-to-date mining operation.



Fig. 10: Ore processing sludge from the Alegria mine, Minas Gerais/Brazil.

During the last couple of decades, the need to protect the environment and the need for standards regarding mine closure became obvious. Environmental protection groups, international organizations, regulators, NGOs, consultants, and the mining industry itself were all searching for common procedures to close a mine. Furthermore, while conducting several mine closures, there was a general consensus that since post-mining remediation can be extremely expensive, measures should be taken to prepare for sustainable mine closure during the active mining process. As a result, nowadays, most mining operations have to provide mine closure

plans at a very early stage of the environmental assessment plan, before being given permission to mine. Concerning water quality, those closure plans must consider very long time scales as the longevity and potential changes in mine water quality must be properly investigated. This leads to the demand for “Post-mining Sustainable Land Use Planning” instead of “Closure Plans”, which assumes that regulators will accept a sustainable land use other than the pre-mining conditions (Robertson and Shaw 1999). Within such plans, it could well be that some parts of an abandoned mine have to be managed (for example slopes of tailings dams or open pits), while others would be sustainable. A sound understanding of both the physico-chemical aspects that could occur after the mine closure and the mass transport within the flooded underground mine is necessary for post-mining sustainable land use planning. Therefore, post-mining sustainability requires many of the methods needed for acid mine drainage prediction. It is not always possible to predict whether a self sustainable state will be reached after mine closure. Also, it is sometimes unclear what self sustainability means: many long abandoned mining areas, such as the Rio Tinto area, are arguably self sustainable, but if a mine operator proposed to leave a mine site in this condition, he would never get a permit for mine closure by the authorities. Similar conditions, from a legal point of view, occur at the German Freiberg/Saxony Rothschnberg or the Burgfey/Eifel (Fig. 11) drainage adits, which are draining polluted water (Zoumis et al. 1999; Mair 2002; Weyer 2003). In all three cases, natural attenuation creates sustainable conditions (see section 11.5 for a description of “natural attenuation”).



Fig. 11: The portal of the 7.5 km long German Burgfey/Eifel mine water adit discharging 15—19 m³ min⁻¹ of polluted mine water of the abandoned Mechernich mining area into the Veybach stream.

Using mine water in an environmentally safe manner after mining ceases is considered an excellent example of post-mining sustainability (see section 11.7). Sustainable uses of mine water include drinking water (Burbey et al. 2000; Teaf et al. 2006), spa water (e.g. Bad Suderode, Feengrotten Saalfeld, Alexisbad/Harz Mountains, Bad Gastein/Austria), turbine water (e.g. Max-Braun-Adit, Biberwier/Tirol, Fig. 12), and recreational purposes (e.g. Passagem de Mariana Gold mine, Minas Gerais, Brazil; Banyan Tree Phuket, Thailand). Many mine sites are examples of post-mining sustainability (Fernández Rubio 2004). Such sustainable sites are not always as spectacular as the Cornish Eden Project (Macadam and Shail 1999; Harris 2004), the Swedish Dalhalla Stone Quarry (Mattsson and Stenbacka 2004), the Yorkshire

Rother Valley Country Park (Brook and Cresswell 1989), the German EuroSpeedway Lausitz (Lusatia), or the German Rammelsberg World Heritage lead-zinc-mine (Roseneck 2001).

One of the world's nicest health resorts is the Banyan Tree Phuket resort at Laguna Phuket in Thailand, which was not affected by the December 2004 Indian Ocean tsunami because it was situated on the lee side of the wave. Tin dredging in the Ohang-Nga Bay and Bang-Tao Bay started in 1907, and led to the mangrove swamp being dredged to a depth of 30 m, resulting in the destruction of the mangrove environment. Even six years after an area was dredged, the natural mangrove habit did not recover at all, and sediment plumes moving into the open sea subjected off-shore coral reefs to siltation and devastation (Aranyakanon and Yongsakul 1981; Chansang 1988). Due to the large tin reserves in the Ko Phuket area, a tin smelter, accounting for 10% of the world's tin production, was erected. Its proximity to a reef, called the Tin Smelter Reef, was the reason for several biological studies, which revealed a good adaptation of the corals to the high metal contents in the water (Brown and Holley 1982; Scoffin and Bradshaw 2000). Because of the increasing conflict between the interests of the tourist industry and the shallow and coastal tin dredging activities, as well as low tin prices, mining ceased in 1985 (Carlin 1999). Chansang (1988) concluded that even after the dredging operation, the "problem of effects, caused by increases in suspended solids and turbidity will remain". For a long time during the active and after the active dredging operations, which stopped in 1992, an intense, well-documented monitoring of the area was conducted by the Phuket Marine Biological Research Centre. In 1977, the United Nations Development Plan stated that the area is a toxic wasteland too severely damaged to support vegetation and consequently "too environmentally damaged to have any development potential". Nevertheless, two years later, the Tourism Authority of Thailand wrote: "although the damage was probably beyond repair, effort should not be spared to try to restore the environment". A \$200 million (U.S.) restoration programme was initiated, starting with importing fertilised topsoil, which was used for planting casuarinas, palm, fruit and flowering trees as well as other vegetation. This measure successfully stopped the erosion of the abandoned mine and tin smelter land. At the same time, large quantities of contaminated soil were removed, pit lakes were flooded with fresh and salt water, and environmental impact studies were undertaken for the new hotel resorts. Still, previous offshore tin mining in Bangtao Bay continues to affect the region as millions of Baht have to be used to replace the sand swept away into the holes on the ocean floor left over by the dredging operations (adopted from Rung-Aroon Wichitwatee, Public Relations & Advertising Manager, Laguna Phuket Media Contacts).

Miller et al. (2005) investigated mine and mine treatment discharges for their potential use in aquacultures in West Virginia, where, especially in the southern parts, mine water is also used as drinking water with no or only little treatment. They concluded that mine waters that meet the standards for fishing waters might be used for aquacultures as long as the CO₂ levels are not too high. Furthermore, abandoned treatment facilities might be converted to aquaculture, as shown by the Guyses Run acid mine drainage plant, which started operation in 2001. In 2004, six aquacultures produced salmonids in Pennsylvania, Maryland, and West Virginia/USA. Minnesota had more than 20 abandoned mine sites for aquacultural use, but several of them had to be closed as the increase of nutrients and turbidity affected the good ecological status of the watersheds (Axler et al. 1998).

The Felsendome Rabenstein near Chemnitz/Saxony is a visitor's mine in which the two lowermost levels have been flooded since the beginning of the 20th century (see section 12.10). It has been a local attraction since 1936. During the last couple of years, the site was converted into an attractive recreation area, and the underwater part is now being used by divers. The "Alte Kalk-Bergwerk" limestone mine in Miltitz/Saxony has been used for various purposes between 1571 and 1967, and diving in that abandoned and flooded, up to 65 m deep underground mine, started in 1997 (Udo Krause, Dresden, pers. comm.). Yet, compared to the

flooded Bonne Terre/USA lead mine, the world's largest fresh water diving resort, 100 km south of St Louis, Missouri, both of those mines are relatively small.

Inert material as well as radioactive waste is commonly disposed of underground as a post-mining use, at sites where the ground water around the mine will not be negatively affected after a final mine closure. Usually, the substances stored in abandoned underground mines are inert materials, such as fly ashes or residues from different industrial activities. Radioactive waste disposal, though it is a post-mining use, cannot be listed under the definition given at the beginning of this section, because it requires maintenance.

Underground mines are also used for geothermal energy production and first investigations date back into the 1970s (e.g. Lawson and Sonderegger 1978). Yet, this investigation in Montana/USA yielded only seven anomalies with discharge temperatures between 5 and 12 °C. In Canada's Nova Scotia, water in the Springhill coal mines with a temperature of 18 °C is used for geothermal energy production (Jessop et al. 1995; Jessop 1998). Furthermore, Saxony/Germany is currently investigating the potential for geothermal energy from abandoned underground mines (Workshop Geothermie Bad Schlema 2004), and a EU project called "minewater project" investigates potential energy sources from mine water in the Netherlands (Heerlen) and Scotland (Midlothian). Another geothermal project using warm mine water has been initiated in the Aachen/Germany coal mining area (Clauser et al. 2005). Other mines where mine water is used for heating purposes are Park Hills/USA, Follfda/Norway, Shettleston/Scotland, and Ochil View/Scotland (Banks et al. 2003). Future geothermal projects at mine sites with elevated water temperatures can be a contribution to reduce the CO₂ emissions into the atmosphere, but their potential for that purpose has not yet been fully developed.



Fig. 12: Turbine house of the Austrian Biberwier/Tirol hydropower plant using mine water of the Max-Braun-Adit located in the Silberleithe hills that can be seen in the background of the photograph. The first turbine to produce energy was installed in 1904; the currently operating turbine was installed in 1936.

If mine water is polluted, a sustainable treatment scheme has to be undertaken to ensure that the environment is protected. The European Union PIRAMID R&D project investigated the sustainable installation of passive mine water treatment schemes (PIRAMID Consortium 2003), and similar studies have been conducted elsewhere. The use of polluted mine water for irrigation purposes, e.g. for sugar beans and wheat, has been investigated by several authors (e.g. Du Plessis 1983; Voznjuk and Gorshkov 1983; Annandale 1998; Annandale et al. 1999, 2001; Jovanovic et al. 2001; Annandale et al. 2002; Jiries et al. 2004). Yet, the long term effects of such irrigation on ground or surface water quality have not been studied in detail.

In the case of the German Rammelsberg lead zinc mine, the mine operator presented a closure plan to the authorities that also included the treatment of the acidic mine water. It was unquestionable that the mine water had to be treated for a given time period but the requisite time period could not be agreed to by the authorities and the mine operator. Therefore, the case went to court and the court ruled that the mine water has to be treated by the mine operator until the quality reaches the limits given by the authorities – if necessary, *ad infinitum* (Anonymous 1994, 1995). This case clearly proves how important post-mining water quality is regarded by the authorities.

A residential area for prosperous inhabitants of Belo Horizonte/Minas Gerais was constructed directly on the shores of the flooded Brazilian Águas Claras mine pit lake. Most pit lakes within the *Quadrilátero Ferrífero* (Iron Quadrangle) in Minas Gerais can be seen as environmentally friendly from a water quality perspective, because the abandoned iron mines are usually situated within itabiritic rocks that do not generate acid mine drainage. Several kilometres away, the abandoned Mangabeiras iron mine was transformed into a Public Park, which is now the biggest green area within the borders of Belo Horizonte (Grandchamp et al. 2001; Amorim et al. 2001).

One of the most spectacular European open-cast mine regeneration projects is the Cornish Eden Project. It is an excellent example of co-operation and an ecologically sound after use of a mine site. Cornwall contains 80 km² of active and abandoned China clay mine workings and numerous abandoned tin mines (Younger and LaPierre 2000). Many abandoned mine sites are of public concern because the metal loads pollute surface streams. Once, the Bodelva Mine was a prosperous China clay mine in the central part of Cornwall, and extracted the clay for about 150 years. A total of $7 \cdot 10^6$ m³ of material was removed from the open pit leaving behind a 30–70 m deep hole. In 2001, the Eden Project began and the remediation and dewatering work of three years could be presented to the public. Today, Rio Tinto, Anglo America, MIRO, Eden Project and English Nature are working together in a collaboration to establish an educational centre for post-mine regeneration at this site (Macadam and Shail 1999; Harris 2004).

5.3 Previous Mine Flooding Studies

While predicting flooding is highly uncertain, the prediction of mine water quality and mine water stratification is even more challenging. Currently, no case is known where the stratification of a mine was predicted precisely and where remediation methods based on stratification predictions were successful. The most comprehensive study of mine water stratification in a single mine known so far was conducted at the Niederschlema/Alberoda/Germany uranium mine (Wolkersdorfer 1996). Between June 1992 and December 1994, some 115 depth profiles in seven underground shafts were measured and the evolution of stratification in shafts was observed. A thorough investigation of flooded mine shafts on a regional scale was conducted by Coldewey et al. (1999), who studied all accessible flooded hard coal shafts in the German Ruhr Area, summing up to 19 shafts. Another comprehensive study of ten flooded underground shafts was conducted by Ladwig et al. (1984) in Eastern Pennsylvania/USA. Playton et al. (1980) investigated seven flooded mine shafts in the Picher/USA Mining District.

In chapter 12, several case studies of flooded and stratified underground mines will be described using examples found in the literature and my own investigations. Only mines that are not backfilled will be discussed. Physico-chemical parameters (temperature, salinity, pH, redox potential, oxygen content), in addition to chemical parameters, will be presented, a preliminary interpretation of the stratification patterns will be given, and an attempt to classify stratified mine shafts will be discussed.

The situation in metal, non-metal, and coal mines differs significantly from flooded salt mines. Nevertheless, one should bear in mind that the controlled flooding and monitoring of the German Hope Salt mine near Hannover was one of the first investigations studying the full range of mechanical and chemical processes during the flooding of an underground mine (GSF – Gesellschaft für Strahlen- und Umweltforschung 1985, 1986; Flach and Heick 1986; Herbert 1989). Therefore, some case studies of flooded salt mines will be presented in chapter 12, too.

Some investigators have proposed that mine water stratification can be used as a remediation measure for polluted mine water (e.g. Heidenreich et al. 1991; Kindermann 1998; Gatzweiler et al. 2002). In fact, mine water stratification can be used to aid remediation methods, but only where thermal or forced convection will be reliably eliminated. Incorrect assumptions in the Straßberg case (12.11) and the sudden breakdown of stratification in the Frazer's Grove mines (12.6) clearly prove the importance of a thorough understanding of mine water stratification.

5.4 Effects Contributing to Mine Water Constituents

5.4.1 The First Flush

In most situations in surface streams, concentrations of water constituents inversely correlate with the stream flow rate (Stumm and Morgan 1996). Such a scenario can often be seen in surface streams after runoff events, where the electrical conductivity, as a measure of total dissolved solids, decreases while the flow increases. However, at most flooded surface and underground mines there is a steep increase in the concentration of mine water contaminants after the mine water reaches the outflow level (Fig. 13, Fig. 14). In the course of time, the concentrations usually decrease until they stabilise at a relatively constant value – which hopefully is the background concentration of that constituent. Though this phenomenon of mine water quality change has long been known it was not before 1997 that it got a name: the “first flush” (Younger 1997), which means that concentrations increase with the increasing flow. Commonly, the compounds leached during the first flush are more easily soluble geogenic or anthropogenic compounds (U.S. Environmental Protection Agency 2003).

Depending on the mine geometry and volume, the type of mineralisation, and the amount of precipitation, the mine water will recover to background values after several years or several decades, sometimes centuries (Younger 1997; Demchak et al. 2004). The reasons for the elevated concentrations of the mine water constituents can be assigned to one of the following three mechanisms:

- production of vestigial acidity
- production of juvenile acidity
- buffering

Vestigial acidity (Younger 1997) comes from the dissolution of the secondary minerals formed during and after disulphide oxidation (section 3.2). Most of those minerals are easily soluble sulphate salts and store a large amount of acidity, sulphate, and metals. They are therefore commonly referred to as ‘acid generating salts’ (Bayless and Olyphant 1993). Because of their high solubility they are quickly dissolved by the rising acid mine water and thus contribute to the peak of the first flush scenario.

Juvenile acidity (Younger 1997) results from three sources: proton acidity from the disulphide oxidation within the fluctuation range of the mine water table, proton acidity from the disulphide oxidation in the un-flooded mine workings above the water table, and mineral acidity from the dissolved metals, such as Fe, Mn, Al, and Zn, which are usually present as hydroxide

or sulphate salts. This phase is characterised by fresh water flowing into the system and reacting with the remaining disulphide and the before-mentioned minerals.

Besides the acid producing component, there is also an acid consuming component that occurs during the first flush scenario. As described in section 4.5, a range of minerals react as proton acceptors and neutralise acidity. Such buffering can lead to neutral or circum-neutral conditions in the mine water, after all the acidity is flushed out or consumed by the buffering minerals (Demchak et al. 2004).

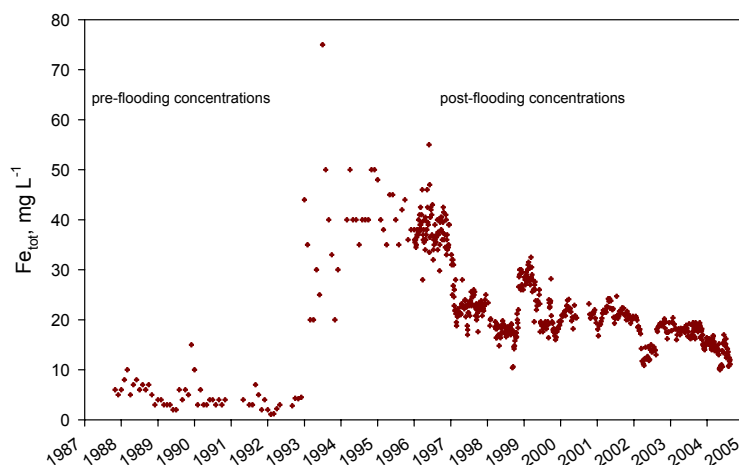


Fig. 13: Time series of changing Fe_{tot} in the Straßberg/Germany fluorspar mine before and after the flooding of the mine in 1993 (data kindly provided by BST Mansfeld GmbH and GVV).

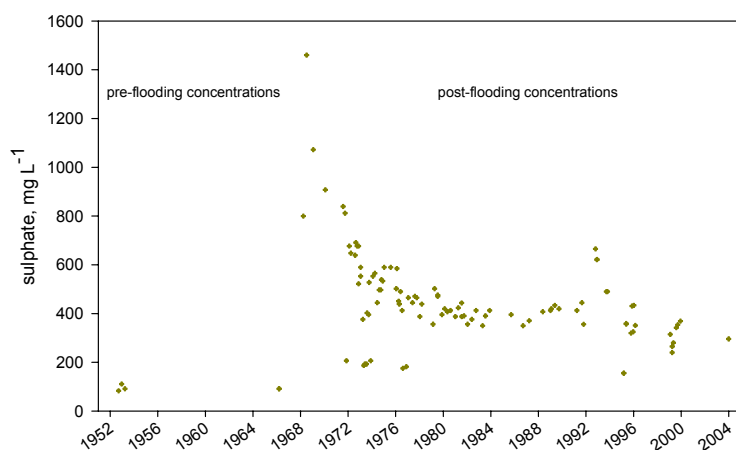


Fig. 14: Time series of changing sulphate concentration at the Rothsönberger Stollen/Germany adit (modified after Merkel et al. 1997; Baacke 2000; Beuge et al. 2001; Yefimochkina 2004).

Most first flush scenarios show a characteristic time lag after which the pollution load reaches its maximum. This time lag can range from several hours to several days, and has not been thoroughly investigated. Yet, physico-chemical measurements in shafts during the flooding process often show that the electrical conductivity in the upper meters of the flooded shaft is less than it is several meters beneath the water surface. Wolkersdorfer (1996) shows that this low conductivity layer (zone № 1) is commonly between 1 and 18 m thick, though in one case it was 190 m. It is followed by zone № 2, where the electrical conductivity decreases within a

thickness of 1 to 30 m; finally comes zone № 3, where the electrical conductivity is relatively constant or slightly decreasing (see also Fig. 113). The reason for this behaviour is that the polluted mine water is commonly overlain by less polluted infiltration water. When the mine water finally discharges into the anthroposphere, first the less polluted mine water drains out of the mine followed by the more polluted water, which leads to the characteristic first flush.

The duration of the first flush t_f mainly depends on the acidity removal aci_{rem} by buffering or dissolution, by the rate r_w at which acid-containing minerals weather, the volume V of the inter-connected workings, the hydraulic connection, conductivities K of the mine workings, and the ground water recharge R_{GW} . According to the findings of the HERO research team (University of Newcastle upon Tyne), the duration of the first flush after the overspill of the mine water can be calculated using the following empirical equation (Younger 2000d):

$$t_f = f(aci_{rem}, r_w, V, K, R_{GW}) \approx (3.95 \pm 1.2) \times t_r \quad (26)$$

with t_f : duration of first flush, min

t_r : time for voids to be flooded (“rebound time”), min

Wood et al. (1999) commented that t_f is a function of time dependent and time independent factors, whereas the lithological setting and the extend of the workings belong to the first, and the removal of vestigial acidity, carbonate dissolution, iron hydroxide armouring, bacterial reduction of sulphate, and change in water flow rate belong to the latter. From their analyses of 32 coal mine discharges, they were able to show that in the long term, pH-ranges between 6 and 7 and total iron decreases to less than 20 mg L⁻¹. A similar approach was used by Younger (2002d), who related the peak concentration of Fe_{tot} to the total disulphide content of the mined coal seams (Tab. 10).

Alkaline mine water can revert to acidic conditions when all the buffering phases were consumed and the production of juvenile acidity cannot be buffered any longer. Such a scenario has been observed at the Yorkshire Bullhouse mine, which became acidic after 60 years of being alkaline (Laine and Dudeney 2000; Younger 2000a). For every mine flooding scenario, it is therefore essential to calculate the potential net neutralisation capacity of the rocks in contact with the flooded mine.

Tab. 10: Disulphide contents of mined coal seams and their relation to observed peak iron concentrations in the United Kingdom. The numbers can be used to predict peak iron concentrations for the first flush period. Modified after Younger (2002d).

Total disulphide content, wt. %	Observed ranges of peak Fe _{tot} -concentrations, mg L ⁻¹
< 1	0.01—0.5
1—2	0.5—100
2—3	100—350
3—4	350—1,200
4—5	1,200—1,500

5.4.2 The Effects of Storm Events

Closely related to the first flush is the increase or dilution of contaminant concentrations at a mine water discharge (see also page 23). The impacts of storm events at abandoned mine sites can be minimised by using a thorough remediation strategy. Interestingly, there are many publications touching the matter of increased pollution and sediment loads in mine water affected streams, but no general work about that interesting subject, though Bayless and Olyphant (1993) describe many of the effects that cause stream pollution after surface runoff at mine sites. Most pollution comes from the erosion of tailings, waste rock piles, or from adit

mines and the subsequent transport of the sediments and metals into the receiving streams, the flooded mine workings, or pit lakes. Several measures to prevent such runoff by covering waste rock piles were described in Integrated Geotechnical Engineering Services Specialists in Unsaturated Zone Hydrology (2003) and U.S. Environmental Protection Agency (2000). Field studies, monitoring of surface run-off at abandoned coal mine sites, and the modelling of the run-off were described by Hodgkinson and Armstrong (1996).

Many regulations include precautions against surface runoff from active mines, and the recommendations should be kept in mind for mine closure activities as well (U.S. Environmental Protection Agency 2003). The necessary measures to be taken before, during, and after a storm event should be compiled in a stormwater management plan, similar to all the other management plans for an active or abandoned mine site (Jacobs and Denham 2001). Such procedures include covering of tailings or waste rock piles, covering of potential infiltration zones above the mine workings, water diversion, regrading of slopes, or revegetation. Some of those measures are described in section 11.4.7. Retention ponds hold back suspended solids. If diversion ditches are constructed to guide the runoff around waste rock piles, they must be large enough to collect all the water, even during high storm or snowmelt events. At the Aitik/Sweden mine, diversion ditches were too small and erosion caused damage to the cover (Salmon and Destouni 2001). Especially in tropical regions, stormwater management is critical, as shown at the Batu Hijau mine, located on Sumbawa Island, Nusa Tenggara Barat, Indonesia. Up to 50 mm in a 15 min interval and 200 mm in a 24 hour interval were measured at the mine site and if no measures were taken to drain the water – even during the post-mining period – severe damage to the surrounding environment could occur. Sediment traps were installed to combat the problem (Jacobs and Denham 2001; Jacobs and White 2001).

Besides the erosion of solids and its transport in the runoff water, rainfall also raises the water table in tailings or waste rock piles and can release the easily soluble secondary minerals in tailings (Al and Blowes 1996). Loadings during and after rainfall events can be up to 20% higher than during normal flow situations. Especially after long rainfalls events, when the tailings become flooded, significantly high discharges might be expected.

Possibly one of the most interesting sites where such phenomena occur is the Tasmanian Mount Lyell mining area, which affects the King River and, at its mouth, the trout farming of Macquarie Harbour (Lake et al. 1977; Locher and Keller 1995). Several hundreds of kilometres downstream, the aquatic life was severely affected by the effects of surface runoff, sediments from the remnants of copper mining, and a high daily load of copper (Parliament of the Commonwealth of Australia 1970; Taylor et al. 1996; Twining and Cameron 1997). The sediments transported by the Kings and Queen Rivers filled up the river mouth with nearly 1.5 m of sediments, so that boats were hindered from entering the river's mouth (Parliament of the Commonwealth of Australia 1970).

Proper regrading and revegetation is an effective means to minimise acid mine drainage (The Pennsylvania Department of Environmental Protection 1998). To prevent runoff, the slopes of tailings or waste rock piles should be regraded to reduce the slope of the piles and to minimise the erosion during storm events (Hawkins 1995). This provides more stable surfaces, supports revegetation, and thus prevents the pollution of surface streams and ground water. Several of the measures described under water diversion (section 5.7) can be used to prevent surface water from flowing over un-capped waste rock piles (Adams et al. 2000). Sandén et al. (1997) measured the amount of colloids during a storm event 500 m downstream of a remediated mine waste deposit. At this sampling location, the flow in the stream increased 2 hours after the rain started. They clearly proved that the amount of solids started to increase 1 hour later, increased by about 100% after another 30 minutes, rose to a maximum of 150% 4

hours after the runoff water reached the measuring location, and stayed that high for at least 24 hours.

In the Picher Mining District (Kansas/Oklahoma/Missouri, USA), measures were taken to prevent surface runoff from flowing into abandoned mine shafts and bore holes (Tar Creek Superfund Task Force Water Quality Subcommittee 2000a; DeHay et al. 2004). In addition, several areas were covered with impermeable soil to prevent storm water from flowing into the abandoned mine workings, but that measure was not very effective and is not recommended to prevent water flowing into abandoned workings. If there is a need to predict surface runoff, Appendices A and B of U.S. Environmental Protection Agency (2003) describe the necessary measures to predict the amount of runoff by using several methods, thereunder graphical and numerical techniques.

Burbey et al. (2000) observed two different effects of rainfall on the discharge water quality. While the main ions in the water, such as Mg, SO₄, and Ca, are relatively uninfluenced by the rainfall, the pH is negatively correlated with increased flow due to rainfall, and the iron concentrations were positively correlated with the rainfall events. They conclude that the rainwater, which contains no iron, flushes the mine water and also lowers the pH.

Sandén et al. (1997) took 47 samples over a period of 24 hours after a 15.5 mm rainfall event during 7 hours. The flow within the stream increased from 0.2 L s⁻¹ to 23 L s⁻¹ and decreased to 6 L s⁻¹ 23 hours after the rain started. Their results were similar to those of Burbey et al. (2000), with an increase in pH, turbidity, and TIC and a decrease of the other trace element concentrations. The main ions and total iron showed two scenarios: they first increased as salt dissolved from the mine waste disposal site, and after 4–5 hours, they decreased again as a result of dilution. Those two investigations show that there is no general behaviour of mining remnants or flooded mines due to storm events. Dilution is very often a result of precipitation. While the contaminant loads stay relatively constant, the contaminant concentrations decrease with increasing flow (Hawkins 1998).

In conclusion, the negative impacts of surface runoff after storm events or snowmelt can be positively influenced by covering the tailings and waste rock piles, regrading the slopes to reduce the slope angles, revegetating the surface of the piles, and diversion of waters. Compared to many other remediation measures, especially mine water treatment, these measures should be an integral part of a mine site remediation to limit AMD/ML.

5.5 Reasons for Mine Flooding

5.5.1 General Aspects of Mine Flooding

When the life of an underground or open-cast mine ends, its maintenance typically stops and the dewatering pumps are switched off. Generally, where no drainage adits exist at the lowest levels, the open pits or underground workings will fill with water and the mined voids will be flooded with ground- or surface water. Eventually, the water will overflow and discharge into receiving water courses. Depending on the open space, the availability of infiltration water, and the type of flooding (uncontrolled, controlled, monitored), inundation can last from several months to more than a decade.

According to the principles of sustainable mining, it is essential that the pre-mining hydrogeological conditions are nearly restored after the mine floods. Furthermore, potential drinking water reservoirs have to be protected, surface streams must not be polluted by the mine water, and the pollutant loads should be reduced to a minimum. Most of those prerequisites can be achieved by reducing the mine's water make, which is the sum of all the water discharging from a mine (Fig. 29). Often the drainage water of a flooded underground mine has a

high potential to pollute the surrounding environment and increase the base flows of receiving streams. Therefore, the mine operator nowadays has to ensure that polluted mine water is treated accordingly and that environmental pollution of the water catchment is minimised. To do this, the mine operator has to decide if it is better to accelerate or deaccelerate the flooding process. When mines are flooded quickly, the disulphides are excluded from weathering as soon as possible and the pollutant loads consequently are quickly reduced, according to equation 26. In such a scenario, the water treatment plant can be switched off after a relatively short time. On the other hand, if the water treatment plant will have to run for a very long time, the flooding process should be deaccelerated. Several options for an adapted mine water treatment are available, and will be discussed in chapter 11.

The main reasons for mine flooding are:

- the mine is not longer economical
- all the raw material has been exploited
- accident, war, or political reasons
- geotechnical stability of the abandoned mine workings
- prevention of disulphide oxidation
- safety reasons (prevention of unintentional visitors)

Controlled mine flooding is conducted in areas where people or the environment might be affected by polluted mine water. There, maintenance of pumps or treatment facilities will take as long as deemed necessary by the mine operator or the authorities. In some mines in the former German uranium mining areas of Saxony and Thuringia, controlled flooding was prepared for extensively; several expert opinions about the mine water rebound and the chemical evolution of the mine water quality were sought, taking nearly a decade to draw final conclusions (Jakubick et al. 2002). Controlled mine flooding does not necessarily mean monitored flooding. Of course, in nearly all mines that are flooded in a controlled way, the water level in the mine and the drainage water quantity are measured. Monitored flooding of a mine is a subset of controlled flooding, and includes the installation of physico-chemical probes, regular measurements of the drainage water quality within the flooded part of the mine and at the discharge points, as well as a geotechnical monitoring system. The Prediction Workgroup of the Acid Drainage Technology Initiative (2000) described such monitoring schemes for coal mines while the ERMITE Consortium et al. (2004) did the same for all types of mining operations. Usually, some pumps will stay installed, allowing to interfere with the raising mine water. Such a scheme was used by the Niederschlema/Alberoda/Germany mine, where the mine water table rose quicker than expected and the mine water table had to be lowered by pumping to avoid the discharge of heavily contaminated mine water (Meyer et al. 2002).

Installations for controlled mine flooding include:

- temperature measurements
- geomechanical installations (geophones)
- conductivity measurements
- piezometers
- flow measurements
- pumps

There are two possibilities for controlled mine flooding. In the first case, enough water in the catchment area of the mine is available to flood the mine. The mine can then be flooded passively, without using water from other catchments. The second case occurs where not enough water is available to flood the open space or where the water that would be available has a

chemical composition that would cause problems. Such situations can be found in the Lausitz/Germany lignite mining areas, where the rivers Spree and Neiße do not have enough water to flood all the open pit lignite mines that were closed after 1990 (Kaden and Schramm 2000; Schlaeger and Köngeter 2001; Grünwald 2001). Other examples are salt mines that are flooded with saturated brine to avoid the destruction of nearby mines or surface subsidence due to salt dissolution (Herbert 1989). Because the flooding of abandoned salt mines is different in many aspects from the flooding of abandoned coal or metal mines, their mine water management shall be excluded from the following discussion. The interested reader is directed to the investigations of the central German Salt mines by Brendel et al. (1982).



Fig. 15: The flooded Reiche Zeche shaft of the Freiberg/Saxony mining field (March 2nd 2006). The brown colour is a result of ochre precipitates as a consequence of the catastrophic flooding of August 2002. Width of picture approximately 2 m.

Uncontrolled flooding usually happens in the case of accidental mine flooding, which can occur due to mine water intrusions (section 5.5.4) or where no damages to the ecosystem or engineering construction are assumed. Other examples are mines where the mine budget is unclear, or crisis or wars begin and pumping has to be stopped. Usually, the mine water pumps are stopped at a certain time and neither a geotechnical nor a chemical or physical monitoring system is installed.

For reasons of completeness, it should be mentioned that there are situations where the flooding of a mine or of a mining region would cause or have already caused detrimental impacts on the eco- and anthroposphere. Such situations will occur in the German Ruhr, the Dutch Limburg and the Polish Silesian coal mining areas, where decades of mining lowered the ground water table and cause surface subsidence under the pre-mining ground water level (Bekendam and Pöttgens 1995; Younger 2002b). Flooding of salt or gypsum mines – be it intentional or accidental – can also cause catastrophic situations, as occurred in Polish and German mines (Schwandt 1993; Seifert and Schwandt 1996; Schwandt and Seifert 1999; Zuber et al. 2000).

Flooding prediction is still a highly fuzzy job, needing experience and knowledge of the underground workings. Although several empirical and numerical models exist (e.g. Heidenreich et al. 1991; Younger and Adams 1999; Frydrych et al. 2000; Banks 2001; Adams and Younger 2001; Spiessl et al. 2002; Kalka et al. 2002), the modelling results are still highly

uncertain and are only truly satisfactory for simple mine geometries (e.g. Aurada 1970; Brown 1982; Hanzlík and Vydra 1985; Rogoz 1994; Banks 1994; Gatzweiler et al. 1997; Younger 2002d; Whitworth 2002). One of the rare examples, where prediction and reality show a good correspondence, is the Whittle mine of the Shilbottle coalfield. Adams and Younger (2001) investigated the hydrological and hydrogeological situation at this mine site and used the GRAM code to predict the flooding scenario in 1998. As has been shown by Younger (2004a) the model predicted that the water would reach the 50 m level by around May 2002 and in fact the prediction was a 'precision landing'. More details on water rebound will be given in section 5.9.

5.5.2 Surface Mines

Because some surface mines were or will be converted to underground mines (e.g. Ronneburg/Germany, Chuquicamata/Chile, Palabora/South Africa, El Valle and Carlés gold mines/Spain, Aguablanca nickel mine/Spain) it might be that both mines are hydraulically connected with each other. For those mines it is necessary to have at least an idea of the relevant processes in surface mines. Therefore, this short section gives some general aspects that are important for flooded surface mines.

Surface mines, also called open pit mines, and opencast mines, are used for near surface raw material deposits if the costs for removing the overburden are less than the expected profit from selling the raw material. Surface mines are used to extract large raw material deposits such as coal, gold, copper, or industrial minerals like gypsum, limestone, or hard rock. After mining ceases, they are often flooded, forming pit lakes (Fig. 16). Though open pit lakes are not the focus of this publication, they should be mentioned for completeness because they are unique water bodies that attract different kinds of creatures: bacteria, fish, birds, and scientists. Since the 1970s, the number of open pit mines have increased steadily due to an increasing demand on raw material resources, the increasing technical possibilities to extract ores also from low mineralised ore bodies such as copper-porphyry type deposits, and costs. Leading surface mining countries are the USA, Canada, Australia, Chile, Brazil, Peru, Mexico, Indonesia, the Philippines, Papua New Guinea, West Africa, and Kazakhstan (Castro and Moore 2000).



Fig. 16: Spreetal Nordost open pit lake in the abandoned Lausitz/Germany lignite mining area. The final water level will be about 8 m above the current water surface.

Depths of open pits range from several meters to several hundreds of meters. One of the deepest open pit lignite mines is located in the Western German area around Cologne while the deepest metal open pit mines are the Chuquicamata/Chile and Bingham/USA copper mines. Open pits near Cologne reach a depth of up to 500 m while the Chuquicamata open pit

mine has a depth of 850 m and is projected to reach 1100 m by 2013 (Rowan McKittrick, Water Management Consultants Ltda., Santiago de Chile, pers. comm.; Ossandón et al. 2001; Arancibia and Flores 2004). The Bingham copper mine is currently deeper than 900 m (Borden 2003). Besides the depth, the aerial size of the open pits can vary considerably from several hundred square meters to even square kilometres.

In contrast to underground mines, the hydrodynamics of open pit lakes, their geochemical development, and the prediction of open pit filling were studied by many researchers using numerical, geophysical, geochemical, and tracer techniques (e.g. Prediction Workgroup of the Acid Drainage Technology Initiative 2000; Castro and Moore 2000; von Rohden and Ilmberger 2001; Howell 2002; Mudroch et al. 2002; Ardejani et al. 2003). A review of pit mine hydrology and geochemistry is given by Howell (2002) and in a special issue of IMWA's *Mine Water and the Environment* journal (Howell 2003).

Flooding of surface mines is a common procedure after production ceases or the operations discontinue. In many countries, guidelines for surface mine flooding were developed (e.g. Länderarbeitsgemeinschaft Wasser [LAWA] 2001) and besides the hydrological prediction, geochemical predictions of surface mine flooding were published (e.g. Klapper and Schultze 1996; Klapper et al. 1996; Prediction Workgroup of the Acid Drainage Technology Initiative 2000; Geller et al. 2000a). The inflow into an open pit mine can be modelled in several ways, using an equivalent well approach, two-dimensional flow equations, or numerical techniques (Finite Element Method, Finite Difference Method, Boundary Element Method). For an operating mine, such models can also be used to estimate the depression cone of the dewatering procedures (Singh and Reed 1988).

After stopping the dewatering pumps or the drainage wells, the open pit gradually fills with ground or surface water and a pit lake will establish. Surface waters (precipitation) and runoff water from the pit slopes can be captured and stream water can be diverted or pumped into the pit lake. Ground water can be water from a ground water body connected to the pit or from old abandoned underground mine workings. Water tables usually equilibrate with the ground water table, but they might be higher or lower depending on the local meteorological conditions. High precipitation causes pit lake water levels to be higher than the ground water table, whereas high evaporation lowers the pit lake level. If highly polluted pit water is expected, pumps might be installed in the pit to lower the pit water table, to prevent the contaminated pit water from interacting with uncontaminated ground water (e.g. the Aznacollar pit, where the pit lake level is kept 6 m below the ground water table; Schultze et al. 2006; Kurt Friese, pers. comm.).

Most of those man-made pit lakes are similar to natural lakes and the hydrodynamic and geochemical evolution of such lakes has been studied intensively during the last couple of decades. GeoRef lists more than 400 and the ISI database more than 100 entries concerning pit lakes, but there are few general publications on mining pit lakes. Geller, Klapper and Salomons discuss many aspects of acidification and remediation as well as water quality management of acidic mining lakes (Geller et al. 1998; Klapper and Geller 2001). They also address numerical and empirical models that were developed to predict water table recovery within and around pit lakes.

Water quality in such pit lakes varies over a broad range, from drinking water quality to highly acidic mine water, and it is known that polluted open pit mine water has caused lethal diseases to birds or fish. Besides water quality, the depth and the physical properties of such lakes vary over a broad range. They can be only several meters deep, or reach depths of up to 500 m. Typically, such pit lakes have a low surface area to depth ratio with values below $10 \text{ m}^2 \text{ m}^{-1}$, compared to natural lakes with surface area to depth ratios above $2000 \text{ m}^2 \text{ m}^{-1}$ (Doyle and Runnells 1997; Castro and Moore 2000; Howell 2002). This might cause a stable

stratification of the pit water body and the depths of the epi-, meta-, and hypolimnion might stay stable for a long time. Some studies have presumed that, due to the reducing conditions at the bottom of stratified mining lakes, these lakes could be used as mine waste disposal sites (Doyle and Runnells 1997; Levy et al. 1997).

Not all mine pits are kept open, or are flooded. Alternative uses might include soil reconstruction, forestry, recreation, agriculture, or wildlife habitats (Brenner 2000; Kennedy 2002). Some open pits are backfilled with overburden from other open pits or with reactive material to prevent acid mine drainage. Another option is to backfill pyrite rich overburden or tailings in the pits to prevent the pyrite from weathering under oxidising conditions. Howell et al. (1998), and Howell (2002, 2003) give guidelines on environmental issues that have to be considered and list procedures on how to abandon open pits.

5.5.3 *Underground Mines*

Underground mines have been used since Neolithic times to extract many different types of raw materials. While the first Neolithic underground mines only reached several meters of depth, Bronze Age underground mines went down more than 100 m, and modern mines, like the planned AngloGold's Western Ultra Deep Levels mine (WUDLs) extend up to 5,000 m (Zschocke et al. 1932; Holgate 1991; Shepherd 1993; Vieira et al. 2001; Lynch 2002; Fig. 18).

During the last decades the number of surface mines steadily increased while, at the same time scale, numerous underground mines were closed. Yet, it can be assumed that in the future underground mines will probably be more attractive, because the depths of surface mines are limited, while the depth in which raw material deposits can be found theoretically reaches the thickness of the upper crust. Once the economically extractable near-surface raw material deposits are exploited, underground mining will re-emerge and the knowledge of ecologically safe closure procedures for such mines will become more important than ever. An example of an open pit mine that will be converted to an underground mine is the Chuquicamata mine in Chile, which will start to work underground in the year 2013 (Arancibia and Flores 2004). Río Narcea's El Valle and Carlés gold mines as well as the Aguablanca nickel mine (Asturias/Spain), have been converted to underground mines since 2004 (Paul Younger, pers. comm.; Río Narcea web page).

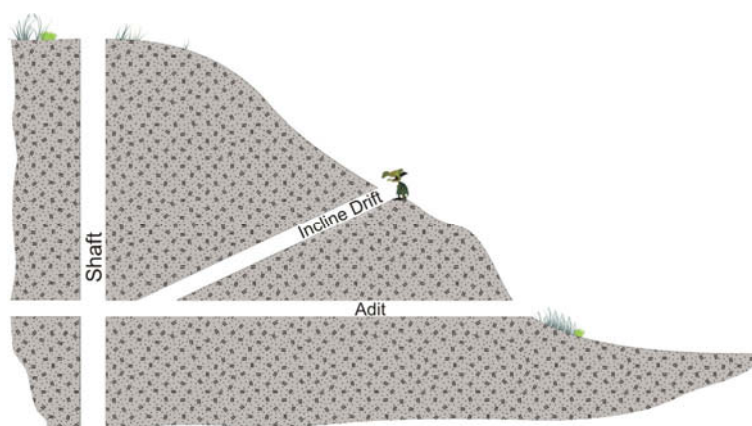


Fig. 17: Principal mine types: shaft mine, incline mine with incline drift, and adit mine.

There are three different types of underground mines: adit mines, incline mines, and shaft mines – and of course combinations of those types (Fig. 17). Their hydrological and hydrogeological characteristics differ significantly and the closure procedures for each require different techniques. From an environmental point of view, adit mines should preferably be

closed by constructing dams (bulkheads) at the mine entrances to allow the mine to flood, re-establishing a water table that will resemble the pre-mining water table. Unfortunately, this closure technique requires reliable engineering constructions whose stability cannot be guaranteed in all circumstances due to highly acidic drainage waters and extreme water pressures. Also, the rising water table can cause unforeseeable non-point discharges of polluted mine water (ERMITE Consortium et al. 2004). Therefore, the construction of dams in adit mines always needs extensive investigations to exclude environmental problems, personal injuries, or damage to engineered constructions. If the situation is unclear, the full closure of such mines should be avoided (ERMITE Consortium et al. 2004). More details about such physical barriers can be found in section 5.7.3.

After a mine has been abandoned, the maintenance of the wooden and steel support stops or pillars start to weather and, depending on the physical properties of the support its stability will gradually decrease. If the rock pressure, caused by the static or dynamic loading, exceeds the supportive forces, the support fails and the workings collapse, eventually giving rise to surface subsidence. In unsupported parts of the mine, especially in the stopes and adits, overburden pressure decreases rock stability and, depending on the extraction ratios, causes roof failures or pillar crushings as well. Alternatively, the buoyancy forces of the rising mine water will cause the pore pressures to decrease and the overburden pressure and the potential for subsidence might be minimised; even uplift can occur. Although such effects have been known since Terzaghi's pioneering soil mechanical investigations (Terzaghi 1925), only minor systematic research has been conducted on this topic (e.g. Oberste-Brink 1940; Goetz et al. 1994; Hunt and Reddish 1997). While Goetz et al. (1994) found proof of positive buoyancy effects, Hunt and Reddish (1997) think that those effects are negligible. Yet, as the mine flooding in Limburg/The Netherlands shows, vertical lift of up to 0.2 m can occur in areas that are affected by the raising water table (Pöttgens 1985; Bekendam and Pöttgens 1995). Oberste-Brink (1940) investigated the surface uplift in the abandoned Witten/Germany hard coal mining area. He noticed uplifts in a mine ranging up to several centimetres.

The geophysical records of flooded mines show that in tectonically weak areas of the mine the number of micro earth quakes usually increases during flooding (Flach and Heick 1986; Rudolph 1997; Ogasawara et al. 2002), and normally come to a halt after flooding is finished. Such micro-seismic events within a mine can induce significant changes in the mine water level. One of those events was observed in the Niederschlema/Alberoda uranium mine on November 21st 1995, where a micro-seismic event dropped the mine water level by 34 cm. Even 14 years after the start of the flooding process, the micro-seismic events induced by the flooding did not cease, though their magnitudes steadily decreased (Krentz et al. 2004). Flach and Heick (1986) observed that micro-seismic events were clearly linked to the flooding, especially when the quantity of flooding water into the mine changes, and that the number of micro-seismic events decreased after flooding. These measurements verify the observations of Goetz et al. (1994) that during the flooding process, areas that were at the limit of their stability will fail and, after flooding, the buoyancy forces will stabilise the mine workings.

One of the main reasons for the flooding of underground mines is the prevention of disulphide oxidation and consequently acid mine or rock drainage. Although not recorded in detail, acid mine drainage causes world-wide damage in the range of several billion Euro. Literature about acid mine drainage is numerous and the reader interested in the details of acid mine drainage is referred to Stumm and Morgan (1996), Younger et al. (2002), Lottermoser (2003), or Jambor et al. (2003) for further reading (see also chapter 4). Disulphides, namely pyrite and marcasite (FeS_2), but also the monosulphide pyrrhotite (FeS) are the crucial minerals for acid mine drainage formation (the chemical formula of pyrrhotite is FeS , yet, most natural pyrrhotites are found to be $\text{Fe}_{1-0.83}\text{S}$, usually written as Fe_{1-x}S , Strunz and Nickel 2001). In the presence of water and oxygen, and catalysed by bacteria (see section 4.4), these disulphides

weather to form acid conditions as well as ferric and ferrous iron (Stumm and Morgan 1996). In addition, the acid further accelerates dissolution of other minerals, resulting in an increased trace metal concentration in the mine water. Those products of the disulphide oxidation cycle can remarkably impact the environment by forming highly acidic mine drainage and after leaving the drainage adits dyeing the streambeds, natural wetlands, and overflow plains with an unpicturesque red to yellow colour. Iron concentrations in highly acidic mine waters can reach several thousands of milligrams per litre and pH-values of -3.6 were measured in the Californian Richmond Mine (Nordstrom et al. 2000). Though, acid mine water can be treated by several techniques involving passive or active treatment options (Younger et al. 2002; Brown et al. 2002). The best option is to prevent the formation of acid drainage at its source: in the mine. Mine flooding isolates the disulphides from atmospheric oxygen and therefore significantly reduces the kinetics of acid formation. Due to the low diffusion coefficient of oxygen in water as well as the low oxygen content and redox potentials of ground water, mine flooding reliably limits the formation of acid mine drainage.



Fig. 18: Unflooded access Gallery in the Western Deep Level/South Africa underground mine. The Gold Reef comes in from the top right side and dips to the lower left corner. Concrete support helps to support the geomechanical stability in the 2500 m deep gallery.

Safety reasons are another motivation for mine flooding and – of course – mine sealing. Non-flooded or unsealed underground mines are frequently visited by people who are not aware of the dangers they might encounter underground, and mine owners can be made legally responsible for personal injuries. Mine gases (mainly Rn , CH_4 , H_2S , CO_2 , CO) are another reason for mine sealing (those gases are usually not much of a problem during the active mining period because of the mine air ventilation). Especially in remote areas of the USA or Australia, deadly accidents are reported frequently, and programmes to close open shafts or adits can be found throughout these countries (e.g. Suman et al. 2000). Mining archaeologists investigating the history of abandoned mines, be they professionals or semi-professionals, are usually well equipped and accidents therefore are rare (Leupolt and Hocker ca. 1999). Mine owners therefore should try to work together with local interest groups to find closure solutions that fit the needs of mining archaeologists, environmentalists, as well as health- and safety regulators.

Economics and the depletion of the deposit bare the most obvious reasons for mine flooding. Dewatering pumps, especially in deep underground mines, are extremely energy consuming and therefore add significantly to the costs of underground mining. It is therefore in the interest of the mining company to stop dewatering pumps as early as possible after the productive period. For underground mines involving techniques that do not include surface subsidence,

including most metal mines, such a procedure usually presents no danger for the people living in the vicinity. In contrast, longwall mining operations, where the roof is allowed to collapse behind the advancing face, induces surface subsidence, as can be seen in many coal mining areas of Britain, Poland, Germany, Australia, and the United States. Switching off dewatering pumps in those areas, which include very heavily populated regions, would mean the flooding of many well-known industrial zones (Younger 2002b). A compromise approach is to collect and discharge relatively uncontaminated infiltration water to avoid high water treatment costs; this approach has been used during the flooding of the Eastern German uranium mines.

The controlled flooding of an underground mine should include the following precautions:

- removal of potential water hazards in the mine
- backfilling of near-surface structures that might subside or collapse during or after the flooding process
- installation of water diversion systems (see section 5.7)
- grouting of tectonic structures that are known to be connected to drinking water reservoirs
- installation of a hydrogeological and geotechnical monitoring system underground and at the surface
- prognosis of hydrogeochemical and hydrological development of mine water

5.5.4 Mine Water Inrushes

Besides mine gas explosions, mine water inrushes are the most common reason for catastrophic accidents in underground mines and still claims many victims. Though mine water inrushes due to insufficient pumping capacities are relatively rare – compared to historical times – they still occur. Recent cases are the Russian Sapadnaja-Kapitalnaja coal mine (October 23rd 2003), the Chinese Shentang coal mine (June 22nd 2002), the Chinese Xuzhou coal mine (January 1st 2000, Anonymous 2000) and other Chinese mines (Anonymous 2002; Bei and China Daily 2004). Singh (1986), Sammarco (1986), Job (1987), Vutukuri and Singh (1995), Kroker and Farrenkopf (1999) as well as Younger (2004b) give in-detail-studies of such accidents and some of them also discuss their prevention.



Fig. 19: Reasons for mine inundation (modified after Job 1987; Vutukuri and Singh 1995).

By statistics, abandoned panels or parts of a working mine pose the highest risk of accidental mine water inrushes (Fig. 19, Job 1987; Vutukuri and Singh 1995). Job (1987) investigated

208 water intrushes in British coal mines between 1851 and 1970. He was able to show that 78% of all those accidents were due to contact with abandoned old workings, whereas he identified 11 distinct reasons for such accidents: ineffective barriers, incomplete old plans, absence of plans, incorrect plans, incorrect interpretation of old plans, failure to obtain the abandonment plans, absence of protective boreholes, ineffective protective boreholes, failure to plug borehole, unknown old workings, and incorrect seam correlation. One of the latest of such accidents was the Quecreek/USA accident (July 24th 2002) where the barrier pillar between this mine and the adjoining Saxman coal mine breached, entrapping 9 miners which could be rescued 78 hours after the accident (Pauley et al. 2003).

5.6 Remining of Abandoned Mines – A Special Type of Remediation

Many abandoned underground mines, waste rock piles, or tailings ponds still contain a great amount of unprocessed ore and acid-producing minerals. This source of acidity may have been left due to the mining method, for example in the pillars used for support, because the concentration of the commodity was below an economic concentration, or because no appropriate mining or extraction method was available to extract it at the time of first mining. It is common though to reopen such mines if the economic or technical conditions that caused the closure of the mine change. Though remining, strictly speaking, means the reopening of an existing abandoned or closed mine, this section will also cover daylighting of underground mines (which means that a former underground mine is converted to an open pit mine, Fig. 20), and the reprocessing of mining waste disposals or tailings. Furthermore, as will be shown, remining might significantly contribute to a reduction of the pollution loads and is therefore described here.



Fig. 20: Remining of an abandoned underground gold deposit in the Kalgoorlie/Western Australia Super Pit by daylighting through Kalgoorlie Consolidated Gold Mines. Some of the old mine workings can be seen in the middle of the photograph. The remining contributes to a reduction of the pollutant loads as potentially acidic strata are excavated, processed and disposed of.

Following the definition given by Hawkins (1995), remining would only describe mining operations aimed to improve pre-remining water quality. Another definition is given by the U.S. Environmental Protection Agency et al. (2001): “Surface mining of abandoned surface and/or underground mines for which there were no surface coal mining operations subject to the standards of the Surface Mining Control and Reclamation Act. Remining operations implement pollution prevention techniques while extracting coal that was previously unrecoverable”. Neither definition would cover the reopening of non-coal mines. However, in this pub-

lication, “re-mining” will be seen in a slightly broader context and covers all of the above definitions.

Remining was also conducted in the German Ruhr Area near Witten/Ruhr, where the history of German coal mining started at the end of the medieval times. At least three mining periods have been recognised there and all of them reflect the stage of technical possibilities at their time (Koetter 2001). Each subsequent period reused the old abandoned workings to mine coal that was uneconomic in the previous period or which could not be extracted because of technological reasons. Remining has been covered by several official publications in the USA and a lot of good guidance books or publications are available (e.g. Smith 1988; Hawkins 1995, 1998; U.S. Environmental Protection Agency et al. 2001).

Unfortunately, at least since the German Rammelsberg court cases (Anonymous 1994, 1995), mining companies refrain from remining abandoned mines because they will become responsible for the historic pollutants caused by the mine that they used for remining (Amezaga and Younger 2002). This situation differs from the Pennsylvanian legislation, where remining (*sensu* Hawkins) started in 1983. There, the mine operator will not be responsible for the pre-mining pollution as long as he does not further degrade the water quality. Should the pollutant loads after cessation of the remining activities be below the previous pollutant loads he will be released of liability concerning this pollution (Hawkins 1994). Due to this legislative support Pennsylvania has become one of the most intensively remined states in the USA and possibly on a world-wide scale. Approximately 91% of all remining projects in the USA, a total of more than 300, are located in Pennsylvania (U.S. Environmental Protection Agency et al. 2001) and the history of the legislative procedures needed for remining in Pennsylvania is well documented in the earlier cited publication. Furthermore, remining can be seen as a remediation technology for abandoned mines which – usually – does not to have paid for by the tax payer but by the industry.

In the case of the Tasmanian Mount Lyell copper-gold deposit, the Australian and Tasmanian Governments formed a new company, Copper Mines of Tasmania, which should have remined and remediated the heavily polluted Mount Lyell mining area (Sustainable Development Advisory Council and Copper Mines of Tasmania 1995; Copper Mines of Tasmania 1995). Yet, even after special legislative measures were taken to regulate the non-responsibility for the pre-mining period, the plan failed because of different reasons and the Mount Lyell area still endures non-remediated tailings and acid mine drainage.

Nonetheless, many remining projects prove that the environmental impact of acid mine water discharges can be positively influenced or at least generally do not worsen (Perry and Evans 1999). The reasons for this are twofold: remining removes the left-over pyritic strata (very often coal or lignite) and due to the more stringent modern reclamation standards, potential pollution sources have to be removed, covered or reclaimed in other ways. Furthermore, daylighting often exposes and disrupts alkaline strata that provide alkalinity; pH improvements from about 3 to 7 are known (The Pennsylvanian Department of Environmental Protection 1998). Anyway, the conversion from an underground mine to a surface mine will usually produce better water quality because surface mines typically leave less pyritic material and thus have better water quality than underground mines (Fig. 21). The few remining projects which resulted in higher pollution loads are negligible (Hawkins 1994, 1995; Prediction Workgroup of the Acid Drainage Technology Initiative 2000; West Virginia Division of Environmental Protection – Office of Mining and Reclamation 2001)

After remining, it is comparatively simple to count the hectares of land reclaimed, the numbers of trees replanted or the amount of tailings or waste rock piles covered. Those measures can therefore relatively quickly be checked against the necessary remediation requirements. Conversely, a thorough statistical investigation of the pre-mining and post-mining situa-

tion is essential to determine whether remining activities changed the pollutant loads. As in other mine water cases, a thorough understanding of the hydrogeological and hydrological situation of the mine site as well as statistical methods are required. Especially the pre-mining baseline pollution load has to be known and evaluated precisely (Smith 1988). Some of the simpler statistical techniques that must be used are normality testing, exploratory data analyses, and ranked correlation coefficient analyses (Hawkins 1994, 1995). Furthermore, time series analyses, T-tests, or ANOVA tests would be necessary to get statistically significant differences in the data sets (see also section 9.3.4).

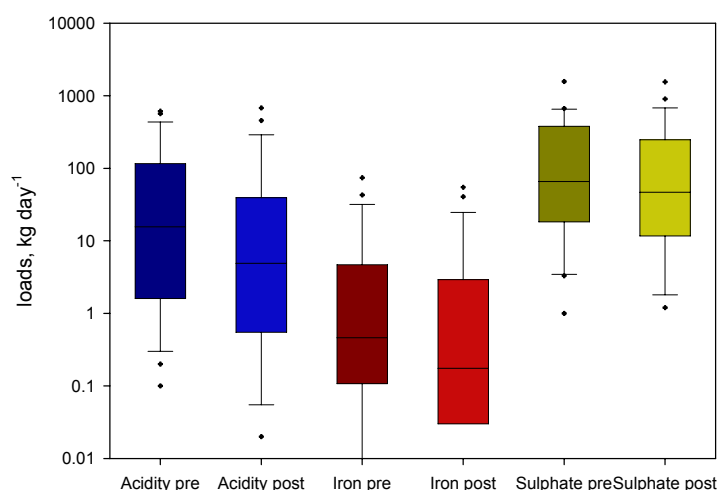


Fig. 21: Box-plot of logarithmic loads showing the mine water quality improvements in selected surface and underground mines after remining procedures were conducted; pre: pre-mining, post: post-mining situation (modified after Hawkins 1995).

From a mine water management perspective, the sealing and elimination of abandoned workings is the most important aspect of remining. During remining, old abandoned workings are exposed at the side walls of the open pit. To prevent pyrite oxidation and subsequently the formation of acid mine drainage in these abandoned workings, it is essential to seal them with low permeability material. Yet, care must be taken so that the seals will not be blown out if extreme pressures build up behind the seals. Should this be a concern, it is advisable to install drainage pipes through the seal to reduce the water pressure behind the seal (U.S. Environmental Protection Agency et al. 2001).

From an economical point of view, remining activities are usually cost effective, especially if the operator would have had to treat the discharge water of a previous underground mine. Skousen et al. (1998) summarises some remining projects, including ecological and economical data. The REMINE software package is used by U.S. agencies to calculate the effects of a remining operation (U.S. Environmental Protection Agency and Office of Water 1988). As a conclusion, remining can be seen to be a cost-effective and environmentally sound measure for the remediation of abandoned underground mines.

5.7 Diverting Water

5.7.1 Why Divert Water?

Compared to most geological processes, which might last millions of years, mining is a short term use of our landscapes that can cause substantial change to our environment. Though the changes might be acceptable, or even desirable, pollution of river catchment areas must al-

ways be excluded, because according to most national laws (e.g. the European Water Framework Directive), no negative influences on our surface and ground water resources are allowed. It is therefore important during a mine's closure phase to prepare the mine and to guarantee a good status of the environment after the mine's abandonment. In the following section, some guidance to minimise post-closure impacts during the closure phase will be given.

Because treatment of mine water might be a long lasting and expensive task, it is advisable to divert surface or ground water from the potentially polluting sources in and around the underground mine. This is already a common practice. Nonetheless, this method is neither practicable nor environmentally advisable in the case of huge open pit mines, where the mined area has to fulfil its function as an aquifer after mine closure. In other cases, the diversion of surface or ground water will generally minimise the amount of water that has to be treated but, there are cases, where the mixing of highly acidic mine water with neutral or well buffered surface water might have a positive effect on the mine water quality. In the latter case, the mixing of the two waters could significantly improve the mine water quality causing the pH to rise and pollutants to precipitate, thus improving mine water quality.

Obviously, the formation of polluted mine water could be prevented:

- the reactants, oxygen or water, are not allowed to get into contact with the disulphides
- water is not allowed to get into contact with the secondary minerals
- water chemistry is changed in such a way that dissolution processes are hindered
- minerals are removed or covered by inert material

To exclude the contact of unpolluted water with potential water contaminants, several methods were used. Most of those methods were introduced to avoid ground water flowing into a working mine (Morton and Mekerk 1993), but they can also be used to minimise post closure pollution:

- underground dams
- diversion ditches or channels
- diversion wells, pipes, or adits
- grout curtains
- grout injection
- cover drilling
- backfilling of open voids
- backfilling with alkaline fly ash to buffer the acid and seal the ore
- mine entrance sealing

Last, but not at least, neither diversion nor mine sealing installations are walk-away options for mine closure. Depending on the circumstances, technical installations might need longer maintenance or control according to a Quality Control Plan. Sometimes, technical measures, such as backfill operations, can offer solutions without maintenance (Potvin et al. 2005).

5.7.2 General Measures to Keep the Water Away

Diversion of water could serve the following main goals:

- prevent the contact of acid producing material with the water
- minimising mine water intrusions into opencast and underground mines
- minimise the amount of water that has to be treated

- mixing of acid mine drainage with alkaline surface water

In the first two cases, the surface or ground water has to be diverted around the potential inflow zones. Such zones could be sink holes, seeping places, shafts, fracture zones, or bore holes (e.g. Scott and Hays 1975; Ackman and Jones 1988; Plotnikov 1988; Younger et al. 2002). If dams are constructed within the void space before inundation, the dams have to ensure that the water will either be prevented from leaving potentially polluting areas or will be guided around such areas.

Mine diversion should be avoided where it will prevent mine flooding that could result in a better ecological status of the catchment area. Such decisions must be based on appropriate hydrogeological and hydrogeochemical investigations and a thorough monitoring programme.

If rocks are highly fractured or karstified and where a significant surface water inflow into the mine voids is known, diversion can be an extremely useful tool to minimise post closure impacts. This method, therefore, is highly recommended if a high acid producing potential in the mine voids is expected during and after mine closure.

Means to divert surface water from mines are channels, pipes, or ditches. Such systems can be extremely large, sometimes several kilometres, and can affect different water catchments. They have to guarantee that the surface water will not percolate into the mine and that the water is diverted around the mine as fast as possible before it might enter the mine. Therefore, such systems must be securely sealed. The capacities of ditches, channels, and pipes must be evaluated by hydrological investigations or calculations. Anyway, the destruction of such systems by storm events should be prevented, because their destruction could cause slope failures or sludge movements (see section 5.4.2). Regular inspection and periodic maintenance for such diversion systems is highly recommended – if not mandatory.

Besides surface water diversions, ground water diversions can be installed. Such systems consist of grout curtains, grout injection, or large subsurface concrete walls, as has been constructed in the Eastern German lignite mining areas (Fahle and Arnold 1996), where they protect nearby wetlands from ground water dewatering and prevent ground water flow into the opencast mines. In the Idrija/Slovenia mercury mine, grouting of abandoned mine workings prevents pollution and helped to prevent surface movements (Dizdarevič and Režun 1998). Similar techniques can be used during the mine closure phase if potential pollution sources are encountered. Cover drilling, normally used for developing new adits or shafts, can also be used to prevent unpolluted ground water from flowing into mine voids (Nel 2000).

From tracer investigations in already flooded mines, it is known, that underground dams constructed before flooding can fail. This might be due to pressure differences on both sides of the dam, which could result in the dam's damage during the flooding process (Wolkersdorfer 2001). As the case of the M. Mayerova Mine/Czech Republic shows, grouting after the flooding of the mine might be a successful way to stop drainage of polluted water (Kipko et al. 1993).

Sometimes, the diversion of alkaline surface water might serve as an on-site remediation measure for acid mine water. By mixing alkaline and acidic mine water, the pH of the water can be raised to ecologically reasonable values as reported in the Montana/USA Stillwater River basin, where fresh surface water enters a highly contaminated mountain stream and thus improves the biological diversity (Gurrieri 1998). Although such techniques can reduce the concentrations of iron, toxic trace elements, metals, or semi-metals by hydrolysis and co-precipitation, they cannot minimise conservative water pollutants, such as sulphate, chloride, or sodium. Such a procedure can sometimes reduce the pollution load and improve the ecological status of the water; it is advisable not to apply the “dilution prohibition” rules of the

waste directives, because this method must be seen as a passive treatment option for polluted mine water. At abandoned sites, it may be the only affordable option.

In the case of surface water diversions, it is important to know the catchment area of the underground mine so that the mine's water make might be calculated. In those cases where exploration boreholes were converted to observation wells (piezometers), the mine's catchment area can be calculated to a certain degree (Plotnikov et al. 1989). However, if not enough observation wells are available, the mine's catchment area can only be estimated since the inflow from "old" deep ground water can cause uncertainties in the calculation. Though numerical codes are an excellent tool to help the researcher in estimating the mine's recharge area (e.g. Singh and Reed 1988; MacLeod et al. 1993; Lewis et al. 1997; Dumpleton et al. 2001; Witkowski et al. 2003), they are nothing more than another way of producing a conceptual model of the mine's surroundings. Dumpleton et al. (2001) calculated two different flooding scenarios for the Nottinghamshire coal field with surface break outs of the mine water 19–20 and 52 years after the end of dewatering. They clearly state that neither scenario can be "identified as an optimum management proposal", showing that numerical models can be regarded as also being a working hypothesis.

The following example shall exemplify how complicated it might be to calculate the catchment area of an underground mine. It is based on a project draft of the German "Wismut" company predicting the flooding of the Niederschlema/Alberoda/Germany uranium mine in the early 1990s.

The mine's catchment area was thought to be 22.6 km², in the river Zwickauer Mulde catchment area. Together with the rivulets Schlemabach, Lößnitzbach, Silberbach, Floßgraben, and Alberodabach, the catchment has a total area of 115 km². Based on geological investigations and hydrogeological observations underground, it was known that most of the water infiltrated through the upper 300 m of rock. A mean precipitation of 825 mm a⁻¹ and a mean surface runoff of 38.7% as well as a mean annual temperature of 8.5 °C were reported (Veit et al. 1987). Thus, by using the equation of Turc (Gray et al. 1970) for calculating the real evapotranspiration, a total annual mine water make of $1.02 \cdot 10^6$ m³ was calculated. However, the annual water make known from mine dewatering was $7 \cdot 10^6$ m³ per year. Using this number to back-calculate the mine's catchment area, an area of 155 km² would be needed to get the actual annual water make of $7.0 \cdot 10^6$ m³. Interestingly, by using Saul's observation, that 20–40% of the annual precipitation above the mined out area can be used to estimate the total water make (Saul 1948), the numbers would be 3.7 – $7.5 \cdot 10^6$ m³. It was therefore suggested that stream water is infiltrating into the mine or that the mine affects an area bigger than the 22.6 km². Having such a big difference between the observed and calculated mine water makes, and the suggestion that stream water infiltrates into the underground mine, flow measurements in the surface water streams to evaluate potential inflow zones would be necessary. Tracer techniques can be used successfully for such an investigation (chapter 10 and e.g. Kimball et al. 1999).

5.7.3 Physical Barriers

Under normal conditions, all parts of a mine are well connected with each other to guarantee quick, safe, and unhindered movement of material, workers, water, and air (Gertsch and Bullock 1998). Therefore, any fluid (which might be water, oil, or air) will pass through the horizontal and vertical connections within a relatively short time. Unused parts of a mine are commonly back-filled after the ore has been extracted (Landriault 2001; Potvin et al. 2005) but only in the case of cemented or hydraulic backfill will the hydraulic connections between the different parts of a mine be significantly reduced. Usually, the former open voids will be a

preferential flow path for the mine water, because the backfill still has a porosity that exceeds that of the surrounding host rock.

Consequently, the water in flooded mines can flow without restrictions into nearly all parts of the mine workings and water pollutants might rapidly be transported through the mine by Poiseuille or free and forced convective flow. Additionally, the pore space of the dewatered rocks and sediments will be filled with water and Darcian flow will occur. Flow will also occur in fractures and faults.

Physical barriers, which are all types of constructions that manipulate or prevent the flow of fluids in an underground mine, delay or prevent the water flow in the horizontal or vertical workings (Skousen et al. 1998). Such installations are commonly called bulkheads or dams and in working mines they are constructed to isolate abandoned workings, to control the water make of the mine, to manipulate the ventilation system, to exclude dangerous gases such as radon or methane or to prevent unwanted access (Fig. 25). Of course, there are other reasons for constructing dams with no direct connection to proper mining behaviours. Physical barriers could be made of steel, concrete, sand, or wood, or combinations of those materials, depending on their function and the hydrostatic pressure they have to resist. Another method to control the water flow into the mine or within the mine is grouting, where different types of cement mixtures can be used depending on the water chemistry or pressure. Only rarely have physical barriers been constructed to control the water flow during and after mine flooding, though their regulative character is obvious and has been studied by several scientific groups. If *in-situ* treatment of a flooded underground mine is planned (section 11.6), bulkheads should be constructed after a thorough hydrodynamic investigation of the mine (chapter 10). It must also be clear that bulkheads can have negative effects if constructed poorly or at an inappropriate position within the mine.

Bulkheads or open dam walls are constructed for reasons of control, emergency, precaution, or consolidation. Chekan (1985) discusses detailed conventional bulkheads to control the water flow and water level in mines. He distinguishes five types of bulkheads:

- Control bulkheads
- Emergency bulkheads
- Precautionary bulkheads
- Consolidation bulkheads
- Open dam walls

Control bulkheads aim to block abandoned parts of still active mines and shall prevent water inflow from behind the dam into the worked mine (Fig. 22). Usually, there is enough time available to carefully plan and consider the layout and position of the bulkheads. They have to withstand high water pressures and usually they are permanent constructions with no access to the abandoned panels or workings. Often, they include pipes with valves and pressure gauges (in American English: gages). Emergency bulkheads are constructed in case of accidents, to protect the mine from unexpected water inflows. Their position and dimension is determined by the place of the inrush. No access to the workings behind the dam is planned and they also have to withstand high water pressures. Precautionary bulkheads are constructed well in advance of potential water inrushes at locations that shall separate the main haulage roads and mine entries. They can be shut in case of an accident and secure the mine's safety. Water pressures behind such constructions might be extreme. Consolidation bulkheads are temporary constructions which keep away water during grouting works or ground consolidation structures. Open dam walls, finally, partly seal an adit and are a means to store mine water for reasons of treatment, conservation, or reuse. Besides those reasons listed under the five bulkhead types in Chekan (1985) some more reasons for constructing bulkheads for water

control might be thinkable. Environmental reasons should be added under the control bulkheads and open dam walls. For advance drilling it might be useful to construct temporary bulkheads mentioned under consolidation bulkheads and for hydrogeochemical investigations precautionary bulkheads could be used. Yet, this is only a classification about the use of bulkheads. More important is a classification according to the construction of the bulkhead in relation to its use. It should be added that bulkheads could also be constructed as flow-through bulkheads to control the water exchange within different otherwise separated water compartments. In the case of expected acid mine drainage, bulkheads have to be constructed by using acid-proof concrete (Bagdy and Sulyokm 1990).

Wolkersdorfer (1996) suggested a method based on the construction of vertical bulk heads to prevent the overall convective transport of contaminants for the Niederschlema/Alberoda uranium mine in Germany. Due to a lack of existing experiences, this method was rejected by the mine operator at that time. Heidenreich et al. (1991) conducted numerical methods to study the flow in a radioactive waste disposal and concluded that physical barriers could positively influence the convective flow regime. One of the rare examples, where physical barriers were constructed before flooding is the Ronneburg/Germany uranium mine. 120 physical barriers shall separate highly contaminated mine water from low contaminated ones and thus minimise the potential environmental impacts and the treatment costs (Gatzweiler et al. 2002).

Lewis et al. (1997) made numerical simulations to find the best places for bulkheads in the Michigan/USA White Pine Copper Mine. They used MODFLOW to model the regional hydrogeological regime and found that nine bulkheads would positively influence the mine water's hydrogeochemistry and flow regime. However, their objective was not to influence the mass transport of contaminants in the mine, but to influence the amount of water draining from the mine. Without bulkheads an annual water make of about $625 \cdot 10^3 \text{ m}^3$ was calculated. Installing the bulkheads resulted in $405 \cdot 10^3 \text{ m}^3$ per year, thus reducing the water make by 35%. Their solution is interesting as the bulkheads aimed to raise the water levels in the different mine compartments and because the cones of depression would be smaller, the water infiltrating into the mine would be reduced.



Fig. 22: An underground dam in the abandoned Königstein/Germany ISL uranium mine to prevent the discharge of highly acidic leachate fluid (January 8th 1992).

In the M. Mayerova Mine near Karlsbad (Karlovy Vary/Czech Republic), $8 \cdot 10^3 \text{ m}^3$ of clay-cement grout were injected into the flooded workings to control the water flow in the mine (Kipko et al. 1993). As a result, the flow was reduced from $5 \cdot 10^{-1} \text{ m}^3 \text{ min}^{-1}$ to $2 \cdot 10^{-4} \text{ m}^3 \text{ min}^{-1}$ and the water temperature decreased from 31°C to 18°C , thus proving the

positive effects of the grouting procedure. This example shows that physical installations can also be installed in mines which are already abandoned and flooded. Depending on the mine water chemistry a proper grout composition must be found. The grout's viscosity and composition has to be selected in that way that it serves several goals: flowing through the disposal pipes without blocking them and building up in the adits that have to be blocked without flowing away with the water stream.

Another physical method to reduce water inflow into flooded mines is the sealing or divergence of surface streams crossing the underground mine. Such measures can minimise the influx of oxygen-rich surface waters and consequently the reaction rates between unoxidised disulphides and oxidising water are reduced. Furthermore, the amount of drainage water is decreased and treatment operations have to deal with lower water flows. Yet, a wrong composition of the grout or the wrong grouting technique could cause unnecessary costs (Ackman and Jones 1988; Morton and Niekerk 1994; Niyogi et al. 1999). Measures to locate potential infiltration zones are electrical conductivity investigations within the stream segments that could infiltrate into the underground mine, tracer investigations, or flow measurements (Ackman and Jones 1988; Kimball et al. 1999). At the abandoned Teichtalstollen near Königshütte/Harz Mountains in Germany a tracer test with microspheres proved that there is a good hydraulic connection between a surface stream and the drainage adit which is used as a drinking water supply (Wolkersdorfer and Hasche 2003).

A different type of physical barrier would be the complete sealing of the surface above a mine, called encapsulation. Preferably, plastic liners or low permeable natural material, such as clay, are exaggerated above the affected area. This method prevents the infiltration of surface water into the worked veins, seams or disposed material and thus the formation of acidity by disulphide weathering could be stopped or minimised. In addition to the clayey material, alkalinity producing material can be used to buffer the acidity. Yet, two aspects have to be kept in mind: the area to be sealed could be extremely large. At a USA surface coal mine in Alton, West Virginia $1.8 \cdot 10^5 \text{ m}^2$ of a combined disposal site for coal refuse and overburden was sealed with a plastic liner. This resulted in a reduction of ground water recharge and a significant diminution in the acidity loads (Skousen et al. 1998). Another problem is to locate the correct area of recharge above the underground mine. Even if this could be delimited to a certain degree, the subsurface recharge area (that means ground water inflow into the mine) still stays unclear until isotopic studies or tracer tests define the full recharge area. Those reasons together with the fact that some parts of the surface cannot be sealed because of slope stabilities or buildings caused such systems being only partly successful.

All of the techniques described above are designed to control the water flow in or around the flooded mine. Thus, the oxygen rich air in the mine is replaced by oxygen poor water and the reaction rate of the acid production consequently reduced. As described above, the flooding of adit mines can cause geotechnical problems due to high potential pressures at the water side of the seals. For situations where a full sealing of an adit might be dangerous, partial physical barriers were developed. They guarantee a free flow of the water, but eliminate the flux of oxygen into the underground mine by insulating the mine against the air and can reduce the acidity by 50—80% within the first months of mine sealing (e.g. Anonymous 1936). Such techniques usually consist of a concrete or brick wall, a U-bended pipe, or a wall combination which allows free flow of water but eliminates the oxygen flow into the mine through the adit (Fig. 23). Because of such a physical barrier, the concentrations of radon behind the wall might increase above the MAC-values, and the oxygen content of the mine air will drop under the values necessary for safe working conditions. Such techniques seem to contradict the needs for bat protection, because bats would need good quality air for living in the underground space. The construction of such air-barriers should therefore be shifted several tens of meters into the adit if bats are known to live in the mine. Yet, as Foreman (1972) already has

pointed out for the mine sealing projects in Butler County, Pennsylvania/USA and Schindler et al. (2004) for the Randolph County, West Virginia/USA, “the effect [air] seals have had on water quality is inconclusive”.

Mine sealing operations usually exclude future options for *in-situ* remediation techniques and they might shift the mine water problem from a point-source to a non-point source problem. Furthermore, bulkheads or dams are engineering constructions which are subject to damages by chemical weathering or by rock mechanical influences – in no way such constructions can be seen as “walk away options” (ERMITE Consortium et al. 2004). Adit mines should not completely sealed, because the breakout of heavily polluted mine water could cause disastrous damages to human life, engineering constructions and the environment. Positive cases of mine sealing operations can be found in the literature (Stoertz et al. 2001), but it is not always clear, if the positive effect is due to the mine sealing operation or the natural attenuation of the mine water.

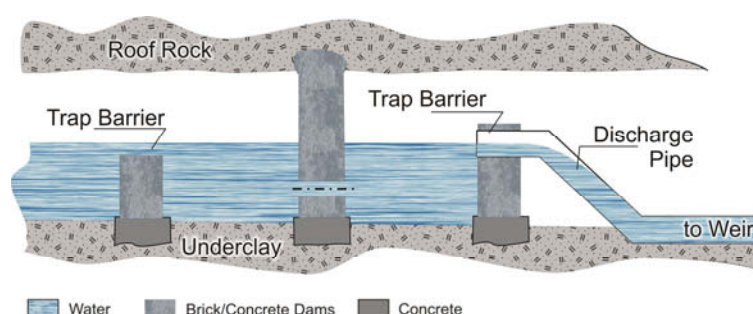


Fig. 23: Partial seal of an abandoned adit mine (after Foreman 1971).

5.7.4 Mine Entrance Sealing

Permanent mine seals become more and more used nowadays, because such systems overcome several problems after mine closure:

- prevent drainage of polluted water
- minimise oxidation of disulphides
- stop the emission of hazardous gases
- prevent unauthorised entrance to the open voids
- protect cultural heritage

Yet, the total sealing of mine entrances might contradict with the needs of protecting endangered species, for example bats (yet they need only small holes to get through), and should therefore be used with care only. As discussed above, the construction of such barriers can be shifted several tens of meters into the adit if bats are known to live in the mine and by doing so, both interests can be met. Furthermore, the total mine seal could activate pre-mining water drainage systems, such as karst features or fractures, which were dry during the operation of the mine. In such circumstances, mine water could drain uncontrolled into the environment and mine water treatment in such cases could be problematic. Such situations can occur when the water level is allowed to raise in an uncontrolled manner to its near pre-mining level.

In the USA, mine entrance sealing studies with the aim to reduce surface stream pollution dates back to the late 1920s and in the late 1930s over 1,000 coal mines had already been sealed with pollution reductions ranging from about 50% to 90% (Leitch 1935; Fellows 1937). One of the most comprehensive description of pollution prevention and control from abandoned mines has been given by Scott and Hays (1975), though most of the techniques

described have been improved meanwhile. Mine sealing, according to this report and other publications, depends firstly on the mine layout (e.g. Foreman 1971). Mines with access through adits or galleries have to be sealed differently compared to mines with access through shafts or inclines.

Before any sealing is conducted, the voids to be sealed should be cleaned of all trash material such as timber, metal, wires, and other material that could hamper an effective sealing procedure. There is no need to remove such material from all the voids in case the mine has to be fully or partly flooded, except wastes that are potentially harmful to the aquatic environment, such as oils, chemicals, or cleaning detergents. Most materials used in mining have a low pollution potential under anoxic conditions. Mine sealing can be conducted in two ways: total mine seal to prevent water from draining into the river catchment or mine seal against oxygen coming into contact with the mined out rocks. Furthermore, sealing measures can be categorised into dry and wet mine seals. Whilst dry mine seals prevent oxygen from entering a dry mine, wet mine seals could be used for both of the before mentioned options (Fig. 26).

Another method, first described by Taylor and Waring (2001) prevents acid mine drainage by injecting reducing gases as CO_2 , CH_4 , and H_2S into the mine voids. This GaRD (Gas Redox and Displacement) System utilises reducing gas mixtures generated by anaerobic bacterial activity to displace oxygen. Those gas mixtures can halt sulphide oxidation and acid generation, and may result in the precipitation of secondary sulphides from the accumulated drainage water.

As will be shown in the examples, not all mine sealing operations are successful, the reasons for it being different. Dam failures, though rarely observed, are mostly due to seal blow-out or leakages around the constructed dam as well as chemical reactions between the dam and the mine water. Most of the failures reported were the result of insufficient hydrogeological or geotechnical investigations, which anyway would have been less cost intensive than the subsequent remediation measures!

Furthermore, the hydraulic head above a mine seal might be extremely high and such sealing systems must therefore be used with good engineering practice. Most mines would have a hydraulic head exceeding that of the highest water dams (Fig. 24). Another issue is the mine water quality. As several investigations show, mine water quality might change during time and chemical reactions between the water and the concrete could destroy the dam construction. Some of the possible dam constructions and mine sealing techniques are illustrated in Fig. 25 to Fig. 27.

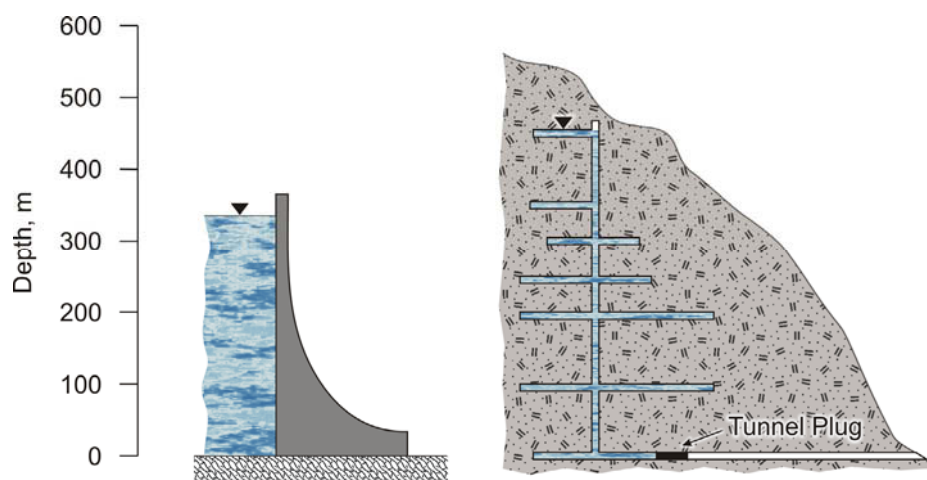


Fig. 24: Comparison of hydraulic head above a mine plug with the world's highest dam (Rogun Dam, Tadschikistan, 335 m).

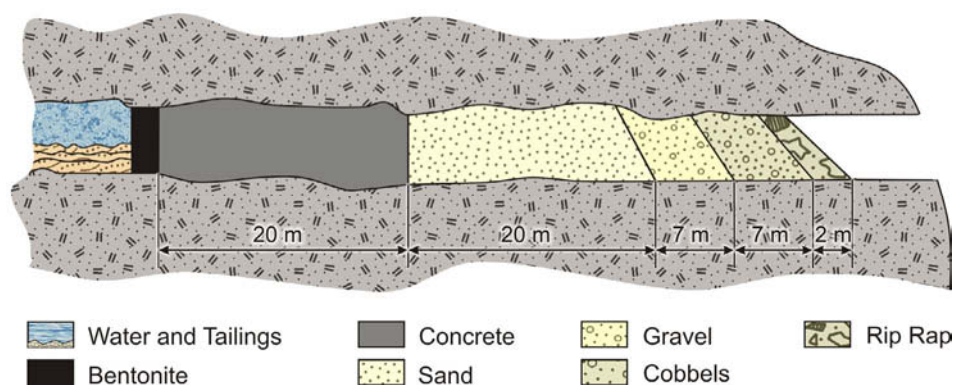


Fig. 25: Example of an underground mine dam (after Lang 1999).

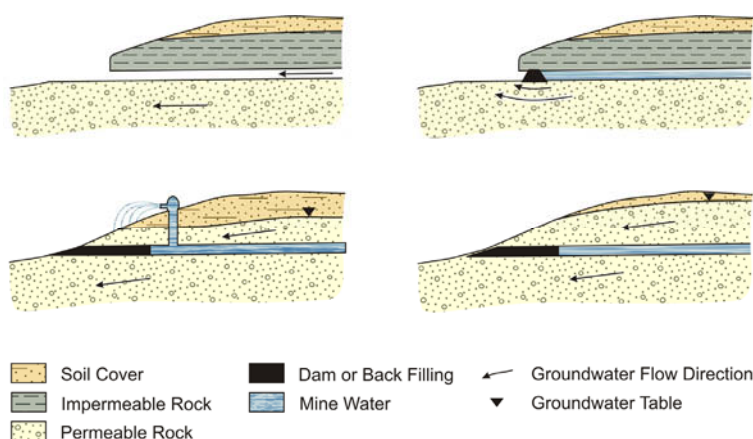


Fig. 26: Possible mine sealing techniques (after Fernández-Rubio et al. 1987).

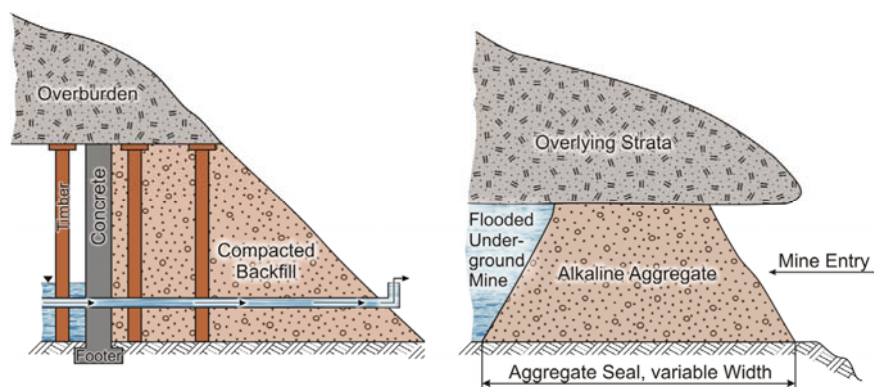


Fig. 27: Possible mine sealings (after Scott and Hays 1975).

Shafts are usually sealed by either backfilling them with inert or alkalinity producing rock material or with a concrete seal. Each technique has its advantages and disadvantages which have to be discussed within the mine closure plan. For monitoring purposes a concrete plug should be constructed with monitoring pipes with a diameter of at least 8 cm (3 inches) to gain access for measuring instruments (see chapter 9). Because backfilled shafts could be a pathway for water infiltrating into the mine, the top of the backfilled shaft should be sealed to provide drainage away from the shaft.

Mine adits which do not drain polluted water and are likely not to drain polluted water in the future should not be permanently closed. Such openings should be used as reserves for en-

dangered animals such as bats. Gates can be constructed safely to gain free drainage of water and free access for the animals. Local environmental protection groups, bat protectionists, or mining archaeologists can help to maintain such localities.

To stop the production of acid mine drainage, total or partial mine seals with dams or barriers can be installed. Usually, such barriers or dams must not withstand the water pressure of a total water rise into all the open voids. It could be enough to prevent oxygen coming into contact with the disulphides thus minimising the production of acidity. Yet, as Barnes and Clarke (1964) state, oxygen is not the only controlling reactant of acid mine drainage/metalliferous leachate and before a mine is sealed, a thorough understanding of the theoretical geochemistry of the site as well as laboratory experiments and field observations are necessary.

A positive mine sealing example is the Beth Energy's underground Mine № 105W near Buckhannon in West Virginia/USA. Two of the three mine portals had to be sealed with five seals to prevent the drainage of acid mine water. At the same time in 1984/85 the mine water was treated with an *in-situ* method by applying apatite rock-dust to the potentially acid parts of the mine and by injecting alkaline substances in the mine water pool. Even after four water changes the mine water, originally with a pH of 3.9, stayed at a pH of around 7 (Hause and Willison 1986) and in 1989, just the iron concentration came back from 1 to 15 mg L⁻¹ after an initial concentration of 144 mg L⁻¹ (Aljoe and Hawkins 1993).

Yet, there are cases where a mine entrance sealing had a negative impact on the drainage water quality and the receiving streams. At the Mynydd Parys mine site in Anglesey/Wales the stream Afon Goch is highly polluted by acid mine drainage with pH values between 2 and 3 (Boult 1996) and elevated concentrations of metals (e.g. Al 81 mg L⁻¹, Fe 431 mg L⁻¹, Cu 39 mg L⁻¹, Zn 45 mg L⁻¹; Younger et al. 2004). In the early 20th century the Dyffryn Adda drainage adit was sealed with a concrete seal and the mine water, containing elevated amounts of copper, was processed to extract the copper from the mine water. During and after this regular water extraction from the underground mine, which was controlled by valves and pipes through the dam, a mine water body with approximately $270 \cdot 10^3$ m³ of acidic mine water and a hydrostatic pressure of $3.6 \cdot 10^5$ Pa built up behind the dam. One reason for the development of that mine water pool was the closure of the valves after the *ex-situ* mining was stopped in the mid 1950s. As no data about the dam was available and the dams safety was not clear, $245 \cdot 10^3$ m³ of acidic mine water were pumped out of the mine to dismantle the dam. After the dam was removed the mine water quality increased slightly as microbiological and geochemical analyses proved (Bryan et al. 2004; Coupland et al. 2004; Johnston 2004). Consequently, the water quality of Afon Goch South has greatly improved in the meantime while the Afon Goch North is no more impacted than it was before.

5.7.5 Quality Control Issues

Quality control in diversion of mine waters is equally important as quality control in sampling and remediation measures. Therefore, a Diversion Quality Control Plan which also helps to support decisions concerning the water diversion should be established. In the Diversion Quality Control Plan, the following requirements should be described:

- which specific data types are needed
- what are the data collection requirements
- which monitoring instruments are needed
- failure of instruments and actions
- which measurements of which parameters are necessary

Periodic inspection and maintenance of the diversion installations must be a main task in the Diversion Quality Control Plan. Water must flow without hindering in the channels and ditches and erosion or overflow of the diversion installations must be repaired as soon as possible. Maintenance must include the cleaning of any waste material in the channels or at the measuring devices. There must be references to the maintenance intervals in the Diversion Quality Control Plan and at least before and after major storm events a control of the diversion systems is necessary to guarantee its proper functioning.

If mine entrances are sealed with dams, the dams have to be controlled regularly, to guarantee their proper functioning. Any leakage of the dam must be reported according to the Quality Control Plan. If physico-chemical parameters of the diverted water courses must be measured, the same quality controls as described in section 10.7.1 are essential.

5.7.6 Examples and Case Studies

Examples of water diversions around mines as well as mine sealings are numerous and can easily be found in the relevant literature data bases (e.g. GeoRef, Geobase, FIZ-GEO). Here, four examples with interesting results will be given to show the effects of mine water diversion and mine sealings.

Biaglow (1988) reported on a hydraulic mine seal in the year 1980 at the Mine Complex 88 at Big Four Hollow Creek near Lake Hope State Park in south-eastern Ohio/USA, which did not result in the predicted pollution reduction. His study was conducted to determine the effectiveness of the mine seal and to clarify whether flooding the mine complex had an impact on the amount of acidic mine drainage to Big Four Hollow Creek. Comparisons with historical data showed that the mine seal initially resulted in a decreased amount of dissolved sulphate and iron loadings. This initial decrease was followed by increased amounts of loading discharging to Big Four Hollow Creek. His results indicate that dissolved sulphate and total dissolved iron loadings are at nearly the same levels observed before completion of the mine seal. 20 years later, in 2000, the site was reinvestigated and it could be proved that on a long term basis the hydraulic mine seal had a positive, but small effect (Stoertz et al. 2001). Since its sealing, the pH in the mine water increased from 2.7 to 5.3, electrical conductivity decreased from 2,700 to 600 $\mu\text{S cm}^{-1}$, and dissolved oxygen is lower than 2% saturation. Water quality in the mine clearly shows removal of oxygen through flooding and a subsequent decrease in the production of acid mine drainage. Furthermore, it could be shown that the mine water quality has improved gradually in the two decades since flooding, due to mine sealing, natural attenuation and the effect of the first flush. On the other hand, leakages of the mine sealing and an accidental seal blow-out resulted in a lower remediation target than expected.

Surface water diversions around active mines are a common measure where the surface water might either be a hazard to the surface or underground mine operations or in cases where the surface water is directly above a productive mining area. In the case of the Meadowgate open pit mine south-east of Sheffield/UK the River Rother was diverted temporarily between 1977 and 1981. After the coal was mined out the whole area was reconstructed and in 1983 the Rother Valley Country Park was opened. During the construction of the diversion no real problems occurred due to the use of engineered constructions. Yet, after the river was back-diverted, the alluvial clay planned to be used as a river bed caused handling and storing problems due to its plastic character (Brook and Cresswell 1989).

In Eastern Germany, at the border with Poland, subsurface concrete sealing walls were used for the first time. They prevent nearby wetlands (Peitzer Teichlandschaft und Spreeaue) from falling dry but also prevent the river Neiße from flowing into the large open pit lignite mines (Jänschwalde Open Pit). Such diaphragm walls are up to 85 meters deep and 11 kilometres long. The first one's construction there started in 1979 and the last one was finished in 2005.

The results of the hydrogeological investigations and the more than 20 years of experience clearly show that besides surface water diversions also deep sealing walls might be used to prevent water from flowing into working and abandoned mines (Mielcarek 1988; Arnold and Kuhlmann 1994; Häge 1995; Fahle and Arnold 1996). From a technical point of view such walls could be as deep as 100 m in case there is a low permeability strata with a thickness of at least 2 meters. Furthermore, no interaction with another cone of depression must be allowed and no dewatering under the sealing strata must take place. An important fact to guarantee the functioning of such walls is the proper hydrogeological modelling of the area thought to be affected.

During investigations of the flooded Straßberg/Harz/Germany underground mine in the early 1990s it became clear that the mine water in the flooded mine is stratified, with a shallow water body of less contaminated mine water and a deeper one with highly contaminated mine water (Rüterkamp and Meßer 2000). Therefore, three drainage adits were constructed to divert the lower contaminated mine water at shallow mine levels. Shortly after the three adit system came fully in force in 1996, the stratification broke down and highly contaminated mine water entered the catchment area of the river Selke. By that time neither the mine operator nor the consultant were aware of the reasons for this breakdown of stratification. However, the situation was similar to that one of the flooded Niederschlema/Alberoda uranium mine, where the effect of convective mixing in such a situation has been described based on a three year study of the flooding regime (see section 12.11 and 12.9; Wolkersdorfer 1996). The Straßberg example clearly shows the importance of hydrodynamic investigations of a flooded mine and how useful tracer tests could be to optimise a proposed diversion scheme (Kindermann 1998; Wolkersdorfer 2001).

One of the biggest remediation projects in the USA was conducted in the Tar Creek area (thereunder the Picher Mining District), belonging to the Tri-State mining district in north-eastern Ottawa County, Oklahoma. In 1983 the area was added to the USA “National Priorities List” and subsequently got known as the “Tar Creek Superfund Site”. To prevent surface water from flowing into the underground workings, diversions and dike structures were constructed. Surface water was stopped from flowing into abandoned mine shafts, open boreholes and surface subsidence areas, of which about 600 out of 2,600 estimated were located on the site. Approximately 75% of the annual inflows coming from the surface where thought to be prevented from flowing into the abandoned workings and subsequently into the receiving stream, the Tar Creek. Yet, in 1991 it became clear that the decreasing concentrations of pollutants might be a naturally occurring phenomena that the discharged mine water volume was not significantly decreased, the surface water quality not significantly improved and that the effectiveness of the well plugging operations were not successful. Investigations in 2002 showed that the mine and ground water quality further improved. Finally, after nearly two decades of remediation, in 2000 the “Governor Frank Keating’s Tar Creek Superfund Task Force” was set up to end and co-ordinate the clean-up process for the most prominent dangers to the environment and people (Griffin et al. 2000; Tar Creek Superfund Task Force Water Quality Subcommittee 2000a, 2000b; DeHay 2003; DeHay et al. 2004). Meanwhile an eight year Watershed Management Plan for the Tar Creek and the Lower Spring River has been set up (U.S. Army Corps of Engineers 2004). Though enormous amounts of money were consumed to remediate the Tar Creek catchment area, it must be concluded that from a hydrogeological point of view most of the work was done by natural attenuation. None of the diversion measures was efficient enough to significantly improve the mine water quality on a long term basis – though the amount of infiltrating waters (and thus the load) on the site was reduced. One of the reasons might be that the Pennsylvanian and Mississippian rocks are too permeable and fractured for diversion measures to be effective.

From the above mentioned examples it could be concluded that diversion measures after mine closure compared to the resulted improvements are too expensive to be used in the future. In fact, it is not clear if the advances in ground and surface water quality could not be the result of natural attenuation processes. It might be disappointing that reactive transport cannot be influenced by diversion measures in such a way that the receiving water courses are protected on a long term basis. However, in all the mentioned cases a lack of hydrodynamic and hydrogeological understanding was involved in the projects. The upstream catchment areas of underground mines can be extremely big and diverting surface water just above the mined out strata is usually not of much help. In cases where tracer tests proved a hydraulic connection between the surface streams and the mine workings, surface water diversion will be successful; in other cases it tends to be a waste of money.

5.8 Mine Dewatering

5.8.1 Introduction

Though mine dewatering is usually not involved when a mine is abandoned, there are cases of controlled flooding, where mine water had to be pumped out of the mine even during the flooding process, as a means of mine water treatment, or for environmental investigations (e.g. Anonymous 1997; Nuttall et al. 2002; Jakubick et al. 2002; Gammons et al. 2003; Nuttall and Younger 2004; Coulton et al. 2004; Wolkersdorfer and Hasche 2004). Furthermore, as in the case of the abandoned coal mines in the southern Ruhr area, pumping has to continue to prevent the productive pits from flooding (Coldewey and Semrau 1994) and for other coal mining areas similar considerations apply (Younger and Harbourn 1995; Younger 2002b). Due to the different mining methods, dewatering of open pit mines differs from dewatering of underground mines. Therefore, the following chapter will focus on mine dewatering techniques, some of which can also be applied to active mining schemes.

Since Neolithic times, dewatering of mines was a major task for the miners, and where no dewatering techniques were available, the mine had to be closed down (Shepherd 1993). Beneath manual water haulage and the use of buckets and leather buckets mounted on water wheels no real innovation was made in underground mining for centuries (Younger 2004b). After the first reliable wooden pumps were invented at the German Rammelsberg/Goslar in the 15th and at Ehrenfriedersdorf/Erzgebirge in the 16th century (Agricola 1556; Kreibich and Eisbein ca. 2003), it was not until 1749, when the first reliable metal pump was introduced in Schemnitz by Joseph Hell. By the *machina pyraulica* (the first usable steam-engine in mines was installed in 1712; Younger 2004b) of Thomas Newcomen, the first water-independent dewatering machine could be used (Kunnert 1974) and the first one with a metal pump was installed in 1788 in the German Tarnowitz Lead-Zinc mine (now Tarnowskie Góry, Poland; Seidl and Stähler 1913). Until then the depth of underground mines was mainly controlled by the depth of the water table or the dewatering techniques available. Nowadays, the dewatering techniques for open pit mines and underground mines are well established and even mines with several thousands of meters depth can be dewatered with the pumps available (Vutukuri and Singh 1993; Duane et al. 1997; Fig. 28).

Another method for deep mine dewatering was the construction of drainage adits. Such systems have long been known and were also used as water supply tunnels in pre-historic and historic times as the Turkish Troyan Water Mine (Biswas 1970; Wolkersdorfer and Göbel 2004). Such drainage galleries needed the close co-operation of all the miners, land-owners, and authorities involved because during the construction of such a drainage adit no profit was made and the construction of such systems took up to 100 years and reached length of several tenths of kilometres (Jobst et al. 1994; Gundelwein 2001). To protect the miners of such systems and to guarantee the water rights of the operators after the drainage gallery reached its

final length, the construction, operation and maintenance of such dewatering systems was regulated by special legislation for such *Wassergemeinschaften* (water communities) and *Erbstollengemeinschaften* (drainage gallery communities). Some of the best known German *Erbstollen* are the *Roederstollen* at the Rammelsberg, the *Schlebuscher Erbstollen* near Sprockhövel, the *Rothschönberger Stollen* of the Freiberg Mining area or the *Tiefer-Ernst-August Erbstollen* of the Clausthal and Zellerfeld Mining (Ließmann 2002) areas some of them being still in use more than 200 years after their construction. Younger (2004b) lists 38 “important” drainage adits for Britain which were constructed between 1617 and 1897.



Fig. 28: Mine dewatering pump and mine water scheme control room in the Vazante/Brazil underground zinc-mine.

In many countries, mine water – if exceeding the regulator’s limits – will be classified as waste water and is controlled in accordance with the relevant waste water directives. Examples of such countries are Finland and Turkey, where all industrial waters including water emanating from mining operations are classified as waste waters (Mäkinen 1996; Wolkersdorfer and Göbel 2001b). Countries where mine water is not considered waste water are Germany and Ireland (Government of Ireland 1997; Plinke and Wildhagen 2003), which have special regulations under which mine water is covered. In the future, when the European Water Framework Directive will be finally set in force in all EU countries, a harmonization of water legislation might be assumed here, too.

5.8.2 Dewatering Techniques

No comprehensive textbook about mine dewatering techniques was published in recent years. Only Younger et al. (2002) describe the basics of mine dewatering and the problems involved. Yet, the most comprehensive book about mine dewatering techniques (“Bergmännische Wasserwirtschaft”) was published more than 50 years ago in German by Kegel (1950) and another one eight years later by Sinclair (1958). In principle, the water management in an operating mine can be divided into water collection, water removal, and water disposal. Yet, the details of all dewatering techniques are beyond the scope of this publication.

Most surface (“open pits”) and underground mines have to be dewatered to gain access to the raw material and to guarantee safe working conditions for the miners. During exploration and working, pollution reduction is usually not the main focus of the dewatering, though several cases exist, where a successful dewatering also serves as a method for pollution prevention (Morton and Niekerk 1994). Drainage water usually is referred to as “mine water”, but according to Wolkersdorfer and Hasche (2001c) mine water can be classified as follows (possibly this definition might only be suitable for German authorities):

- flooding water: ground water that is used for the flooding of an open pit/underground mine or water that is discharging after the flooding of the open pit/underground mine
- free or pore water: processing water that is accumulated together with precipitation in tailings ponds
- leakage water: surface water that infiltrates into waste dumps as well as waste rock piles and leaks out at the bottom of waste dumps, waste rock piles, or within the open voids of working mines
- geogenic water: ground water from deeper aquifers that are typical for hard coal mining

According to the definition above, only leakage water and geogenic water will be discussed in this chapter. Drainage water amounts can range between several litres to several cubic meters per second (Fig. 29) and they can vary extremely in their chemical composition (Younger 1997; Younger et al. 2002; Lottermoser 2003). Some drainage waters might even be used as drinking water (see section 11.7).

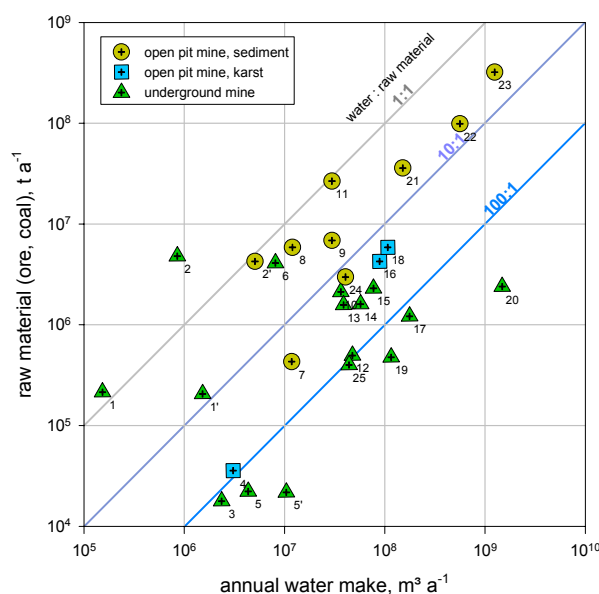


Fig. 29: Selected worldwide ratio of water make and raw material production (modified, updated, and data added after Fernández-Rubio and Fernandez Lorca 1993). 1 and 1' Agnew (Australia, Ni), 2 and 2' Lock (Australia, coal), 3 and 4 Burra (Australia, Cu), 5 and 5' Beaconsfield (Tasmania, Au), 6 German Creek (Australia, coal), 7 Scottish National Coal Board (UK, coal), 8 Bowmans (Australia, lignite), 9 Thorez (Poland, coal), 10 Picher (USA, Pb, Zn), 11 Morwell (Australia, lignite), 12 Fejér Co. (Hungary, bauxite), 13 Chingola (Zambia, Cu), 14 Dorog (Hungary, coal), 15 Tatabánya (Hungary, coal), 16 Pine Point (Canada, Pb, Zn), 17 Konkola (Zambia, Cu), 18 Kingston (Australia, lignite), 19 Bakony (Hungary, bauxite), 20 Kailuna Hebei (China, coal), 21 Atabasca (Canada, tar sand), 22 Rheinbraun (Germany, coal), 23 Lausitz (Germany, coal), 24 China (coal), 25 Vazante (Brazil, Zn)

The following methods are commonly used in open pit mining, depending on the mine layout and the local geological conditions: filter wells, disposal wells, inverted wells, vacuum drainage, horizontal drains, sealing walls (grout walls), cut off walls, and guard wells (Plotnikov et al. 1989; de la Vergne 2003; Kecojevic et al. 2003). By using such methods, even extremely large and deep open pit mining areas can be dewatered successfully, as can be seen by the German Rhenish lignite mines, which are dewatered to a depth of 350 to 500 m, with a depression cone of approximately 2,500 square kilometres (Briechle et al. 1998).

In contrast to open pit mines, underground mines are usually dewatered by the sump pumping method or by adit dewatering. A sump at the deepest or at shallower levels collects all water seeping into the mine. As a general rule, mine workings below 140–500 m depth from the surface can be expected to be dry in many mines (Saul 1948; Wolkersdorfer 1996; de la Vergne 2003). This is mainly because hydraulic conductivities in such a depth are very low and as Karrenberg (1981) states are ranging between 10^{-6} – 10^{-9} m s⁻¹ (Karrenberg 1981; Stober 1995). Furthermore, it can often be observed that water inflows in non-karstified rocks stop within a relatively short time. Close to the sump pumping stations submersible pumps are installed that pump the water to the surface. This water can be chemically contaminated or could contain suspended matter and therefore needs treatment after having been pumped out of the mine. While turbidity can be removed using settling lagoons, chemically polluted water has to be treated with passive or active methods according to the kind of chemical pollution (see chapter 11).

When planning a dewatering scheme within a water catchment, all measures have to ensure that the good ecological status of the surface waters will not be negatively affected. Therefore, the selection of which dewatering systems shall be used needs careful consideration and depends on several factors, such as: hydrogeological and hydrological conditions, lifetime of the mining operation, total water make, availability of dewatering equipment, and drainage water quality. Furthermore, the following aquifer parameters have to be known: type/state of aquifer (confined or not confined), drainable porosity, transmissivity and storage coefficient, static water level, seasonal variations, depth and thickness, as well as water recharge (Morton and Niekerk 1994).

As several studies show, sump dewatering should be given preference against adit dewatering, because after closure, mines with adits usually drain polluted water over a longer life span than sump dewatered mines (Commonwealth of Pennsylvania 1973; Mentz et al. 1975 in Scott and Hays 1975; Hawkins 1994; Younger 1998; Callaghan et al. 2000). In the case of the Shoff and Yorkshire mines, both situated in the Pennsylvanian Bigler Township/USA, the differences between updip (adit dewatering) and downdip (sump dewatering) mining were evaluated. Except that the two mines were on opposite sides of the Clearfield Creek and worked in different ways, the geological situation, coal quality, period of operation, discharges, and the hydrological conditions were similar (Commonwealth of Pennsylvania 1973). In 1974 the mine water was sampled eight times to evaluate water composition. It could be shown that the water quality of the flooded downdip Yorkshire Mine was significantly better than the mine water draining from the updip Shoff Mine (Fig. 30). Mentz et al. (1975) also compared the economical differences in both mining methods. Obviously, pumping costs are higher in downdip mines, but if offset against the remediation costs downdip mining has to be given priority. Thus they concluded that downdip mining is advantageous relative to updip mining because the mine water quality in flooded shaft mines in the long term is better than in adit mines.

The reason for this behaviour is that sump dewatered mines are usually flooded after mining comes to an end, while mines with adits are subject to oxidation above the water table, thus releasing metals and acidity for a long period of time. Flooded mines, as discussed in section

5.4, insulate the disulphides from the oxygen and – to a certain degree – microbiological activity and therefore reduce the release of acidity and consequently pollutants. Examples of such adit mines with heavily polluted discharges can be seen by the *Rothschönberger Stollen* in the Freiberg/Saxony mining area (Baacke 2000; Weyer 2003; see section 12.12), the Gernrode/Harz Mountains Hochwarthe Adit or the Burgfey Adit in the Eifel Mountains (Mair 2002).

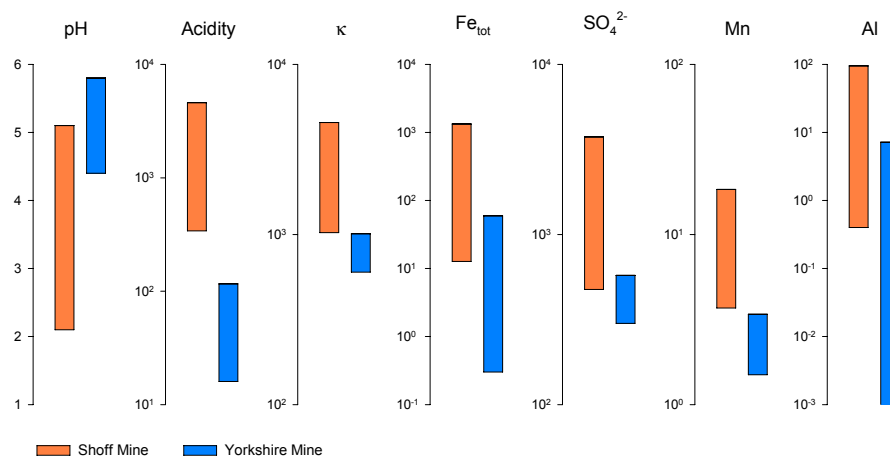


Fig. 30: Up-dip versus down-dip mining: comparison of selected mine water parameters of the Shoff (up-dip, adit-drained) and the Yorkshire (down-dip, flooded from shaft collar) mines (after Mentz et al. 1975). $n = 8$; pH without unit, κ (specific electrical conductivity) in $\mu\text{S cm}^{-1}$; all other in mg L^{-1} .

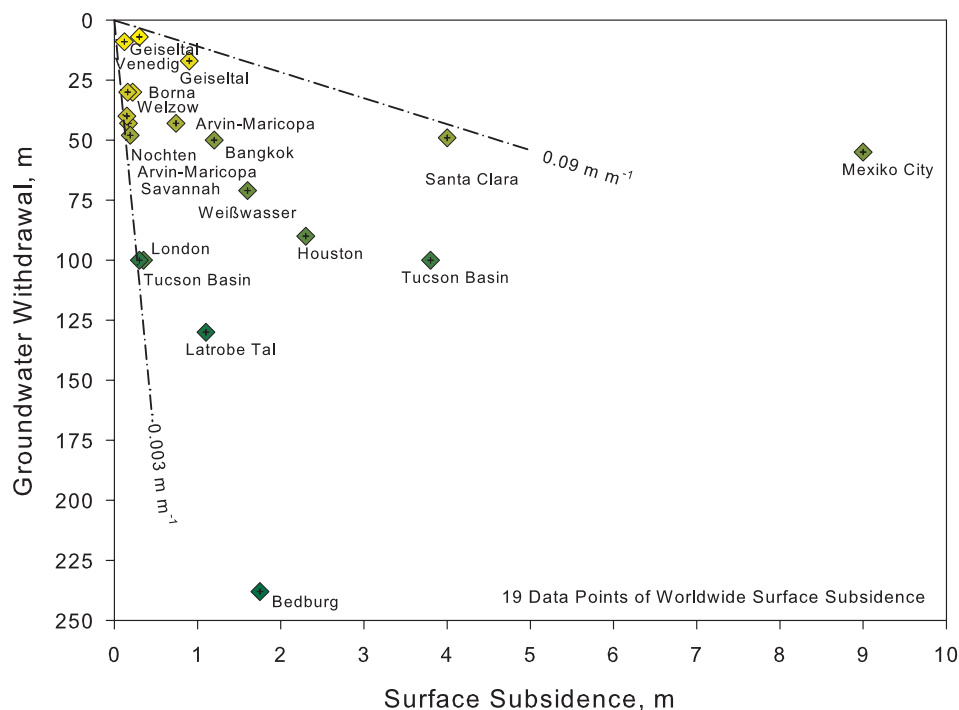


Fig. 31: Surface subsidence of sediments due to the dewatering of mines and groundwater abstracted for drinking water supply (modified after Wolkersdorfer and Thiem 1999).



Fig. 32: Settlement lagoon and drainage pumps of an open pit iron mine near Belo Horizonte/Brazil.

5.8.3 *Disposal and Treatment of Dewatering Effluents*

As discussed above, not all mine waters are a source of pollution. Depending on the mine type and the water quality, drainage water has to be treated (see chapter 11) or can be discharged to the river catchment without further treatment.

If mine water will not change the good ecological status of a river catchment, no further action will be needed. Those mine waters can be used for irrigation purposes, drinking water or health resorts (Banks et al. 1997; Annandale et al. 2002). In the German lignite mining areas several hundred million of cubic meters of geogenic drainage water can be discharged without any further treatment because it is ground water of good quality (Regionaler Planungsverband Westsachsen 1998; Landesumweltamt Nordrhein-Westfalen 2002). Nonetheless, unpolluted mine water, when diluted with more polluted surface water will change the geochemical equilibrium of the receiving water course. This might result in dissolution of previously precipitated contaminants and must be considered in the mine drainage plan. In the worst case, the surface water quality downstream of the mixing zone can be poorer than upstream of the inflow. Where mine water with a good ecological status has a higher suspension load than the receiving streams, a settlement lagoon before the discharge point would be a neat solution (Fig. 32).

All other pumped drainage waters have to be treated according to the appropriate standards, depending on the mining licence. Mining operations can largely be classified as metal mining, coal/lignite mining, salt mining, and quarrying. Except in salt mining, all mine types, regardless if they are surface or underground mines, can encounter polluted mine waters. One of the most problematic mine waters is acid mine water (usually called AMD [acid mine drainage] or ARD [acid rock drainage]) or neutral zinc-rich water which can already occur during the active mining period. Active treatment plants are usually installed for cleaning highly polluted mine water during the mine's lifetime. Those techniques are well established and the construction of LDS (low density sludge) and HDS (high density sludge) treatment plants (section 11.3) as well as other techniques (e.g. ionic exchange, reverse osmosis, nano-filtration, distillation, electro-dialyses) are well known. A combination plant has been working at the Polish Dębieńsko coal mine since 1975 (Motyka and Skibinski 1982; Tar Creek Superfund Task Force Water Quality Subcommittee 2000b; Brown et al. 2002; Turek 2004). An interesting mine water treatment technique is a distillation process described by LaRosa et al. (1971). They used coal refuse together with molten iron to produce energy which is then used to drive

either a distillation or a reverse osmosis plant. While the process worked well in the laboratory, it has never been used on site.

After treatment, the drainage water can be discharged into surface streams or lakes and the residuals will be disposed of or pumped back into the underground mine voids. Only in some rare cases, the mine water's residuals are used as a raw material. One example was published by Hedin (2003), who was able to sell 1000 t of iron oxide recovered from an abandoned coal mine in the Appalachian mountains/USA. Another one is the treatment plant at the abandoned Hungarian Mecsekérc uranium mine (Pécs/Hungary), where uranium is extracted from the mine water and sold at the world market (Gombkötö et al. 2002). Unfortunately, the extraction of pure metals from mine drainage remains economically non-viable.

An alternative to minimise the impacts of mine drainage water are passive treatment methods for polishing already treated waters or for treating less polluted mine water (section 11.4). Those systems can remove different mine water pollutants, at least iron and turbidity and should therefore be installed wherever applicable. Another option to reduce the amount of polluted mine drainage water is the separation of waters flowing in or around the mine. The commonly used methods nowadays (sump dewatering, adit dewatering) usually drain water of different quality because their separation is often complicated (see section 5.8). With respect to the Water Framework Directive mine operators should introduce dewatering techniques, where the amount of polluted and to be treated water is as small as possible. Many cases are already known, and especially for open pit mining such techniques can easily be used, whereas in underground mines the installation of different pumping stations would be necessary.

An interesting case of acid mine drainage/metalliferous leachate occurs in north-western Pennsylvania/USA, which is already threatened by several thousands of mine water discharges polluting the surface streams in this area. Acidic, iron-rich water was percolating through gas wells that were abandoned about a century ago as a kind of artesian discharges (Hedin et al. 2005). Those wells reach 300—500 m down into gas bearing Devonian sands which are connected to formation brines. Yet, the acidic water was not saline and during the investigations it was shown that the casings of the wells must be corroded or missing in the shallower aquifers. Those aquifers are connected to shallow coal mine workings and when reaching one of the gas wells the water upwells to the surface. Because the wells are so numerous, the authors concluded that sealing the wells would not be feasible as the water would just flow to another well or to another place with a lower hydraulic pressure with fewer possibilities to treat that water.

5.8.4 Drawdown Management

Usually, mine abandonment includes that the water table is allowed to recover and near pre-mining conditions will establish again. Yet, there are several reasons for maintaining the water level below the potential discharge point, if negative environmental impacts might be expected. Such a situation can occur when the raising water level is expected to flush tailings or waste rock piles, polluted mine water could enter drinking water reservoirs, if the rising mine water might flush weathered minerals in long abandoned near surface mine workings, or if subsided areas might be flooded.

Mine dewatering causes the natural ground water table to drop in the close vicinity of the mining operation. This may lead to the drying out of springs, wells, lakes, or rivers. It is advisable to start the ground water level monitoring well in advance of the remediation or mining operation, where it is possible to convert exploration drill holes into ground water monitoring wells (Plotnikov et al. 1989). Water wells have to be constructed in accordance with good practice procedures (Driscoll 1987). Thus, the drawdown can be precisely monitored

throughout the lifetime of the mine and the remediation process. From a hydrogeological point of view mine dewatering is a huge steady-state pumping test – if only the data gathered during that test would be used for the closure phase of the mine!

Besides the monitoring, the hydrogeological modelling of the expected drawdown is of importance. The radius of drawdown – and consequently the radius of a potential impact on the water courses – usually depends on the degree of the water table lowering by the mining operation, the aquifer's hydraulic conductivity, and homogeneity, and the pumping rate. Simplified, a mine can be seen as a deep well and therefore, the “equivalent well approach” can be used to calculate the drawdown caused by a mine. Up to date computer codes are able to model the ground water drawdown and the impacts on the surface and ground water at acceptable precision level. A prerequisite for modelling is an ample experience with such models. Yet, nothing substantially changed since the basic investigations of Theis and his principles for calculating the impacts of wells on dewatering are still valid (Theis 1935).

Areas affected by mine water drawdown can reach several thousands of square kilometres, as in the German lignite mining regions. Such large mining operations have to ensure that surface water reservoirs are protected and especially natural wetlands are watered during drawdown. Several mines installed water re-injection wells to keep the affected area as small as possible. Such regions can be found in the Western German surface lignite mines, near the Dutch/German border where water injection wells keep the negative effects of ground water drawdown to a minimum. In special cases, diversion pipe lines or channels have to be constructed for that purpose. Where springs that are used for drinking water supplies dry out, the people affected have to be supplied with water pipelines. Such a situation occurred in the now Polish Silesian mining area at the end of the 19th century. In an effort of local authorities, the mining authorities and the different mining companies, wells were drilled, good quality mine waters collected and kilometres of pipe-lines constructed in 1878. Thus, the population in the Silesian industrial area could be supplied with enough good quality drinking water (Geisenheimer 1913).



Fig. 33: Damaged house because of surface subsidence above an abandoned iron ore mine in Roncourt, Lorraine/France.

Drawdown management has to ensure that the environmental impacts as well as negative impacts to engineered constructions will be minimised. Surface subsidence can reach several meters (Fig. 31), but as long, as only small subsidence differences within a single engineered construction occur, no damages to those constructions will arise (Poland 1984). Modern mining and drawdown operations based on high quality hydrogeological and geological investigations can reduce those negative impacts to a minimum. In places, where subsidence differ-

ences and damages to buildings are expected (Fig. 33), buildings have to be secured by technical installations or the buildings foundations (Fig. 34).

Examples, where mine water drawdown has to be maintained during and after the remediation phase can be categorised into pollution prevention and flooding prevention conditions. Pollution prevention could either mean that mine water is kept away from potentially polluting material or that polluted water is kept away from drinking water resources. Flooding prevention, on the other hand, means that the mine water table has to be kept below the surface because during longwall mining the surface subsided and uncontrolled mine water rebound would mean that large, developed or agricultural areas would be flooded with polluted mine water.

Pollution prevention was and is one of the reasons for maintaining the mine water level below the potential discharge point at the abandoned Niederschlema/Alberoda/Germany underground uranium mine (Gatzweiler and Mager 1993; Hagen and Lange 1995; Rudolph 1997). Because the abandoned near surface mine workings are within a geological zone of arsenic rich ores, the mine water was kept about 60 m below the final flooding level. Thus, the quality of the mine water was kept at a reasonable good quality until a water treatment plant was constructed that could also cope with the elevated arsenic concentrations.

Ground water protection is the reason for pumping at the Aznalcóllar/Spain pit, where the pit water table is lowered by 6 m below the ground water table (Kurt Friese, pers. comm.). For the same reason, the Horden Colliery/UK is not allowed to flood up to the discharge point, as the polluted mine water might negatively influence the Basal Permian Sands aquifer (Coulton et al. 2004). An cone of depression is also kept at the abandoned Königstein/Germany *in-situ* leaching uranium mine. There, the upwelling mine water would impact the 3rd Sandstone aquifer which supplies the cities of Pirna and Dresden with drinking water. Currently, a final flooding level of 140 m a.s.l. is planned which would hold the water 170—250 m below the surface until the higher contaminant concentrations of the first flush will have passed (Jakubick et al. 2002; Schreyer et al. 2002). This cost intensive strategy has been questioned by some authors who used modelling tools (e.g. PHREEQC) for suggesting cheaper remediation methods such as diluting the mine water or using the surrounding strata as a passive remediation method (Merkel 2002).



Fig. 34: Technical solution to secure a private house from destruction resulting from surface subsidence (courtesy Alexander Kattner, March 25th 2003).

Many mining areas in the world subsided due to longwall mining. To prevent build-up areas from being flooded or impacted by polluted mine water, the dewatering pumps possibly must run forever. Such areas are the abandoned coal mining fields of Durham/United Kingdom, Limburg in the Netherlands, Aachen and the Ruhr area in Germany, or Silesia (Schlesien) in Poland (Bekendam and Pöttgens 1995; Younger 2002b). No solution of how those areas could be protected by cheaper methods is known so far because from a human point of view the flooding of such build-up areas and a resettlement of tens of thousands of inhabitants is unacceptable.

The last cases described in this chapter are the Pennsylvanian Swank's and Shoff mines in Clearfield County/USA. During an investigation conducted in the 1970s ten drift openings of the Swank's and thirteen of the Shoff mine were localised and to stop the pyrite oxidation and subsequently the pollution of the Clearfield Creek the sealing of all those drift adits was suggested. Yet, the consultant recommended "a manually operated drawdown system will be installed in the southern end of the mine. This will consist of a pipe to the lower portion of the mine pool with a valve that will act as a safety device to lower the mine pool if required" (Commonwealth of Pennsylvania 1973; see Fig. 23).

5.9 Prediction of Ground Water Rebound

Flooding of an abandoned underground mine ideally means that all mine dewatering pumps will be switched off after the completion of mining and the ground or surface water is allowed to flow into the open voids. The flooding process will stop when the mine water surfaces equals the ground water surface or reaches a point of discharge – ideally a drainage adit. Usually, the prediction of the rebound scenario is imperative to allow the planning of certain remediation measures or mine water treatment options. As mentioned before, rebound prediction is a fairly fuzzy job and as so often in the mine water research rules of the thumb and experience can help a lot to calibrate numerically based conceptual models (compare chapter 8).

Because conceptual models are a compelling necessity in all hydrogeological investigations, and consequently mine flooding scenarios, the definition of Younger and Adams (1999) shall be given here:

A conceptual model is an assembly of simplifications about a complex, real system, which achieves a valid representation of that system, including all major features, whilst avoiding unnecessary detail.

Or in the words of Rushton (2003):

Conceptual models describe how water enters an aquifer system, flows through the aquifer system and leaves the aquifer system

Such a model can start at the back side of an envelope and almost certainly every hydrogeological project started with a first sketch at a piece of paper in the size of this book. Drawing sketches to understand the hydrogeological or flooding scenario is useful in every mine water related research. Such a sketch helps much to understand the fundamental principles of the investigation area and should be a must for every research. Of course, colourful graphics and computer print outs are helpful to understand complicated geometries and numerical results, but they are useless if not based on an evolution of conceptional models. Finally, everybody should keep in mind that numerical models are – at least in hydrogeology – always approximations of reality (Box 1979) and even when they look very real today they are not reality itself!

Younger and Adams (1999) and Younger (2002d) classified the models available for the calculation of ground water rebound into two categories: semi-distributed lumped-parameter

models and physically-based, distributed models. From an environmental and a legal point of view rebound prediction shall be conducted well in advance of the mine closure. Such an investigation for underground and surface mines must include answering the following questions:

- to which elevation will the mine fill up?
- which time will it take to fill the mine and reach a water table equilibrium?
- what will the water quality be during and after rebound?
- is there a potential to impact buildings?
- will mine gases (e.g. Rn , NH_4 , H_2S , CO_2) emanate from the flooded mine workings (e.g. Pokryszka et al. 2005)?
- which drinking water aquifers might be impacted?
- could ground subsidence occur?

Based on the results of such a calculation it might be necessary to backfill parts of the underground or surface mine to prevent the release of acidity or toxic compounds. Another result of such a rebound calculation could also be a complete backfill of a mine to prevent the accumulation of a large volume of polluted mine water. Such a complete backfill of an open pit mine has been conducted at the Ronneburg open pit uranium mine in Germany (Zurl et al. 1992; Laubrich et al. 1996; Geletneky 2002; Paul et al. 2002), the Blindwells open pit coal mine in Scotland or other mines worldwide. Rebound of open pit lakes can very often be estimated by using a simple water balance model, or an analytical model. If the hydrochemical situation has to be predicted as well, more complex numerical pit filling codes or coupled ground water flow models are advised to be used (see section 5.5.2; a description of the necessary procedures can be found in Younger and Adams 1999).

In general, the most important question to be solved is the duration of the rebound time t_r . t_r is a function of many different parameters and can be written as follows:

$$t_r = \phi(V_m, V_r, r_c, A, P, E, R) \quad (27)$$

with V_m : volume of mine voids
 V_r : volume of matrix
 r_c : coupling ratio between matrix and mine voids
 A : area of dewatering
 P : precipitation
 E : evaporation
 R : surface runoff

A central task for the prediction of the rebound time is knowledge about the mined and dewatered volume of rock. Such a calculation is comparatively simple for open pit mines, as the cone of depression, the volume of the open pit and the water make during the mine's lifetime are – usually – well known. Furthermore, the climatic, hydrologic, and hydrogeological conditions are to be known, as pit filling depends on ground water inflow, surface water runoff, precipitation, evaporation, and seepage out of the mine. An often overlooked fact is that the amount of water infiltrating into the mine changes while the mine water table rises.

Therefore, even if all available data has been considered and the predictions were accurate as in the Cornish Wheal Jane tin mine (Hamilton et al. 1994b), technical and meteorological problems can result in catastrophic mine water outfalls as it happened at this mine on January 13th 1992 when 30,000 m³ of heavily polluted mine water spilled out of the mine (Hamilton et al. 1994a). Furthermore, it might be that ecological or economical reasons on a regional scale

do not allow all the surface or ground water to be used for pit filling. Such a situation can be seen in the German Lausitz lignite mining area, where not enough surface water is available to flood the open pits in a reasonable short time because the river Neiße and Spree water must not be used for pit flooding (Arnold and Baumgarten 1996; Grünewald 2001; Schlaeger and Köngeter 2001).

For the calculation of rebound times, it is important to consider that the water inflow into an abandoned mine will change while the mine water table rises: it is quick at the start of the flooding and slows down to its end. This is mainly because the volume of the cone of depression gradually is reduced. Furthermore, the difference of the hydraulic head between the original ground water table and the mine water table decreases and therefore, the total flow into the mine continuously diminishes during the flooding process. Yet, there are several exceptions from this general rule: deep mines within a comparatively impermeable rock with a drainage adit and workings above and below that adit will not encounter much of a change until the water table reaches the level of the drainage adit. The reason is that most of the water in those mines comes from the – usually – lower permeable strata above the drainage adit. One of such cases is the Niederschlema/Alberoda/Germany mine (see section 12.3.3 for details). A comparison of historical, pre-mining data with the current mine water make clarifies that after the flooding process is finished and if the Markus-Semmler drainage gallery will be used as the drainage gallery, nearly pre-mining conditions can be expected. As the pre-mining water make ($5\text{--}9 \cdot 10^6 \text{ m}^3 \text{ a}^{-1}$) nearly equalled the water make during the active mining period ($8\text{--}10 \cdot 10^6 \text{ m}^3 \text{ a}^{-1}$) the typical decrease during the flooding process will not clearly be noticed. The reason for that behaviour is that most of the water make comes from the huge dewatered area above the Markus-Semmler drainage gallery and only smaller amounts from the interconnected deeper aquifers. A flooding up to the level of the Markus-Semmler adit will therefore not change much of the current situation (Wolkersdorfer 1996).

Banks (2001) summarises the two characteristic types of inflows into mines, which can be used to understand the reasons for the water make at mines such as the Niederschlema/Alberoda mine:

- A head-independent flow, derived from direct infiltration of precipitation through covering strata, worked veins or mine voids which are open from the surface. This inflow will possibly vary seasonally (or even daily) with rainfall, but in the long term can be regarded as a constant.
- A limited number of head-dependent flows, entering the mine from adjacent saturated ground water reservoirs via permeable horizons, fractures, or abandoned exploration boreholes. The natural ground water head in the reservoir at some distance from the mine drives the inflow from each of these features.

Impacts of an aquifer which is used for drinking water production have been predicted for the flooding of the Durham coalfield (Younger 1993). Three centuries of coal mining resulted in the dewatering of a large area which is hydraulically well connected by mine workings. Due to ground water rebound that water will percolate through the abandoned workings and also connect different drinking water resources. From the flooding of the adjacent coal fields it was known that the raising mine water will be enriched with pollutants and will have a high acidity. The following potential consequences were identified by Younger (1993):

- ground water pollution in the adjacent Basal Permian Sands aquifer
- leaching from landfills intersected by the rising water table
- compromised integrity of foundations
- increased flows in old sewers below the new rest water level
- corrosion of other buried services

- surface gas emissions
- long-term subsidence risk from old workings

Therefore, the Horden mine water treatment plant and pumping scheme was put into operation in 2004 to keep the mine water well under the Basal Permian Sands. The plant pumps 150 L s^{-1} from the abandoned Horden Colliery main shaft and treats the water in a HDS treatment plant (Coulton et al. (2004), Fig. 87).

An extensive rebound prediction was also conducted for the Slovenian Mežica lead zinc mine. Based on tracer tests, laboratory investigations, and data from the active mining period a total flooding time of 1245 days was predicted (Veselič et al. 2001; Prestor et al. 2003). Mine water rebound started in December 1994 and 119 days later, in April 1995, the water drained through the +418 mine level. Veselič et al. (2001) and Prestor et al. (2003) recalculated the data and concluded that the area of the cone of depression and the hydraulic conductivity of the rock matrix were overestimated. In the next paragraphs it will be shortly described that the researchers possibly did not overestimate those values, but overestimated the influence of the dewatered rock matrix on the rebound course.

Some of the largest abandoned mining regions in Europe are the Aachen/Germany and South-Limburg/Netherlands coal mining districts, where in 1992 mining ceased after 800 years of active mining. During that time a cone of depression with an area of 400 km^2 built up but since 1992 the mine water rose by about 200–240 m, thus diminishing the dewatered area. The whole ground water rebound was well monitored and controlled by using a step-by-step approach and up to now that procedure suited well for that mining area (Heitfeld et al. 2002).

As a conclusion of this chapter it must be stated that rebound prediction, though several good analytical, empirical, and numerical codes are available is still complicated. Even if all conceivable data and experiences were put into the model the rebound prediction might fail. A comparison of the methodological procedures shows that complicated, numerical models might as well fail as simple empirical models. If the failures described above and in the literature (e.g. Veselič and Norton 1997; Younger and Adams 1999) are weighted up, their reasons can usually be seen in the following data areas:

- misinterpreting the cone of depression (usually too large)
- over estimating the significance of the volume of the dewatered rock matrix
- insufficient data about the volume of the mine voids
- wrong conceptual model

At least the second point can be explained from a hydrodynamic point of view by using a mine voids-matrix coupling ratio c_r . First, the mine voids are filled rather quickly while the rock matrix will stay unsaturated for a comparatively longer time. After the mine voids are completely filled the rock matrix saturates again until the hydraulic regime reaches an equilibrium – which, of course, will not be the same as the pre-mining condition. The reason for the slow saturation of the rock matrix can be seen in the fact that the rock matrix consists of three different types of porosities with fast and slow flow occurring in those porosities: flow porosity, residual porosity, and diffusion porosity (Norton and Knapp 1977). Therefore, the volume of the voids is the main factor for the rebound and the volume of the rock matrix accounts only as a minor part. It is understandable that the coupling of those processes has to consider Darcy flow in the far field of the cone of depression, as well as non-Darcy flow in the voids and the parts of the cone of depression with high hydraulic gradients together with turbulent and non-turbulent situations – this is what makes the situation complicated (Darcy flow is defined as a laminar flow for a viscous fluid through isotropic porous media).

This chapter – at first view – seems to be disappointing for those willing to predict mine water rebound. Indeed the situation is not that bad and a nearly correct prediction is better than no prediction at all! If all necessary data are considered, a good conceptual model with a first empirical calculation is made and after enough data are available, a more sophisticated numerical model is used, so that predictions are, to a certain extent, satisfying. Future research in this field must focus on the rock matrix–mine voids interaction, because this mechanism was disregarded to a large extent in the existing models.

5.10 Unexpected Mine Water Discharges During Flooding

As has been seen in the previous chapter the prediction of ground water rebound can fail and in such cases any preventive measures are just not available in the first place. Such mine water emerging in unexpected or inappropriate places, with unforeseen contamination loads or at underestimated total flows can range from small seepages to catastrophic mine water outbreaks. Some of those failures have already been addressed in section 5.7 or will be described in the case studies (chapter 12). Furthermore, the facts that can be learned from those failures can be found in many recommendations or hints throughout this work without referring to them in all instances. This chapter will focus on unexpected discharges from abandoned mines during or after the flooding process. Failures or accidents during the operational period of the mine had already been addressed in earlier chapters and dam failures, though one of the accidents with the most public interest, are out of the scope of this book.

One of the more prominent failures during the flooding of an abandoned mine is the Cornish Wheal Jane tin mine in the United Kingdom. In January 1992, at a time when the water was allowed to rise in the abandoned workings due to pump maintenance works, the water level reached the Nangiles adit and 40,000 m³ of polluted mine water discharged into the Carnon River, the Restranguet Creek and Carrick Roads of the Fal Estuary (Younger 2002e). Though the accident found a wide interest in the media and was described as a catastrophe for the region, investigations two years later showed that natural attenuation and the robustness of the organisms were strong enough to overcome the accident and most parameters came back to pre-accident conditions (Somerfield et al. 1994; Hunt and Howard 1994). After the accident, an active and a pilot scale passive treatment scheme were constructed at the site (Hamilton et al. 1994a, 1997; Johnson and Hallberg 2002; Walder et al. 2005; Younger et al. 2005).

Another accident was the mine water outbreak at the Anna Stollen (Anna adit) Nassereith/Austria in June 1999. This adit had been abandoned about 50 years ago and the mine water was drained through a concrete pipe through the Nassereith conglomerate. In the course of time parts of the adit and the pipe became blocked with fine grained material from the conglomerate. During normal precipitation periods the system worked fine but after the extreme wet spring of 1999, the system was no longer able to drain the mine water. Consequently a large mine water pool built up behind the blockage and the system finally failed giving rise to a huge mine water outbreak and mud flow which partly flooded the nearby valley (Amezaga et al. 2004; Weber 2004; Eckhart and Riepler 2004). Luckily the mine was a lead-zinc mine within carbonate host rocks and the chemical water quality therefore was of no threat to the local community or the environment. A similar accident had already occurred at the same location in July 1948 and again in June 1955, the reasons being always the blockage of the Anna drainage adit. Interestingly, the 1948 “mine water explosion” could have been foreseen and prevented as the mining authority noticed in 1935 that the adit was blocked 13 m behind the mine portal. In a report to the mining authority Gustav Hiebleitner calculated that the total volume of water that had accumulated behind the blockage was some 30,000 m³ (Mining Authority Hall Zl. 1112/1935; 2075/1948; 1342/1955).

A comparable accident happened in August 2002 when the *Rothschönberg Stollen*, the central drainage adit of the German Freiberg/Saxony mining area collapsed after a catastrophic rain-fall event lasting for nearly 4 days. This collapse led to a partial flooding of the famous mining academy's teaching and tourist mine (Fig. 15), the mine water level having risen by about 20 m. By an intensive remediation and re-construction measure the functioning of the drainage system could be re-established (Kugler and Weyer 2002; Weyer 2003).

During the research for this chapter it became clear that many "unexpected" mine water discharges after flooding are known, but – obviously – people tend not to publish such failures, or try to interpret them in a positive manner. One of such cases is the abandoned Dresden/Germany Coschütz/Gittersee uranium mine. First, flooding was slower than expected (Gatzweiler et al. 2002) and then the mine water discharged at unforeseen places with an unexpected contamination load (Thieme 2005).

What can be learned – again – is that old workings and forgotten connections between the modern and past workings are the most dangerous threat in the mining industry. It is therefore essential that mining authorities keep track of mine layout maps and that all kind of investigations are conducted to locate forgotten workings in long abandoned mining areas. The Quecreek accident could probably have been avoided would certified maps of the abandoned Harrison № 2 have been available (Pauley et al. 2003). A recently developed and improved method to identify unknown acid mine water discharges are multi-spectral airborne investigations. They were successfully used in the Upper Arkansas River Basin/USA, where natural acid drainage in altered rocks occurs, and in several Pennsylvanian river basins for locating acid mine water discharges of abandoned underground mines (Yager et al. 1998; Williams et al. 2002; Sams and Veloski 2003; Hammack et al. 2003b; Ackman 2003; Hammack et al. 2003a; Sams et al. 2003; Al-Fouzan et al. 2004; Sares et al. 2004).

5.11 Mine Closure Legislation and Guidelines – Selected Aspects

Mine closure legislation differs from country to country, though some general conclusions can be drawn from the existing investigations of mine closure. Detailed studies are given in McCourt (1999), Anckorn and del Olmo (1999), Hámár et al. (2002), or Kroll et al. (2002a, 2002b), most of them with a European focus. Anyhow, mine closure legislation covers a huge area of legal aspects, such as waste, soil protection, water protection, socio-economical impacts, and other relevant legal issues. Furthermore, regulations, standards and laws change quickly and therefore an in depth discussion of those matters would be outdated before the publication of this book and would cover several hundreds of pages. For example, Burchi and Mechlem (2005) collected the most important treaties and other legal instruments about ground water in international law and came up with a 584-page volume! A collection of international and cross-boundary laws and directives related especially to mining has been published by Pring (1999). Though his work is not focused on water, he gives many references, where international water law impacts mining issues, for example the "Ramsar Convention on Wetlands of International Importance".

Though not meant as a legislative piece of paper, the Berlin Guidelines on "Mining and the Environment" since its publication in 1991 (United Nations Department of Technical Cooperation for Development and Development Policy Forum of the German Foundation for International Development 1992) has resulted in a lot of recommendations and international directives which were partly incorporated in national legislation (Natural Resources Management Unit Division „Protection of the Environment and Natural Resources“ 1999). As Berlin I did not address mine closure these recommendations did not much effect mine closure legislation (see appendix III). Yet, Berlin II also addresses abandoned mines and several aspects of mine closure were discussed at this round table meeting, thereunder mine water issues.

On a world wide basis, most water laws do not explicitly refer to mining. Some exceptions are recommendations and directives of the European Union, and the Palestine's West Bank, Kenya, Australia, African Incomati and Maputo Watercourses (Burchi and Mechlem 2005). This fact does not mean that mine closure or mine water issues are not regulated in other countries; it just means that there is no special reference to mine water in the water laws (Plinke and Wildhagen 2003)!

In many countries, mine closure planning nowadays starts with the mine management plans and it is compulsory to include the mine closure plans already in the environmental impact assessment (EAI): in a way to say "from cradle to cradle". This movement towards sustainable and environmentally sound mining is mainly due to the influence of the United Nations Environment Program and the World Bank Group which connects new lendings for mining enterprises to the principles of environmentally friendly procedures and their guidelines and directives (Strongman 2000). One of the key aspects and the six most important safeguard policies can be related to:

- environmental assessment (including Environmental Management Plans)
- natural habitats
- dams
- indigenous people
- involuntary resettlement
- cultural property

Sheldon et al. (2002) summarised the future issues of mine closure in the following way:

The traditional approach toward environmental protection and reclamation has largely been for mining companies to take care of any surface safety hazards and secure the pits, flooding open pits and, where needed, providing for pumping or drainage of water from underground workings nearby and sealing the entryways, shafts, and adits. This is no longer sufficient. Good practice today requires removing unwanted plant and equipment, stabilising and securing waste dumps and impoundments, detoxifying hazardous materials, protecting ground water, addressing any acid rock drainage issues (which in the worst cases can be a severe problem), and reclaiming, rehabilitating, and revegetating land in a manner compatible with local vegetation. Such environmental protection and clean-up apply not only to the mine site but also to all locations affected by the operation, including any rivers, ports, and other transshipment points, some of which may be a hundred miles or more from the mine site. For example, in the case of the Ok Tedi mine in Papua New Guinea, the transshipment port is over 200 miles from the mine, and the area impacted by the disposal of mine waste and tailings extends over 1,200 miles from the mine site to the Torres Straits.

Currently, the Global Alliance, a union of different acid mine water initiatives and mining companies, is preparing the GARD: "Global ARD Guide (GARD guide) – Guide for the Prediction, Prevention and Mitigation of Neutral and Acid Drainage from Sulphide Wastes" which will be finished by about 2007.

Independent from any legal issues, if the BATNEEC principle is used throughout the whole mine closure process, the legislative requirements will be met in most instances on a world-wide scale. BATNEEC means "best available technology not entailing excessive costs" and is for example used by the Irish Environmental Protection Agency (Environmental Protection Agency 2001; Flynn et al. 2005). As BATNEEC is not a static procedure, but depends on the improvements of existing and challenges of new technologies, the BATNEEC principle guar-

antees that most environmental aspects are met during mine closure. Yet, it could be feasible in some instances to set up more stringent rules than those that could be derived from BATNEEC principles. If, for example, mine water might pollute drinking water reservoirs, an even better and more expensive technology might be demanded by the authorities.

Legal issues of mine waters in Europe have intensively been studied by the PECOMINES and ERMITE projects (Jordan et al. 2004; ERMITE Consortium et al. 2004). Those publications partly cover legal aspects of mine closure as well. A publication which falls into a similar category has been conducted by the French Bureau de Recherches Géologiques et Minières (BRGM) and covers mainly waste issues from the European mining industry (Charbonnier 2001).

Another aspect, which will become important in the future, is environmental liability (Betlem and Brans 2002; Bernasconi and Betlem 2003). The European Commission has issued a Proposal on Environmental Liability (COM[2002]17) and each mine operator will do its best to exclude any future liability that could be traced back to his mining operation. Mining rehabilitation is mandatory in Republic Act 7942 or the Philippine Mining Act of 1995. Remediation and rehabilitation of abandoned/mined-out areas are thus accorded top priority in policy agenda to address negative impacts of mining projects. Several legal aspects of mine closure in Iberoamerica were discussed in the first Iberoamerican seminar on mine closure held in La Rabida in 2000 (Villas Bôas and Barreto 2000). Experts from Mexico, Peru, and Chile reported about the experiences with new mine closure legislation with respect to environmental protection.

As can be seen, the regulative or legislative procedures that are necessary to close working mines are numerous and well adopted in most developed countries. By international treaties and the role of international organizations they are also followed in many developing countries. Over and above the efforts of INAP, ICMM, the Global Alliance to produce “Global Acid Mine Drainage/Mine Leachate Remediation Guidelines” will lead in this direction. If this is true for the global players in the mining industry, small scale mining in all countries stays a real problem for the environment, as Armando Zamora has summarised (Zamora 1999). Therefore, the problem of closed or abandoned mines is to a lesser degree associated to current mining, but to past mining periods and small-scale mining (Balkau 1999). As discussed in other chapters mine water pollution from abandoned mines is a worldwide problem and no matter how highly developed a country is, all of them have their yellow or red streams somewhere in their backyard.

6 Do's and Don'ts of Mine Closure

6.1 General remarks

During a mine closure process, whatever reason caused the closure, the focus of nearly every work is to close the mine as quick, environmentally sound, and cheap as possible. Even in those cases where a proper mine closure plan was available, people rarely think of the possibility that years later problems might evolve and a new or extended remediation project has to be initiated.

Mine water researchers, mining or industrial archaeologists, and economic geologists desire that shafts or underground workings would not be backfilled or fully closed. Yet, the safety reasons for the public and the mine operator are obvious and therefore, the wishes of the first group cannot be fulfilled at all times. Fortunately many closed or abandoned mines are converted to visitor mines so that research is still possible and deeper understandings of flooding processes, the life and work of the historic miners, and the genesis of the ore deposit are still possible. Also from a mining heritage point of view a total remediation of all the mining's remnant should be avoided because those remnants tell us much about our own past (Shepherd 1993; Lynch 2002; Anonymous 1999b; Carvajal and González 2000). Of course, some of the most polluted abandoned mine sites could also be seen as study area for mass transport and past mining activities such as the Spanish Rio Tinto or the German Harz Mountains area (Dobler 1999; Leblanc et al. 2000; Carvajal and González 2000). Many countries, for example Malta or Sweden, have heritage issues included for mine closure operations (Wolkersdorfer and Göbel 2001a; Lindgren and Destouni 2002)

Of course, from the view of a mine water researcher there would be many wishes of how to equip a mine that is going to be flooded, the simplest being a continuous monitoring of the temperature or electrical conductivity development during and after the flooding process. Such measurements would help to increase the understanding of flooding processes and thus could help to develop strategies of environmental safe mine flooding. There are already many mines where a monitored flooding process was of use for later remediation strategies (see chapter 12). Because many mine operators, understandably, just want to close down their mine this wish will probably remain a long way off.

Another fact, yet it will not be covered here, are the social-economic effects associated with mine closure. Miners and their families lose their regular income, people are moving away and formerly prosperous regions can lose most of their citizens. Cases like that can be seen all over the world and when no-one else lives in the former mining place the likelihood of nobody taking care of the abandoned and remediated mine workings might become a problem. Fortunately, modern mine operations require closure strategies well before the first machinery that starts digging the hole and therefore community interests can be covered well in advance of the closure (Robertson et al. 1998; Robertson and Shaw 1999; Hoskin 2000; Hoskin 2003).

This chapter will list some facts that could be done during a mine closure process and that help later researchers to gain access to the old workings without reducing the safety needs of the operator. Regrettably, this list is far away from being complete and all types of suggestions are welcome and will be included in a later edition.

6.2 Shafts

Shafts should not be fully backfilled if it is not necessary for reasons of ground stability or the safety of engineering constructions. Usually, a concrete plug with pipes that gain access for sampling equipment or borehole cameras will suite all the needs concerning safety. The cases

where fully backfilled shafts caused ground subsidence above the backfill are numerous (e.g. Skinner and Beckett 1993; Klapperich et al. 2001; Busch et al. 2002; Sroka et al. 2003; Meier et al. 2004) thus proving that this technique therefore cannot be called “state of the art”. The pipe should have a diameter of at least 8 cm (3 inch) because most down hole equipment would fit through that hole. As those small pipes tend to twist in such plugs within time or during cementation it is recommended to use larger diameter pipes of at least 20 cm (7 inch). The current procedure in many areas of pumping concrete into abandoned mines *ad infinitum* does not meet the needs for mining heritage and later scientific investigations of abandoned mines. “Severe danger” is not given in all the cases where shafts were filled with concrete just because other techniques were not known to the responsible person. It is necessary to establish and use grouting techniques that save both the mining heritage and engineering constructions on the surface. Where concrete plugs are not wanted for whichever reason polyurethane foam (PUF) has been successfully used for shaft sealing (Suman et al. 2000).

Furthermore, from the safety view, using unclassified material or material that could rot should always be avoided! In the Picher Mining District/USA unused cars, railroad ties, or oil tanks were used for shaft sealing (Suman et al. 2000). Some abandoned shafts in the Mulreesh–Portnealon lead-zinc mining area of Islay/Scotland (Wilson and Flett 1921) were covered with corrugated iron and waste material and a small shaft in Tyrol/Austria was just covered with meanwhile rotten planks. And – of course – shafts should never be left open as animals or people could fall into the openings (Fig. 35). Even when fenced the latter method is highly dangerous and does not pose a long term protection.



Fig. 35: Unsecured, open shaft in the abandoned Coolgardie/Australia gold mining field.

One of the main dangers for sampling equipment and downhole probes are leftovers floating in the mine water and especially in the flooded shaft. Common barriers already found in shafts are wood, steel cables, plastic foils, files, gas masks, plastic pipes, and whatever was used underground. Such objects floating in the mine water could easily be avoided if all on-

setting stations would be secured with structural steel lattices. To avoid high amounts of suspended solids it is useful to construct 1—1.5 m high dams at distances of 5—10 meters before the onsetting stations. Of course, the rotting of wooden shaft furniture or pipes in the shaft compartments cannot always be avoided (Fig. 36), but the floating of wire or shot firing cables in which probes could get caught *is* avoidable.

A positive case is the clearing-out and rehabilitation of the Virginia shaft at the abandoned Mechernich/Germany lead mine. This abandoned mine shaft was restored in 1986 to get access to a drainage gallery under a waste disposal site and after waste water was seeping into the drainage gallery the shaft was secured and the access to the underground gallery was made possible again (Benning et al. 2002; Mair 2002).



Fig. 36: Rotten shaft furniture in a flooded shaft of the closed Schwaz/Austria dolostone mine. The water surface is just a little bit below the left corner of the shaft.

6.3 Adits

Closure of adits was already addressed in section 3.7.4 but in another context. For safety reasons all adits have to be closed carefully but they should be closed in that way that future researchers or bat protectors could get access to the adit at a later stage. Bat protection and mining research (which covers all kind of scientific research within an underground mine) does not exclude each other but the experiences concerning the co-operation between both groups are not very promising in many countries. Again, the Mechernich/Germany mine is a good example of co-operation between environmentalists and mine researchers (Thies 2003).

From the experiences of many visitor mines and mining authorities the closure of an adit with a door causes “door researchers” to test the stability of such constructions. It is interesting to see how many different methods were developed by those groups to get access, including the use of explosives and sledgehammers. As a general rule such doors must be as stable as possible to prevent unauthorised access. Door hinges and padlocks must be inside the adit and they should be protected against humidity. It is a good practise to leave a sign near the entrance of how access to the adit could be gained. Examples of such procedures can be found at many caves throughout the world where a relatively common code of honour has already been agreed between most cave researchers.

If polluted mine water is draining through the adit huge amounts of iron oxide sludge could accumulate behind the adit’s sealing. Such a case has been observed at the abandoned Gern-

rode/Germany fluorspar mine. One meter behind the adit's sealing a low wall was constructed to prevent waste from building up behind the drainage pipe (Tamme 2002; Schindler 2003; Wolkersdorfer et al. 2003). Within two decades iron-hydroxide at a length of approximately 500 meters built up behind that wall. In August 2001 and December 2003 the adit was inspected by a consultant company and the authorities. The persons walking in the sludge initiated a sludge avalanche that polluted the receiving stream at a length of at least 3 kilometres. The reason for the sludge accumulation can mainly be seen in a blockage of the drainage pipe which caused a rise in the water level and thereafter the sludge accumulation. This example shows how important the regular inspection of such sealing systems is. Therefore, those systems must be accessible for inspection at all times, which is only possible if there is no total sealing.

6.4 Choosing the final Water Level

A key question for every mine operator is where to establish the final water level of a flooded mine – in other words: where shall the underground beach be established? It has been described in an earlier chapter that the final water level shall be as high as possible to prevent the disulphide oxidation. In spite of that, disulphide oxidation in flooded mines preferentially occurs within the range of seasonal fluctuation of the water table, due to alternating high and low oxygen concentrations as well as redox values within the highest and lowest water levels. Short term fluctuations occur at times of storm events or after longer periods of precipitation. While the water level is lowered disulphides are subjected to oxidation and the processes described in section 4.2 will produce acidity, metalliferous leachates, and easily soluble secondary minerals (Fig. 37). In arsenic rich-environments the varying redox conditions could facilitate the release of arsenic (Berg et al. 2001) which will also be the case for antimony. When the mine water table rises, those secondary minerals will be hydrolysed and mobilised. Consequently, acidity (“juvenile acidity”, section 5.4) as well as metals will be released into the water because of those wetting-drying-wetting cycles (Frostad et al. 2002) until – after

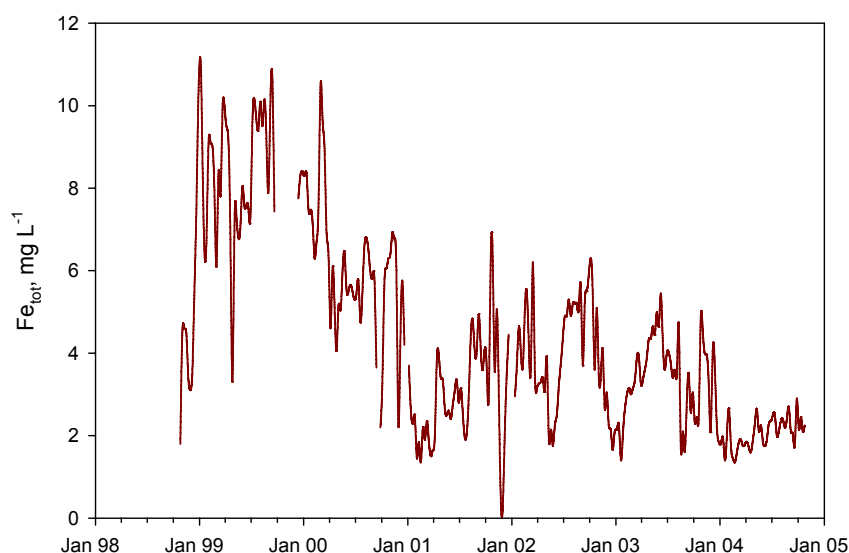


Fig. 37: Seasonal variation of Fe_{tot} in the Glasebach/Germany visitor mine (sampling point A1b; data kindly provided by BST Mansfeld GmbH and GVV). The sampling point is directly under a spoil heap which shows strong seasonal fluctuations of the total iron concentration (each data point represent the arithmetic mean of three measurements a week).

decades – all of the disulphide minerals are either exhausted or coated. As a result, the contaminant load of the discharged mine water increases up to a factor of 10 and adversely affects the receiving water courses (Younger 2000d). Apparently, the long term prediction of such discharges becomes rather uncertain and therefore, such a situation belongs to the conceivably worst scenarios after mine abandonment (Wood et al. 1999).

Water table fluctuations are usually minimal at adit mines but can reach several decametres in shaft mines, which are dewatered by pumping. Water table fluctuation will also occur at adit mines which are drained through drainage pipelines because they are capable to discharge only a certain amount of water. A similar situation occurs if sealed adit mines become leaky and mine water trickles through the mine seal. Furthermore, if the mine water discharge has a big seasonal variation, which can go as far as no water will drain out of the mine, oxidation within the range of fluctuation will occur as well. If the mine water starts to discharge again, larger contaminant loads will occur. Such situations can be found in regions with big seasonal differences of precipitation and temperature, for example in Spain or Australia. For inspection reasons, observation boreholes should be drilled into the mine water body and if long term observations are necessary, they must be constructed in that way that they function throughout the whole observation period planned.

In karst areas fluctuations of the water level might cause dissolution of the lime- or dolostone and consequently the formation of sinkholes (Zhou 1997; Li and Zhou 1999). Those sinkholes can cause disastrous destructions of engineering constructions and can cause water intrusions with many casualties (Newton 1984). Though water fluctuations in limestone areas are less dangerous with respect to producing acidity, as most of the acidity is buffered by the carbonates, they must be considered for these reasons. Water level fluctuations can also be induced by seismic events. In the abandoned Niederschlema/Alberoda/Germany uranium mine, a flooding induced micro-seismic event on November 21st 1995 caused a water level fluctuation of 34 cm recorded within an hour. During the August 18th 1959 Montana/USA earth quake, three shafts in the Pennsylvanian Wilkes-Barre area experienced water level fluctuations of up to 3 cm (Hopkins and Simpson 1960). Yet, those unexpected and random water level fluctuations need not be considered for mine water management procedures as the duration of the fluctuation is usually of short period and the effects therefore are insignificant.

For the calculation of the final water level hydrogeological and hydrological data is needed, including measurements of precipitation, ground water recharge and the observation of ground water level fluctuations by means of piezometers. Though the piezometric water level fluctuations, identified through hydrograph recession analyses, cannot directly be assigned to the final flooding scenario, they give an idea of the potential fluctuations which can be numerically calculated (chapter 8). Besides the hydrogeological data, knowledge about the location of disulphide bearing strata and acid producing waste rock or tailings is necessary. Based on those data the overall strategy of the mine closure scenario must be to avoid water table fluctuations within the potentially acid producing material of the mine and its left-overs. Never, the fluctuating water level should be within the reach of spoil heaps or tailings disposal sites. If acid-producing material is to be deposited within the abandoned mine or the open pit that material must be well deposited under or above the lowest potential water table to exclude the discharge of polluted mine or ground water. If necessary, technical installations, such as drainage pipes, groutings, or dams can be used to insulate the acid producing material against oxygen and water (Prediction Workgroup of the Acid Drainage Technology Initiative 2000; U.S. Environmental Protection Agency et al. 2001). Fluctuating water levels usually cause worse drainage water quality than totally dry or fully submerged acid producing material (Perry et al. 1997).

7 Hydrodynamics

7.1 Flow Patterns in Flooded Mines

7.1.1 Introduction

This section describes the types of flow that can occur in flooded vertical, horizontal or cave-like mine voids (shafts, inclines, adits, galleries, stopes). It is based on theoretical work about hydrodynamics and flow in enclosures and explains the flow in some real examples. Currently, hydrodynamic investigations in flooded mines are an evolving area of research and therefore there are not many examples available that try to combine theory and *in-situ* investigations.

In hydrogeological investigations, Darcy flow is the most common type of flow considered (eq. 75). This flow is defined as a laminar flow for a viscous fluid through isotropic porous media. Yet, if the Reynolds number for porous media exceeds approximately 10, the flow becomes non-linear (which means turbulent) and the flow consequently becomes non-Darcian. Because the subject of non-Darcy flow, especially when it comes to convective flow, is very complex, some basic hydrodynamic understanding will be required. Detailed descriptions of the theoretical hydrodynamic background can be found in the relevant literature (e.g. Gebhart et al. 1988; Landau and Lifschitz 1991; Batchelor 2000; Oertel 2001; Drazin 2002; and many others).

7.1.2 Type of Flow in Flooded Voids

Three different types of non-Darcy flow or combinations of them might occur within the open voids of flooded underground mines: laminar or turbulent Poiseuille flow, laminar or turbulent convective flow, or almost no flow except Brownian molecular forces or diffusion. The dimensionless numbers for describing the transition from laminar to turbulent flow are the Reynolds number Re for the Poiseuille flow and the Rayleigh number Ra for convective and Rayleigh-Bénard flow:

$$Re = \frac{L \times v \times \rho}{\mu} \quad 28$$

$$Ra = \frac{L^3 \times \rho^2 \times g \times \beta \times \Delta T \times C_p}{\mu \times k} \quad 29$$

with L : characteristic length, m
 v : kinematic viscosity, $m^2 s^{-1}$
 ρ : density of fluid, $kg m^{-3}$
 μ : dynamic viscosity of fluid, $Ns m^{-2}$
 g : gravitational acceleration, $m s^{-2}$
 β : coefficient of thermal expansion, K^{-1}
 ΔT : temperature difference, K
 C_p : specific heat capacity, $kJ kg^{-1} K^{-1}$
 k : thermal diffusivity, –

According to Landau and Lifschitz 1991 the transition from laminar to turbulent flow in enclosures occurs at critical Reynolds numbers Re_{Kr} between 10—100. Many books and articles about hydrogeology refer to Re_{Kr} as being between 6 and 10 in porous aquifers e.g. Nuttall et

al. 2002; Langguth and Voigt 2004; Hölting and Coldewey 2005) but hydrodynamic experiments about the transition from laminar to turbulent flow clearly showed that many variables are responsible for Re_{Kr} and that fixed numbers cannot be given with absolute certainty, because Re_{Kr} always depends on the fluid and the nature of the flow that has to be investigated (e.g. Fedotov et al. 2002; Frigaard and Nouar 2003; Sharp and Adrian 2004). Only for the flow in a horizontal cylindrical pipe without wall roughness and water Re_{Kr} is about 2000 (Sharp and Adrian 2004), but can reach values as large as 10^5 (Landau and Lifschitz 1991).

Free convective flow in vertical enclosures only occurs in systems where the Rayleigh number Ra is larger than the critical Rayleigh number Ra_{kr} (Landau and Lifschitz 1991), which is between 658 for two free surfaces and 1708 for no free surface. Small diameter bore holes or wells can therefore not be used for measuring or estimating convective flow in natural systems or in flooded mines!

Furthermore, the flow regime might be characterised by free flow (buoyancy driven, also called natural flow) or forced flow. Free flow develops, when the density differences $\Delta\rho$ within the fluid are the only driving forces of the flow, while forced flow develops, when external forces cause the fluid to flow. However, free and forced flow does not exclude each other and in many cases of practical interest both effects are of significant magnitude (Gebhart et al. 1988). Such cases, with both flow characteristics involved, are called mixed convection flow. In free convective flow dominated systems, the dimensionless Grashof number Gr replaces the Reynolds number Re :

$$Gr = \frac{L^3 \times \rho^2 \times \beta \times \Delta T \times g}{\mu^2} \quad (30)$$

with L : characteristic length, m
 ρ : density of fluid, kg m^{-3}
 β : coefficient of thermal expansion, K^{-1}
 ΔT : temperature difference, K
 g : gravitational acceleration, m s^{-2}
 μ : dynamic viscosity of fluid, Ns m^{-2}

In flooded underground mines all combinations of the above listed flow types can be assumed to exist at the same time. Changes of flow type might occur within short distances and short time intervals, too. Even if Darcy flow in flooded mines is of minor importance, such flow characteristics will occur wherever the water flows through backfill, downfall, or goaf material (the latter case was numerically modelled by Jäger et al. 1990).

To describe the flow regime in flooded voids, the Navier-Stokes differential equation for incompressible fluids is applied. An explicit solution of this equation can only be given for very simple cases, but in fact, a mine cannot be called a simple case. For such situations, numerical methods have to be used to model the fluid flow. Unfortunately, the number of cells needed to appropriate model turbulent flow grows with the Reynolds number as $Re^{3/4}$ and the memory needed for such calculations in the order of $Re^{9/4}$. Numerical CFD simulations of flooded underground mines conducted by the author in 1996 as closed cavity simulations (FIDAP 7.52; Sun Workstation) and another simulation by Unger (2002) (ANSYS FLOTTRAN 2.2, Convex Workstation) clearly proved that no convergence for turbulent flow was reached within an adequate time. Furthermore, the model cases computed in laminar flow situations only converged when calculated in dimensionless matter.

Besides channel flow within the voids, fracture and porous flow through the rock matrix occurs as soon as all the voids are filled with water (Norton and Knapp 1977; Jäger et al. 1990; Wolkersdorfer 1996). The permeability decreases with the mine's depth (Nordstrom et al.

1989; Wolkersdorfer 1996) and also depends on the thickness of the disaggregation zones around the voids ranging between 0 and 3 m with maxima of 8 m (Müller-Salzburg 1978; Jacobi and Everling 1981; Militzer et al. 1986; Stoll and Bauer 1991). Whilst those fractured zones, where the water-rock interactions occur, are important for the chemical composition of the mine water, they are of minor importance for the flow within the voids.

Water levels in a pipe network are only similar when the density of the water is similar in the whole pipe network. Therefore, reasons for forced flow within flooded mines can also be wrongly calculated weir heights for controlling the water flow within adjacent mine parts (see section 12.11.3) or density differences in interconnected shafts. During the flooding of the Friedenshall/Germany salt mine Pelzel et al. (1972) observed water level differences of several meters in the shafts. They concluded that those differences are due to the density differences between the saline mine water the interconnected shafts. The difference in the water level Δh of two interconnected shafts with water levels h_1 and h_2 can be calculated according to the following equation:

$$\Delta h = \frac{\rho_1 \times h_1 \times \left(1 - \frac{\rho_2}{\rho_1}\right)}{\rho_2} \quad (31)$$

with Δh : head difference between the shafts, m
 h_1 : height of water column in shaft 1, m
 ρ_1 : density of mine water in shaft 1, kg m⁻³
 ρ_2 : density of mine water in shaft 2, kg m⁻³

Sander and Herbert (1985) conducted depth dependent temperature and velocity measurements in several flooded German salt mines. Their investigations focused on potential isolation effects of radioactive waste deposits against the anthroposphere. Yet, the results from their investigations are important for understanding temperature and electrical conductivity measurements in other mines as well. It becomes clear that zones with nearly constant physico-chemical parameters are dominated by turbulent flow and zones with gradients in the temperature or electrical conductivity are diffusion dominated (they call those layers 'static layers'). A typical situation is shown in Fig. 38. Zones 1 and 3 are characterized by constant temperature and fluctuating velocity. This is an indication for a turbulent flow situation. Because the water does not discharge from the shaft there must also be a convection occurring in this layer. On the other hand, zones 2 and 3 are characterised by a temperature gradient and no velocity fluctuations, which indicates a more or less static situation. Sander and Herbert (1985) conclude: "from laboratory experiments and *in situ* tracer tests it may be deduced that the transport of matter: (a) in static layers takes place by diffusion, i.e. very slowly; (b) in turbulently mixed layers is chiefly a function of flow, i.e. very fast; at the boundaries of the layers the flow reverses; (c) across the boundaries of the layers is essentially due to diffusion, even if each of the two adjacent solutions is in a state of turbulent flow."

In the Niederschlema/Alberoda, Brixlegg, and Straßberg mines, turbulent flow must be assumed from the results of time-dependent temperature measurements. As can be seen from Fig. 39, temperature and electrical conductivity oscillations are a characteristic feature in those mines. Wolkersdorfer (1996), Kolitsch et al. (2001), and Johnson and Younger (2002) conducted *in-situ* measurements of mine water velocities in flooded shafts, where they observed characteristic oscillations of velocity and temperature. Such oscillations with extremely irregular and unsystematic temporal variation in velocity and temperature are characteristic for a developed turbulent flow regime with high Reynolds or Rayleigh numbers Re and Ra (Landau and Lifschitz 1991). Bau and Torrance (1981a, 1981b, 1983) studied open and closed fluid loops, which can be compared to the situation in flooded underground mines

with two or more shafts. They found out that those systems, when turning turbulent, are of chaotic nature and that the flow starts depending on critical heating situations and values as well as perturbations and eventually can reverse. Furthermore, they concluded that “free convection loops provide a means for circulating fluid without the use of pumps” (Bau and Torrance 1981b).

Results of tracer tests in flooded underground mines showed that the flow in mine voids tends to be nearly always turbulent, caused by the rocks’ roughness, the mine support, broken roof, and waste material left back by the miners. Otherwise it could not be explained why the tracer breakthrough usually arrives quicker than predicted. Velocities in flooded mines are reported to be in the range of $0.3\text{--}1.7\text{ m min}^{-1}$ (29 tracer tests, 95% confidence interval of all velocities except the maximum and minimum; Wolkersdorfer 2002b). However, to react in the same way as the fluid, the tracer particles must be small compared to the convection cells, which Landau and Lifschitz (1991) show for general cases of fluid-suspended particles reaction under fully developed turbulent flow. Unfortunately, according to the suggestions of Mandelbrot (1982) (and, of course, in his other outstanding works) the solution of the Navier-Stokes equation must be a fractal – and consequently of chaotic nature. This explains that “turbulence is one of the oldest, most complicated, and frustrating chapters in physics” (Mandelbrot 1982)!

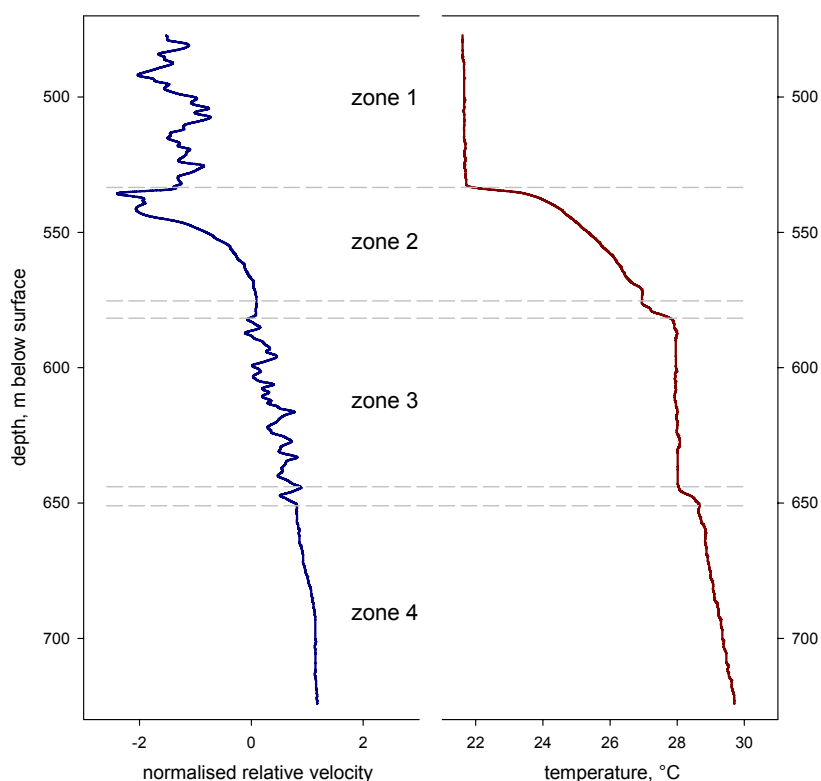


Fig. 38: Depth dependent velocity and temperature measurements in the shaft of an anonymous flooded salt mine (modified after Sander and Herbert 1985). Zones 1 and 3 are characteristic for turbulent convective flow and zones 2 and 3 for diffusive flow (static layer). Between zones 2 and 3 as well as 3 and 4 are intermediate layers which isolate the main zones from each other.

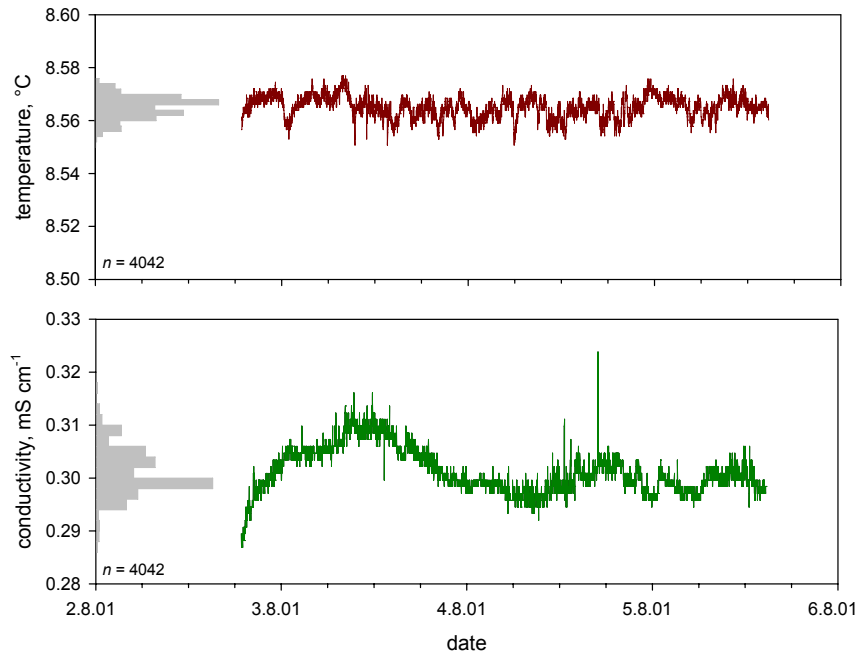


Fig. 39: Time dependent temperature and electrical conductivity fluctuations. 68 hour continuous measurement in the Georgi Unterbau blind shaft. The histograms at the left side are an indication for the near Gauss-distribution of the measured values.

7.2 Stratification in Flooded Underground Mines

7.2.1 Description of the Stratification “Problem”

Water stratification or layering occurs when water is separated into horizontal layers of different physical or chemical characteristics (Fig. 40). Whilst the stratification might be a consequence of changes in chemical (e.g. contents in sulphate, sodium chloride), physical properties (e.g. temperature, viscosity, density) or turbidity the stratification itself is always due to density differences of the fluid (Fig. 41). A classical case would be a layer of fresh ground water above a highly saline mine water (Fig. 121).

The stratification might be stable or unstable, depending on the temperature or density differences within the single layers of water (Gebhart et al. 1988). Generally, stratification can be assumed stable when low temperature differences or high density differences *within* the stratified layer occur. Because water is a Newtonian fluid, pressure differences can usually be neglected. Stable stratification, written in simple terms, exists, when the following conditions apply:

$$\frac{d}{dx} \rho_{\infty}(T_{\infty}, p_{\infty}) \leq 0 \quad (32)$$

with x : depth, m
 ρ : fluid density, g cm⁻³
 T : temperature, K
 p : depth dependent pressure, Pa

To compare different types of stratification, the dimensionless Richardson number Ri is used. This number describes the ratio between power and energy and determines whether two layers in a fluid will mix with each other or not:

$$Ri = \frac{g \times \frac{d\rho}{dx}}{\rho \times \left(\frac{dv}{dx}\right)^2} \quad (33)$$

with g : gravitational acceleration, m s^{-2}
 x : depth, m
 ρ : fluid density, g m^{-3}
 v : velocity in the fluid, m s^{-1}

Stable stratification will occur in cases where $Ri > 0.25$, meaning that no heat or particle interchange between two different layers will occur. If $Ri < 0.25$ internal waves spontaneously occur and break down.

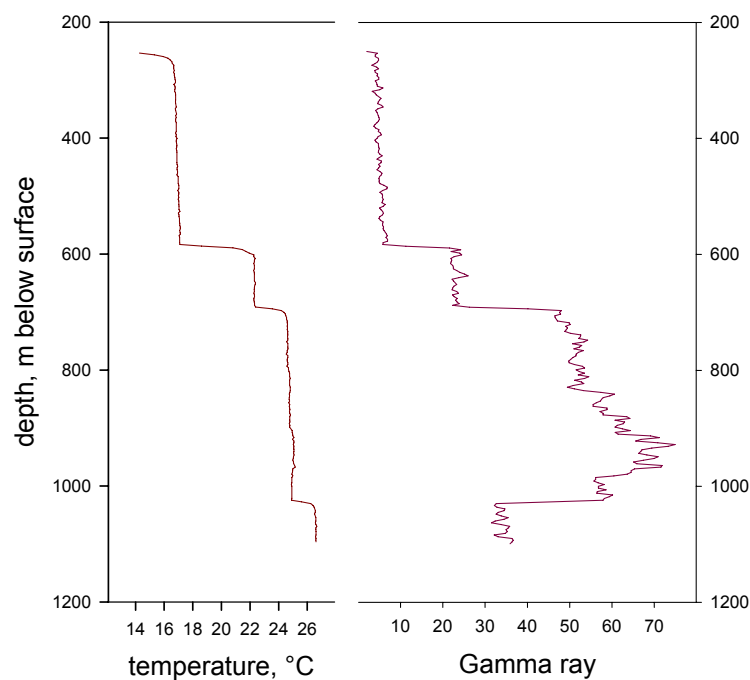


Fig. 40: Temperature stratification in West Rand Consolidated Mines' Deep Shaft. The gamma ray supports the temperature stratification and proves that different types of water bodies lie above each other (data courtesy Alan Harley).

Under such circumstances, the density difference $\Delta\rho$ between the two different stratified layers being very large and the velocity differences Δv being comparatively small, no exchange between the layers will occur. Examples for such cases are temperature differences above 10 K, total dissolved solid differences of above 3% (static stratification) or big differences of turbidity (dynamic stratification), resulting in density differences of 2 kg m^{-3} , 20 kg m^{-3} , and 200 kg m^{-3} , respectively (Kranawettreiser 1989). Uerpmann (1980) experimentally studied such a case, the Niederschlema/Alberoda/Germany 371 II b shaft had a stratification that was stable for a period of at least two years, and the extreme situation of the Atlantis II deep in the Red Sea also proves how stable such stratifications can be (Blanc and Anschutz 1995). On the other hand, the stratification can be broken down by external forces, such as earth quakes, storm events, inflow of fluids with significantly differing density, changes in the overall flow regime, or opening of new adits or voids (e.g. due to a broken dam, construction of a drainage adit, or flooding of a new level during the rebound process).

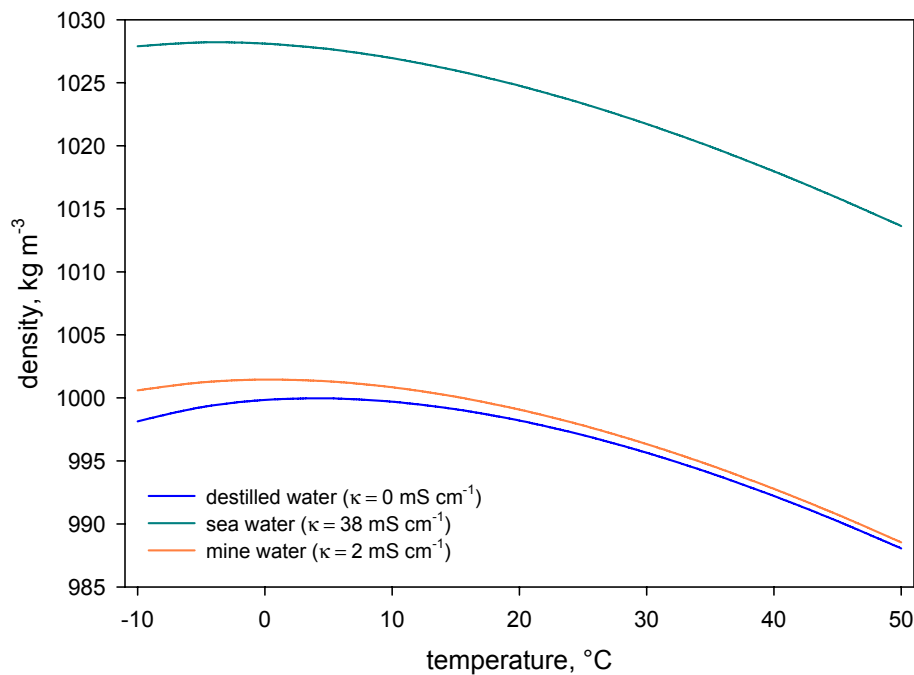


Fig. 41: Calculated density of water as a function of the electrical conductivity κ and temperature T . Calculation is based on Fofonoff and Millard Jr. (1983).

In some of the above mentioned cases, an intermediate layer between the two layers will be established. This intermediate layer is a very effective barrier concerning the vertical transport of dissolved ions, gases, or temperature, and in limnology it is called metalimnion, sometimes also thermocline (Geller et al. 1998). Depending on the thickness Δx of this intermediate layer and the velocity difference between the two layers the upper and lower layers will interfere with each other or not (Faust et al. 1983; Kranawettreiser 1989). No interaction between the two layers will be observed when

$$\Delta x > \frac{(\Delta v)^2}{24 \times g \times \frac{\Delta \rho}{\rho}} \quad (34)$$

with g : gravitational acceleration, m s^{-2}
 x : depth, m
 ρ : fluid density, g m^{-3}
 v : velocity in each of the fluid's layers, m s^{-1}

Water stratification is a common feature in surface lakes and has been studied there intensively (see section 5.5.2). Stratification in natural lakes (e.g. Tietze 1978; Boehrer et al. 2000; Boehrer 2000) as well as mining lakes (e.g. Goodman et al. 1981; Sumioka 1991; Klapper et al. 1996; Murphy 1997; Stevens and Lawrence 1998) is an important case study in all limnological surveys, as the exchange of nutrients, pollutants, or gases is controlled by the seasonal variations of the epilimnion, metalimnion, and hypolimnion distributions. An intensive laboratory and theoretical investigation of stratified saline fluids has also been conducted by Barry et al. (2001). They investigated dependencies of the turbulent length scale and the turbulent diffusivity for mass. Yet, whether their results can be transferred to flooded underground mines needs further studies and comparisons.

A feature yet to be investigated in flooded mine shafts is double-diffusive convection, though there are some indicators (e.g. staircases) for double-diffusion to occur in some of them (e.g. Wismut's 296 II b shaft: Wolkersdorfer (1996) page 127). Convective flow in fluids occurs when density variations are present. Double-diffusive convection takes place, when two or more components of different molecular diffusivities contribute in an opposite sense to the vertical density gradient. Typical scenarios are the dynamics of magma chambers, growing crystals, or salt water/fresh water systems. Yet, the classic example is heat and salt in water, causing "salt-fingers" (Brandt and Fernando 1995).

Double-diffusive convection would explain many features that can be observed in flooded shafts, e.g. a linear temperature distribution and a step-like electrical conductivity distribution. However, as Kelley et al. (2003) point out, staircase profiles can also be a result of mixing striations or internal waves as recently investigated in Lake Constance (Boehrer et al. 2000; Boehrer 2000). Furthermore, the examples and theoretical contributions given in Brandt and Fernando (1995) suggest that double-diffusive convection cannot be excluded for flooded mine shafts. This could also explain such features as the "steps" in the electrical conductivity as shown in Fig. 44.

Slight horizontal or vertical, jet-like inflows into a stratified water body within a shaft, reservoir, or pit lake can destratify the whole system in a comparatively short time even if no forced flow for the whole system is created. Such systems could build up if supply pipes or ore rolls connect different parts of a mine. Small density differences between the jet and the ambient water, exemplified by densimetric Froude numbers Fr_d lower than 10, are sufficient for an effective mixing and can therefore breakdown or prevent the stratification within a flooded mine. Due to that procedure a water body with a formerly quasi-linear temperature distribution will be converted in two clearly separated water bodies as can be seen in Fig. 42 (Stefan and Gu 1992).

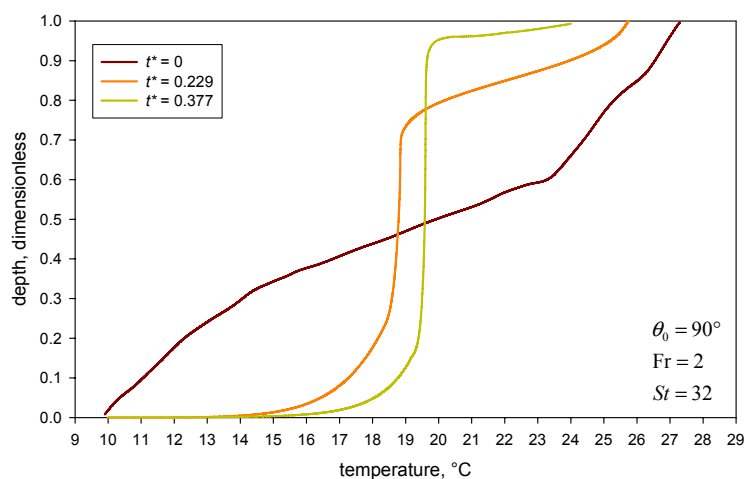


Fig. 42: Temperature distribution in an enclosure prior and after jet-induced mixing. t^* : dimensionless time, θ_0 : jet angle, Fr : Froude number, St : stratification number (after Stefan and Gu 1992).

7.2.2 Stratification in Underground Mines and Shafts

Stuart and Simpson (1961) and Barnes et al. (1964) were the first who reported stratification in flooded underground mines of the Pennsylvanian anthracite fields in the USA (Fig. 98). A first extensive investigation of mines in this region was conducted by Erickson et al. (1982). Another North American mining area where stratification was found to occur was the Picher

Mining District in the Tri-State district (Playton et al. 1978). Though the fact of mine water stratification was known as early as the 1950s, its importance for potential remediation methods or radioactive waste disposal sites was not considered by then. Uerpmann (1980), who measured the stratification in five flooded salt mine shafts, proposed to make use of mine water stratification for securing radioactive waste by implying a special mine lay-out. Similar studies with similar results were conducted by Heidenreich et al. (1991) and Czolbe et al. (1992) who also used numerical models to predict the distribution of radioactive pollutants in flooded mines for radioactive waste disposal. Nearly all published cases with mine water stratification are described in chapter 12.

Stratification does not imply that there is no measurable flow in the mine. During several mine water tracer tests it became clear that the water flows within the stratified parts of the mine. Also comparisons with experimental data and field measurements show that flow occurs in the stratified parts of a water body.

A characteristic feature of stratified mine shafts is the fact that the single mine water bodies are separated from each other usually at the onsetting stations of adits connected to the shaft. Three reasons can be made responsible for this stratification: water inflow from the connected adit, fluid loops through interconnected shafts and adits, and flow disturbance due to changing physical properties of the shaft. A numerical model with FIDAP 7.5 showed that fluid loops can occur and that the direction of those loops changes rapidly with minimum changes of energy input (temperature changes within the measured range of temperature in the shafts). The same appearance was described by Javeri (1995) who investigated potential waste disposal in flooded underground mines using the code TOUGH2.

König and Blömer (1999) found out that with increasing length to diameter ratios of L/d of a shaft the number of convection cells and the velocity increases. They modelled a single-shaft closed cavity thermo convection problem with dimensionless Rayleigh numbers $Ra = 0.03 \cdot 10^6$, $6.2 \cdot 10^6$, and $243 \cdot 10^6$ (see equation 29) using the Navier-Stokes equation and a Boussinesq approximation in the software package Hot-Spring.

In the case of the Ronneburg/Germany uranium mine (for details see below) it is hoped that stratification will occur without intervention (Gatzweiler et al. 2002). Kalka et al. (2002) modelled the density driven vertical convection in the Ronneburg mine caused by the geothermal gradient and found out that those effects are neglectable there. Paul et al. (2002) measured the temperature-depth and salinity-depth profiles in an observation well of the Ronneburg north-west mining field. Their temperature-depth profiles show a quasi linear course with a gradient of $42\text{--}44\text{ K}^{-1}$, thus indicating that diffusive transport dominates and vertical convection at the Ronneburg site might in fact be neglected. However, the geothermal gradient is not the only driving force for free convective flow, but density differences in the fluid due to effects such as mineralization or turbidity. Therefore, the geothermal gradient cannot be used as the only measure to estimate if free convection will arise or not.

Most of the measurements conducted so far indicate that stratification in mines can clearly be traced by temperature as well as electrical conductivity changes. Similar results were obtained by Erickson et al. (1982), who correlated electrical conductivity and sulphate concentrations concluding that acid mine drainage contamination can be mapped by continuous electrical conductivity measurements. Those electrical conductivity and temperature measurements in flooded shafts are a first indication of the chemical and hydrodynamic processes occurring in the mine water pool. Generally, it can be concluded that higher conductivities in the lowermost mine parts are an indication for a higher mineralisation and often pollution while the lower conductivities in the higher parts of a mine water pool indicate lower polluted water or fresh infiltration water.

This dichotomy can be observed in nearly all mines with stratification. It can also be observed in the Austrian Nikolaus-Bader shaft where “pollution” in its usual understanding can be excluded, because none of the exposed rocks or sediments (tillite, dolomite, and limestone) has an acid producing potential or contains significant amounts of potentially toxic metals (Fig. 43). Therefore, the higher conductivities, and thus anion and cation concentrations, are a result of an exclusion from an overall convective loop and from a mixing with the fresh(er) water in the higher levels of that shaft and consequently – in the case of a mine water pool – the higher mine workings.

During a tracer test in the Niederschlema/Alberoda/Germany mine, suspended material within the 366 b shaft was slowly moving to the water’s surface and then reversed the motion, flowing downward. Several of those convective cells were situated near each other, building a flow scheme similar to Bénard cells for natural convective flow. As Bénard cells are typical for thin water layers with surface tension (Gebhart et al. 1988), it might be assumed that the cell structure seen in the Niederschlema/Alberoda mine was either restricted to the upper surface of the flooded shaft, or the convection cell’s symmetry was congruent to Bénard cells. However, the temperature differences in the Niederschlema/Alberoda shafts clearly indicate that Rayleigh-Bénard instability highly likely occurs. Numerical simulations of Javeri (1995) also resulted in convective cells as they were observed *in-situ*. Independent from the question, if the cells are Bénard cells *sensu strictu* or not, convective cells in flooded shafts clearly exist. The continuous temperature, conductivity, and pH measurements during the rise of the mine water in the Niederschlema/Alberoda № 383 shaft can be seen as an indication for such small convective cells (Fig. 44).

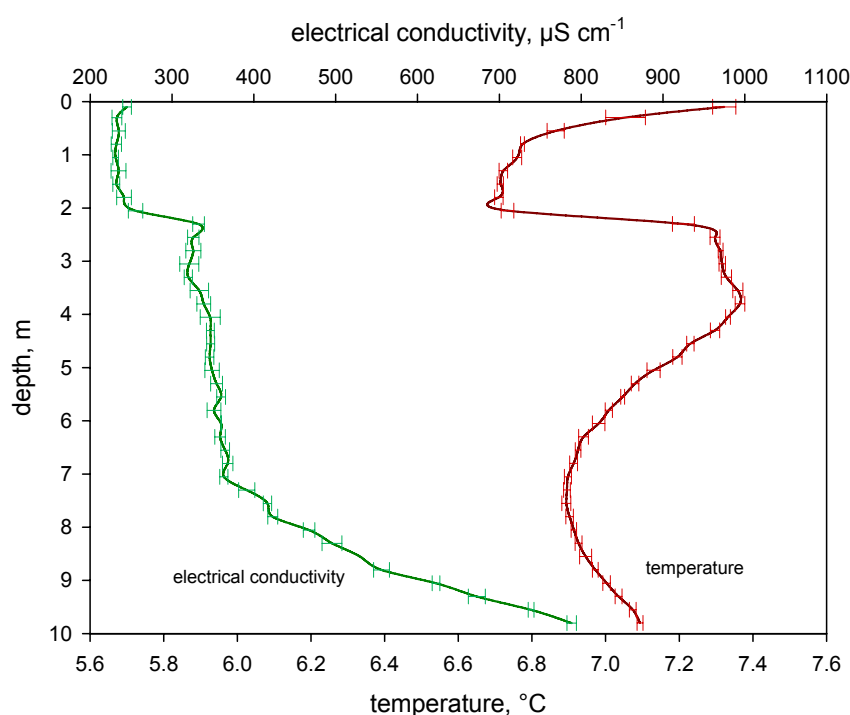


Fig. 43: Temperature and electrical conductivity stratification in the 10 m deep Austrian Nikolaus-Bader shaft, an abandoned small-scale mining site.

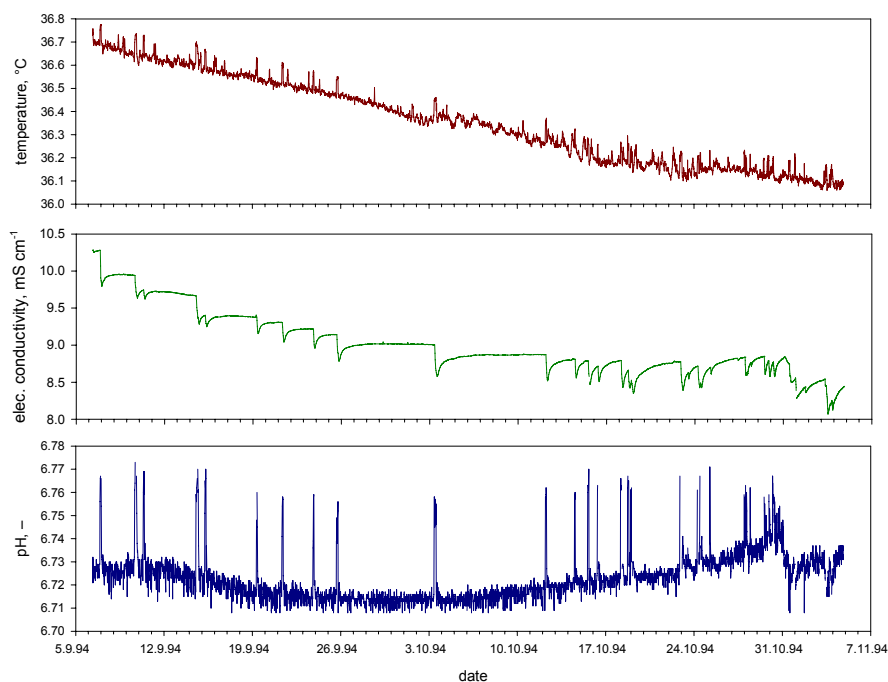


Fig. 44: Continuous temperature, electrical conductivity, and pH measurements in the Niederschlema/Alberoda/Germany № 383 shaft between September and November 1994 while the mine water was rising between levels -945 and -900. Head above probe on September 6th: 42.0 m, November 4th: 60.4 m.

8 Mine Closure Flow Models and Geochemical Modelling

8.1 Introduction

The following sections are a simple classification of models and shall give the novice a hint of what can be used. A basic definition of a model is given in section 5.9. This chapter is not meant to be a detailed classification of model types used in mine water modelling. A thorough description of models used in hydrogeology or the mine water field is somehow painstaking, as there is a large overlap between the different models – and even definitions what a ‘model’ is. Furthermore, some models have different capabilities and could be grouped into more than one category (see Christopher Wells’ “Groundwater Modeling Tools” summary at the In-foMine homepage).

Most modellers know that “all models are wrong but some are useful” (Box 1976, 1979). Therefore, one of the first decisions when modelling mine water flow or rebound is: which model shall I use? There are nearly as many models as model developers and to answer the question which model to use is nearly impossible. Many people start with the model selection and afterwards they compile their data, yet, the way round is less “pebbly” and should therefore be used instead. There seems no logical reason for using a sophisticated numerical finite element or finite difference model for a mine and its environment if the data available is not sufficient or too fuzzy. Simple model approaches sometimes yield as good results as an x-thousand cell CFD-model (computational fluid dynamics) would give. Yet, in case you belong to those people who think that CFD means “colourful fluid dynamics” there is no reason for you to read this section and you should directly jump to chapter 9.

Modelling of flows in flooded underground mines helps to understand the release of pollutants from abandoned mines. Such “mine water” models can be used for studying the release of naturally occurring pollutants, such as metals released from acid mine drainage, as well as anthropogenic pollutants emanating from underground waste repositories for nuclear and non-nuclear waste. As described in section 10.1.3 the network of shafts, workings, and galleries can be seen as a karst conduit network and most of the considerations for flow situations from karst aquifers are similar to those of flooded underground mines. For simplification reasons a flooded underground mine can also be looked upon a network of pipes which eases the effort necessary to model each feature found in a mine. Collon-Drouaillet et al. (2005) used this modelling approach for the South Basin of the Lorraine/France iron-ore deposits. In this chapter some of the potential modelling techniques that were used so far will be described. Because each mine differs from the other and most models were extremely site specific and were adapted to the special requirements of a single mine it would not make sound sense to discuss those approaches here.

Several approaches to model the flow in mines have already been described above, in chapter 7. Yet, no discussion about the possible modelling techniques has been given there. This chapter will describe the principal possibilities to model flow in flooded mines. For each of the possibilities a short example will be given. Because every mine has its special geometry and for every model a special conceptual model (section 5.9) with special questions is involved, it is currently not possible to give recommendations which modelling approach should be used. As the geometry of mines is usually more complicated than a “normal” aquifer and even more complicated than a turbine or motor, mine models are comparatively complicated.

In principle, three approaches can be used to model flows: analytical, empirical, and numerical. Analytical models for a real flooded underground mine would be rather complicated – if possible at all and no comprehensive analytical model of a flooded mine is known so far.

Thanks to the improvement in computer technologies, 3-D-visualisation nowadays is a convenient means to show the complicated interconnections in underground workings and the chemical or physical developments that might appear during or after the flooding process. Yet, those sophisticated pre-processors must not deceive into believing that the real work has to be done by the modeller not the pre-processor!

Governing equations of flow in a conduit and matrix network have to include water flow, heat transport, and solute transport. They are derived from the conservation of mass, momentum, and energy laws. Those laws mean that the mass, momentum, and energy in cell one of a model must stay the same in cell two of the model. Furthermore, as temperature, concentration, and pressure are changing in real world mines, the equations of state for density and viscosity have to be included in mine water flow modelling approaches.

In hydrogeological investigations the governing equations for modelling water flow are Darcy's law and the law of mass conservation. For those standard situations, many analytical solutions and numerical codes are available and were used successfully in many different conditions (U.S. Environmental Protection Agency 2003; Hiscock 2005). Yet Darcy's law is restricted to flow in porous media and to laminar flow as well as small flow velocities. All those prerequisites are not fulfilled in flooded underground mines, as they are neither "porous" nor is the flow always laminar and, as can be seen in Tab. 19 the flow velocities might be rather big. However, several researchers simplified the real situation so that a Darcy approach could be used and the mine operators or regulators benefited from numerical codes such as MODFLOW. Some of those investigations are described hereafter. However, several software packages were especially developed for mine water flooding investigations.

8.2 Empirical Models

Empirical models are used when the physical properties or chemical reactions involved are not known detailed enough to analyse them by analytical methods or to set up a numerical model. In contrast to expert systems or expert knowledge, empirical models are not necessarily based on a detailed knowledge about the processes involved. Usually, the data for the empirical model comes from laboratory or field experiments and their systematic connections to the governing equations involved are not clear at first instance. In semi-empirical approaches parts of the model might be empirical; others can be based on a supporting theory (Boudjema et al. 2003).

One of the best known empirical equations to hydrogeologists or hydrologists is the Darcy law (eq. 75). In 1856 Henry Philibert Gaspard Darcy investigated the public water supply system in Dijon/France (Darcy 1856). In an attempt to improve water purification through sand filters he measured the piezometric heads before and after a vertical sand filter filled with siliceous sand of the river Saone. He conducted 23 experiments with 4 different column heads under atmospheric pressure and 12 experiments with changing pressures at the filter outlet and concluded that "pour un sable de même nature, on peut admettre que le volume débité est proportionnel à la charge et en raison inverse de l'épaisseur de la couche traversée" (Darcy (1856), Appendice – Note D: Détermination des lois d'écoulement de l'eau à travers le sable). Darcy's empirical law, if described in modern words, is a simplified averaged momentum balance equation for a Newtonian fluid in porous media, consequently the Navier-Stokes equation. It can also deduced analytically (Bear and Bachmat 1991).

Other empirical models are the discharge equations for thin plate weirs or for flumes as described in section 9.2.1. Nearly all of the equations given there, though their physical background is known very well, are empirically deduced from experiments with the equations only valid for the specific weir or flume type.

Empirical models use different statistical, stochastic, or mathematical techniques to reproduce the “real world” parameters (Fig. 45; Box et al. 1978). Their aim can be to describe a physical law or to predict the future or past development of a measured dataset. Such techniques known to most researchers in the mining field are simple or multivariate regression analyses, curve fitting algorithms, parameter fitting packages (see section 9.3.4), or polynomial spline curves. Other methods are Fourier series analyses, wavelets, NURBS (non-uniform rational B-spline), or neural networks.

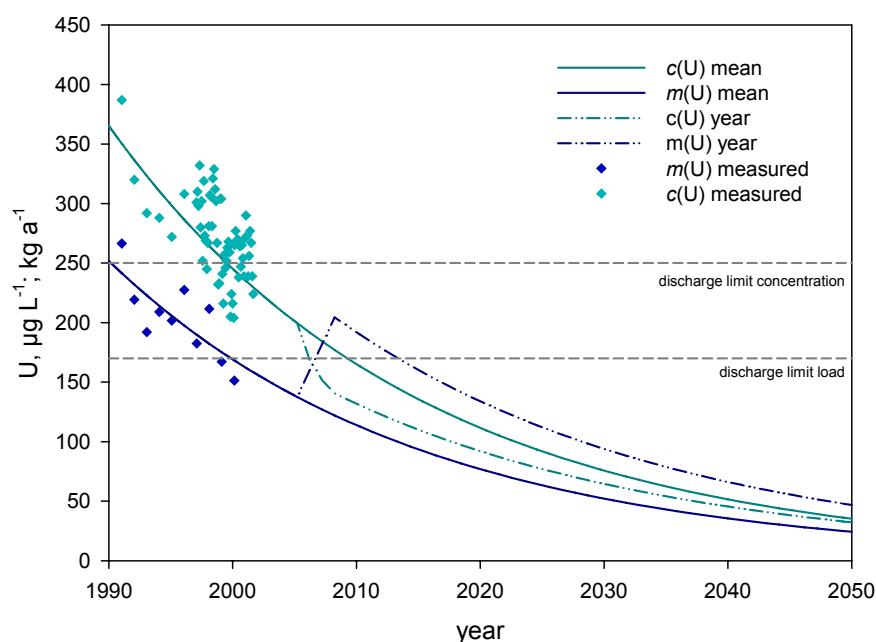


Fig. 45: Result of a black box model for the uranium concentrations and loads of an abandoned uranium mine related to dewatering boreholes to lower the uranium concentrations. $c(U)$: uranium concentration; $m(U)$: uranium mass load. Forward predictions without advance dewatering boreholes (full line) and with dewatering boreholes (dotted line). While the concentrations decrease after drilling, the total load will increase. The discharge limits for the uranium concentration and loads are predefined by the authorities.

Though splines might be useful to fit simple existing data with only one variable, their use is restricted if it comes to prediction models or to more complicated hydrodynamic situations. They are usually used to model discrete data but empirical mine water models based on splines are not known to date. Yet, if the knot position and spline coefficients can be deduced precisely enough and represent the data adequately, most discharge curves or mine flooding predictions could be based on those data.

Fourier series analyses are commonly used in the interpretation of geophysical data sets. Al-Fouzan et al. (2004) describe an airborne mine water investigation by using frequency-domain electromagnetic (FDEM) conductivity investigations. To remove the noise in their data they used Fourier filtering. Yet, empirical mine water models based on Fourier series have not been published so far, though this method can be useful to extract seasonal variations or systematic fluctuations. By using a Fourier series analysis fluctuations in the head of a mine water discharge in the Ehrenfriedersdorf/Saxony tin mine were analysed. The hypothesis of the mine operator was that those fluctuations are due to switching on and off mine water pumps, but the Fourier series analyses could clearly assign those fluctuations to tidal effects.

Wavelets are a modern method to analyze measured time series data according to their scale where the amplitudes or wavelengths of the important cycles are time dependent (Graps 1995; for example they are used in compressing pictures according to the JPEG 2000 standard of the Joint Photographic Experts Group). The method originated from signal and image processing, but is nowadays used in a broad range of applications. In contrast to Fourier time series analyses wavelets are used for time-frequency representations with the advantage of a good time and frequency representation. Results of such wavelet analyses are the variabilities of a time series within time (Torrence and Compo 1998; Lord et al. 2000). So far, wavelet analyses of time series in mine water applications have not been done. Yet, they can be used to interpret time series of concentrations, or to find characteristic patterns in those time series that could help to understand the changes in mine water chemistry after a certain – not obvious – treatment option has been applied. One of the most comprehensive commercial software packages, which also has the capability to use Fourier transforms and many other options to analyse measurements, is SeraSolve's AutoSignal (SeaSolve Software 2003).

8.3 Analytical Models

Analytical or semi-analytical models for real underground mines have to find an explicit solution for the general transport equation. For obvious reasons this is an impossibility – at least with the mathematical methods known today. Yet, for comparatively simple cases or for solute transport in porous media such solutions have already been presented (Grisak and Pickens 1981). Grisak and Pickens (1981) were the first who presented an analytical solution for tracer flow through a single fracture. Pfingsten and Mull (1990) provided an analytical solution for flow in fractures with connected fractures and Chan et al. (1987) compared analytical solutions for a single fracture with the numerical 3-D-code MOTIF (model of transport in fractured/porous media) while Reimus and Haga (1999) provided analytical solutions for a double porosity conceptual model. Further analytical solutions for fracture and multiple fracture flow were presented by Tang et al. (1981) and Tsang and Tsang (1987).

Interesting case studies were conducted by Bau and Torrance (1981a, 1981b, 1983). They conducted experiments with open and closed fluid loops which were heated and cooled at different locations along the loop. Though their intention was to find analytical solutions for the flow within solar panels, their results have essential implications for the flow in flooded underground mines as discussed earlier.

Simple mine geometries and consequently simple conceptional models can be used if only parts of a mine shall be investigated. Such approaches were used by several authors (e.g. Fawcett et al. 1984; Singh and Atkins 1984, 1985). Fawcett et al. (1984) investigated the water inflow into coal mines by assuming that a mine acts like a big well. They used the Theis and Jacob approaches for confined and leaky aquifers (Theis 1935, 1940) and thus were able to estimate the amount of water flowing into a mine.

A novel analytical solution for open pit lake filling was presented by Fontaine et al. (2003). For real cases they came to the conclusion that the lateral extent of the cone of depression is not met exactly enough and that the time of the maximum extent of dewatering as well as the temporal effects on springs and seeps cannot be modelled in all detail. For such requirements they conclude that a full numerical code and substantially more data are necessary.

Tracer transport in fractures was investigated by Małoszewski and Zuber (1984, 1985). They found an analytical solution for the diffusive mass flux from fractures into the matrix with a Dirac tracer injection. In a later publication they also provided an analytical solution for a single fracture with diffusion processes into the matrix which they called Single Fissure Dispersion Model (SFDm; Małoszewski and Zuber 1990).

8.4 Numerical Models

Numerical codes are nowadays commonly used to model mine water rebound, reactive transport, and the hydrogeochemical development of mine waters, or ground water. Those numerical models, most of them based on Darcy's law for porous aquifers and laminar flow, have to a large extent been applied to ground water investigations (Deming 2002; Hiscock 2005; Hölting and Coldewey 2005). Examples for commercial and non-commercial numerical codes used in mine water rebound and flow predictions are MODFLOW, VULCAN, FEFLOW, SHETRAN VSS-NET, ANSYS FLOTRAN, MIFIM, KASOMO, GRAM or SPRING of which some can also deal with non-Darcy flow (Fig. 46). Most of the models are either finite element or finite difference models with no preference to either methods and only a smaller number of models is based on other numerical methods. Advanced models cover nearly every aspect encountered in a mining environment and complex models are able to simulate multi-phase flow and multi-species transport. Yet, all those codes have to deal with a major problem in mine water research: the lack of consistent geometrical and hydrogeochemical as well as hydrogeological data. Because all those models require boundary conditions to solve the continuity equation, many starting parameters are based on weak empirical data or in the worst case have to be guessed.

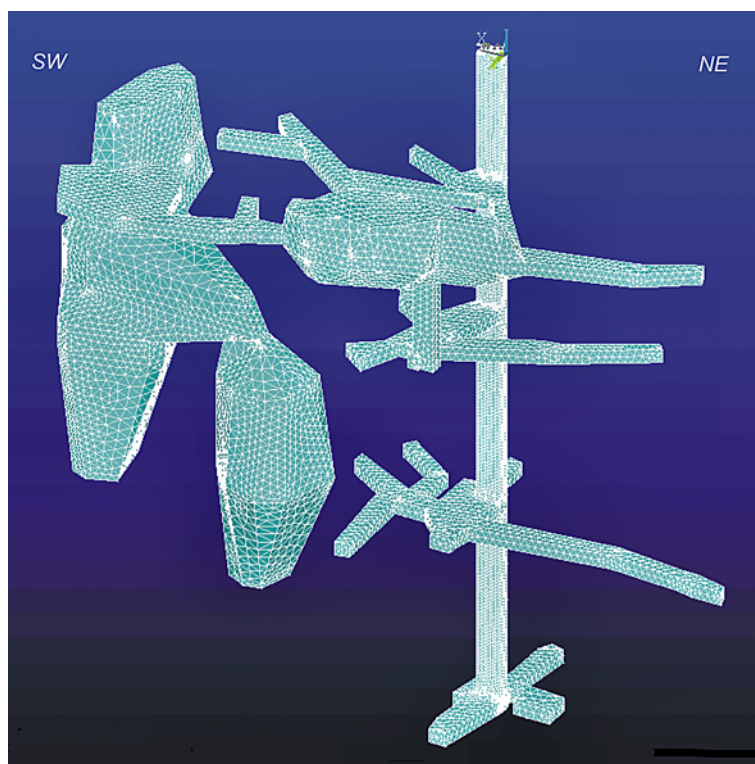


Fig. 46: ANSYS-FLOTRAN finite difference mesh for the numerical model of the Austrian Georgi Unterbau mine (modified from Unger 2002).

As described earlier, Darcy's law for flow in porous media, laminar or turbulent Poiseuille flow, laminar or turbulent convective flow, or almost no flow will occur together in a mine environment. Furthermore, different physical properties of the rock matrix, the fissures, and the mine workings must be considered. Though some codes have the capability of dealing with Darcy as well as non-Darcy flow situations, the implementation of both flow types might become a trial of patience. Exhausting calibration procedures, often based on trial and error measures, are necessary to obtain trustworthy results. Because mined strata are not a standard condition for most standard numerical codes, the reason for those long lasting calibration pro-

cedures might be seen in the non-standard use of those codes. To overcome several of those restrictions, black box (“lumped” parameter) modelling approaches were used widely in solving mine water rebound and flow problems (e.g. Sherwood and Younger 1994). Especially when many input parameters are involved, it is sometimes unclear, which parameter should be fitted to get the best result. In a pipe network, for example, where the velocities within the pipes are too large, the head difference, the roughness of the inner walls, or the diameter of the pipes can be slightly modified. A change in any of those parameters would result in lower velocities and it is often unclear which screw is the right one to be driven. In such circumstances, modellers tend to say that the parameters are unidentifiable.

Depending on the mine layout and the mining technique several approaches can be used. Since the first numerical codes were used for modelling mine water rebound and mine water flow, the computer technology with respect to hard- and software as well as the numerical approaches experienced a steady development. Most models are site specific and adapted to a special situation, and only a few of them were used for modelling other situations as well. Describing them here would therefore just be like a spot-light in one direction because the code development is rather quick. Yet, some principles shall be picked out.

To overcome some of the above mentioned problems, Frydrych et al. (2000) used a mixed-hybrid FEM model for calculating the flow field and the transport of the chemical compounds of the Stráž uranium mine (Czech Republic). For the time discretization, which covered only 7 of the expected 30 years flooding period, they used the Rothe method. A total of 20,000 trilateral elements had to be calculated and they modelled 2560 days with 10-days-time-steps. Novák et al. (1995) used another numerical model at the same uranium mine and both numerical models resulted in similar TDS distributions in the Stráž block, though showing differences in the details.

Heidenreich et al. (1991) used a numerical model which was originally developed to predict the formation of salt caverns. In the version used by the authors, it was extended to cover also the needs for radioactive waste disposal modelling. This model copes with free and forced convection, diffusion, heat transport in the fluid and rock matrix, salt dissolution, natural and radioactive heat sources and radioactive decay. Both, the Navier-Stokes and Darcy equations can be used, depending on the question to be solved. Their model included two key scenarios with one shaft and two levels: one with a blind shaft between the two levels, the other one without the shaft. Their model clearly proved that the blind shaft between the two levels increased the speed of radionuclide transport and should therefore be sealed in case the mine shall be used as a radioactive waste disposal. Furthermore, Heidenreich et al. (1991) describe the commonly known situation of flooded salt mines where a fresh water lens overlays the brine without a significant mixture between the two layers over time. They concluded that such a situation might be advantageous for radioactive waste disposal and therefore modelled a flooded shaft with a brine overlain by fresh water. As the gravitational forces exceed those of the thermo convection that layering stayed stable for a long time. All together the numerical model of the authors is compatible with the “multiple” and “single shaft” approach deduced from mine water tracer tests (Wolkersdorfer 2006).

8.5 Models Based on Monte Carlo Methods

Monte Carlo simulations use randomly generated values from a pre-defined range of potential input parameters based on the probability distributions. They can be combined with nearly all the modelling approaches described here and are commonly used to evaluate the sensitivity of certain parameters in a given model. This modelling approach is also used to evaluate minimum or maximum results of models through multiple runs of the same model with randomly chosen values according to the Monte Carlo method. Its scientifically based introduction dates

back to 1946, when this method was used in the Manhattan project (Metropolis and Ulam 1949), yet Monte Carlo methods were used before with a lesser statistical background. In the mine water field many modelling approaches are based on Monte Carlo simulations as many variables are of high uncertainty.

Aurada (1970), for example, modelled the ground water rebound after the cessation of the Mansfeld/Germany mining area. Based on a computer code (*Grubenwasseranstau Mansfelder Mulde*) he set up a numerical simulation, where he randomly selected hydrogeological parameters within the range of the observed values such as volume of voids, expected precipitation, interconnections within the mine, and the potential discharge point. The model consisted of 12 “lamellas” representing the hydrological provinces of the mining area, each of which was filled with water by the code. After lamella 12 was filled the code restarted using another, randomly chosen scenario. From his conceptual approach, his model is very similar to the GRAM model (Burke and Younger 2000).

GRAM (Groundwater Rebound in Abandoned Mineworkings; Sherwood and Younger 1994; Burke and Younger 2000) is an analytical model which uses exactly the same conceptual idea as *Grubenwasseranstau Mansfelder Mulde*. It can also be run with a Monte-Carlo sub-routine for predictions under considerable uncertainty (Paul Younger, pers. comm.). Its first application was the prediction of the ground water rebound of the Durham/United Kingdom coal field, which has a size very similar to the Mansfeld mining area. Based on the investigation of the mine geometry and the hydrogeology of the surrounding area, the mine is subdivided into ponds which are then filled by the model. All uncertain model parameters are based on ground water levels, mine water pumping rates, precipitation, volume of voids, interconnections, and discharge areas.

8.6 Geochemical Modelling

Since the outstanding work of Nordstrom (1977) about acid mine drainage, geochemical models are regularly used to interpret and predict mine water environments and mine water treatment scenarios. Alpers and Nordstrom (1999) summarised and discussed such models that can be used in mine water investigations and Nordstrom et al. (1979) compared the results of different numerical models. The most up-to-date publication about the main concepts and key developments in the field of geochemical modelling including examples has been given by Nordstrom (2004a). A good, easily understandable guide to numerical modelling of aquatic systems, using PHREEQC, has been published by Merkel and Planer-Friedrich (2002) in German and English (Merkel et al. 2005b) and Berk (2000) described how to interpret water-rock interactions in mine waste rocks by using PHREEQC. Commonly, the models are referred to as “geochemical models” or “thermodynamic models” However, „thermodynamics“ means in this context chemically thermodynamics. Whatever they are called, they use a database with chemical thermodynamic data, a numerical computer code to solve the governing equations, and they need an experienced scientist who understands the concepts behind such a numerical approach.

Geochemical models are a tool to understand the species distribution in an aqueous solution as well as the theoretical mineral assemblage that would result in a water with the given chemical composition and physical properties. Modern codes can calculate the mass balance, speciation, reaction-path, sorption processes involved, and kinetically controlled reactions (like pyrite oxidation). PHREEQC, nowadays, is the programme used by most hydrogeologists. The most recent version also includes the Pitzer equation for solutions with higher to extreme activities, such as brines. Many researchers who are using numerical codes for interpreting water analyses take the results as 100-percent-correct and representative for the water they analysed. Yet, the chemical thermodynamic database is based on experimental results

which are inherent in that way that all experimental data have an error range. Nitzsche et al. (2000) or Meinrath et al. (2004) varied different constants in the database within their error range and were able to show the uncertainties of such numerical codes – and some of the results might be surprising for those who tend to discuss about the 2nd or 3rd decimal place in the results of the programmes (Fig. 47). Error bars of up to 20% might be obtained if the uncertainties of the codes are considered.

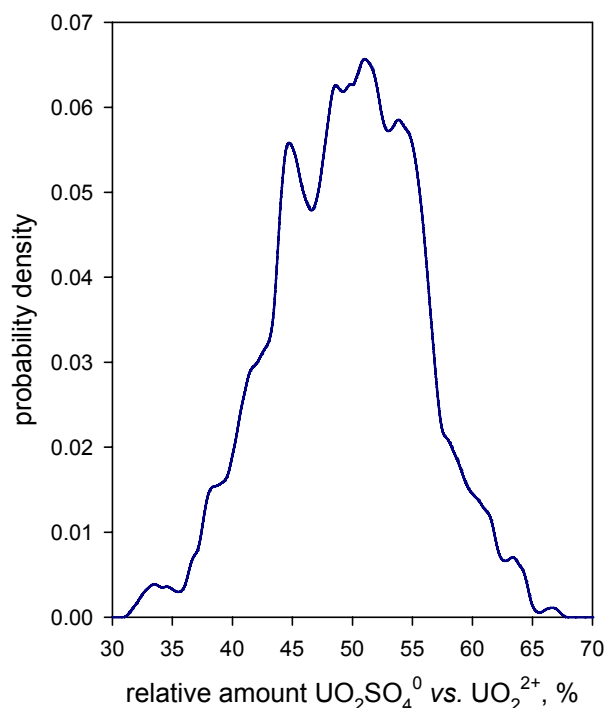


Fig. 47: Results of geochemical modelling for the UO_2SO_4^0 vs. UO_2^{2+} -fractions based on database uncertainties. Acid mine water at a pH of 2.24; $[\text{SO}_4^{2-}] = 3.1 \cdot 10^{-3}$ mol; $[\text{U(VI)}] = 4.2 \cdot 10^{-5}$ mol (modified after Nitzsche et al. 2000).

Numerical codes in the mine water field can be used for a wide range of applications. On one hand they help to understand the processes observed in the field or laboratory, on the other hand they are a useful tool to predict future conditions based on the investigation of different scenarios. When the results of a chemical analysis of a mine water are interpreted by using a numerical approach, it is possible to understand with which mineral phases the water was in contact and it can be further concluded from which part of the mine the water might come from. An example of this approach is the mine water discharging from the abandoned Gernrode/Harz Mountains fluorspar mine. Two of the questions that had to be solved were, how the complicated mine water hydrogeochemistry can be interpreted and how the mine water could be treated. The hypothesis for the chemical composition of the mine water was that “pure” mine water is to a certain degree mixed with highly mineralised mineral water. Because the underground part of the mine was not accessible, no analyses of the water within the mine were possible. Therefore, historical mine water analyses and the analyses of mineral springs in the surrounding of the mine were used. Based on those analyses and the current mine water composition a mixing model was set up in PHREEQC and the mine water was mixed with different mineral waters at different mixing ratios. Based on those results it was concluded that the mine water discharging from the drainage gallery comprises of 15–30% of highly mineralised water and that a separation of that water and the mine water would ease the treatment of the mine water discharging from the drainage gallery (Schindler 2003).

None of the numerical modelling of aqueous speciation can ever substitute for an experienced scientist. There are many publications discussing the fact that a numerical code calculated the over saturation of some 'nice' iron phases and the authors were expecting precipitation of those phases, but they were not able to identify them in the sediment or water! Everybody in the mine water management field, managers, researchers, authorities, stakeholders, and politicians must know that the results of geochemical modelling can only be relied on if the results are properly interpreted by an experienced modeller. Furthermore, modelling should be based on observations in the field or in the laboratory. Croxford et al. (2004) and Coulton et al. (2004) gave an excellent example where geochemical modelling was used to optimise an active mine water treatment process. Based on the results of the model laboratory and field experiments were conducted and the final treatment plant was thus optimised.

To overcome the problems with missing kinetics, Hamm et al. (2005) used a simplified approach by just modelling the predominant reactions that they identified through kinetic laboratory tests. Furthermore, they coupled the kinetics to a simplified pipe network model for investigating the impact of backfill material on the water quality of an abandoned iron mine in the Lorraine/France.

9 Monitoring and Sampling

9.1 Introduction

This chapter's objective is to show how important reliable measurements of hydrogeochemical and physico-chemical parameters of mine water are for the design of mine water treatment facilities and remediation actions. There are numerous scientific investigations which go into all the details of a specific mine site. These investigations are essential for the understanding of the interactions of the compartments involved and the influences of the mine water discharges on the aquatic environment or the water catchment. Yet their aims are far beyond the aims of semi-routine remediation or monitoring purposes of the mine operator or regulator.

All three, working mines, mines under the closure process, and abandoned mines will be covered by the following descriptions. Mine water can significantly impact the good status and ecological potential of water catchments and therefore, the measurement of chemical and physical parameters has to be given special consideration. Prerequisite for a catchment monitoring programme is a site assessment that collected all relevant data for the mine sites within the catchment.

All the techniques used for analysing mine water discharges evolved from ground or surface water monitoring. There are several pitfalls in sampling and monitoring mine water discharges, which this short guide will help to avoid. Sampling programmes should avoid acquisition of unrepresentative data which can result in an inappropriate treatment or remediation design.

Mine water discharge qualities can alter significantly during the period between sample acquisition and analysis. Concentrations of the relevant elements can decrease or increase compared to that of the analysed sample. Therefore, before any treatment or warning decisions can be made, it is essential to obtain as much information about the hydrogeochemical characteristics of the mine water discharge as possible. In addition to the flow measurements and physico-chemical measurements, chemical analyses of the mine water will have to be conducted.

Treatment designs must not be based on water samples taken without on-site measurements of some critical parameters. Furthermore, mine water samples usually need pre-treatment such as filtering or acidification and they have to be transported to the laboratory as quick as possible in cool and dark containers (ca. 4 °C).

In addition to obtaining chemical data on the discharge, it is advisable to consult people with knowledge of the mine and local residents who often have information about extreme situations or other data that could be relevant for the mine water investigation. This information supplements the experience of the consultant.

When establishing a site monitoring programme, it is recommended to conduct a step-wise approach. In a first screening investigation simple parameters (e.g. electrical conductivity, pH, total Fe) can be used as indicators for the worst discharges in the catchment. Based on those results, further water samples can be taken and the number of monitoring stations can be increased or decreased accordingly.

This chapter will give some hints and lists some experiences gained in several years of mine water research. For in depth studies it is highly recommended to consult national or international guidelines (e.g. DIN, ISO, ISO-EU, EPA, DVWK) or the relevant literature, which varies from country to country and is regularly updated.

Since every mine water is generally different in some way from any other mine water, no specific recommendations for a monitoring programme can be given. It is the obligation of the mine owner, its consultant, and the competent authorities to set up a priority list for the parameters to be measured. A first indication of the parameters that should be measured can be found in parts of the appendixes of the European Water Framework Directive (in principal the inorganic parameters), the regional Water Directives and in the case of uranium mining the relevant radiological parameters have to be considered. As a hint the following parameters should be measured as a minimum: (hot) acidity (base capacity K_B), alkalinity (acid capacity K_A), pH, electrical conductivity, temperature, redox-potential, Fe, Mn, Al, sulphate, flow.

Furthermore, though sampling, monitoring, or taking photographs at most mine sites is comparatively uncomplicated, the situation can significantly change if it comes to coal mining. Coal mines, and in some places also other mines such as salt mines, are a source of methane and therefore measurements in coal mines can only be conducted with intrinsically safe instruments. Most countries are very restrictive concerning the use of electronic equipment in potentially methane bearing mine sites and the relevant regulations always have to be followed!

9.2 Flow Measurements

9.2.1 *Direct Flow Measurement*

For calculating the contaminant mass loads in a water catchment area with polluting surface or underground mine water discharges, flow measurement is essential for all treatment actions. Especially for flow measurements at mine sites, some guidance documents were published in recent years but are not commonly available (Minister of Public Works and Government Services 2001; Science Applications International Corporation 2001). Precise flow measurement in open and closed channels is a science of its own (Durgin 1981; Herschy 1995; Miller 1996; Bohl 1998; Brassington 1999; Bonfig 2002), and even simple techniques have several restrictions. Those restrictions must be considered, as mine water discharges can significantly change in their flow: dry streams can become wet and vice versa due to changing hydrologic conditions. Therefore, the flow measurement technique used could be suitable at one time and unsuitable at another time. Each measurement technique has special limitations which influences the accuracy of the flow measurement. For the selection of the correct technique it is therefore essential to know the expected flows and the limitations of the technique. Most investigations are planned for one hydrologic year – or even less – and due to the irregular characteristics of precipitation and hence surface flow it might be that the data gathered are not representative for a longer period of time. This fact has to be kept in mind when conducting direct measurements of flow, where the maxima, minima and mean flow conditions including the seasonal variations shall be recorded at the highest precision obtainable. If long-term gauging stations are situated close to the mine water discharge, missing flow and precipitation data can be correlated to a certain degree with existing data. Precipitation data for a special location can be determined by applying one of the following methods: arithmetic mean, Thiessen Polygon, isohyetal method, hypsometric, or multiquadratic methods (Shaw 1994).

Most researchers and practitioners in the mine water field look upon flow measurement as being a simple task. This chapter shows that this is not the case. Therefore, and to avoid misinterpretations due to a lack of basic physical understanding, this chapter will describe some of the necessities when conducting flow measurements.

For each method many international and national standards (e.g. ISO, DIN, ČSN ISO, Ö-NORM, BS, EN) are available. The International Organization for Standardization (ISO) for

example lists more than 100 different methods for flow measurement. Those standards describe in detail the limitations of each method and under which conditions the method is applicable. Furthermore, some standards include equations to calculate the errors and uncertainties of the flow measurements which have to be considered when comparing different measurements. In the following paragraphs only the methods commonly used in mine water research or remediation projects are described. They assume a channel or pipe flow with a uniform laminar or turbulent flow at the cross section of the flow measurement.



Fig. 48: Flow measurement with Na-fluorescein at the inlet of a lake which receives polluted mine water. Shown is the Dirac-impulse injection of the Na-fluorescein (dark colour when it flows out of the bottle, lighter colour when it mixes with the stream water; photograph Andrea Berger).

In many investigations, pollutants are only given in concentrations (mass of a substance per volume or mass), but equally if not even more important are the total mass loads which are calculated by multiplying the flow with the concentration:

$$M = c \times Q \quad (35)$$

with M : mass load, g s^{-1}
 c : concentration, g L^{-1}
 Q : flow, L s^{-1}

Low concentrations of pollutants at high flow conditions can consequently be as contaminating as high concentrations at low flow conditions. This fact is often overlooked, as people are used to comparing the concentrations of parameters only.

Usually, the national hydrologic stations within the catchment area are insufficient to calculate the total loads of mine water discharges. It is therefore an important task to evaluate the already existing measurement network and – if necessary – to install new flow measurement gauges (in American English: gages) which fit the needs for the planned treatment or monitor-

ing project. Details of how to interpret catchment hydrology and data evaluation are described in Linsley et al. (1988).

Two options are used for flow measurements: continuous (electronic or drum recorders) and discontinuous measurements. Usually, discontinuous flow measurements will be good enough, but cheap and reliable methods are available to measure and store flow rates continuously. In recent years data storage devices became gradually smaller and cheaper and can therefore be installed without major investment. Because high flow conditions are usually the limiting events in any remediation or mine water treatment operation the continuous measurement of flows should be given preference over discontinuous measurements.

Mine water flow measurement is conducted by one of several methods. The techniques used can be divided into primary and secondary flow measurements, whereas primary means a direct measurement of the flow with one variable (e.g. water level: V-notch weirs; pressure: pitot tubes) and secondary involves two variables (e.g. water level and travel time of sound: ultrasonic flow meters). Nearly all the primary methods can be combined with secondary methods. Here is a selection of the most common flow measurement methods in the mine water field:

- bucket-and-stopwatch (up to 500 L min^{-1})
- impeller current meter (Fig. 52)
- V-notch weirs (Fig. 49)
- H-flumes (Fig. 56), Parshall flumes (Fig. 58)
- Venturi tube
- pump run-times and ratings
- water meters (Fig. 59, Fig. 60)
- pressure meter measurements of boreholes
- transit-time acoustic (ultrasonic) flow meters

Where the flow is measured with weirs, flumes, and tubes equations relate the head of the water level above the measuring device to the flow through that device. The water levels are usually measured automatically or manually with level gauges or by a technician using contact free instruments (e.g. ultrasonic methods), floats, or gauge meters. All flow equations relate the flow to the water head at a given distance from the measuring device. Therefore it is essential to install the level gauge or to measure the water level at a proper upstream distance from the installation. Those distances are either given by the manufacturer or, as for weirs, outside the upstream water level influenced by the drawdown.

The best option to measure the water flow must be chosen according to the site specific situation, the man power available, and the water quality. Heavily contaminated water might damage measuring devices or could change the diameter of a channel or weir (iron hydroxide precipitation). If contaminated mine water shall be measured, H-flumes or contact free methods therefore should be given priority. Some mine waters might be contaminant or iron free, but could contain high amounts of carbonates, which might interfere with the measuring devices.

In some circumstances the flow at a chosen location can be estimated with regression or estimation techniques. This requires a certain number of gauging stations and precipitation values from which the flow at another location might be deduced. Besides the standard regression and estimation techniques it is advisable to use geostatistical methods as well (e.g. Davis 1986; Linsley et al. 1988; Lerner et al. 1990; Shaw 1994; Simmers 1998; Burbey et al. 2000).



Fig. 49: V-notch weir measuring the flow of the Blaydon Hazard Shaft site near Newcastle upon Tyne, United Kingdom.

Though flow measurement in underground mines is very important for calculating mass transport, it can often be observed that the weirs or channels used in mines are unsuitable for a proper flow measurement. Bad designs are various (Thomas 1959; United States Department of Agriculture 1979) – in underground mines many of them are simply caused by the rough conditions underground (Fig. 50). Therefore, such technical installations must be calibrated by using independent flow measurement devices. Here is a list of the mistakes commonly found: weirs are constructed directly in the gulleys and the water just flows through the weirs without overflowing them (quasi-submerged flow); the blades are mounted in the wrong direction; channels are not cleaned of sediment; metal parts are bent or are covered with precipitates; the weir or flume is of poor design due to a lack of hydrodynamic understanding. Theoretically, submerged weirs or flumes are covered by equations to calculate flow, but their accuracy is low and a lot of calibration measurements at different flow conditions are needed to get a usable head/flow relationship estimation – on the other hand most ‘submerged flow measurement devices’ are due to poor construction and were originally meant as free flow weirs! A submerged weir for flow measurement is not good practice and must be avoided whenever applicable.

Even if there is a simple method to cross check flow measurements, it is – even in scientific investigations – often not done. Based on the example of an abandoned Austrian Silver mine those mistakes shall be discussed. The mine was investigated at the turn of the 21st century and several of the flow measurement channels were badly maintained (Fig. 50). The flow was measured 10 times at 5 different sampling locations and the mine water coming from sampling locations 1–4 flowed through sampling location 5. The total flow ranged between 22 and 60 L min⁻¹ and the relative difference between the sum of locations 1–4 and location 5 ranged between 1 and 30%. No dependence of that error on the total flow could be observed. 50% of the differences were in an acceptable range of $\pm 10\%$ and the rest exceeded that limit. Neither data about the accuracy of the measurement nor about the uncertainty was available. As we will see, it would have been a simple task to add all the single flow measurements and compare the results with each other.



Fig. 50: Badly maintained channel for flow measurements. The channel inflow is not laminar and the channel itself is filled with sediment.

To detect the “wrong” flow measurements, it is advisable to measure both the flow and the electrical conductivity of the water, which is easily possible even in uncomfortable underground situations. If the flow measuring cross-sections are not too far apart from each other and if no precipitation or dissolution of minerals appears within the area investigated, electrical conductivity can be seen as a conservative parameter for the mine water. Tab. 11 shows how such an accuracy check might be done (data has partly been adopted to show the principle of this method). In the first three lines the parameters of the measurement in February are given. Those data were accurate within the expected uncertainties. The situation was different in May. The difference in the total flow was 29% and therefore a mistake was assumed. By calculating the “conductivity load” and comparing those with the data from February it became obvious that there was a mistake at sampling location SL 4. After recalculating the data the error apparently was a typo in the flow. Instead of 54.4 L min^{-1} it should read 44.4 L min^{-1} . After correcting that typo the error became -7% and again was in the acceptable range.

Tab. 11: Example of a flow measurement in a closed underground mine showing how a wrong flow measurement can be identified and corrected. SL: sampling location; Diff: difference; electrical conductivity in $\mu\text{S cm}^{-1}$; flow in L min^{-1} , “load” in $\mu\text{S L cm}^{-1} \text{ min}^{-1}$.

Date	SL 1	SL 2	SL 3	SL 4	SL 5	Sum 1, 2, 3, 4	Diff	Diff %
February 22nd								
flow	3.50	0.83	0.25	16.50	21.80	21.08	0.72	3%
electrical conductivity	318	466	619	374	388			
“load”	1113	387	155	6,171	8,458	7,826	633	7%
May 30th								
flow	1.16	2.80	0.30	54.40	45.50	58.66	-13.16	-29%
electrical conductivity	859	163	521	167	204			
“load”	996	456	156	9,085	9,282	10,694	-1,411.94	-15%
relative “load” difference	11%	-18%	-1%	-47%	-10%			
May 30th								
flow, corrected	1.16	2.80	0.30	44.40	45.50	48.66	-3.16	-7%

This method can also be used to calculate missing flow data. In the abandoned Austrian Georgi Unterbau mine there is a huge inflow into the drainage gallery at the crossing of the Grünwies fault. The amount of water flowing into the mine can become extremely high after rainfall events and thus cannot be measured. Yet, it is possible to measure the upstream water volume in the drainage gullet and the conductivities in the upstream, downstream and inflowing mine water. The missing flows can therefore be calculated as follows:

$$\kappa_d \times Q_d = \kappa_i \times Q_i + \kappa_u \times Q_u \quad (36)$$

with κ : electrical conductivity, $\mu\text{S cm}^{-1}$
 Q : flow, L s^{-1}
 d, i, u : downstream of inflow, at inflow, upstream of inflow

There are two unknowns, Q_d and Q_i , but because

$$Q_d = Q_i + Q_u \quad (37)$$

equation 36 can be written as follows:

$$\kappa_d \times (Q_i + Q_u) = \kappa_i \times Q_i + \kappa_u \times Q_u \quad (38)$$

with only one unknown variable left. Now, after multiplying and rearranging the equation becomes

$$Q_i = Q_u \times \frac{\kappa_u - \kappa_d}{\kappa_d - \kappa_i} \quad (39)$$

and the inflow and the downstream mine water volume can easily be calculated (Tab. 12). Instead of the electrical conductivity, the temperature or the concentration of conservative water constituents such as Cl or Br could be used.

Tab. 12: Example of a flow calculation by using equation 39. Data modified after Unger (2002).

Parameter	Upstream	Inflow	Downstream
κ	377.7 $\mu\text{S cm}^{-1}$	328.3 $\mu\text{S cm}^{-1}$	352.4 $\mu\text{S cm}^{-1}$
Q measured	1.97 L s^{-1}	—	—
Q calculated	—	2.06 L s^{-1}	4.03 L s^{-1}

9.2.1.1 Bucket-and-stopwatch

One of the simplest method for flow measuring is presumably one of the most frequently used methods when conducting investigations at abandoned mine sites: bucket-and-stopwatch. To conduct such flow measurements the mine water must flow through a pipe or a channel and it must be possible to fill the container without an overflow which would appear under high flow conditions.

The flow that can be measured by this method varies roughly between a maximum of 500 L min^{-1} and a minimum of 0.1 L min^{-1} . To measure the flow a container and a stop-watch are needed, whereas the container volume can vary between 50 mL and 100 L and the time should be measured at the 100th of a second. The measurement is made by determining the time that is needed to fill the bucket or graduated cylinder. At least three measurements should be done, but as much as 10 could be necessary under changing flow conditions. Commonly, this method gives mean errors of 5—10%, but under certain flow conditions better results are obtained.

$$Q = \frac{V}{\Delta t} \quad (40)$$

with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 V : volume of container, m^3
 Δt : time to fill container, s

Tab. 13 gives an example of a bucket-and-stopwatch measurement which includes the uncertainties of the time measurement (0.1 seconds) and the measurement of the bucket, which had 1 L marks. It is assumed that the volume of the bucket can be determined within about 0.5 L during the filling process. By using those uncertainties, 3 flow volumes can be calculated: the mean flow calculated from the measured volume of the bucket and the measured filling time, the minimum flow from those numbers, and the maximum flow. Interestingly, the standard deviation of the mean measurements and the mean uncertainty from the minimum and maximum values are within the same range. This is not always the case, but in that example it proves that the researcher worked very accurately.

Such tables and error calculations are found extremely seldom in the geosciences and in mine water research. A more detailed description of simple error calculations can be found in section 9.3.4 and the following.

Tab. 13: Example of a bucket-and-stopwatch measurement of flows at a mine water discharge. Flows in the mining sector are often measured in L or m^3 per minute and therefore, the results are given in that unit. Uncertainties are calculated in the last 3 columns using the min. and max. flow based on the accuracies of the volume (0.5 L) and time measurements (0.1 L). Bucket volume: 10 L.

time, s	flow, L min^{-1}	min flow, L min^{-1}	max flow, L min^{-1}	uncertainty, L min^{-1}	uncertainty, L min^{-1}	uncertainty mean, L min^{-1}
9.76	61.5	57.8	65.2	3.7	3.7	3.7
10.25	58.5	55.1	62.1	3.5	3.5	3.5
10.20	58.8	55.3	62.4	3.5	3.6	3.5
8.98	66.8	62.8	70.9	4.0	4.1	4.1
10.17	59.0	55.5	62.6	3.5	3.6	3.5
mean 60.9					uncertainty 3.7	
standard deviation 3.5 5.7%					uncertainty 6.0%	

9.2.1.2 Area-Velocity Method

A simple method to estimate the flow is the area-velocity method. To measure the velocity of the stream, a floating object or a strong dye are put into the stream and the time needed by the floating object or the dye to travel a special distance is measured. As the velocity in a given stream profile is not constant but increases from the stream's bottom to the surface the floating object typically should dip into the water by $\frac{2}{3}$ rd of its diameter. The flow can then be calculated according to the following equation:

$$Q = A \times \frac{\Delta x}{\Delta t} \times f \quad (41)$$

with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 A : characteristic cross section of stream, m^2
 Δx : measure distance, m
 Δt : time for object to float the distance Δx , s
 f : factor for average velocity in area (0.66—0.9)

The dimensionless coefficient f can be calculated as follows:

$$f = \begin{cases} 0.3 \leq d \leq 1.8: & f = 0.64 + \frac{0.2 \times d}{3} \\ d \geq 1.8: & f = (1.4 - 0.06 \times \sqrt{d})^{-1} \end{cases} \quad (42)$$

with d : water depth in middle of stream, m

Another common method for determining the mean velocity are the Manning and Chezy equations which include the roughness coefficients n and C depending on the type of channel (Gray 1970; Herschy 1995). Because their tables are rather long and give details for different naturally occurring stream conditions which are not representative for mine water discharges, their method will not be further discussed here.

In some cases the flow might be measured in partly filled pipes, half-pipes or ditches. In such situations only the water depth d and the width of the water l can be measured (Fig. 51). As the equation for calculating the area A of the circular segment is not readily available in most equation collections, it will be given here:

$$A = \frac{1}{2} \times \left(\frac{l^2 + 4 \times d^2}{8 \times d} \right)^2 \times \left\{ 4 \times \arctan \left(\frac{2 \times d}{l} \right) - \sin \left[4 \times \arctan \left(\frac{2 \times d}{l} \right) \right] \right\} \quad (43)$$

$d \leq l$

which, luckily, can be simplified to

$$A \approx \frac{d}{6 \times l} \times (4 \times l^2 + 3 \times d^2) \quad (44)$$

with maximum errors of -0.1—9.4% for all cases where $d < l$ and where

- A : characteristic cross section of stream, m²
- l : width of stream or water surface in pipe, m
- d : water depth in middle of stream or pipe, m

In case a dye is utilised the dye must be quickly poured into the water to avoid dispersion (see section 10.5.1 for details of tracer injection). The mean length of the dye cloud should be used for measuring the time and the factor f can be set as 1. If the mean length of the dye cloud cannot be evaluated exactly enough, the first and last appearance of the tracer cloud could be measured instead and the mean of those two times be used as an estimate.

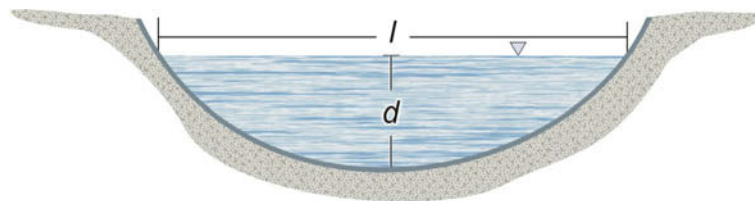


Fig. 51: Necessary parameters for calculating the cross section A of a partly filled pipe, half-pipe or ditch after equations 43 or 44.

9.2.1.3 Current (Flow) Meters

Probably the most common instrument for flow measurements in the field is a current meter (Fig. 52). Though all techniques are different from the physical characteristics involved they are described together because the results gained are evaluated in the same way. For the

reader wishing to go into the details of closed channel flow measurement, the excellent in-detailed book of Miller (1996) is recommended.



Fig. 52: Flow measurement with an impeller current meter in the discharge channel of a natural lake receiving polluted mine water. The left operator records the time and the right one holds the impeller current meter and the electronic device for counting the turns of the impeller.

Several types of current meters are available and can be used to measure the velocity of a water stream at a predefined point in the water stream:

- impeller current meter
- heat pulse
- electromagnetic velocity meters
- Doppler velocity meter

Their characteristics and use in the field differ significantly, but in most mine water related investigations either of the different current meters can be used when calibrated properly. If uncertain which technique is the most reliable in the actual investigation it is advisable to compare different instruments to make sure that the flow measurements are accurate. For a rough calculation and estimation of the methods accuracy, the area-velocity method should be used.

As a prerequisite for current meter measurements the flow within the cross-section of the stream must be laminar. For the calculation of the flow in a given cross section, the cross section is divided into several laminae and within each lamina the stream velocity is measured at several depths, which are depending on the specific method used. Both, the number of laminae and the number of velocity measurements per lamina are depending on the width and the depth of the stream to be measured (Fig. 53). Usually, where mean mine water discharges shall be evaluated, three laminae and three velocity measurements in three depths are a good choice. U.S. Department of the Interior – Bureau of Reclamation (2001) lists eight different methods to measure and calculate the flow in a stream:

- two-point method
- six-tenths-depth method
- vertical velocity-curve method

- subsurface method
- depth integration method
- two-tenths method
- three-point method
- one-point continuous method

For details about each of those methods this reference should be consulted. For most calculation methods the total flow is calculated from the sum of each flow in a single lamina:

$$q_i = \bar{v}_i \times A_i \quad (45)$$

$$Q = \sum_1^n q_i = \sum_1^n (\bar{v}_i \times A_i) \quad (46)$$

with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 q_i : flow in lamina i , $\text{m}^3 \text{s}^{-1}$
 $\bar{v}_i$: mean velocity in lamina i , m s^{-1}
 A_i : area of lamina i , m^2
 n : number of laminae

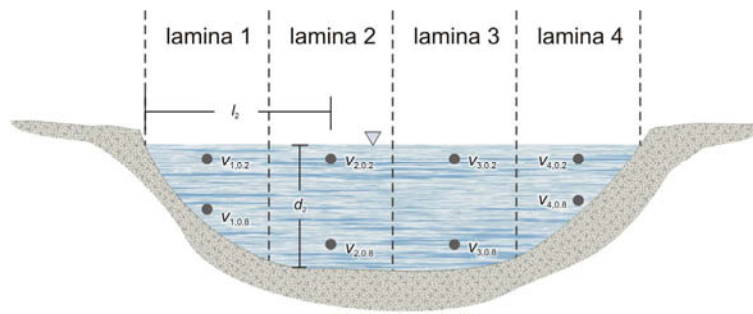


Fig. 53: Calculation of flows in a stream or channel according to the two-point method with four laminae.

As can be seen, the only problematic variable is the mean velocity $\bar{v}_i$, for which the above mentioned methods are used to calculate the best representative estimation. One of the simplest methods is the “simple average method” based on a two-point velocity method. In the middle of each lamina the velocity is calculated according to the two-point method as shown in Fig. 53:

$$\bar{v}_i = \frac{v_{0.2} + v_{0.8}}{2} \quad (47)$$

with $\bar{v}_i$: mean velocity in lamina i , m s^{-1}
 $v_{0.2}$: velocity at $0.2 \times d_i$ in lamina i , m s^{-1}
 $v_{0.8}$: velocity at $0.8 \times d_i$ in lamina i , m s^{-1}

Then, $q_{i,(i+1)}$ is calculated as follows:

$$q_{i,(i+1)} = \left(\frac{\bar{v}_i + \bar{v}_{i+1}}{2} \right) \times \left(\frac{d_i + d_{i+1}}{2} \right) \times (l_{i+1} - l_i) \quad (48)$$

with $q_{i,(i+1)}$: flow in lamina $i, i+1$, $\text{m}^3 \text{s}^{-1}$
 $\bar{v}_i$: mean velocity in lamina i , m s^{-1}

- d_i : depth in the middle of lamina i , m²
 l_i : distance from stream edge to middle of lamina i , m²

No general recommendations can be given for every case where mine water discharges might be measured by a current meter method, as the appropriate selection depends on the stream depth, velocity, and its width. By comparing two or more methods with each other, the best method must be found. Yet, if more than two velocities are measured in a laminae, the different velocities have to be weighted. Other than in equation 47 of the “simple average method” the velocities cannot be calculated by using an arithmetic mean. In case of non-laminar flow, which is possible behind pipes, bends or obstacles, the current meter method should be avoided and the salt-dilution method should be used instead (section 9.2.2).

9.2.1.4 Weirs

By far the most common method used for making repeated flow measurements in mine water environments are thin plate weirs. They can be installed relatively easily under many different flow conditions, their maintenance is comparatively easy and the flow can be calculated in a simple way by measuring the head above the crest of the weir. Some of the weir types are portable weir boxes and can be used in channels or gullies, others are portable weir plates that can be put into either pre-installed weir holders or hammered into loose soil. They can be easily cleaned from ochre precipitates and any malfunction can usually be quickly identified. By using head-discharge tables of one of the equations available, the flow over a standard weir can be calculated in an easy effort. Though correction methods for submerged flow (also called non-modular or drowned flow) over sharp crested weirs exist the water must usually flow freely over the notch as such corrections methods can only be used after careful calibration under different flow conditions. A good summary of the prerequisites for installing weirs is given by U.S. Department of the Interior – Bureau of Reclamation (2001). Yet, the flow measuring devices most commonly installed or used in a wrong way are also weirs.

Weirs or flumes can only be used when the relation between the velocity of the discharged water and the depth of the channel or gully does not exceed a critical value. This dimensionless value is called the Froude number Fr and is follows:

$$Fr = \frac{\bar{v}}{\sqrt{g \times d}} \quad (49)$$

- with $\bar{v}$: mean velocity in channel upstream of weir, m s⁻¹
 g : gravitational acceleration, m s⁻²
 d : mean depth of channel upstream of weir, m

If Fr is bigger than 0.5 flow measuring structures lose their accuracy and good flow measurements with weirs or flumes should only be done in cases where $Fr < 0.5$. Thus, in a channel with a depth of 0.2 m for example, the stream velocity must not exceed 0.7 m s⁻¹ for getting reliable results.

In free flow condition, which is the normal working condition of weirs, there must be air between the overflowing water and the weir plate. If there is no air, the pressure condition for which the weir was calibrated is not any longer valid and the flow measurements therefore are inaccurate. Another inaccuracy will appear if the head is measured at the wrong position upstream from the weir. For most weirs the head must be measured at 4—5 times the expected maximum overflow upstream of the weir, as the water starts its drawdown at approximately 2 times the head away from the weir. Also, the water level above the crest of the weir must not be too low as will be exemplified in appendix II.

The weir basin behind the weir must regularly be cleaned from accumulated sediments as they also alter the flow conditions. Furthermore, the flow to the weirs should be laminar; therefore their installation should be after a flow section which is long enough to calm turbulent flow situations. The channel sides should be at a distance of at least twice the expected head over the weir to minimise the influence of the channel borders and the downstream water level should always be at least 6 cm below the notch.

U.S. Department of the Interior – Bureau of Reclamation (2001) lists 12 prerequisites which are necessary if a weir is to be installed correctly:

- The upstream face of the weir plates and bulkhead must be plumb, smooth, and perpendicular to channel flow.
- The weir crest needs to be level for rectangular and trapezoidal shapes. The bisector of V-notch angles must be plumb.
- The edges of the weir opening must be located in one plane. The corners of the weir opening must have proper specified angles.
- The top thickness of the crest and side plates should measure between 1 and 2 mm.
- All weir plates need to have the same thickness over the entire overflow crest boundary. Plates thicker than 2 mm should be bevelled to an angle of at least 45° on the downstream edge of the crest and sides. The downstream edge of V-notches should have a 60° angle to prevent water from clinging to the downstream face of the weir.
- Upstream edges must be straight and sharp. Burrs and scratches should be removed by machining or perpendicular filing; abrasive cloth or paper should not be used.
- The bottom edge plates and upstream fastener projection should be located at least the distance of two measuring heads from the crest. All upstream faces must be free of oil and grease.
- The overflow sheet or nappe touches only the upstream faces of the crest and side plates.
- The weir head measurement is the difference in elevation between the crest and the water surface at a point located upstream. The upstream point is at a distance of four to five times the maximum expected head on the weir.
- The maximum downstream water surface level should be at least 2–6 cm below the crest elevation.
- The weir head measurement, depending on the weir type, should be at least 2–6 cm to prevent the overflow sheet from clinging to the downstream weir face.
- The weir approach should be kept clear of sediment deposits and other debris.

In general, 3 types of weirs are used: V-notch (Thomson) weirs, rectangular notch (Poncelet) weirs and trapezoid (Cipolletti, sometimes also Cipoletti) weirs (Fig. 54). All of them have to be sharp crested where the sharp edge has to be at the upstream side. Several researchers investigated different experimental set-ups of full width and contracted weirs, resulting in a set of equations (Ackers et al. 1978), of which Kolupaila (1961) already listed some 100 references.

Kindsvater and Carter (1959) analysed which parameters influence the discharge over a thin-plated weir and came to the conclusion that at least eight variable parameters are necessary to describe the flow over such a measuring structure:

$$Q = \phi(L, B, P, h, \gamma, \rho, \mu, \sigma) \quad (50)$$

with Q : flow rate

- L : width of the notch
 B : width of approach channel
 P : crest height of weir above mean bed level
 h : gauged head related to weir crest
 γ : specific weight of fluid
 ρ : density of fluid
 μ : dynamic viscosity of fluid
 σ : surface tension of fluid

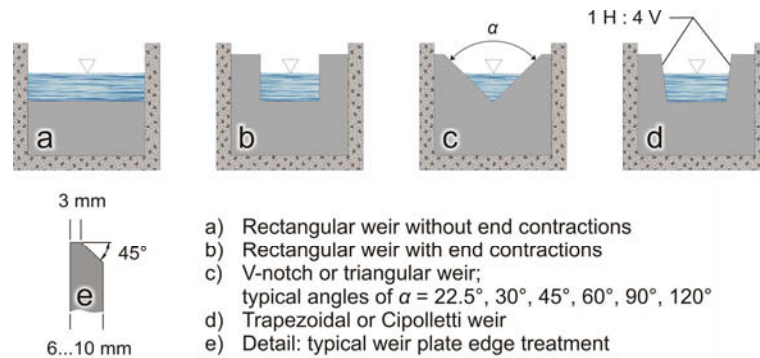


Fig. 54: The three different weir types commonly used in mine water research: rectangular notch weir, V-notch weir, and trapezoidal (Cipolletti) weir (modified after Minister of Public Works and Government Services 2001).

They summarised several of the above parameters to the coefficient of discharge C as follows

$$C = \phi\left(\frac{L}{B}, \frac{L}{h}, \frac{h}{P}, \text{Re}, \text{We}\right) \quad (51)$$

- with Re : Reynolds number (measures the relative influence of viscosity)
 We : Weber number (measures the relative influence of surface tension)

Because the influence of Re and We for “large” heads on “large” weirs can be seen as negligible, the coefficient of discharges thus can be simplified to

$$C = \phi\left(\frac{L}{B}, \frac{h}{P}, h, L\right) \quad (52)$$

and the general equation for calculating the flow over all sharp crested weirs according to some 250 experimental set-ups by Kindsvater and Carter (1959) becomes:

$$Q = C_e \times L_e \times h_e^{\frac{3}{2}} \quad (53)$$

- with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 C_e : effective coefficient of discharge, $\text{m}^{\frac{1}{2}} \text{s}^{-1}$
 L_e : effective weir length = $L + k_b$, m
 h_e : effective head above weir crest = $h_1 + k_h$, m
 L : measured length of weir crest, m
 h_1 : head measured above the weir crest, m
 k_b : correction factor to obtain effective weir length, m
 k_h : correction factor, $9.144 \cdot 10^{-4}$ m

Though this general equation looks rather simple, different solutions are given in the literature and the only way to get a precise head-discharge dependency is to calibrate each notch before it is used. Another option is to install manufactured, pre-calibrated weirs. In most circumstances, a rough estimation of the discharge would sufficiently meet the requirements for mine water investigations. A comparison of the equations given in the literature showed that for high flows the errors are usually very big, because the equations do not consider the coefficient's dependency on the head above the weir crest. Because mine water discharges can reach several cubic meters per minute this inaccurate coefficient might result in an underestimation of up to 13% compared to the real flow. Therefore, the simplifications given in several publications for measuring the flow discharged by a mine or a mine water treatment scheme should not be used (e.g. U.S. Department of the Interior – Bureau of Reclamation 2001; Younger et al. 2002).

Depending on the weir type, equation 53 can be solved in different ways, of which some are presented in appendix II. For most purposes, a simplification of the equations fulfils the needs of mine water researchers, but all of the simplifications have an error of ± 1 —5%. This error mainly is a function of the head above the crest of the weir and normally increases to both ends of the 'boundary' conditions for each weir type. It also increases if weirs are installed that do not meet the restrictions for its use or if weirs are installed in non linear flow conditions. If high precision measurements are needed, the equations in appendix II should be used and – if necessary – be fitted to the site specific conditions as described in U.S. Department of the Interior – Bureau of Reclamation (2001). All of the following SI unit based equations were computed in that way that the original equations were used, discharges for different flow conditions were computed and based on those results the simplified equations were fitted to the results by using a state of the art curve fitting model (TableCurve 2D and 3D). Before using those equations it is recommended that another flow measurement technique is applied to quantify the measurement uncertainties and to evaluate the real discharge coefficients.

The following nomenclature is used for equations 54 to 59 (see also Fig. 126):

- h : gauged head related to weir crest, m
- b : crest breath, m
- P : crest height of weir above mean bed level, m
- Q : flow over weir, $\text{m}^3 \text{s}^{-1}$

The unit for all of the coefficients is $\text{m}^{-0.5} \text{s}^{-1}$.

V-notch (Thomson) weirs (90° , $\frac{1}{2} 90^\circ$, $\frac{1}{4} 90^\circ$; $h > 0.1$ m; Fig. 49, Fig. 55)

$$Q_{90} = 1.369 \times h^{2.5}; \alpha = 90^\circ \quad (54)$$

$$Q_{\frac{1}{2}90} = 0.684 \times h^{2.5}; \alpha = 53^\circ 8' \quad (55)$$

$$Q_{\frac{1}{4}90} = 0.348 \times h^{2.5}; \alpha = 28^\circ 4' \quad (56)$$

accuracy 90° : -4 — 0%

accuracy $\frac{1}{2} 90^\circ$: -6 — 0%

accuracy $\frac{1}{4} 90^\circ$: -10 — 0%

Compared to a 90° V-notch weir, the $\frac{1}{2} 90^\circ$ and $\frac{1}{4} 90^\circ$ V-notch weirs have a flow which is half and a quarter of the flow over a 90° V-notch weir at the same gauged head, respectively.

Rectangular notch (Poncelet) weirs (full width)

$$Q = 1.773 \times \left(1 + 0.148 \times \frac{h}{P} \right) \times b \times h^{1.5} \quad (57)$$

accuracy: 0—3%.

Rectangular notch (Poncelet) weirs (contracted)

$$Q = 1.758 \times b \times h^{1.5} \quad (58)$$

accuracy: -4—1%

Trapezoid (Cipolletti) weirs

$$Q = 1.869 \times b \times h^{1.5} \quad (59)$$

accuracy: -1—1%



Fig. 55: A V-notch weir in the Felsendome Rabenstein/Germany underground visitor mine showing that the sharp crested side has to be at the upstream side of the weir.

9.2.1.5 Flumes

Flumes change the dimensions in an open channel to accelerate the water flow by converging the channel sides or by raising the channel bottom. If standard flume dimensions are used the flow can be calculated from the water head upstream the flume at a predefined distance from the critical depth. Flows between 0.3 L min^{-1} and $5,600 \text{ m}^3 \text{ min}^{-1}$ can be measured with flumes.

Flumes can be categorised as short- or long-throated flumes, whereas short and long refers to the linear parts in the calibration curve not to their constructive length (Fig. 57). In contrast to short-throated flumes long-throated flumes can simply be calculated with computer programmes and it is easy to arrange their dimensions to nearly every flow and constructive situation. Short-throated flumes, according to the suggestions in U.S. Department of the Interior – Bureau of Reclamation (2001) should not any longer be constructed due to their complicated calibration and their constructive restrictions. Detailed descriptions for different flumes are available in Ackers et al. (1978). Up to date, flumes in working or abandoned mines is rarely used because they need more careful installation than weirs. If installed appropriately, their use is as easy as that of weirs.

H- and Parshall-flumes are short-throated measurement structures and each needs its own calibration table. Due to hydraulic restrictions the dimensions of a given flume cannot be up-

or downscaled by multiplying its properties to fit the new flume size. Both flume types have the advantage of coping with high sediment loads and due to their construction and the velocity increase involved they have a certain “self cleaning” capacity. Their disadvantage is that the flow measurement is not as accurate as with long-throated flumes and that they need special construction considerations which are not always possible in the restricted space of underground mines or surface channels.

H-flumes

H-flumes (Fig. 56) were invented by the U.S. Soil Conservation Service for continuous and accurate flow measurements in small catchment areas with the requirement to measure small flows as well as higher flows (United States Department of Agriculture 1979; Hudson 1993). They are named “H” after the eighth experimental setup for testing new flume types. Three different types for three different flow conditions ranging from about 0.3 L min^{-1} to $199 \text{ m}^3 \text{ min}^{-1}$ were investigated and a total of seventeen construction dimensions ranging from 0.4 to 4 feet are available (the numbers here must be given in feet, because each of the fourteen calibration tables is especially made for one type of H-flume; the equation given in table 3.2 of Younger et al. (2002) cannot be used for all types of H-flumes; see Fig. 57). They need free fall out at the end and must not be submerged, though a calibration for submerged H-flumes is available as well. Tab. 14 shows the wide range of flow conditions in which H-flumes can be used. As in the case of V-notch weirs self registering and monitoring devices can easily be installed. Ackers et al. (1978) give detailed equations and coefficients for calculating the flow through H-flumes. Those equations and the according coefficients depend on the upstream heads and submerged or free flow conditions. If manufactured H-flumes cannot be installed, the references in chapter 2 of United States Department of Agriculture (1979) must be followed, as erroneously constructed flumes will result in wrong discharge measurements.



Fig. 56: 2-ft-H-flume near Mettmann/Germany (courtesy Umwelt-Geräte-Technik GmbH Müncheberg).

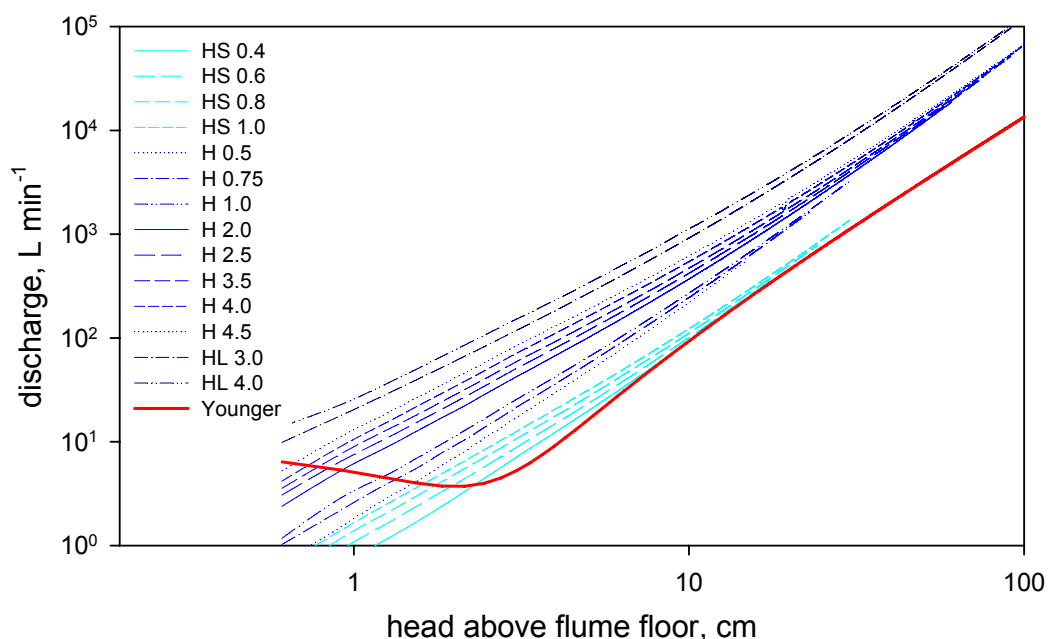


Fig. 57: Graphical comparison of the 14 different HS-, H-, and HL-flume types used for flow measurements. The simplified H-flume equation of Younger et al. (2002) is given for comparison. It can only be used with heads over about 6—7 cm and for HS-type flumes.

Tab. 14: Selection of the commonly used H-flume types for flow measurements (after United States Department of Agriculture 1979). Each H-flume needs its own calibration table. GPM: U.S. gallons per minute. GPM and metric units are rounded. For comparison the minimum height of a V-notch weir at the same maximum flow is given. As V-notch weirs should not be constructed with heads above 30 cm, those data are given for comparing reason only. 3.0-HL-flumes, which should be substituted by H-flumes, are not recommended by United States Department of Agriculture (1979).

H-flume type	Width, feet	Flow, GPM	Flow, L min ⁻¹	V-notch, cm
HS	0.4	0.07 – 36	0.3 – 140	8
HS	0.6	0.10 – 99	0.4 – 370	11
HS	0.8	0.13 – 205	0.5 – 780	15
HS	1.0	0.17 – 360	0.6 – 1,400	20
H	0.5	0.18 – 149	0.7 – 560	13
H	0.75	0.27 – 430	1.0 – 1,600	21
H	1.0	0.31 – 862	1.2 – 3,300	28
H	1.5	0.49 – 2,392	1.9 – 9,100	42
H	2.0	0.62 – 4,928	2.3 – 18,700	56
H	2.5	0.81 – 8,617	3.1 – 32,600	69
H	3.0	0.94 – 13,644	3.6 – 51,600	84
H	4.5	1.39 – 37,699	5.3 – 142,700	126
HL	3.0	2.60 – 25,637	9.8 – 97,000	108
HL	4.0	2.24 – 52,510	8.5 – 199,000	144

An H-flume was constructed at the Shilbottle/UK passive treatment scheme where large amounts of iron-precipitates were expected. It was found that the flume worked reliably during its operation time and a certain self cleaning capability could be observed. On occasions when it became clogged (usually with plant litter in the autumn), it could be manually cleaned in seconds by pushing the clogging matter out through the open “jaws” of the flume (Paul Younger, pers. comm.). This is a big advantage over weirs for which laborious digging out behind the plate is needed.

Parshall flumes

As for H-Flumes, Parshall flumes (named after Ralph L. Parshall who developed them in 1922) are short-throated measurement structures which are regulated by ISO 9826-1992 (E). Being a modification of a Venturi flume their characteristic is that the flow is contracted into the inlet and has vertical side walls (Fig. 58). They are described here, because in some mine water applications they were installed and can still be found. Yet, as mentioned earlier, they should be avoided due to their constructive restrictions. 22 variations of Parshall flumes with throat widths between 2.5 cm and 15.2 m were calibrated in the laboratory and can be used in mine water applications for flows between 5 L min^{-1} and $5,600 \text{ m}^3 \text{ min}^{-1}$ (Ackers et al. 1978; Herschy 1995). It is not possible to use other dimensions unless the new dimensions are calibrated again. The head difference between the upstream and downstream part is usually low and as for an H-flume they are not as much affected by precipitation or high sediment loads. Therefore, they could also be used in mine water applications where the gulleys have a moderate gradient and due to their construction they can be submerged up to 70% without the need of corrections. This fact is an advantage over Venturi flumes (Science Applications International Corporation 2001).

The head measurement has to be conducted exactly at the position at which the flume originally was calibrated. A shift in that position causes erroneous readings and consequently wrong flow measurements.



Fig. 58: Parshall flume in the flooded South Crofty/Cornwall tin mine.

9.2.1.6 Pump Run-Times and Ratings

Usually, in underground mines which have to be dewatered, the pump run times are recorded relatively accurately. Those data together with the pump characteristics can be used to estimate the expected flow rates (Fernández-Rubio and Fernandez Lorca 1993). Often, it might be acceptable to use the manufacturer's information about the pumping capacities, yet wear and tear or cavitation might alter the factory parameters and an independent flow measurement should be conducted before that data are used. Furthermore, fluctuations in the power supply slightly change the pump capacity and must be considered when using pump run-times for estimating the mine water flow. Another problem might be clogging in the pipe network due to ochre precipitates, which results in wrong flow estimates. Finally, it should be noted that flow estimates that are based on pump run times are often over-estimated by around 20%, in some rare cases up to 50% (Paul Younger, pers. comm.).

9.2.1.7 Water Meters

Flow measuring with water meters is a very reliable method for quantifying mine water discharges. To use such instruments the mine water must flow through a closed pipe which usually must be filled 100%. The flow through the pipe is either measured mechanically with a propeller (Fig. 59) or an electro-magnetic flow meter (Fig. 60). Propeller water meters are available for pipe diameters ranging between 1 and 183 cm (0.5 — 72 inch) and a flow of up to $10 \text{ m}^3 \text{ min}^{-1}$ with an accuracy of 1—10%. They can cause problems if water with high sediment loads has to be measured or when abrasion of the mechanical parts occurs. Magnetic-inductive flow meters are available from 0.1 to 800 cm diameter and have a typical accuracy of 0.2—3% (mainly depending on the pipe's diameter) and if maintained properly (coating of sensors) they are one of the best flow measurement devices in mine water research.

Other methods are laser-Doppler-anemometers, acoustic flow meters, or pitot tubes. Laser-Doppler-anemometers can precisely measure the velocity profile in a pipe or a channel, but for regular flow measurement in the mine water field they are currently too expensive. They require regular maintenance and service, and gases as well as sediments could influence the acoustic and laser signals (Durgin 1981; Zosel 1997).



Fig. 59: Mechanical, propeller based water meter at the abandoned *Dienststelle Marienthal* bunker near Marienthal/Germany.



Fig. 60: Magnetic-inductive flow meter in the *Schwarze Pumpe*/Germany mine water treatment plant.

9.2.1.8 Dilution Method

By its character dilution techniques are tracer techniques used for flow measurement. Due to their high accuracy – if performed by experienced personnel – they can be used to calibrate other flow measurement techniques. Furthermore, they are the methods of choice for turbulent flow conditions or in streams or gullies with no defined cross sections, as there is no need to measure channel geometry or floating times (Herschy 1995; Käß 1998; U.S. Department of the Interior – Bureau of Reclamation 2001; Science Applications International Corporation 2001). Yet, backflow eddies or seeping sections should be avoided as they negatively influence the results. All types of conservative, water soluble tracers, which are described in section 10.2.5, are suitable for the salt-dilution method but for reducing the costs of tracer analyses sodium chloride or sodium fluorescein are recommended because both tracers can be detected cheaply with on site measurement techniques: electrical conductivity and mobile fluorometers. Spence and McPhie (1997) successfully used that method for mine water impacted streams in the Canadian Northern Territories, where many streams are shallow, braided, and rocky. They analysed the tracer in the laboratory and used potassium chloride instead of sodium chloride, because the natural potassium concentrations are usually small compared to sodium. Having compared a variety of stream flow measurement methods they concluded that the “dilution method compares favourably to other measurement techniques both in accuracy and operational ease”.

Two different methods can be used: the “constant rate injection method” and the “sudden injection (‘gulp’) method”. In the first case, the tracer is injected at a constant rate with a predefined concentration and in the second case a predefined mass of tracer is suddenly poured into the water. Both methods have their special field of application and Kimball et al. (2001) (and other publications of that research team covering exactly the same subject) for example showed how the “constant rate injection method” can be used to measure the inflow of mine drainage into a mountainous stream. At the Gernrode/Germany fluorspar mine, for example, the “gulp method” was used to measure small inflows into the Steinbach creek and to compare and calibrate the results of current flow meter measurements (Fig. 48, Fig. 61).

Because the constant rate injection method needs a lot of technical installations it is mostly used for scientific based catchment investigations where different inflows have to be quantified. Either Mariott bottles or constant rate pumps can be used to inject the tracer (Käß 1998). On the other hand, the flow calculation is comparably simple, because it is not necessary to measure the whole breakthrough curve but only the plateau which develops after enough time

and the distance between tracer injection and detection. In that case the discharge can be calculated in that way (U.S. Department of the Interior – Bureau of Reclamation 2001):

$$Q = q \times \frac{c_1 - c_2}{c_2 - c_0} \quad (60)$$

with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 q : discharge of tracer into the stream, $\text{m}^3 \text{s}^{-1}$
 c_0 : background concentration of tracer, g L^{-1}
 c_1 : tracer concentration of injected tracer, g L^{-1}
 c_2 : tracer concentration in stream after full mixing, g L^{-1}

If only the flow of a mine water discharge or a technical flow measurement device shall be measured, the gulp method with tracer integration can be used instead. A given amount of tracer is quickly poured into the stream, channel, or gullet. In a distance far enough away that full mixing can be guaranteed, the tracer cloud is measured from the time of the tracer injection until all the tracer passed the measuring cross section. Thus, a breakthrough curve is obtained of which the concentration-time-integral must be evaluated. By dividing the amount of tracer injected by the integral, the discharge Q can be calculated:

$$Q = \frac{M}{\int_{t_0}^{\infty} c_t \times dt} \quad (61)$$

with Q : flow rate, $\text{m}^3 \text{s}^{-1}$
 M : mass of injected tracer, kg
 c_t : concentration of tracer at time t , kg m^{-3}
 t : time, s



Fig. 61: Flow measurement with the salt dilution method in the turbulent flowing German Gernrode/Harz Steinbach creek. The salinity probe is hanging in the middle of the stream and the data logger records the electrical conductivity every second (photograph Andrea Berger).

Before the test is evaluated, the electrical conductivity probe must be calibrated with the salt used in the tracer test and the stream water which shall be traced. This procedure is necessary to calculate the tracer concentration depending on the electrical conductivity of the water. Each rank of salt reacts differently and even the same rank of salt might get other results in

the same stream at different times. Thus, drifts of the electrical conductivity probe and chemical reactions between the salt and the stream water can be compensated. As for most tracer tests there is no straightforward equation to calculate the amount of tracer needed for the dilution method. For most situations in the mine water sector, 0.5—2 kg of rock salt, depending on the electrical conductivity and the flow of the water, should be a good estimation.

In a typical flow situation the breakthrough curve will have a distribution as shown in Fig. 62. Should the background concentration of the tracer used change significantly during the investigation, the graph will deviate from the ideal curve and the test should be repeated.

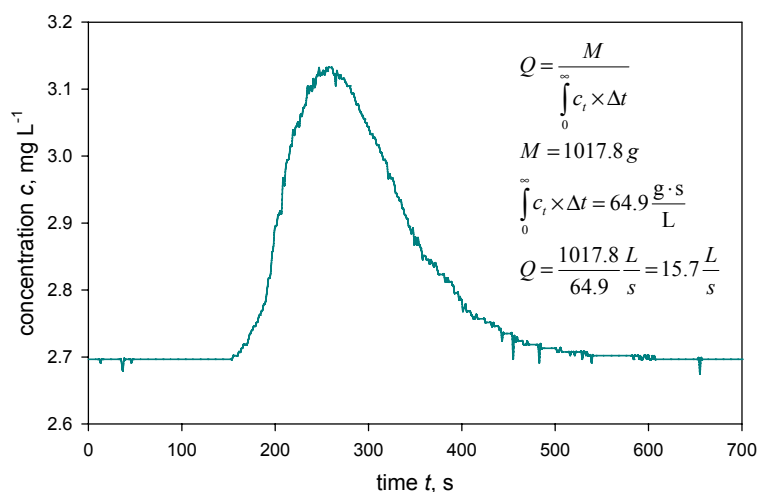


Fig. 62: Breakthrough curve and evaluation for a salt-dilution tracer test in the German Gernrode/Harz Steinbach creek. The distance between the tracer injection and the sampling site was 53 m and the complete mixing was tested in advance by using a dye tracer.

9.2.2 Mine Water Tracing

Monitoring of flows within a flooded mine by tracer tests has not been a common method hitherto, because until recently mine water tracer tests were complicated and the results sometimes unsatisfactory. In the last years, mine water tracer tests proved to be a decision-support tool for treatment or remediation measures or diversions within a mine (for details see chapter 10). Although tracer tests are site specific tests they are used for catchment-based decisions in case measures are necessary to decrease the contaminant load in the whole catchment area.

Before a mine water tracer test is conducted, it is essential to monitor the hydrogeological regime of the mine. The overall flow situation must be known as precisely as possible, because tracer tests will only be successful if the injection and sampling sites have been chosen carefully. Nothing is more frustrating than conducting an extensive and expensive tracer test without getting usable results!

Of the range of artificial tracers used so far, microspheres (polystyrene beads of 0.6—100 μm diameter) seem to be one of the best options (section 10.2.5.2). Most of the other artificial tracers are either not stable in acid mine water (e.g. Na-fluorescein), they are too expensive (e.g. bromides) or their use is restricted (radioactive tracers). Besides artificial tracers, natural (environmental) tracers, such as isotopes or rare earth elements have been used successfully to trace the surface and underground flow regimes of mines. Always, the background level of the tracers in the waters to be traced must be evaluated before the start of the tracer test.

Before the tracer test starts at least 2—4 months of work are necessary to investigate the hydrogeological situation, to get the necessary permits and to conduct laboratory experiments with the potential tracers. Successful tracer tests in mines need 2—8 weeks of time to make sure that as much of the tracer as possible is detected. After the tracer test 1—2 months must be allowed to process the samples and to present the results. All together 4—8 months are a realistic time for a well based tracer investigation in a mine.

Two different tracer injection apparatuses for tracer investigations in flooded underground mines were developed. The first one, LydiA (*Lycopodium* Apparatus), was invented for injecting the tracer into large diameter shafts. The other one, TinA (Tracer Injection Apparatus), can be used to inject the tracer through mine plugs or boreholes of down to a diameter of 3" (Fig. 63, Fig. 78). In conjunction with a special auto sampler (MeFiSTo: Multiple Filter Storage Tool), particle tracer tests can be conducted more reliably and more cheaply than in previous years (Wolkersdorfer et al. 1997a, 1997b).



Fig. 63: Tracer Injection Apparatus (TinA, left) and *Lycopodium* Apparatus (LydiA, right). The Diameter of LydiA is 120 mm.

Studies with stable isotopes should be given special consideration as natural or artificial tracers. Such studies provide potentially powerful methods to deduce flow paths and residence times, thus helping to understand hydrochemical and hydrodynamic investigations. Due to the complexity of stable isotope studies they will not further discussed here. Anyhow, tracer tests should only be considered trustworthy if they were conducted by experienced consultants or research groups. Some examples of successful mine water tracer tests are described in chapter 12.

9.3 Physico-Chemical Measurements

9.3.1 Sampling Procedures

Sampling procedures for ground waters are described in detail in the literature (e.g. Lloyd and Heathcote 1985, American Public Health Association et al. 1998), but the special prerequisites for mine water sampling are only rarely summarised in commonly available publications (e.g. Ficklin et al. 1999). We are still far away from regular process analyses (Kellner et al. 1998) and mine water diode or sensor arrays as well as mine water chips, as they are available for wastewater treatment (Bourgeois et al. 2003), are not even under development. Therefore, this chapter will highlight some of the necessities but also describe standard sampling procedures that are helpful for conducting good quality mine water samples. Readers who are interested in the details of analytical chemistry or the instruments used are referred to the general literature (e.g. Schwedt 1995; Kellner et al. 1998; Otto 2000) or the literature cited in the following paragraphs and sections.

Mine water discharges usually show changes in chemical composition with time and the impacts on the catchment can therefore differ (Fig. 64). Only rarely does the mine water quality remain constant except when it has been discharging for many decades.

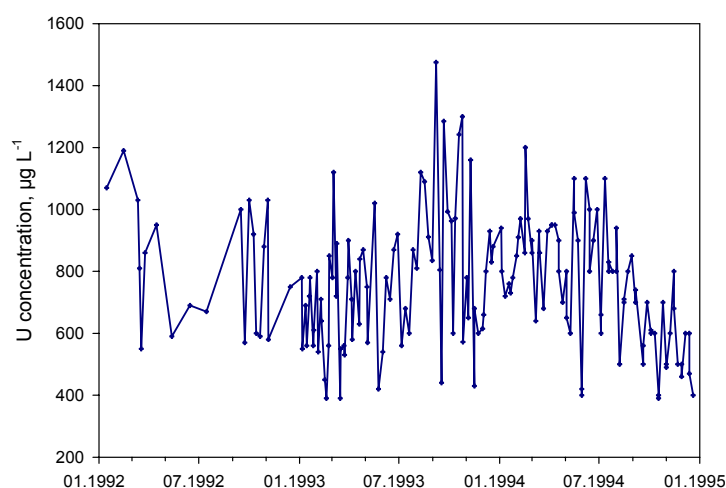


Fig. 64: Changing uranium concentrations at the measuring point m-107 of the abandoned Niederschlema/Alberoda uranium mine (data from Wolkersdorfer 1996).

As discussed in section 5.4 the first flush occurs when water from a flooded abandoned mine reaches the surface and the concentrations of water constituents decrease thereafter. However, changes of mine water quality are not restricted to the time after the flooding of a mine; they also exist during the active working time or after the end of the first flush. In the case of the flooded Freiberg/Saxony underground mines, which are dewatered by the 50 km long drainage adit *Rothschönberger Stollen*, Fe_{tot} was about 2 mg L^{-1} during the active mining period, rose to 170 mg L^{-1} in 1969 (start of mine flooding), came down to about 20 mg L^{-1} in 1975 and has presently reached 1 mg L^{-1} . The flow, in 1997, varied between 2 and $6 \text{ m}^3 \text{ min}^{-1}$ (Merkel et al. 1997; Baacke 2000). Therefore, both, the mine water quality and flows for at least one hydrogeological year must be measured.

In the first set of samples for an unclassified mine water discharge, as many elements as possible should be analysed to identify potentially hazardous constituents. Because mine water usually does not contain large amounts of organic pollutants, it is adequate to measure only the inorganic parameters. Nevertheless, some mines discharge organic substances and pollute

the receiving stream. If this is assumed for an unclassified mine water discharge, the sum parameter TOC (total organic carbon) as well as other organic substances such as BTEX should be analysed. Depending on the results of the analyses, further investigations may have to be considered.

From the list of inorganic parameters, as a minimum the major ions (Na^+ , K^+ , Li^+ , Ca^{2+} , Mg^{2+} , NH_4^+ , Cl^- , SO_4^{2-} , HCO_3^- , NO_3^-) as well as ferric and ferrous iron, manganese, zinc, and aluminium should be measured. Further elements should be chosen after geological and mineralogical studies of the mine site as well as the former use of the land property or mine (keep in mind potential underground waste disposal). As most modern laboratories provide ICP-MS devices, it may be advantageous to have a screening of the full range of elements that can be measured by this method. The outcome of such desk and laboratory studies will be the basis for the future monitoring programme.

Anions like Cl^- , SO_4^{2-} , and NO_3^- are typically determined by ion chromatography (IC). With this method also cations can be analysed, but it is recommended to determine the cations by ICP-AES or Ca^{2+} and Mg^{2+} by conventional titration and Na^+ and K^+ with flame photometry. Although the major ions (except Na^+ and Cl^- in salt mining) usually not considered as pollutants, those parameters are essential for thermo-dynamic modelling.

For every monitoring point, at least two water samples should be taken: one for the main ions and the other one for the trace elements (often referred to as “heavy metals” – but this term is even more confusing than “trace elements”, because many “heavy metals” are neither heavy nor are they metals at all!, Duffus 2002). All water samples should be filtered through a $0.2\ \mu\text{m}$ filter, though most investigations and recommendations use the $0.45\ \mu\text{m}$ filter. The reason for filtering the sample with $0.2\ \mu\text{m}$ is twofold:

- Water contains particles, colloids, and solute constituents (Ranville and Schmiermund 1999). Usually colloids are in the range from 0.001 to $1\ \mu\text{m}$ and thus solute constituents have a diameter of less than $0.001\ \mu\text{m}$ (Fig. 65). To separate solute constituents from non-solutes, a filtration with $0.001\ \mu\text{m}$ pore size is required. Since this is very difficult in the field, $0.2\ \mu\text{m}$ pore size is an operational measure (replacing the former $0.45\ \mu\text{m}$ operational measure). However, if aluminium is to be determined, a special treatment by using a $0.1\ \mu\text{m}$ filter is strongly recommended, since aluminium tends to form colloids that are smaller than $0.2\ \mu\text{m}$ in size.

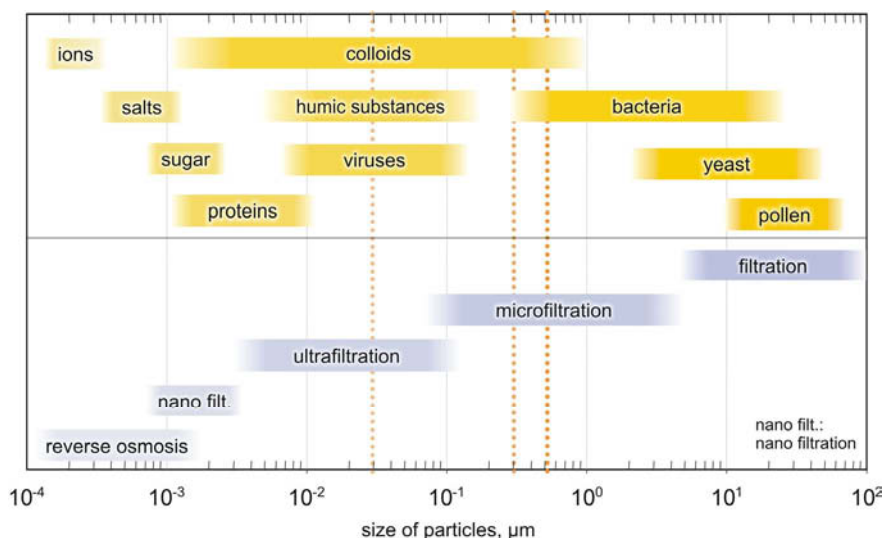


Fig. 65: Sizes of common water constituents and filtration techniques. The dotted lines gives the 0.02 , 0.2 , and $0.45\ \mu\text{m}$ filter sizes used for filtrating water samples.

- Water contains bacteria with an average size from 0.2 μm to 6 μm and by using 0.2 μm pore size filters, microbiological reactions are minimised in the filtered samples. On contrary, it is not recommended to filter all samples especially those for the major cations and anions, since filtration with 0.2 μm will consequently change the gas concentration particularly that of carbon dioxide and thus the distribution of inorganic carbon species.

Ranville and Schmiermund (1999) showed that the 0.45 μm filtration has consequences when conducting geochemical modelling with PHREEQE, as some phases are supersaturated in the < 0.45 μm filtrate and under saturated in the < 500 Dalton (approximately 0.005 μm) fraction (the latter can be seen as the true “dissolved” fraction of mine water). They conclude that geochemical modelling should be restricted to those constituents that are not significantly attenuated by the filtration process. Shiller (2003) compared results of 0,02 μm filtrations with those of 0,4 μm and 0,45 μm filtrations. He used a field applicable filter tool for syringe filtration and also noted some significant differences between the filter sizes he used. The differences were based on the different filtration techniques on the one hand and on the analysed element. Interestingly, the discrepancies between the 0,4 and 0,45 μm filtered results were sometimes larger than those between the 0,4 μm and 0,02 μm filter results.

After filtration, the sample for the trace elements must be acidified according to laboratory instructions or the QA/QC manual. Usually, 2—3 drops of HNO_3 (ultra-pure!) per 50 mL of sample are used to stabilise the trace constituents by maintaining a pH of < 2. For speciation analyses, or for analyses of Sb, Ru, and Sn HCl (ultra-pure) has to be used instead (American Public Health Association et al. 1998). These procedures are necessary to prevent chemical reactions during transportation and storage owing to precipitation, adsorption or contamination (Lloyd and Heathcote 1985). All containers (made of laboratory glass, PE, or PTFE) must be clean and pre-treated, labelled properly with the place name, name of sampling point, date, name of operator, and treatment (e.g. filtered, acidified).

Mine water is usually monitored at its discharge point into the surface hydro- or anthroposphere. However, it must be remembered that mine water quality does not only change in time, but also in depth. Many mines are known, where the water was stratified, and treatment technologies were constructed wrongly, because the stratification detail was disregarded. This was proved by the Straßberg/Germany mine, where mine water stratification is not an unchanging characteristic. Similarly to the physico-chemical properties, the stratification can change significantly within very short times – even hours! Nuttall et al. (2002) were able to show that during a pumping test the reasonably good mine water quality changed to a highly contaminated mine water (section 12.7).

Such observations can also be made in wells. Wells interfere with the aquifer or the shaft and disturb the natural situation in regard to the flow field and the chemical composition of the ground- or mine water. Furthermore, wells show stratification similar to the stratification in shafts (Lloyd and Heathcote 1985). However, this stratification, though looking similar to that in a shaft, has a different physical background than the stratification in shafts. This is the consequence of the length–diameter ratio in a well, which is comparatively large, and therefore wall roughness plays a more important role than in a shaft with a smaller length–diameter ratio. Many abandoned shafts are backfilled and equipped with cased wells. As a result, the sampling procedures required for wells have to be applied to such shafts as well. A usual procedure when sampling wells is to pump out three times the volume of the well before taking a water sample. Yet, it is more appropriate to pump as long as the on-site parameters remain constant. Those parameters are temperature, electrical conductivity, pH, or the oxygen content, which have to be measured in flow-through chambers. Especially, if the well is cased with a steel casing, there is also a reaction between the mine water and the steel. Samples

taken in such a situation are not representative and must therefore be avoided, as the steel-water reactions significantly change the physico-chemical parameters of the water (Lloyd and Heathcote 1985).

Samples taken must be representative for the water or sediment that has to be investigated, because treatment and remediation measures are based on the assumption that the results of any monitoring programme are representative for the catchment. To ensure representative samples, the geological, ecological, and hydrological situation in the catchment area must be known precisely. During the implementation process of the EU Water Framework Directive numerous catchments were studied and a lot of representative material is already available. Besides the real sampling points in the catchment area, which represent the regional part chosen, there should also be blind sampling points being used for quality control reasons (see section 8.9). Herbert and Sander (1989) described that sampling in highly saline mine waters of salt mines (brines) needs special consideration. The following precautions have to be taken to make sure that the water sample is representative:

- avoid evaporation during sampling
- avoid contamination and an alteration of the sample by the mine air
- avoid particles in the sample
- sample under original temperature and pressure conditions of the solutions in the rock matrix

During and after the Hope/Germany flooding experiment (section 12.14) Herbert and Sander (1989) developed a patented sampling tool which is able to take samples and on-site parameters under “real” *in-situ* conditions. All the relevant on-site parameters can be measured without an alteration of the sample before or during the measurement.

For every dataset the name of the sampling point, dates and times of the sampling as well as the technicians or scientists having taken the samples, and the weather conditions must be recorded. Similar to that, the brand of all meters, filters or any other tool have to be written down to ensure that potential mistakes can be tracked back to its reason. Finally, all chemical data should be tested for accuracy to reveal unreasonable values. Experienced hydrogeologists normally have a feeling for some reasonable or unreasonable data, for example if pH in a carbonate aquifer is significantly deviating from near neutral to basic conditions. If one cannot count on a experienced hydrogeologist, statistical methods should be used (e.g. Kellner et al. 1998) and if there are missing data, several procedures are available to remove that lack (e.g. Lloyd and Heathcote 1985).

One of the standard methods to check the accuracy of a water analyses is a cation-anion balance calculation or a test on electro neutrality of the analyses. Both methods are very similar and it is just a matter of liking and habit which one to use. Different authors also tend to use the expression “ionic balance” for “electro neutrality” and vice-versa:

$$\text{electro neutrality} = \frac{\sum \text{cations} - \sum \text{anions}}{\sum \text{cations} + \sum \text{anions}} \times 100\% \quad (62)$$

$$\text{ionic balance} = \frac{\sum \text{cations} - \sum \text{anions}}{0.5 \times (\sum \text{cations} + \sum \text{anions})} \times 100\% \quad (63)$$

with cations and anions given in equivalents and the electro neutrality and ionic balance expressed in %. It can easily be seen that

$$\text{ionic balance} = 2 \times \text{electro neutrality} \quad (64)$$

making both methods equal in their meaning. If both anions and cations are missing in the same amount, this method will not detect the error in the analyses. Therefore, ion balancing should always be accompanied by other statistical methods as described in the next paragraphs.

One of those other methods is comparing the electrical conductivity with the *measured* total dissolved solids (TDS). Some hand-held probes, such as the Ultrameter 6P (MYRON L Company, Karlsbad/USA), *calculate* the TDS depending on the temperature and the electrical conductivity. Those TDS values cannot be used for evaluating the accuracy of the electrical conductivity measurement. Depending on the water chemistry the following relations exists:

$$\kappa = \text{TDS} \times 0.6 \dots 0.725 \quad (65)$$

with κ : electrical conductivity, $\mu\text{S cm}^{-1}$
 TDS : total dissolved solids (analysed), mg L^{-1}

Some chemical-thermodynamic codes calculate the electrical conductivity based on the chemical analyses. Very seldom researchers use those results though they would be a very easy measure to check the accuracy of a water analyses. Another mathematical method is based on the ion mobility u and their concentration c in the water (Rossum 1975; Coury 1999; according to a pers. comm. by Peter T. Kissinger, Chairman and CEO Bioanalytical Systems Inc., the second part of Coury's article never was published) and even for mine waters with complicated chemical compositions those calculations proved to be suitable. This method can easily be used in every spreadsheet or database and therefore it shall be described here. Simply, the electrical conductivity of a solution with strong and weak electrolytes can be written as:

$$\kappa = F \times \sum_i^n (|z_i| \times u_i \times c_i) \quad (66)$$

with κ : electrical conductivity, S cm^{-1}
 F : Faraday constant, $9.6485 \cdot 10^4 \text{ C mol}^{-1}$
 z_i : charge of species i , no unit
 u_i : mobility of species i
 c_i : concentration of species i , mol cm^{-1}

Because the mobility is concentration dependent, this equation cannot be directly used in the calculation without further rearrangements. In experiments it was found out that the concentration dependent parameters composing the equation can be calculated by using tabulated "limiting ionic molar conductivities" (also called equivalent ionic conductance), the ionic charge of the species, and its molar mass. By using a small program that can be programmed with MS Excel spreadsheets or in Visual Basic add-ins chemical analyses can quickly be checked for their accuracy. Details and limitations of that approach and the relevant literature are described in appendix I.

9.3.2 On-site Parameters

On-site parameters are water parameters that cannot or only with a high effort be preserved during transportation or sample handling. They can change quickly after a sample has been taken. Those parameters are meta-stable, redox- or pressure-dependent and include e.g. pH, redox potential, temperature, ferrous/ferric iron contents, base and acid capacity (in the mine water literature usually called acidity and alkalinity), dissolved oxygen, and specific electrical conductivity. Therefore, those parameters have to be measured *immediately* after the sample has been taken, in other words: on-site (Fig. 66) or in flow-through chambers. Only in cases where a short time between sampling and laboratory analyses is guaranteed, can specific elec-

trical conductivity, acidity, and alkalinity be measured reliably in the laboratory (Cravotta and Hilgar 2000).

A quality assurance/quality control (QA/QC) plan must define if the parameters have to be measured in filtered or unfiltered water and which filter shall be used (see section 8.3.1). Temperature, electrical conductivity, pH, redox potential, total base capacity (acidity), total acid capacity (alkalinity), and oxygen content *always* have to be measured in the unfiltered sample, whereas ferrous/ferric iron should be measured in both filtered and unfiltered samples. For iron it must be clear what the parameters “dissolved iron” and “total iron” mean. Both expressions are often wrongly used by some authorities and consultants for the sum of ferrous and ferric iron (that is total iron). The expression “total metal” should be restricted to the sum of the species only! The reason for this confusion is due to the relevant DIN and ISO regulations (DIN 38406-E1-1 May 1983; EN ISO 11885 November 1997; DIN 38406-32-1 May 2000). Only EN ISO 11885 clearly – but unfortunately not precisely from a chemical and physical point of view – defines “dissolved” metals: “The fraction of metals in a water sample that passes a 0.45 μm filter”. Concerning aluminium, DIN ISO 10566 April 1999 clearly writes “filterable aluminium” instead of “dissolved” aluminium. Future version of those regulations should become more precise on this point (Tab. 15).



Fig. 66: Mine water and tracer sampling in an underground mine.

On-site parameters may never become constant because the physico-chemical properties of the mine water can change rapidly, in seconds or minutes. To overcome this problem it is advisable to take at least three readings within short time intervals (minutes) and calculate their mean value (Tab. 16). At this point all readings must be recorded with their corresponding units and without any field calculation results. Furthermore, the readings should be recorded in full, since all the digits of the result (e.g. pH 7.10 instead of 7.1 are a measure of the precision (not accuracy!) of an instrument.

Most of the on-site instruments used were originally constructed for laboratory use and were adapted to the conditions outside the laboratory. Although good outdoor equipment is now available, most types are made for water with low contamination levels when compared to mine water. Furthermore, they are equipped with plugs or probes that are extremely sensitive for the use in the rough field environment. Whilst the calibration procedure of a probe is simple in a laboratory, the on-site situation could be difficult – especially when working in poten-

tial methane-prone mines. Some companies provide excellent and easy to handle on-site equipment and also modern “pencil”-probes can be used to get reliable on-site parameters.

Tab. 15: Expressions to be used for analysing iron in mine waters. The same expressions should be used for other metals and semi-metals. The kind of filter (0.02, 0.2, 0.45 μm) must always be recorded.

Constituent	Sampling Procedure	Term	Common Tem	DIN/ISO use
Fe^{2+}	Filtered	Fe^{2+} filtered	ferrous iron filtered	–
Fe^{3+}	Filtered	Fe^{3+} filtered	ferric iron filtered	–
$\text{Fe}^{2+} + \text{Fe}^{3+}$	Filtered	Fe_{total} filtered	total iron filtered	dissolved iron
Fe^{2+}	Unfiltered	Fe^{2+} unfiltered	ferrous iron unfiltered	–
Fe^{3+}	Unfiltered	Fe^{3+} unfiltered	ferric iron unfiltered	–
$\text{Fe}^{2+} + \text{Fe}^{3+}$	Unfiltered	Fe_{total} unfiltered	total iron unfiltered	total iron

Tab. 16: Example of a series of on-site measurements with calculated means. The redox potential is not corrected to standard H_2 -electrode and 25 $^{\circ}\text{C}$! The redox potential is not stable and should have been measured for a longer period. From the other parameters it might also be assumed that the total chemistry and the water temperature of the water was changing during the sampling, which took no longer than 5 minutes.

Parameter	Unit	#1	#2	#3	#4	#5	mean
O_2	mg L^{-1}	0.34	0.30	0.35	0.31	0.34	0.33
temperature	$^{\circ}\text{C}$	18.2	17.4	16.8	16.6	16.4	17.1
electrical conductivity (25 $^{\circ}\text{C}$)	$\mu\text{S cm}^{-1}$	2648	2687	2704	2730	2750	2704
total dissolved solids	mg L^{-1}	1934	1975	1991	2012	2027	1988
redox potential	mV	115	101	92	79	66	91
pH	–	7.19	7.10	7.08	7.04	7.02	7.09

Those mobile on-site instruments that can be calibrated (e.g. pH, O_2 , electrical conductivity) have to be calibrated regularly, at least every day when in use and in the case of dissolved oxygen, before every measurement. The instruments require regular maintenance since membranes could be poisoned or damaged and precipitates can cover the probes (e.g. ochre, bio films). Plugs may be dirty or batteries low. Serviceability of all instruments should be ensured before every field trip!

A major problem with mine water is that it often consists of ground water or seepage water from waste rock piles or tailings that might containing elevated concentrations of carbon dioxide or low amounts of dissolved oxygen. When this water reaches the surface, it comes into contact with the low carbon dioxide concentration and high oxygen concentrations of the atmosphere. Thus, depending on the flow regime (laminar/turbulent) it might take a few minutes or hours to achieve a new equilibrium. Since oxygen and carbon-dioxide are key parameters, the master variables like pH, redox potential, and dissolved constituents e.g. iron might change dramatically and consequently the entire water chemistry will change. Redox reactions in particular are catalysed by microorganisms although very slowly. Thus, it is unlikely to see thermo-dynamic equilibrium in mine waters. This should always be considered in the evaluation of data which are based on the assumption of thermo-dynamic equilibrium (e.g. redox potential).

Absolute temperatures can be determined with an accuracy of about 0.1 K and relative temperatures with a precision of about 0.01 K (Lloyd and Heathcote 1985). After the temperature equilibrium between the medium and the thermometer is reached, the temperature reading can be taken. Because electronic thermometers are usually very small, those readings can be taken

very quickly. Alternatively, mercury or alcohol filled thermometers can be used, yet their temperature equilibrium is reached after several seconds up to a minute.

Most probes for electrical conductivity neither require a special treatment (and storage) nor a calibration. However, the accuracy of the probe should always be tested with a standard solution of known electrical conductivity. Some high quality probes also have the capability to be calibrated within the range of use. Readings usually are automatically recalculated to a temperature of 25 °C and can be taken after a short time interval. If the indicated value of the readings on the gauge is varying, this is normally an indication for a malfunction of the probe or the amplifying device (Fig. 97).

pH, nowadays, is commonly measured with pH-probes. Easy usable field probes should have an automatic temperature compensation, automatic buffer recognition and are battery powered. The following stages are necessary to achieve a representative field measurement:

- the pH probe has to be calibrated before any use at least once a day with two, or preferably three or four standards (4, 7, 10 for normal water; 1.68, 2, 4, 7 for acid mine water)
- the pH probe must be stored in a 3 molar KCl solution if not in use (or whatever is recommended by the manufacturer)
- the measurement has to be performed in a flow-through chamber to guarantee both a slight movement of water along the probe and absolutely no contact with the atmosphere
- readings can be taken after 10 to 15 minutes with an accuracy of 0.01—0.05 pH units

If pH probes are not available, colorimetric pH-measurements with either indicator dyes, litmus paper, or modern multi indicator strips should be used. Yet, for dilute waters, dyes and indicator strips can give very erroneous results (Arthur Rose, pers. comm.). Nordstrom et al. (2000) recently reported about extremely acid mine waters with pH-values as low as −3.6. If such negative pH-values are expected, normal pH-probes cannot be used in the field without prior calibration against standard solutions. Furthermore, the pH has to be defined by using the Pitzer method for the calculation of the theoretical pH of the standard solutions and consequently the calibration of the glass membrane electrodes. Details of that approach have been described by Nordstrom et al. (2000).

In addition, redox potential and dissolved oxygen measurements have to be performed in flow-through chambers to avoid degassing or uptake of gaseous components from the atmosphere. Redox probes are subject to surface poisoning which can be overcome by mechanical cleaning of the platinum electrode with corundum powder. Redox probes cannot be calibrated and should be tested to be fit for purpose by standard solutions (e.g. Zobell solution). Since achieving equilibrium might take several hours (obtainable reproducibility about 50 mV), it is recommended to use at least two probes, one for oxidising solutions and another one for reducing conditions. It is further recommended to plot continuous time series to decide whether stable values were reached. Lloyd and Heathcote (1985) present time series of pH- and redox measurements showing that both parameters did not reach stable values before 30 minutes of measuring. If redox measurements in shafts or boreholes are conducted, the eigenpotential of the system must be considered (section 12.1).

Another factor, often overlooked, is the necessary conversion of the redox potential. The international agreed basis for the redox potential is the standard-H₂-electrode. As such electrodes cannot be used in the field, the redox potential is usually (but not always) measured with an Ag/AgCl-electrode (3 mol L^{−1} KCl) and the measured redox potential has to be converted to the potential of the standard-H₂-electrode. Because the redox potential is also temperature dependent, the temperature must also be measured. Instead of using tables, the fol-

lowing simplified empirical equation and the coefficients in Tab. 17 can be used, where the first two terms ensure the temperature correction and the square root the conversion to the standard-H₂-electrode (accuracy of equation: ± 0.5 mV between 5 °C and 65 °C):

$$E_{0(25^{\circ}\text{C})} = E_t - 0.198 \times (T - 25) + \sqrt{a - b \times T} \quad (67)$$

with E_0 : converted redox potential, mV
 E_t : measured redox potential at temperature T , mV
 T : measured temperature, °C

Tab. 17: Coefficients to be used for the redox calculations according to equation 67.

Sensor Type	Coefficient a	Coefficient b
Mercury calomel KCl	67798	324
Ag/AgCl KCl 1 mol L ⁻¹	62612	279
Ag/AgCl KCl 3 mol L ⁻¹	50230	295

Dissolved oxygen (often abbreviated DO) is usually determined by a potentiometric electrochemical probe based on the Clark-sensor (Clark et al. 1953). The sensor has to be replaced regularly (see manual) and calibration for 0 and 100% O₂ has to be completed before each measurement, whereas temperature and pressure calibrations are usually done by the instrument itself (Schwedt 1995). Readings can be taken after a few minutes with an accuracy of 0.1 mg L⁻¹. Common electrochemical O₂-probes have an operation-range from 0.5 mg L⁻¹ to O₂-saturation, thus readings below 0.5 mg L⁻¹ are meaningless. Because the sensor consumes oxygen during the measurement, the sensor has to be lapped around during the whole measurement or kept in a flow-through chamber with a flow of about 10 cm s⁻¹ (Lloyd and Heathcote 1985). Continuous oxygen measurements in mine waters are problematic due to sensor drifts, temperature and pressure dependency, accumulation of biofilms, and Fe-oxide precipitation. During the last years, new ceramic and fibre optic optode-based O₂-probes were developed (Klimant et al. 1995). Those luminescent dissolved oxygen (LDO) sensors are coated with a luminescent material. Light from an light emitting diode (LED) is transmitted to the sensor surface which excites the luminescent material which upon relaxation emits light of a shorter wavelength. Both, the intensity of the steady state luminescence and the average relaxation time are inversely proportional to the dissolved oxygen concentration and are therefore measured and correlated to the dissolved oxygen concentration. To eliminate sensor drifts due to ageing of the luminescent material internal references are used (Rommel 2004; Jackson and Hach Company 2004; Hach Company Product Information; Van Essen Instruments Company Information). Although those sensors proved to be suitable for waste water treatment their reliability for mine water investigations was not verified yet. Körtzinger et al. (2005) were able to show that such innovative oxygen probes can be used in extreme conditions of high salinity in the ocean and up to a pressure of 20 MPa. It might therefore be assumed that such probes can also be used for measuring depth profiles of flooded mine shafts.

Two other parameters that are essential in mine water research, especially for the design of a treatment plant, are the base and the acid capacities (acidity, alkalinity) of the mine water. Different definitions and instructions of how to measure those parameters are known and can be used (e.g. Hedin et al. 1994a; Cravotta and Hilgar 2000): hot acidity with hydrogen peroxide (American Public Health Association et al. 1998; International Institute for Environment and Development 2002) cold acidity with hydrogen peroxide (Fishman and Friedman 1989), or cold acidity without hydrogen peroxide (Ott 1986, 1988). Acidity and alkalinity are usually evaluated by acid-base titration. In normal surface or ground waters the dominant species contributing to acidity and alkalinity are the species of the carbonate system: H₂CO₃, HCO₃⁻, and

CO_3^{2-} . Therefore, the measured alkalinity and acidity, strictly spoken, are related to those species. If much mineral acidity exists the concentrations of the bicarbonate system cannot be evaluated from the alkalinity and acidity titrations, but have to be replaced by the TIC measurements. A detailed discussion of net alkalinity and net acidity can be found in Kirby and Cravotta (2005a, 2005b).

Furthermore, if mineral acidity is present, it is possible to estimate the total base capacity (acidity) in mol L^{-1} from the pH and the concentrations of Fe, Mn, Zn, and Al (equation modified after Hedin et al. 1994a):

$$\text{Acidity} = \gamma_{\text{H}_2\text{O}} \times 10^{(3-\text{pH})} \times \sum_1^n \frac{z_i \times [\text{Me}_i]}{M_i} \quad (68)$$

with z_i : charge of species i , no unit
 pH : pH-value, –
 $[\text{Me}_i]$: concentration of species i (e.g. Fe, Mn, Zn, Al, Co, Cu, Cd), g L^{-1}
 M_i : molar mass of the species i , g mol^{-1}
 $\gamma_{\text{H}_2\text{O}}$: activity of mine water, no unit

Compared to other forms of this equation which can usually be seen in publications, it is written here in its generalised form for being used for all types of species that are contributing to the acidity. Furthermore, the activity of the water is included here. The reason is that the calculated acidities according to the figures or tables given in Hedin et al. (1994a) or Cravotta and Hilgar (2000) regularly overestimate higher acidities, which might be due to the missing activity of the mine water and possibly complexing of ferric iron and Al by OH^- and HSO_4^- (Kirby and Cravotta 2005b).

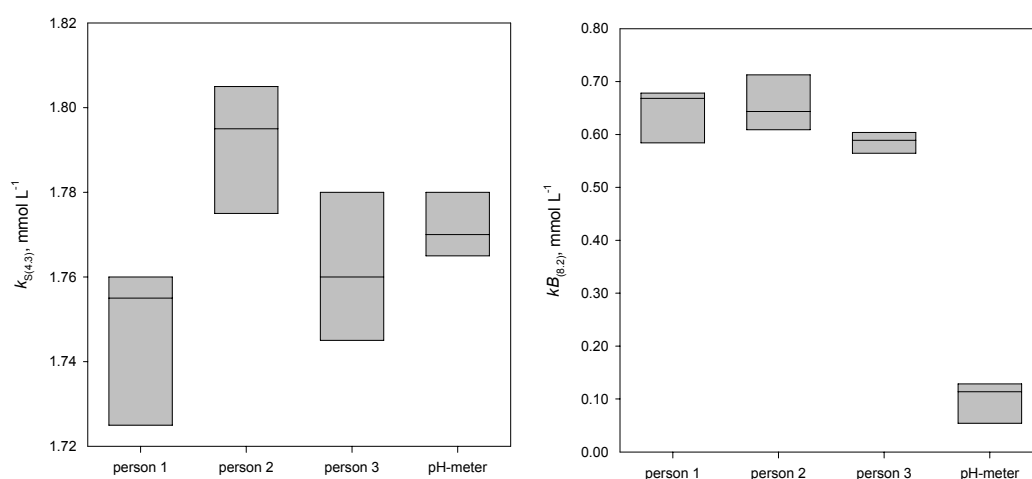


Fig. 67: Repeated measurements of acid capacity $K_{A(4.3)}$ and base capacity $K_{B(8.2)}$ of mine water (Drainage Adit *Rothschönberger Stollen*, Freiberg/Germany). 1—3: titration by experienced technician using indicators to determine the titration end point; 4: titration by using a pH-meter for determining the titration end point. Each titration was repeated three times.

Since the different methods give different results, it is essential, to measure both parameters by using a pH-electrode and recording the titration end point (Fig. 67). Acid and base titrations in the field have the disadvantage that they cannot be performed without the sample being in contact with the atmosphere. Thus, it cannot be excluded that the properties of the sample will change due to degassing of carbon dioxide or to chemical and physical reactions be-

tween the sample and the atmosphere. A recommended alternative method is taking a sample in the field, storing it in a gas-tight container and determining the total inorganic carbon (TIC) in the laboratory. This can be achieved by lowering the pH to 2 and purging the CO_2 from the sample before the detection via non-dispersive IR-photometry in the gas phase. The TIC values can then be used to calculate the acidity and alkalinity.

During the field studies of the German Gernrode/Harz mine water it became apparent that the acid/base titrations that were used for evaluating the acid capacity K_A and base capacity K_B (alkalinity, acidity) of the mine water seemed to be faulty because the calculated bicarbonate was either too high or too low (Fig. 68, Fig. 69). Consequently, four different mine drainage waters (Straßberg/Harz, Fürstentollen/Halsbrücke, Rothsönberger Stollen/Rothschönberg, Verträge Gesellschaft Stollen/Freiberg) were tested to find the reason for this observation.

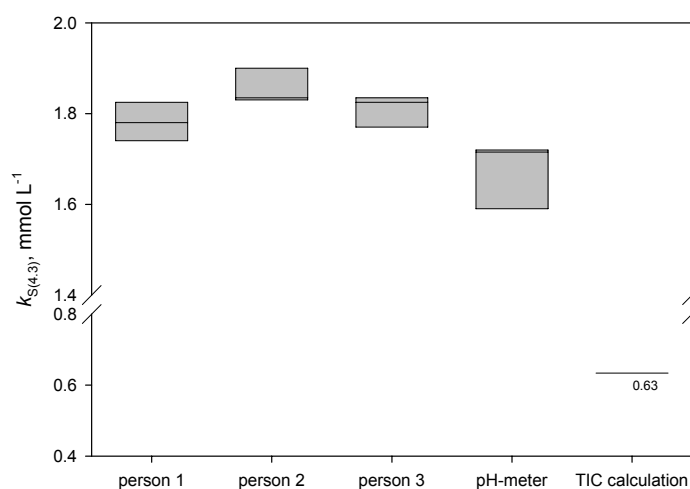


Fig. 68: Results of acid capacity measurements $K_{A4.3}$ (alkalinity) with three different methods: 1—3: persons 1, 2, 3 using indicators; 4: titration end point by using a pH-meter; 5: TIC calculation (total inorganic carbon): determination in the laboratory, $n = 3$ except for TIC (data from Lieber 2003).

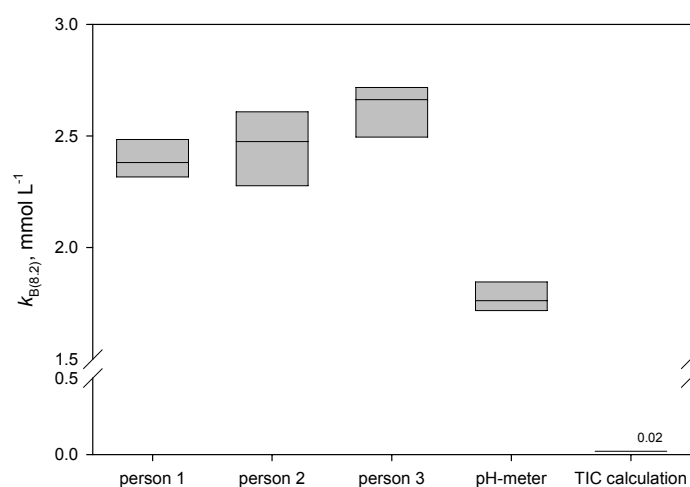


Fig. 69: Results of base capacity measurements $K_{B8.2}$ (acidity) with three different methods: 1—3: persons 1, 2, 3 using indicators; 4: titration end point by using a pH-meter; 5: TIC calculation (total inorganic carbon): determination in the laboratory, $n = 3$ except for TIC (data from Lieber 2003).

Each mine water was titrated in the field with HCl (end point pH 4.3) and NaOH (end point pH 8.2) three times with two different methods (organoleptic test with indicator, pH-meter) and by three persons. Furthermore, TIC (total inorganic carbon) was determined for each mine water and compared with the field results. It became clear that, in some cases, the different methods used for evaluating the acid and base capacities produce significantly different results. Furthermore, the laboratory TIC determination of $K_{A4.3}$ differs significantly from the results of the field determination, whereas the acid titration with indicators and the pH meter did not show significant differences. No calculations of acidity based on equation 68 was conducted, because no metals were analysed.

The situation is reversed in the case of $K_{B8.2}$. TIC-determined $K_{B8.2}$ is close to the value determined with the pH-meter, while the organoleptic determination significantly differs from the value determined with the pH-meter.

From the results of this study it must be concluded:

- $K_{B8.2}$ and $K_{A4.3}$ have to be determined in the field by using a pH-meter instead of indicators
- further studies have to be conducted in comparing TIC-determined acid/base capacities and field determined ones.

If all the above mentioned parameters are measured on-site, and samples for main and trace elements be taken, a 30—60 minute sampling period must be allowed. An additional 10—30 minutes must also be allocated for precise flow measurements.

9.3.3 *Presentation of Results*

During a mine water study within a catchment area a lot of information is collected and stored in different forms, e.g. graphs, tables, or drawings. All this information has to be presented in a report to understand the hydraulic, hydrological, hydrogeological, hydrochemical, and geological conditions investigated. Usually, this information is compiled in a written report, describing the investigations, results, the conclusions, and finally the recommendations derived from the conclusions and results. Therefore, writing a final report is essential to provide the contract awarder and the authorities involved with answers to the questions they had at the start of the project. Usually, a report is the only summarising work presentation within a project.

During writing of a report the style of language used and the form of the document must be given special consideration. Not only the results of an investigation are important, but the words and the style of the report. Conclusions and recommendations can only be clearly understood when being presented in an understandable way. It is recommended to use the following structure for a report about the mine water conditions in a river catchment (modified and adapted after Brassington 1999):

- summary or extended summary
- introduction
- topography and drainage situation of the river catchment
- geological and hydrogeological conditions
- methods
- results (chemistry, heads, flows)
- conclusions and recommendations
- appendix with large format graphs, tables or drawings

Results should be presented in three different ways: in tables, where all measured parameters are listed, in clear graphs which show the relevant changes of important parameters, and summary results after processing the data with statistical methods (e.g. correlation, principal component analyses, descriptive statistics).

Tables usually contain all the data measured or obtained within the catchment or in the laboratory. No interpretation shall be done in those tables and all the data must be given in the way it was measured. Explanatory tables and data tables should always be separated from each other to prevent others from being influenced by personal opinions. Furthermore, it is not good status and not according to QA/QC measures to leave out “outlier” data or to interpolate missing data without giving reference to that practise.

Especially, if a large number of data has been collected within the catchment or the mine, it is advisable to produce graphs which quickly show differences and changes of selected parameters. Several graph types for this goal are available and are used widely. For a quick classification of the water types, Piper- (Fig. 70) or Durov-Diagrams (Fig. 71) give an *impression* of the water composition (Piper 1944, 1953; Lloyd and Heathcote 1985; Klapper and Geller 2001), though such graphs are inappropriate for characterising typical mine waters (ERMITE Consortium et al. 2004). Excellent introductions and explanations of how to interpret Piper- or Durov-diagrams can be found in Hounslow (1995) or in Lloyd and Heathcote (1985). Yet, it must be clear that different waters with the same relative concentrations of the main ions plot into the same field and can therefore not be distinguished from each other in diagrams that use relative numbers. Especially for contaminated mine water, a Ficklin-Diagram (sum of Zn, Cu, Cd, Co, Ni, and Pb concentrations; Ficklin et al. 1992; Fig. 72) will give an indication about the contaminant potential of the waters investigated. Another diagram that can be used to classify mine waters is a net alkalinity versus $\text{Cl}^-/(\text{Cl}^- + \text{SO}_4^{2-})$ -diagram introduced by Younger and improved by Rees et al. (Younger 1995; Rees et al. 2002; Fig. 73).

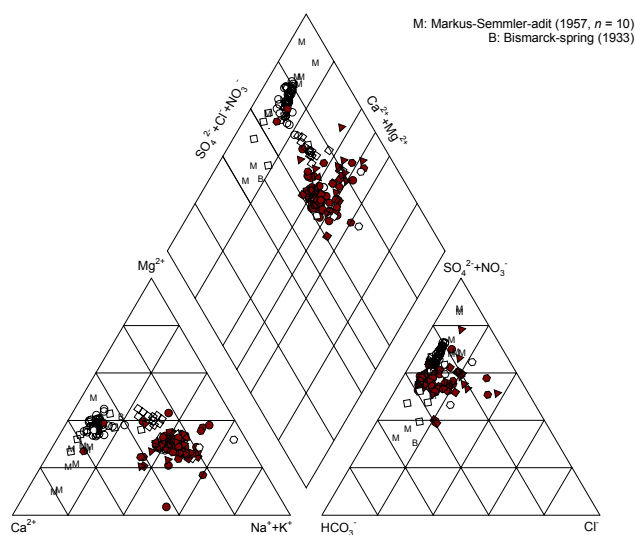


Fig. 70: Example of a Piper Diagram for mine waters in the abandoned Niederschlema/Alberoda/Germany uranium mine (from Wolkersdorfer 1996). Coloured symbols are mine waters, open symbols infiltration waters.

The range of a single parameter is best presented in box-plots which show the mean, median, 5, 10, 90, and 95% quantiles and outliers in a transparent way (Fig. 74). Histogram diagrams can be used to quickly visualise data clustering (Fig. 75). Always, good scientific practice is giving error bars. In a report about the total iron concentration within a waste rock dump a

graph was given that, according to the explanation, showed a clear increasing iron concentration with depth and a correlation coefficient r^2 of 0.9 (Fig. 76 left). The original data were listed in a table and in the appendix the accuracy of the method used was tabulated as well: $\text{Fe}_{\text{tot}} \pm 0.3 \text{ mg kg}^{-1}$. Furthermore, the text explained that each sample was analysed from a 0.25 m long core. Plotting the data with the error bars clearly proves that there is no depth correlation at all because the Fe_{tot} values overlap between 1.11 and 1.36 mg kg^{-1} . Therefore, the correct statement should have been: “The Fe_{tot} values of the waste rock pile range from 1.06 to 1.41 mg kg^{-1} and do not show a depth dependence within its error intervals”.

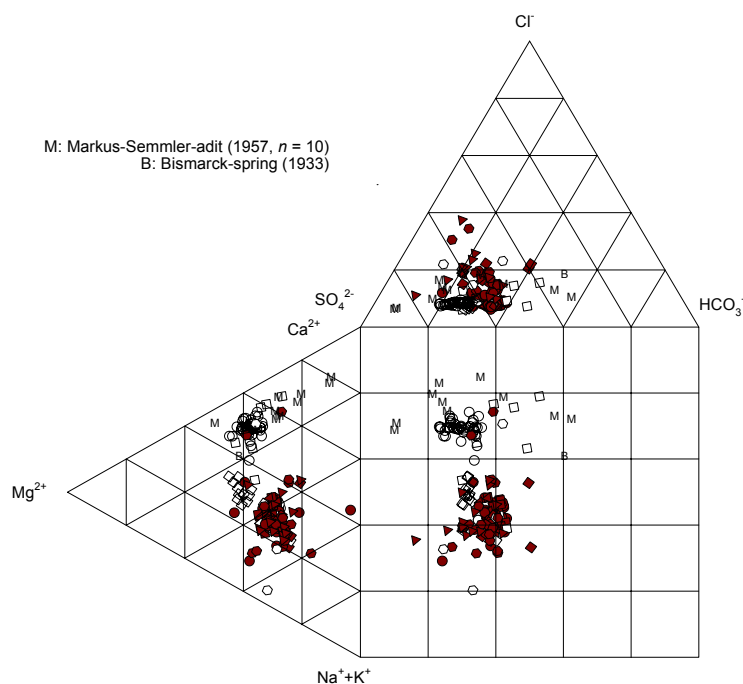


Fig. 71: Example of a Durov Diagram for mine waters in the abandoned Niederschlema/Alberoda/Germany uranium mine (from Wolkersdorfer 1996). Coloured symbols are mine waters, open symbols infiltration waters.

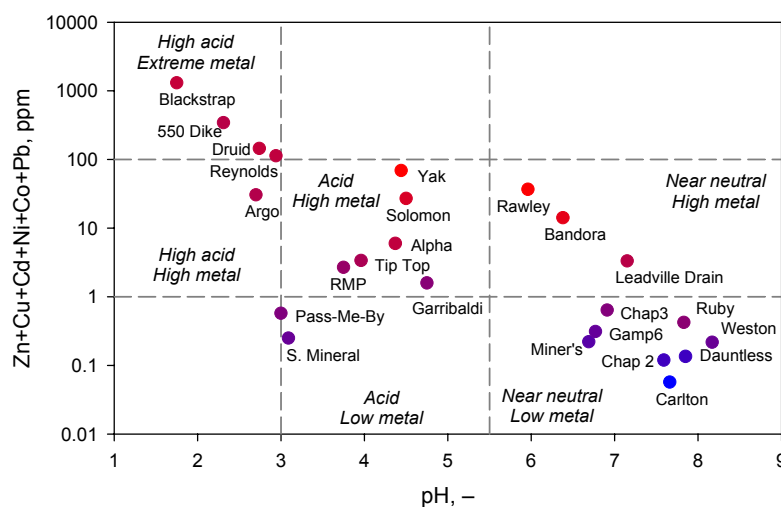


Fig. 72: Ficklin diagram for mine waters of western Colorado (modified after Ficklin et al. 1992).

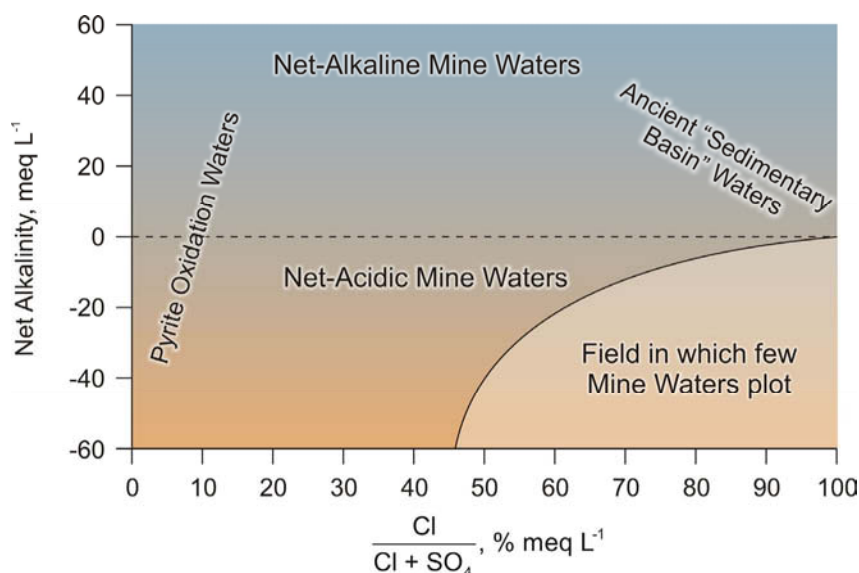


Fig. 73: Mine water classification diagram with net alkalinity against ratio of chloride to sum of chloride and sulphate according to Younger (1995) and Rees et al. (2002).

If different water types have to be classified, cluster analyses or principal component analyses should be used (e.g. Davis 1986). Especially for cluster analyses it is important to find the relevant parameters which can be identified with a principal component analyses before the clusters are calculated.

With the improvement of GIS-technologies, and GPS, GLONASS, or GALILEO availability, the monitoring project should be managed by GIS systems throughout the whole team. Especially on maps produced by those systems, stiff diagrams or pie charts are helpful for comparing different sampling sites. It is advisable to agree to a common GIS and georeference system before the project starts, as some systems are not fully compatible with each other. The EU is establishing a European geographic references system which, in some countries, differs significantly from the national reference systems.

Detailed explanations for writing reports, presenting data, or for evaluating hydrological data are given by Lloyd and Heathcote (1985), Shaw (1994), and Hounslow (1995).

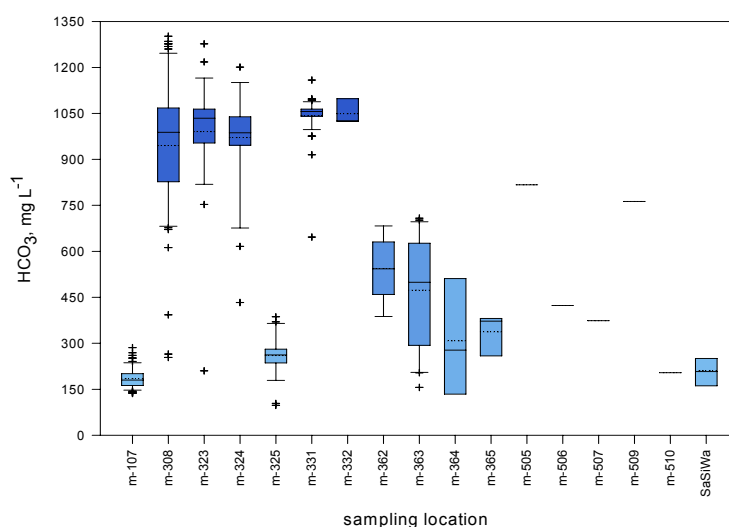


Fig. 74: Example of a box-plot showing 312 data points at 17 sampling locations.

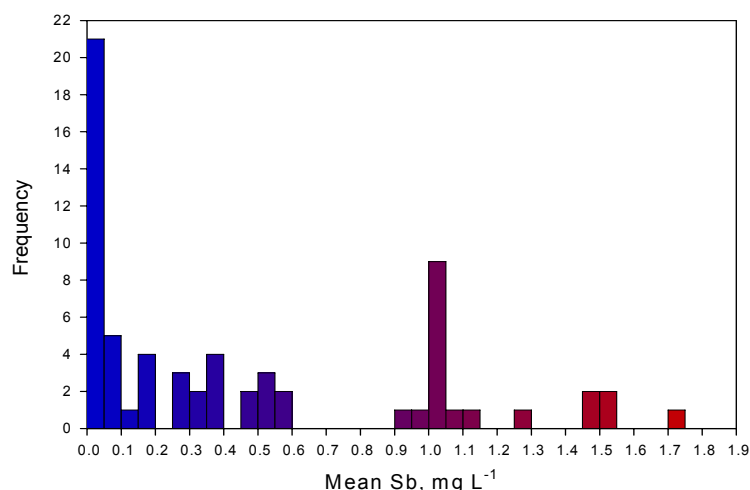


Fig. 75: Example of a histogram, showing the frequency distribution of antimony and the clustering of the data. $n = 66$.

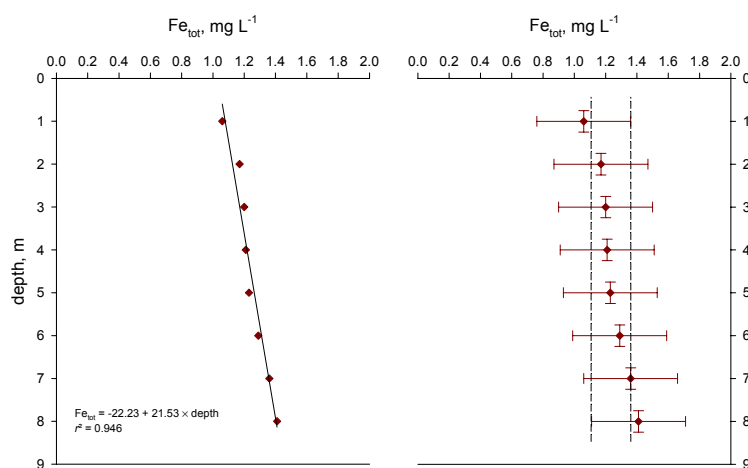


Fig. 76: Faked depth dependencies of Fe_{tot} in a waste rock pile.

9.3.4 Regression and Data Prediction

This chapter is focused on regression and data prediction rather than statistical interpretations in general. In the broader sense it falls into the field of data mining or artificial neural networks – without going into the details of those powerful tools which are commonly used to find relations between parameters that are not obvious at a first glance (e.g. Savic and Walters 1999; Blockeel et al. 1999; Bourgeois et al. 2003; Hewett 2003; Bessler et al. 2003).

Regression and data prediction is a commonly used method by many scientists and consultants. It is an essential method if missing or future values have to be estimated or predicted. A simple example is the estimation of stream or mine water flows by measuring the water height in a given cross section. Usually, 3–10 flow measurements at different water levels are conducted and then a calibration curve is computed.

Many reports present scatter plots including lines which shall emphasise relations between the plotted parameters. Such lines are commonly based on statistical investigations and are given as regression curves or are described as correlation between the investigated parameters. Unfortunately, many of those curves are given with a lack of statistical understanding and by

using software tools which are easy to use but which are not properly suitable for such data presentations. Currently there is only one comprehensive software package available that fulfils *all* the needs of a proper statistical investigation for curve fitting and equation discovery: TableCurve 2D produced by SYSTAT Software Inc. Richmond, California (Systat Software Inc. 2002; for 3D-Data, TableCurve 3D is available as well). Nonetheless, even if proper tools are used for explaining relations between two or more parameters this does not mean that there is a geochemically based relation between those parameters. In cases where analytical solutions or relations are known, those relations must be used for predicting future or dependent values. Often, such analytical solutions are not known and a black box method (lumped parameter model), a Bayesian, or a Monte Carlo method will be used.

Another fact that has to be considered when computing predictions by using PC based software is the floating point precision of the algorithm (not that of the final result). Intel's floating point co-processor has a 19-digit precision, which is used by the common standard graphical tools (e.g. Microsoft Excel). For many predictions this precision might be good enough, yet, TableCurve besides the Intel co-processor, uses its own math emulator with a 38-digit precision. One of the most powerful tools of TableCurve is its ability to investigate the prediction by using several statistical and graphical tools besides the regression coefficient r^2 . Thus, the best prediction can be selected by using graphical and statistical methods.

A thorough regression or data prediction analysis needs to investigate different potential relations between parameters. Sometimes, those relations are not obvious from the correlation investigations, because in the case of multiple dependencies of two parameters, the regression coefficient would be small, but the two different relations could be detected by a regression analysis. As in the case of all statistical analyses or numerical modelling approaches, the interpretation of the results needs a thorough understanding of the governing reactions in the system. Of course, there are cases in the field of data mining where the relation of two or more parameters is not immediately obvious, but if mine water management has to be done precisely and understandable for decision makers, such un-obvious relations should be a start for more research. If regression analyses or predictions into the future are made, values should always be given in ranges better than a single value. This means, for example that the iron concentration after 10 years of flooding should not be given like 5 mg L^{-1} , but either $3\text{—}7 \text{ mg L}^{-1}$ or $5 \pm 2 \text{ mg L}^{-1}$.

9.3.5 Interpretation of Data

Data interpretation is the most important part of a consultancy work. From the interpretation of the data the conclusions are drawn and the final recommendations derived. If only the results are presented and described, a catchment or mine water management study would not have been worth the money and time spent. A good interpretation needs also good data, which means that the data gathering process was based on QA/QC procedures, the data can be verified and back-traced and that all necessary standards for data acquisition, sampling or laboratory investigations were followed.

Unfortunately, how to interpret data cannot be described here – because it cannot be learned from text books. In general, if a report is clearly structured, the illustrations and tables are presented in an understandable form, and the text is written in an understandable language, it can be concluded that the interpretation of data was given the same priority. This does not mean that all the interpretations are trustworthy, but the likelihood of getting precise results is higher than in cases of “bad” reports.

10 Tracer Techniques for mines

10.1 Why to Conduct a Mine Water Tracer Test?

10.1.1 Objectives of Mine Water Tracer tests

Usually, tracer tests in the vicinity of mines are conducted to find connections from the surface to the mine or vice versa. Typical examples are the tracer tests that accompany numerical models for performance assessment at potential radioactive waste disposal sites (e.g. Lee 1984). Nevertheless, some other cases are known. All tracer tests in mines can be grouped on the basis of their objectives:

- mine water intrushes/inundations (Skowronek and Zmij 1977; Goldbrunner et al. 1982; Wittrup et al. 1986; Qiang et al. 1992; Lachmar 1994)
- optimise mining strategy (Adelman et al. 1960; Reznik 1990; Williams and Kirschner 1992; Kirschner and Williams 1993)
- underground waste disposal: non radioactive (Fried 1972; Himmelsbach and Wendland 1999)
- underground waste disposal: radioactive (Abelin and Birgersson 1985; Brewitz et al. 1985; Galloway and Erickson 1985; Cacas et al. 1990; Lewis 1990; Birgersson et al. 1992; Hoehn et al. 1998; Sawada et al. 2000; hundreds of studies were conducted in the Stripa mine/Sweden, the Grimsel/Switzerland experimental site, and the Yucca Mountain/USA site)
- subsidence (Mather et al. 1969)
- remediation strategies (Aldous and Smart 1987; Doornbos 1989; Aljoe and Hawkins 1993, 1994; Davis 1994a, 1994b; Wolkersdorfer 1996; Wolkersdorfer et al. 1997a; Canty and Everett 1998)

Tracer tests have also been conducted to study “heat mining” in geothermal projects (e.g. Gulati et al. 1978; Horne et al. 1987; Kwakwa 1989; Randall et al. 1990; Aquilina et al. 1998). Because their use is similar to studies in fractured rocks (see Himmelsbach et al. 1992; Käß 1998) and energy mining is usually done by using boreholes only, these studies will not be considered further.

Many tracer tests have not been published in the literature or are only available as master’s theses (e.g. Anderson 1987; Bretherton 1989; Diaz 1990; Hasche 2001). This is often because they were either unsuccessful or the results confidential. So far, only a few tracer tests have been conducted in flooded underground mines to trace the hydrodynamic conditions within the flooded mine itself (Aljoe and Hawkins 1994; Wolkersdorfer et al. 1997a; Wolkersdorfer and Hasche 2001a). This was mainly because no suitable method was available for injection of the tracer into the mine water at predetermined depths or without contaminating the mine water above the injection point. Most tests therefore injected the tracer at the surface, to flow into the mine through fractures (e.g. Lachmar 1994) or they used boreholes e.g. Galloway and Erickson 1985; Cacas et al. 1990).

As will be shown in section 11.6, *in-situ* remediation methods usually fail because they lack a thorough understanding of the flow regime within the flooded underground mine. It is therefore of importance to know the flow of the mine water within the flooded underground mine and it is necessary to understand where underground dams have to be installed before flooding, if *in-situ* treatment options shall be successful. Tracer tests in flooded underground mines are therefore imperative for such *in-situ* treatment options.

10.1.2 Aims of Mine Water Tracer Tests

Tracer tests are well established in ground water studies where they are commonly used to investigate the hydraulic parameters or interconnections of ground water flow (Käß 1998). Most of the techniques used are well described and depending on the aims of the tracer test and the hydrological situation a range of tracers or methods can be chosen.

Published results of tracer tests in abandoned underground mines are still uncommon, as already stated by (Davis 1994b, 1994a). Therefore, in mine water tracing, less experience exists and the expected results of an individual mine water tracer test could not always have been predicted. Summarised, the aims are as follows:

- testing the effectiveness of bulkheads dams
- investigating the hydrodynamic conditions
- tracing connections between a mine and the surface
- clarifying reasons for inundations
- contaminant mass flow
- estimating the decrease or increase of contaminants

Historically, the first tracer tests conducted in mines were simply to reveal connections between ground or surface waters and the mine (e.g. Skowronek and Zmij 1977). One of the first tracer tests in a deep flooded underground mine to investigate the more complex hydrodynamic conditions of the flooded mine was conducted in 1995 (Wolkersdorfer 1996).

In many mining situations it would be useful to use more than one tracer in a multi-tracer approach. Yet, because the number of tracers that can be used in mines is restricted such tracer test have rarely been conducted so far. Possible candidates for multi-tracer are dyes, salts, and solid tracers. Wolkersdorfer et al. (2002) conducted a multi-tracer test in the Austrian Georgi-Unterbau mine see section 12.13. They used rock salt NaCl, sodium fluorescein uranine, and different microspheres in the flooded underground mine. Because the number of tracers that can be used in one single mine water tracer test is so limited, my working group prefers fluorescent microspheres. Up to eight different colours can be injected and so far up to six have been used in one mine water tracer test at the flooded Straßberg/Germany fluorspar mine see section 12.11.

In future studies, tracer tests should become a prerequisite for the evaluation of remediation strategies used for reclaiming abandoned mine sites. Kimball et al. (1999) already gave an example of how to use tracer tests and to carry out synoptic sampling of trace metals in surface streams to evaluate the environmental impacts of mine effluents to watersheds. That can also be done in other mine closure cases where the hydrogeological situation is unclear and the water flow in the flooded underground mine shall be evaluated as a basis for remediation strategies.

10.1.3 Possible Mine Water Tracers

Underground mines generally consist of many shafts, adits, raises, and stopes which are comparable to the features found in karstic terrains. Therefore, a flooded mine can be looked upon as a karst aquifer in a conceptual model, and numerical models of flow in karst aquifers (e.g. Liedl and Sauter 2000) might also describe the hydrodynamic situation in underground mines. Furthermore, the tracer techniques developed for karst aquifers (e.g. Gospodarič and Habič 1976; Käß 1998) should be appropriate for flooded mines, too.

According to the literature including our own investigations several classes of tracers were used in mine water tracing so far Tab. 18. Usually only successful tracer tests – if at all – are

reported. Therefore, nothing can be said about tracers that are unsuitable for mine water tracing, though some researchers investigated tracer behaviour in the mine water environment. Aldous and Smart (1987) conducted research into the absorbance of seven fluorescent tracers on iron hydroxide and kaolinite sulpho rhodamin B, sulpho rhodamin G, rhodamin WT, lisamine yellow FF, pyranine, Na-fluorescein, Tinopal CBS-X. As the behaviour of tracers varies in different environments no general conclusions can be drawn. Yet, from their studies sulfo rhodamin G and pyranine would be the best tracers in sediment laden mine water – but none of those tracers has been used so far due to analytical or environmental reasons. For Na-fluorescein, which easily adsorbs to organic materials, we know of successful tests as well as unsuccessful ones. As a conclusion it must be recommended to test the most appropriate tracers for every single case, because the chemical compositions of mine waters can vary in such a broad range.

Tab. 18: Selection of tracers that have already been used in mine water tracer tests references are a selection only.

Tracer	References
NaCl	Mather et al. (1969); Hawkins and Aljoe (1992); Wolkersdorfer (2002b); Wolkersdorfer and Hasche (2004)
Na-fluorescein	Mather et al. (1969); Parsons and Hunter (1972); Nunes et al. (1990); Wirsing (1995)
Rhodamin WT	Canty and Everett (1999)
Club Moss Spores	Wolkersdorfer and Hasche (2001b); Wolkersdorfer (2001); Wolkersdorfer and Hasche (2004)
Microspheres	Wolkersdorfer and Hasche (2001b); Wolkersdorfer (2001); Wolkersdorfer and Hasche (2004)
Sulpho Rhodamin B	Aldous and Smart (1987)
Tritium	Mulenga et al. (1992); Zhou et al. (1993)
CFC chlorofluorohydrocarbon	Parry et al. (2000)
Rare Earth Elements	Worrall (1999); Wolkersdorfer (2002a); Merten et al. (2004)

The chemical composition of mine waters very often tends to be extreme: TDS, pH (e.g. Iron Mountain: pH –3.6; Nordstrom et al. 2000) or metal concentrations are usually high (Banks et al. 1997) and consequently limit the number of tracers that could be used successfully. Even conservative tracers, as Na-fluorescein, might be unsuitable under certain mine water conditions e.g. low pH, high suspension load, wooden support, free chloride radicals.

Therefore, laboratory tests using the chosen tracer and typical mine water compositions are essential for obtaining positive results. All tracers have to be selected based on their expected or known behaviour in the mine water in question. An important consideration when conducting such tests is the duration of the laboratory test. From our experience, mine water flows with mean velocities of 0.3 to 1.7 m min⁻¹ 95% confidence interval of 40 tracer tests investigated, excluding the maximum and minimum values; see Tab. 19 which can be used for a rough calculation of the expected residence time. The stability of the tracer in the mine water must be guaranteed for the whole duration of the tracer test. Preceding the Niederschlema/Alberoda and Straßberg tracer test, 45 and 28 days of laboratory tests, respectively, were conducted. Concurrently, the tracer was sampled and visually analysed to identify alterations (Wolkersdorfer 1996; Wolkersdorfer and Hasche 2001a).

Tab. 19: Distances and mean velocities of worldwide tracer tests in underground mines. Table is given for comparisons only, details concerning geological setting and hydraulic parameters are given in the literature cited. Mean of all 40 tracer tests: $0.3\text{--}1.7\text{ m min}^{-1}$ 95% confidence interval of 40 tracer tests; ¹ result probably wrong.

Distance, km	$v_{\text{eff}},$ m min^{-1}	$v_{\text{eff}},$ m d^{-1}	reference	Location
0.015	0.0001	0.1	Wolkersdorfer unpublished	Rabenstein/Sachsen
0.17	0.0003	0.4	Wolkersdorfer unpublished	Rabenstein/Sachsen
0.2	0.001 ¹	1.4	Aljoe and Hawkins (1993)	Buckhannon/WV
0.044	0.004	5.8	Aljoe and Hawkins (1993)	Buckhannon/WV
0.093	0.004	5.8	Wolkersdorfer unpublished	Straßberg/Harz
0.093	0.006	8.6	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
0.13	0.01	14.4	Aljoe and Hawkins (1994)	Buckhannon/WV
0.077	0.01	14.4	Canty and Everett (1998)	Red Oak/Oklahoma
0.78	0.01	14.4	Wolkersdorfer (1996)	Niederschlema/Alberoda
0.238	0.014	20.2	Wolkersdorfer unpublished	Straßberg/Harz
0.048	0.02	28.8	Wolkersdorfer et al. (2002)	Brixlegg/Tirol
0.01	0.03	43.2	Wolkersdorfer et al. (2002)	Brixlegg/Tirol
1.182	0.07	101	Wolkersdorfer unpublished	Straßberg/Harz
0.35	0.1	144	Mather et al. (1969)	Aberfan/South Wales
1.181	0.1	144	Wolkersdorfer unpublished	Straßberg/Harz
3.539	0.1	144	Wolkersdorfer unpublished	Straßberg/Harz
0.077	0.12	173	Canty and Everett (1998)	Red Oak/Oklahoma
0.077	0.14	202	Canty and Everett (1998)	Red Oak/Oklahoma
0.283	0.15	216	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
1.773	0.15	216	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
0.171	0.17	245	Canty and Everett (1998)	Red Oak/Oklahoma
6.564	0.17	245	Wolkersdorfer unpublished	Straßberg/Harz
1.7	0.2	288	Parsons and Hunter (1972)	Ebbw Vale/Monmouthshire
0.229	0.23	331	Canty and Everett (1998)	Red Oak/Oklahoma
3.6	0.3	432	Aldous and Smart (1987)	Forest of Dean/Gloucestershire
4.798	0.3	432	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
0.15	0.4	576	Mather et al. (1969)	Aberfan/South Wales
0.172	0.4	576	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.216	0.5	720	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.22	0.5	720	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.2	0.6	864	Mather et al. (1969)	Aberfan/South Wales
3.18	0.7	1008	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
0.5	1.3	1872	Aldous and Smart (1987)	Forest of Dean/Gloucestershire
2.25	1.5	2160	Wolkersdorfer and Hasche (2004)	Straßberg/Harz
0.776	1.6	2304	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.736	1.8	2592	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.78	2	2880	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
2.159	5.7	8208	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
2.723	7.9	11376	Wolkersdorfer et al. (1997a)	Niederschlema/Alberoda
0.5	11.1	15984	Aldous and Smart (1987)	Forest of Dean/Gloucestershire

10.2 Tracers and their Use

10.2.1 Introduction

The following chapters are meant as an introduction to the broad field of tracer investigations and consequently, the language and the theory are kept as simple as necessary. It is meant for the novice with basic knowledge of environmental sciences or geosciences and covers mainly

tracers that can be used in mines. Therefore, some rarely used tracers, such as the promising DNA-tracers (Sabir et al. 1999), will not be discussed. Yet, all the necessary prerequisites for conducting a tracer test are given and the relevant references referred to. Many people in the mine water field want to use tracers for their studies once they hear what tracers can do. However, very often the basic knowledge about the properties of tracers and the way in which tracer tests have to be planned or conducted are not clear to everybody. A range of publications about tracers or tracer tests has been cited in earlier chapters and will also be referred to in the following sections. For those readers who want to know more details it is certainly recommended to read one of those publications (e.g. Aley and Fletcher 1976; Moser and Rauert 1980; Käß 1998; Field 2003).

10.2.2 What are Tracers?

Tracers marker substances are substances that are used to evaluate characteristic hydrogeological parameters. They are artificially injected into ground, surface or mine waters artificial tracers or are already in the water natural or environmental tracers. During a planned and to-be-authorised field test, the artificial marker substance is injected into the water and sampled at predefined points and times for analysis.

In principle, tracers can be divided into natural “naturally” or unintentionally contained in the water and artificial tracers “artificially” injected or intended to be in the water. As a rule, the former are already contained in the water to be examined whereas the second are injected into the water during the field test. Environmental isotopes, environmental chemicals, organisms, or physical qualities of the water are regarded as natural/environmental tracers. They are described as “natural” for they normally are not artificially injected into the water. Artificial tracers can roughly be divided into water insoluble e.g. dyes, phages, micro particles, spores and water soluble tracers e.g. pigments, salts, radioactive substances.

Considering tritium as an example, the difference between artificial and natural tracers can be clarified. Tritium arose from atmospheric nuclear weapon tests and reached ground water naturally in precipitation. Here tritium is an *environmental* isotope and therefore is regarded as a *natural* tracer. However, tritium can artificially be injected into the ground water within the scope of a tracer test. Tritium is then a radioactive tracer and therefore regarded as *artificial* tracer (Geyh and Schleicher 1990).

An increasing sensibility of the population and the authorities during the last decades led to increasing problems in conducting tracer tests. For acceptance reasons the words “tracer” or “marking substance” should therefore be used instead of the proper tracer names.

10.2.3 What are Tracers used for?

Tracers are used to characterise flow paths in many different disciplines like geology, hydrology, chemistry, physics, water-supply and distribution, hydraulic engineering, limnology, or biology. The list of applications is long and almost every conceivable case can be found in the literature. Originally, tracers were used to pursue underground watercourses. First reports to this reach back until roman times. Today, tracers are also used in flooded mines, at site investigations of radioactive repositories, surface waters, technical processes, or in living beings.

It is the prime aim of tracer tests to trace back flow paths, flow velocities, mass transfer, and hydrogeological parameters. By evaluating tracer tests the following information and characteristic quantities can be determined:

- hydraulic connections flow pathways
- ground water flow directions

- ground water velocity average interstitial velocity and mean residence time
- hydrodynamic dispersion
- flow rates, mass transfer
- variability of hydrogeological properties
- transport behaviour of dissolved or particulate water constituents as well as their interactions

10.2.4 Requirements for Tracers

A tracer is described as a conservative or ideal tracer provided that it behaves in the subsurface medium in the following way:

- inert behaviour not prone to chemical reactions with immobile solids
- unchanging physical qualities
- transported at the same velocity as the flowing water.

In addition, as many of the following criteria as possible should be accomplished:

- no natural occurrence of the tracers in the transport medium
- good analytical detectability even at great dilution
- water-soluble or dispersible
- robust chemically stable
- no interactions between tracer and medium
- economic reasonably priced in acquisition, handling, analysis
- physiologically safe not toxic
- without a lasting effect on the ground water composition

No tracer satisfies all the aforementioned conditions equally well. Rather, the most suitable tracer has to be found according to the question to be solved. From the view of the transport behaviour tritium would fulfil most of the ideas of a conservative tracer; yet, for reasons of cost effectiveness and analytical procedures, fluorescent dyes are the most common tracers used.

10.2.5 Types of Tracers

10.2.5.1 “Natural” or “Environmental” Tracers

Environmental Isotopes Stable Isotopes

Environmental isotopes stable Isotopes are nuclides of elements that occur in waters without having been injected on purpose to conduct a tracer test. They are therefore natural tracers. Usually, their concentrations are low and the analysis and the interpretation subsequently turns out to be difficult. In distinction to artificial tracers they offer the advantage of allowing interpretation of bigger areas and larger time scales. Besides geohydraulic parameters, environmental isotopes allow the age and temperature evolution of the waters to be traced (e.g. Lippmann et al. 2003). Fractionation of certain stable isotopes also allows the identification of certain biogeochemical processes (e.g. bacterial sulphate reduction; Fauville et al. 2004).

Environmental Chemicals

Environmental chemicals are substances which enter the water cycle accidentally or by slow contamination. They might be the same substances as in the case of artificial tracer tests, but because they reach the water cycle unintentionally they are categorised as “natural” tracers.

Sodium fluorescein or pyranine are two commonly used dyes in cosmetic articles, though a test with several different cleaning detergents showed that at least sodium fluorescein seems to be substituted by other chemicals nowadays. Seeping boron acid from house waste disposal sites can be considered a good tracer (Neal et al. 1998). Increased boron concentrations in water analyses usually point to nearby house waste disposal sites or, in certain areas, to naturally elevated boron concentrations e.g. (Col and Col 2003). Chlorides are found in the vicinity of streets or airfields, where salt is used for de-icing, or downstream of potash pits.

Very good environmental tracers are sulphur hexafluoride SF₆ or chlorofluorocarbons CFCs, that are abundant in nearly every water. They can also be used as artificial tracers.

Metals and semi-metals e.g. Cd, U, As, Cu, Sb can be considered as natural tracers downstream of mine water drainage tunnels. One of many examples is the river Erft Eifel/Germany which receives mine water from the abandoned Mechernich mine (Schalich et al. 1986) or in many other mining areas world wide. Such natural tracers are commonly used in the exploration for ore deposits (Grimes et al. 1995).

One of the most expensive and outstanding “natural” tracer tests happened on August 12th 1901 (Anonymous 1901), when lightning hit the Pernod factory at Pontarlier and about 1 million litres of absinth were flushed down the river Doubs. Two days later, André Berthélot noticed the Absinth at the Loue Spring, 50 km north-north-east of Pontarlier.

Organisms

Organisms are rather exotic tracers, although both higher and lower organisms were used so far. Common natural tracers are bacteria which can be used to trace the environmental influence of excrement. Eels or ducks also were used successfully as tracers, in which examples they should be listed as “artificial” tracers, because the organisms were marked (Käß 1998).

Physical Influences

A physical influence which plays a major role in ground water investigations or the judgement of ground water contaminations is the change of the water temperature. By new fibre optic measuring methods it is possible to receive a high spatial resolution of temperature differences. Schreck et al. (1998, 1999), and Stoddart et al. (2005) used this method to monitor temperature variations in mine dump sites. Berk (1987) was able to show that downstream of a coal mine waste dump, the ground water temperature increased by 3—6 K, consistent with the exothermic pyrite weathering reaction.

Changes in electrical conductivity were used to trace the sources of highly mineralised waters emanating from potassium mine waste rock piles in the German Rhine river valley.

10.2.5.2 Artificial Tracers

Water-soluble Tracers

Dyes

Dyes are the most frequently used artificial tracers, amongst which fluorescent dyes are today the first choice when carrying out tracer tests. The dyes used are largely non-poisonous apart

from the rhodamines and can easily be detected by fluorometers, fluorescence chromatography, or with the support of HPLC high pressure liquid chromatography. Transportable fluorometers and MORES micro optical remission sensor micro sensors using different techniques, including fibre optical methods, are now available and allow continuous monitoring of the fluorescence intensity. Non fluorescent dyes are of minor significance since they are often sorptive and their analytical verification is less precise compared to that of fluorescent dyes.

Both the easy analytical verification and relatively cheap have considerably contributed to the distribution of fluorescent dyes as tracers in hydrogeological investigations. They are also used to dye drift particles e.g. spores, microspheres to increase their detectability.

Salts

The salt most frequently employed for tracer tests is rock salt NaCl. Besides this, potassium chloride, lithium chloride and bromide salts are used. Though most salts are comparatively cheap and their detectability is reasonably good, large quantities of up to several tons of salt may have to be used for a mine water tracer test, and therefore cost might hinder their use as tracer in mine water tracer tests. For a tracer test at the abandoned Straßberg/Germany fluor-spar mine, 20,000 L of saturated NaCl brine were injected (Fig. 77).



Fig. 77: Sampling of saturated NaCl brine for the Straßberg/Harz/Germany tracer test.

Radioactive and Neutron Activatable Tracers

Radioactive tracers can easily be detected with field or laboratory equipment. However, at present it is difficult to get permits for conducting tracer tests with radioactive tracers although they show a variety of positive qualities with which geo-hydraulic parameters are determinable. Neutron activatable tracers are non-radioactive elements that can be used in low concentrations and are neutron activated after they were sampled in the field (Jester and Raupach 1987). Commonly used elements are ^{80}Br , ^{116}In , ^{165}Dy , ^{152}Eu , ^{140}La , ^{56}Mn . Because their detection is very sensitive they can even be used in very low concentrations.

Except that tests with radioactive substances require radiation protection regulations be considered, their handling is not different from that of other tracers used in hydrogeology. The advantage of neutron activatable tracers compared to radioactive tracers is that they are not radioactive before neutron activation.

Water insoluble Tracers

Solid Tracers

For one of the first tracer tests ever reported a solid substance was used: chaff, to examine the springs of the river Jordan in the year 10 AD. However, solid tracers are less suitable for conducting quantitative tracer tests. Should only a qualitative statement be necessary, it may indeed be useful to use solid tracers. A semi-quantitative method has been developed for mine water tracer tests. This LydiA-method, which uses filter nets and auto-samplers, is described elsewhere in this text book (Fig. 78).



Fig. 78: *Lycopodium* Apparatus (LydiA) for injecting solid tracers into flooded underground mines. Diameter 120 mm.

Phages

Phages are viruses with sizes between 20 and 300 nm. They are inactive when outside of bacteria and consequently, they do not have any active metabolism which means that their number cannot increase during the tracer test. Their qualities as tracers are very good although the analytical proof can only take place in special laboratories. Due to the number of different phages known, they are suitable for tracer tests with multiple injection sites. As in the case of all solid tracers, the number of particles found after the sample preparation is counted and used for the interpretation.

Bacteria

Because of hygienic issues concerning drinking water protection zones, bacteria play an important role in such tracer tests e.g. 50-day-time-of-travel-line. Numerous bacterial genera have proved suitable for hydrogeological tracer tests. They can be used in ground water, for column tests, surface waters, and in coastal waters (e.g. Harvey et al. 1989, 1993; Vega et al. 2003).

Micro Particles Microspheres

Micro particles are microscopically small polystyrene beads of 0.6—100 μm diameter which are coloured with fluorescent dyes Fig. 79. In hydrogeological tracer tests, their surface should be neutrally charged to prevent sorption. For special investigations microspheres with positive and negative surface charges are on sale as well. Originally, polystyrene beads were developed for physiological investigations. However, several investigations confirmed that they can also be used as hydrogeological tracers in which the good experiences of the numerous physiological investigations can be used with advantage.

Recently, micro particles containing a rare earth element cocktail were developed. Those tracers can be utilised independently of dyes and in cases where dyes might be destroyed under certain physico-chemical conditions rare earth elements are not subject to this destruction. Microspheres were used in several mine water tracer tests with generally positive results e.g. Straßberg/Harz Mountains, Königshütte/Harz Mountains, Ehrenfriedersdorf/Erzgebirge, Georgi Unterbau/Tyrol, Schlema/Saxony [the latter: Werner Käß, pers. comm.].



Fig. 79: Selection of microspheres used in the Straßberg/Harz 2003 tracer test.

Spores

The spore tracer method is one of the most important methods in karst examinations. Club moss *Lycopodium clavatum* spores with diameters about 50 μm are used for that purpose. They can well be coloured with different fluorescent and non-fluorescent dyes so that in a test several injection places might be examined. The evaluation is laborious since the spores found must be counted microscopically. Besides the examination of karst areas, spores also were used successfully in mine water tracer tests to investigate hydraulic connections and parameters as well as effective velocities underground e.g. Niederschlema/Alberoda/Erzgebirge, Straßberg/Harz Mountains.

10.3 Preparation

10.3.1 *Hydrogeological examination*

The aim and purpose of a tracer test must be known precisely. Before a tracer test is carried out, a preliminary hydrogeological investigation is imperative and all available geological and hydrogeological data of the investigation area have to be consulted to develop a first conceptual model. During the data collection and evaluation special attention has to be given to heterogeneities of the aquifer and the mine lay out. For that purpose, geological maps, existing boreholes, or pump data must be evaluated, because the heterogeneities have a decisive influence on the selection of the injection and observation sites. There are several hydrogeological text books available (e.g. Fetter 1988; Jordan and Weder 1995; Domenico and Schwartz 1998; Brassington 1999; Deming 2002; Hiscock 2005; Hölting and Coldewey 2005). It is beyond the goals of this book to repeat what they have already written about good hydrogeological practise.

Occasionally, sufficient information to establish a conceptual model of the investigation area or to plan the order of potential injection at observation sites does not exist. In those cases further geological or hydrogeological investigations are essential. If necessary, pumping tests or measuring the ground water levels will provide further information (Doherty 1990). Crucial to all tracer tests are one or several field inspections to get an impression of the examination area and the accessibility of the injection and sampling sites. If necessary, the client or the mine operator has to modify sampling and injection sites to fulfil the needs of a proper tracer test.

The following data, or at least an idea of their order of magnitude, should be available for the planning of a tracer test:

- ground water contour maps, and interfered ground water flow directions
- ground water flow velocity
- dilution
- lithology and hydrogeochemical parameters
- water use

Inadequately prepared tracer tests are the most frequent reason for lacking results and preliminary investigations therefore are an essential part of a tracer test – even if the area of investigation is only known little.

Based on the results of the laboratory tests, the cost of the tracer material, and the analytical requirements for the tracer, the most suitable tracer has to be chosen. It shall be mentioned here that in addition to the above described selection of the appropriate tracer, a detailed hydrogeological investigation should clarify the most suitable injection and sampling points “conceptual model of test site”. Many tracer tests, even if a suitable tracer has been chosen, were unsuccessful due to a lack of knowledge about the hydrogeological situation leading to poor selection of injection and sampling sites.

10.3.2 *Selection of Injection and Sampling Sites*

A decisive criterion for whether a tracer test turns out to be successful or not is the correct selection of the injection and sampling sites. The injection sites at least one, of course must be distributed thus that the tracer flows through the examination area and that the purposes of the investigation are reached within that area. Although this demand seems trivial at first, unsuccessful tracer tests show that this prerequisite is not always met!

Since tracer tests often bring new and surprising knowledge, it makes sense to choose one or two more sampling sites rather than to fail in finding the tracer injected. As a rule, tracer tests are cost-intensive, and a failed tracer test due to not enough sampling sites is dissatisfying for both contractors and customers! Underground and surface watersheds frequently do not correspond and this fact has to be considered when selecting the sampling sites. Provided that there are uncertainties, what the correct location of the sampling sites concerns, a “small” numeric model with assessed hydrogeological relevant parameters can be helpful.

One of the most unpleasant situations for both contractors and customers is, if no tracer is found *at all*. Therefore, the tracer should be found *at least at one* sampling site even if the working hypothesis “conceptual model” should not be supported by that sampling site e.g. a bore hole or shaft directly downstream from the injection site which in any circumstance must be included in the overall flow. Such a sampling site can also be at a location which according to the pre-considerations might not be possible at all. Tracer tests during which no tracer is found cannot be evaluated: it means complete sorption or degradation of the tracers, too low transport velocity, or no hydraulic connection between injection and sampling sites. At least the last possibility should be ruled out by a sufficiently large number of sampling sites!

Besides the technical criteria, the spatial position to each other and the accessibility plays an important role for the choice of the injection and sampling sites. Sites which would be necessary from a technical point of view but can only be reached with disproportionately big effort should remain unconsidered when making the choice. Furthermore, it is commonly known that the client wants to have more injection and sampling sites the week before the test starts!

10.3.3 Which Tracers shall be used?

Just as important as the selection of the injection and sampling sites is the choice of the correct tracers. This choice depends on many factors which are mainly determined by the terms of reference. Besides this, the hydrogeological and geochemical conditions, the required sensitivity, available analysis methods, and restrictions of use e.g. permit of the authority, water protection areas play a role for the tracer selection. Occasionally, it is required to use several tracers in a multi-tracer approach. However, compromises always have to be made to solve all upcoming problems as optimally as possible e.g. costs, transport, permits, analytical procedures. Käß (1998, 2004) lists the following questions which have to be considered when selecting the optimal tracer:

- sorptivity and ion exchange
- pore distances
- chemical and physical water parameters
- influence of light
- water use
- analytical criteria
- effort of obtaining, transporting, injection, sampling, analytics
- injection sites
- sampling sites
- long-term behaviour of the tracers

Although an optimal tracer should behave conservatively, depending on the terms of reference, it might be feasible to use a non conservative tracer. Such a case might be an investigation about the transport behaviour of characteristic substances. Tab. 18 gives a selection of tracers that have already positively been used in mine water investigations.

10.3.4 How much Tracer must be used?

In the literature numerous equations are specified to calculate the optimal tracer amount independently of the used tracer and the investigation area. In detail some factors might be contradictory, and therefore, the choice of the correct tracer amount might be tricky from time to time. All equations only give preliminary values which have to be seen as starting points for site specific considerations.

A comparison of two equations for otherwise the same boundary conditions shall clarify this statement:

$$m = L \times k \times B \quad 69$$

with m : tracer amount, kg
 L : Distance to the most important sampling point, km
 k : Coefficient for the tracer substance, if not sodium fluorescein 0.1—20,000
 B : Factor for general set up of tracer test 0.1—10
 $L = 2.4$ km; $k = 1$; $B = 3$ yields $m \approx 7$ kg for sodium fluorescein (Käb 2004)

$$m = L \times K \quad 70$$

with m : tracer amount, kg
 L : Distance to the most important sampling point, km
 K : Coefficient for the aquifer 0.2—2
 $L = 2.4$ km; $K = 1.5$ yields $m \approx 3$ kg for sodium fluorescein (UNESCO 1; Käb 1998)

k , B , and K are tabulated values that can be found in Käb (1998). They will not be further discussed here, because the two above equations shall just exemplify how tricky the calculation of the correct tracer amount might be.

A list of values which are differing in the range of three magnitudes is given in Käb (1998) and Field (2003). In general the empirical values deduced from the experiences of the working group play an important role in the calculation of the necessary tracer amount. Meanwhile Käb (2004) came to the conclusion that expert knowledge is more important than a list of equations and therefore he only refers to equation 69. With respect to this shortage of reliable equations to calculate the necessary tracer amount, Field (2003) published the computer code EHTD (Efficient Hydrologic Tracer-Test Design) which needs 12—13 aquifer and tracer parameters and can then be used to calculate the necessary tracer amounts for various hydrogeologically relevant scenarios. Over and above, EHTD calculates optimised tracer sampling intervals and predicts the breakthrough curve. Yet, a mine water scenario is not included in the code so far.

Therefore, in mine water tracer tests those empirical data can be backed by the following calculation which gives a good approximation of the solid tracers needed for a tracer test in a quasi-closed system (the formula given in Wolkersdorfer et al. (1997a, 1997b) is faulty, as r_r is given in % instead of the rate):

$$m = \frac{1}{55,000} \times \frac{V_m}{r_r \times q} \times f \quad (71)$$

with m : total mass/volume of spores/microspheres, g
 V_m : volume of enclosed fluid in the mine, m³
 r_r : recovery rate of spores or microspheres, – (not more than 0.02—0.1 might be expected)
 q : pumping capacity, L min⁻¹

f : factor (spores: 1; 15 μm microspheres with a concentration of $1 \cdot 10^6/\text{mL}$: 0.25), $\text{g L m}^{-3} \text{ min}^{-1}$

Equation 71 considers water of volume V_m in a mine or another enclosure without a discharge into a receiving water course. During the tracer test the water is pumped out of the mine at a pumping capacity q which must be relatively small compared to the total water volume. The recovery rate for spores or microspheres is based on experience and usually does not exceed 2—10 %.

Yet, we also used other equations to verify the necessary tracer amount. So far we had good experiences with the following procedure: given the detection limit of the tracer is $c_{\min}$, then the discharging mine water must have at least this tracer concentration. The total volume of the water in the mine is V_m and the flow rate at the discharge point(s) is Q with a total potential duration of the tracer test t . Furthermore, we assume a complete homogeneous mixing of the tracer with the mine water. At least this assumption is very optimistic but it proved to be a good starting point. Furthermore, the concentration factor f_c , depending on the sampling equipment must be known. If the tracer is directly measured in the discharging water, this factor is 1, if the water is filtered through a filter (e.g. MeFisTo System) then it depends on the ratio of the water pumped through the filter systems q to the discharged water. The expected recovery rate shall be r_r and the sampling interval t_s .

Then the following physically based equation can be used to estimate the necessary tracer amount:

$$m = \frac{(V_m + Q \times t) \times c_{\min}}{r_r \times f_c} \quad (72)$$

$$f_c = \begin{cases} 1 & : \text{direct tracer measurement in discharging water} \\ q \times t_s \times r_r & : \text{aliquot pumping of discharged water} \end{cases} \quad (73)$$

So far, this conservative approximation was successful in all the tracer test we conducted in fully flooded mines.

10.3.5 Tracer Test Plan

When applying for a permit from the responsible water authorities a tracer test plan should be made available. This plan should list in detail the injection and sampling sites at which temporal distributions shall be sampled and by whom. At the same time, the test plan has to define to which end the tracer test is carried out and it shall predict which results will be obtained. Normally, the test plan also contains details on the costs of the complete tracer test including its evaluation. The success of an application and the tracer test itself depends fundamentally on whether the test plan is complete or not.

During sampling the local or national recommendations for sampling ground water monitoring wells or surface waters have to be considered. Because national and international standards are changing regularly, even locally, the researcher planning to conduct a tracer test is well advised to get into contact with the local authorities which are also responsible for permitting the tracer test.

According to Käß (1998) and other relevant guide lines (Schwarz et al. 2002) a tracer test plan should at least contain the following details:

- district and place name
- description of the injection sites
- person in charge of the tracer test with phone numbers

- customer with full address
- experimental purpose
- injection time (start of tracer test)
- type of tracer
- water quantities to flush the injection site and the tracer
- listing of the sampling sites and sampling intervals

Furthermore, the following details are meaningful:

- work schedule for continuous sampling
- necessary measurements on site
- work schedule for analytical laboratory
- emergency plan

10.4 Approval Procedure

10.4.1 Legal Regulations

Since the legal regulations for permitting tracer tests differ from country to country, the local authorities must be contacted before a tracer test is designed. In some countries tracer tests need a formal, sometimes lengthy approval procedure, while in other countries a telephone call with the water or mining authorities might be sufficient. A typical procedure for applying a permit for a tracer test will be described here for Germany according to Pascaly (1997).

According to § 2 (1) of the Federal Water Budget Law (“Wasserhaushaltsgesetz” WHG) the use of waters requires an official permission and the injection of a substance represents a use in the sense of § 3 (1) № 5 WHG. § 3 (2) regulates that measures causing harmful and not only insignificant changes of the physical, chemical, or biological water parameters, be they permanent or temporal, are also a water use.

The official, revocable permission is granted by the authority in accordance with § 7 (1) WHG. This permission allows using waters (including ground water) for a certain purpose in a specific way predetermined by its manner and degree. To this, in accordance with § 4 (1), the permission can be made under a condition and regulations. Conditions are also permitted to prevent or compensate for adverse effects to others.

All activities of the responsible authorities are under the principle of § 1 WHG which demands that the waters have to be managed for the welfare of the general public and in accordance with that to allow for the welfare of individuals. Avoidable impairments of ecological function must be avoided. Additionally, in water conservation areas, the water protection ordinance has to be considered.

The responsible authorities are determined by legal state regulations. In Saxony, for example, the State Environment Agencies (Staatliche Umweltfachämter) or the lower water authority are responsible for granting the permission. The authority checks the tracer test plan on consistency with the legal specification that is the avoidance of harmful changes and impairments, as well as on the compliance with the regulations or – if necessary – the existing water protection ordinance. It does not only depend on toxicological effects to humans and the environment but also on other possible chemical (e.g. solution contents, taste), physical (e.g. colour, temperature), or biological changes in the waters.

10.4.2 Necessary Details within the Application

As a basis for its permission the water authority needs an application with a detailed description of the tracer test plan. It is advisable to describe the reasons for the test, the tests boundary conditions and the type of tracers going to be used including their environmental compatibility (Arbeitskreis "Human- und ökotoxikologische Bewertung von Markierungsmitteln in Gewässern" 1997; Behrens et al. 2001).

To understand the procedures of the tracer test and to give the permitting authorities a means to assess possible environmental risks, the following details should be presented: amount of tracer (including its potential dilution), volume of flushing water, amount of water taken (for flushing reasons or analyses), duration of the test and potential degradation rates of the tracer. Because of the many details required by the authorities it is recommended to involve the authority as early as possible. Usually, they will then ask for more detailed explanations of the tracer test plan in the context of a meeting or written statements. Furthermore, the responsible special authority is frequently asked to give recommendations or statements to the permitting authority for proving the necessity of the tracer tests.

10.5 Execution

10.5.1 Injection Time and Injection Type

Time and type of the tracer injection arise from the results of the preliminary examination and the tracer chosen. During the tracer injection contamination of the samples or persons in the field and later in the laboratory must be prevented. Therefore, persons who are entrusted with the injection shall not take the samples or carry out the later analyses in the laboratory either. Provided that this is not avoidable, care has to be taken that the clothes are changed and the body as well as the hair are carefully cleaned. Furthermore, when several sampling points are involved in the test, all cross-contamination which might come from probes for temperature or electrical conductivity measurements must be excluded by cleaning the probes carefully with distilled water. Unfortunately, the cover photograph of the Field (2003) publication does not show good practice in tracer injection, because the equipment of the person injecting the tracer will be polluted with the tracer and therefore, carrying off of the Na-fluorescein tracer cannot be fully excluded.

Independently of the tracer chosen, all containers carrying the tracer have to be flushed out repeatedly to make sure that the tracer is fully injected into the mine- or ground water. In the case of a solid tracer this approach above all ensures that the tracer reaches the water body to be examined completely. To avoid contamination in the field, it is recommended that fluorescent tracers shall be already diluted in the laboratory. If necessary, several 10s to 100s litres of water must be poured into the injection point, especially when the injection site is dry.

Provided that tests take place in the unsaturated zone, it has to be considered that the tracer must first reach the ground water before it can be transported by the water. Without additional water injections the tracer would remain in the pore space of the unsaturated zone. Of course, this is not valid if the tracer test is carried out to examine the unsaturated zone itself. When evaluating tracer tests it is assumed that the injection has the form of a Dirac function:

$$\delta(t) = \begin{cases} 0, & t \neq 0 \\ \infty, & t = 0 \end{cases} \quad (74)$$

$$\int_{t_1}^{t_2} dt \delta(t) = 1$$

Therefore, the tracer must be completely injected as quick as possible. Older recommendations to inject the tracer slowly cannot be advisable, because such a technique would result in a “smudging” of the tracer cloud and could feign a non existing dispersion and aggravate the evaluation of the tracer test considerably.

Yet, Kimball et al. (2002) conducted tracer tests in mine water impacted surface streams in Colorado and Utah where they injected soluble salt tracers in a constant rate and calculated the dilution of the tracer. Thus they were able to locate mine water inflows, flows below the stream bed, and into fractures.

10.5.2 Sampling

As in the case of hydrogeological investigations, correct sampling is essential when conducting tracer tests. Good instructions are given by the Saxonian Agency for Environment and Geology and other sources (Ficklin et al. 1999; Sächsisches Landesamt für Umwelt und Geologie 1997). Therefore, the following instructions are just the most prominent ones which should be followed when taking samples for tracer tests. These recommendations, mainly, are the result of the observations experienced when working together with students, mining-engineers, and personnel not involved in regular sampling for such tracer tests.

Already when labelling the sampling containers and the sampling protocols exact working and recording excludes later inconsistencies. All details of the sample have to be written down in the protocol as well as on the sample containers (usually bottles). Particularly the knowledge of irregularities (e.g. extreme water-levels, damaged glasses, not sufficient amount of sample, tracer spilled) can be of great importance at the later evaluation. Since mishaps can happen to everybody, the record keeping of such occurrences is very important.

Another issue are the containers used for water samples. They must be appropriately selected for each parameter that shall be measured in the laboratory. PVC or glass bottles are a good selection for the main ions and the trace elements. TOC samples must be taken in glass bottles and TIC samples in PVC bottles sealed with aluminium foil. All containers have to be washed with hot water and phosphate free lab grade soap, rinsed with de-ionised water thereafter and washed again with dilute acid before they are used again (e.g. 10% nitric acid). After acid washing, the containers have to be finally rinsed with de-ionised water. Containers prepared in that way can directly be used for water sampling. All other containers (e.g. new ones) have to be rinsed out three times with the water to be sampled before they will be filled with the sample.

As indicated above, the sampling has to be carried out according to the tracer test plan. In agreement with the customer or the authorities the test plan can be changed, provided that unforeseen events require this (e.g. appearance of the tracers in till now unsampled springs, shafts, or drippers; no more tracer analytically detectable; breakthrough not complete).

A well drawn up sampling plan considers the travel time between the withdrawal points as well as possible difficulties which might appear situational (e.g. break down of equipment, weather conditions). As a rule, the sampling intervals are short at the beginning and increase during the tracer test. Without knowledge of the hydrogeological conditions no rule of thumb can be given. However, it is better to take too many than too few samples. Provided that a numeric model was carried out before the tracer test, the sampling intervals can be deduced from the results of the modelling. Furthermore, the software code EHTD provides tools to calculate an optimised sampling scheme (Field 2003). Based on the predicted tracer mass as well as the aquifer's hydraulic and geometric parameters, the initial sample collection time and the sampling intervals are approximated (Fig. 80).

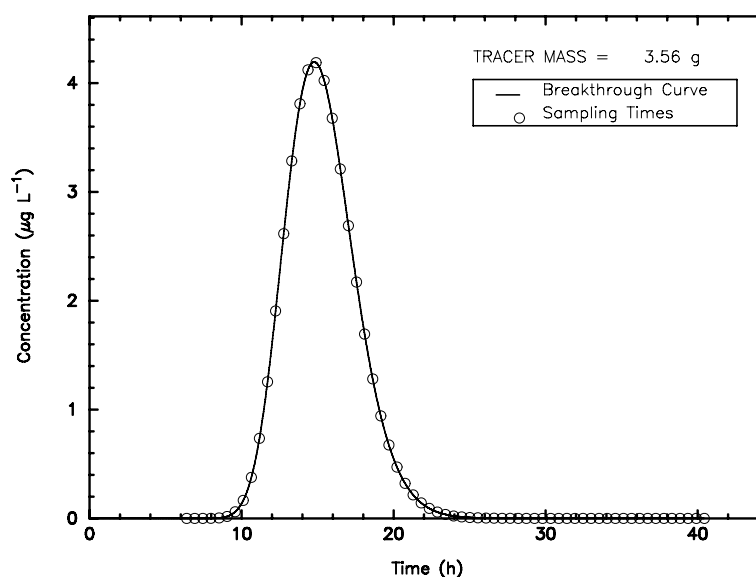


Fig. 80: Sample output of the Efficient Hydrologic Tracer-Test Design (EHTD) software code for a tracer test at the Dyers Spring (data from Field 2003). The circles indicate the optimised sampling intervals.

All staff should be instructed to record the actual sampling times and not those from the sampling plan. Many causes can arise that make an earlier or later, from the original plan divergent sampling time necessary. A “well-meaning” correction is of little help in those cases!

The necessary size of the sampling containers must be discussed with the analytical laboratory. In the case of fluorescent tracers brown glass bottles are recommended since fluorescent dyes decompose by light influence (Käb 1964). For tracer tests with bacteria, phages, spores, or microspheres a careful selection of the appropriate laboratory is imperative, as those tracers are not easily detectable.

As in the case of water samples for the chemical analysis the tracer samples have to be transported and stored carefully packed and cooled. Since the tracers can change in the course of the time, samples must usually be brought back to the laboratory and analysed as quickly as possible.

10.5.3 Tracer Analyses

For good results a laboratory should be chosen of whose quality the contractor could convince himself before the tracer test. The cheapest laboratory will likely not have the best results at the same time (and vice-versa)! If necessary, some selected laboratories should be compared with each other by a ring test to find out which one has the best reproducibility. Ring tests are conducted by sending a sample of known composition or tracer concentration to different laboratories. The results of the different laboratories are then compared with the expected result and the best laboratory is chosen on the findings of that ring test.

It is important that a laboratory book with numbered pages is available and a precise protocol about every test is taken. Experience shows that reliable laboratories have laboratory books that are in good order and protocols that are written down properly! No big confidence should be placed into laboratories with “piece of paper economy” and pieces of memo blocks sticking on the sample containers!

Concerning the analytical methods for the different tracers the appropriate technical literature should be consulted. Even an abstract of potential tracer analyses methods would greatly exceed the scope of this publication. A good reference is Käb (1998).

10.5.4 Documentation and Presentation

Basis of the evaluation of a tracer test is a reliable documentation of the complete test. Already small tracer tests can result in several hundreds of datasets which must expressively be recorded and presented. A professional, electronic documentation in a database or in case of smaller tests in a spreadsheet analysis represents a basic aid for the following work steps. Provided that several institutions collaborate in the tracer test, a common database structure is recommendable. All researchers involved must be urged to follow the database structure provided by the leading team.

Lists or tables are regarded as the simplest data presentation, where the place, the time, and obviously the tracer amount can be documented. Cartesian coordinates (usually in the system UTM WGS84, because that data are provided by modern satellite navigation systems as GPS, GLONASS, or GALILEO) have to be included when several injection and sampling places are involved. Particularly advantageous is the connection of the data with GIS-applications, where differences between the different injection and sampling points are immediately obvious. All modern GIS-applications permit a coupling of existing databases with graphical and topographical information.

To get a first impression of the tracer test, the data can be presented in time-concentration-diagrams (“breakthrough curve”). With few work steps first information is already obvious as Fig. 81 illustrates. The most important, characteristic quantities of the tracer test should also be shown there.

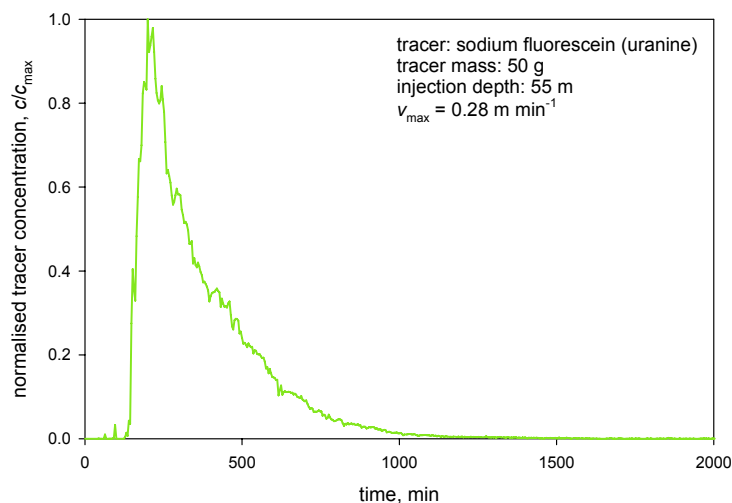


Fig. 81: Normalised breakthrough curve of a characteristic tracer test in the shaft of a flooded underground mine.

10.6 Evaluation and Characteristics of the Tracer Test

10.6.1 Porous Aquifer

Because the basics for nearly all tracer tests go back to the theory about porous aquifers, the governing transport equations for the evaluation of porous aquifers will be discussed in detail here. The same equation can also be used for the evaluation of mine aquifers.

Tracer tests in porous aquifers normally can be interpreted easily with the existing tools since the experiences there are already very extensive. Basis of the flow in the porous aquifer is Darcy's law (one-dimensional; stationary flow; homogeneous, infinitely extensive aquifer):

$$\frac{Q}{A} = v_f = k_f \times \frac{dh}{dx} \quad (75)$$

with Q : flow, $\text{m}^3 \text{s}^{-1}$
 A : sectioned area through which flow passes, m^2
 v_f : Darcy velocity, m s^{-1}
 k_f : hydraulic permeability, m s^{-1}
 $\frac{dh}{dx} = i$: hydraulic gradient, –

and

$$v_a = k_f \times \frac{i}{n_e} \quad (76)$$

with v_a : flow velocity, m s^{-1}
 n_e : effective porosity, –

Due to the microstructure of the aquifer and its inhomogeneities a deviation from the ideal flow path is regularly observed. This deviation is expressed by the hydrodynamic dispersion:

$$D_h = D_m + D_e = \alpha \times v_a + D_e \quad (77)$$

with D_h : hydrodynamic dispersion, $\text{m}^2 \text{s}^{-1}$
 D_m : hydromechanical dispersion, $\text{m}^2 \text{s}^{-1}$
 D_e : effective diffusion coefficient, $\text{m}^2 \text{s}^{-1}$
 α : Dispersivity (dispersion length), m
 v_a : flow velocity, m s^{-1}

Consequently, the mass transport in a homogeneous, three-dimensional aquifer is described by a combination of advection and dispersion:

$$\frac{\partial c}{\partial t} = D_L \frac{\partial^2 c}{\partial x^2} + D_T \frac{\partial^2 c}{\partial y^2} + D_V \frac{\partial^2 c}{\partial z^2} - v_a \frac{\partial c}{\partial x} \quad (78)$$

with c : concentration, g L^{-1}
 t : time, s
 x, y, z : distance in the x, y , or z direction, m
 D_L : longitudinal dispersion coefficient, $\text{m}^2 \text{s}^{-1}$
 D_T : transversal dispersion coefficient, $\text{m}^2 \text{s}^{-1}$
 D_V : vertical dispersion coefficient, $\text{m}^2 \text{s}^{-1}$
 v_a : flow velocity, m s^{-1}

As the tracer distribution usually approximates a Gauss normal distribution (Fig. 81) and as the tracer injection is in the form of a Dirac function, the following equation describes the two-dimensional case of the tracer distribution at position x, y and time t :

$$c_{x,y,t} = c_{\max,t} \times \exp \left(-\frac{(x - v_a \times t)^2}{4 \times D_L \times t} - \frac{y^2}{4 \times D_T \times t} \right) \quad (79)$$

In reality, breakthrough curves seldomly conform exactly to the Gauss distribution, as can be seen in the example of the Rabenstein/Germany tracer test (section 10.5.1). By using software

to interpret the results of tracer tests simulated breakthrough curves can be obtained and from those the appropriate parameters can be deduced by parameter approximation.

10.6.2 Fractured Aquifer

The researcher must know that there are completely different flow conditions effective in porous and fractured aquifers and that ground water flow behaves entirely differently in the two cases. Although the evaluation of breakthrough curves of those two aquifers is similar, breakthrough curves of fractured aquifers are different from those of porous aquifers. One of the most striking differences consists in the recovery rate of the tracers. While at porous aquifers often more than 50% of the tracers are recovered, the values at fractured aquifers usually are less than 50%.

Due to the heterogeneity and the anisotropy of fractured aquifers, preferential flow directions are characteristic for fractured aquifers. Consequently, the assumption that there would be a homogeneous, virtually porous system, is only partially correct. However, the breakthrough curves usually do not show a Gauss distribution and therefore confirm that this assumption only has a low degree of truth.

Fractures represent preferential flow paths for water, pollutants, and tracers within a relatively homogeneous rock matrix. In case that there is no fracture connection between the injection and sampling sites, consequently no tracer is recovered at the sampling site. To model the flow in fractured aquifers, different model approaches exist (e.g. artificial or random fracture distribution by fracture generators, e.g. Wendland and Himmelsbach 2002). Whichever approach is used, the researcher must know that it is practically impossible to describe fractured aquifers adequately exactly. This always has to be considered at the evaluation of tracer tests in fractured aquifers.

In a system with several parallel fractures, variable fracture widths and roughnesses cause a special influence, as this might cause that only certain, few fractures are preferred by the water. Normally, this effect, called “channelling”, results in a breakthrough curve with multiple peaks of which each represents a preferential flow path (Fig. 82).

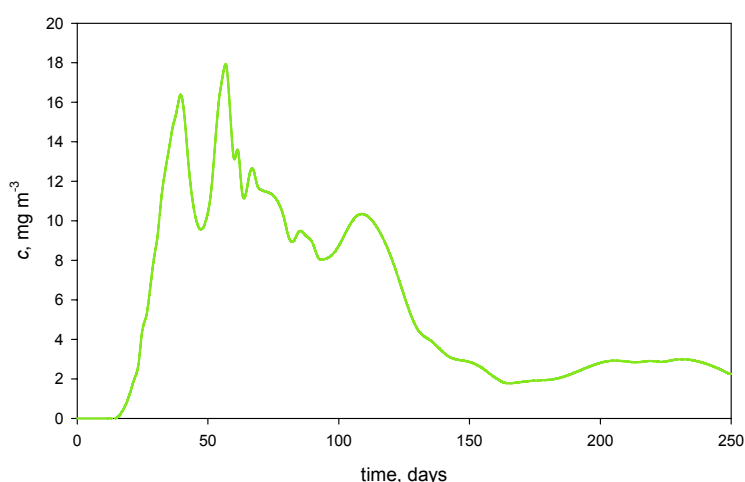


Fig. 82: Breakthrough curve of a tracer test with preferential flow paths (data modified and simplified after Käß 2004).

Another effect in fractured aquifers arises when the matrix porosity shows a significant value. In such cases, tracer from the matrix is slowly transported into the fractures, which usually show a good hydraulic conductivity. Additionally, the breakthrough curves are tailing out,

which indicates that tracer from the matrix still interacts with the water in the fractures (dual or double porosity aquifers).

Many tracer tests in fractured rocks show a causal connection between the distance of the injection and the sampling sites and the flow velocity: the flow velocity frequently increases with an increasing distance. There are similar observations in the case of distance and the hydraulic dispersion.

Often, tracer tests in fractured aquifers turn out to be more difficult than those in karst or porous aquifers since the result of the experiment depends more on the correct choice of the injection and sampling sites than in the case of the other two aquifer types. Despite numerous research projects in the last decades, the experiences with tracer tests in fractured aquifers, compared with tracer tests in porous aquifers, are still low. A thorough interpretation of such tracer tests therefore is still difficult, but the single fracture dispersion model (SFDM) yields sufficient accuracy (Małoszewski and Zuber 1985; Himmelsbach et al. 1992; Bäumle et al. 2001; Bäumle 2003). At present, research in this field is being conducted in many different countries (e.g. Germany, USA, Slovenia, France, Austria, and United Kingdom).

10.6.3 Karst Aquifer

Tracer tests in karst areas have a long tradition reaching back to Roman times (Käb 1998) and they are an imperative tool for understanding the hydrogeological conditions of karst terrains. Nearly all of the potential tracers have already been used in karst tracer tests investigations, and in Slovenia they were positively tested in several multi-tracer approaches (Gospodarič and Habič 1976; Kranjc 1997). Other, more recent applications are described in the proceedings of the International Symposia on Water Tracing (e.g. Hötzl and Werner 1992).

Predominant features of karst areas are dolines, sinkholes, caves, ponors, karren, roofless caves, or poljes. Most of those features have already been described in the fundamental work of Cvijić (1893) and karst is named after its typical appearance in the Kraš Region of the Slovene Dinarides (Bates and Jackson 1987). Karst aquifers are occurring in all water soluble rock formations, namely limestone, dolostone, gypsum, anhydrite, and even quartzite. Due to the dissolution of the forenamed rocks by the acids in the ground water cavities are formed and the ground water can flow through those cavities (Šušteršič 1996). In the underground, the diameters of those structures range from several micrometers to several decametres, mostly offering an excellent hydraulic connection for water flowing within that conduit network. As many karst researchers and karst aquifer tracer tests showed, this network of channels gives rise to different flow paths (Brown and Ford 1971) and tracer tests proved that water uses different pathways with different effective velocities reaching maximum velocities of 10—585 m h⁻¹ (Gospodarič and Habič 1976; Hölting and Coldewey 2005). In the past years the triple-porosity-model has become more and more prominent in the carbonate aquifer research (Worthington 1999) and future tracer tests will have to consider that approach.

As has been seen before, karst aquifers are characterised by high to extremely high flow velocities via discrete flow paths. Modelling of karst aquifers, which is thought to be extremely complicated, has to a special degree been possible in the Tübingen and Göttingen working groups. They developed the computer code CAVE (Clemens et al. 1999; Liedl and Sauter 2000), which has also been used to model water rock interactions in flooded underground mines (Spiessl et al. 2002).

Evaluations of karst tracer tests are done in the same way as described before. Yet, a characteristic difference of karst water tracer tests is the appearance of multi peak breakthrough curves and comparatively long tailings. They are due to the different flow paths through the karst conduits and are usually interpreted by applying the multi-dispersion-model (MDM) as introduced by Małoszewski et al. (1992).

10.6.4 Mine Aquifer (Underground Mines)

Underground mines are very similar to karst aquifers (Sammarco 1995; Burbey et al. 2000) and like them mines can be seen as triple-porosity-aquifers with one-, two-, and three-dimensional porosity elements. The one-dimensional elements would be the main adits, shafts, and workings, while the two-dimensional elements are the fractures, faults, and bedding planes and the three-dimensional element is the rock matrix. If a mine is worked in a karst aquifer the karst conduits would have to be counted as one-dimensional elements as well. Yet, no publication is known so far where the triple-porosity model *sensu strictu* has been applied to mine water research.

Carbonate aquifers can be described through the triple porosity model. This model clearly shows how important all three of these porosities are (e.g. Worthington 1999). A comparison of the hydraulic conductivities in a flooded underground mine together with the basics behind the first-flush scenario clearly proves that the most important factor for mine aquifers still is the “conduit” porosity. Of course, the long tail of the first flush is due to natural attenuation and the comparatively lower hydraulic conductivities of the minor porosities (fractures, matrix, panels). Yet, from an environmental point of view the large volumes are of greater importance for the characteristic of the first flush.

Due to the similarities of mine aquifers and karst aquifers the first quantitative tracer tests in deep flooded underground mines used the techniques established for those types of aquifers (Wolkersdorfer 1996). Also Liebisch and Remus (1966) concluded that the methods applied in karst tracer tests could be used in mines and used Na-fluorescein for a tracer test in the Mansfeld/Germany mining area. Above and beyond the similarities some major differences can be listed (most of those differences are not valid for pre-historic and medieval mines):

- channels (e.g. adits, shafts, workings) in modern mines are usually not curvilinear and typically meet at right angles
- diameters usually do not change for a longer distance
- the water level changes are comparatively small
- voids are not subjected to dissolution as in carbonates

In detail, mine aquifers comprise of a combination of karst aquifers, porous aquifers and fractured aquifers (compare section 7.1). All the larger openings that were used for man or material haulage can be looked upon as a karst aquifer. Typically the openings are large, the storage capacity is small, the effective velocities are big, and the water flow is non-Darcian. Shafts, boreholes, and old abandoned mine workings above the mine water table can be looked upon as epikarst. Between that zone and the mine water table comes the vadose zone and finally, below the mine water table, the phreatic zone is exposed. As the Straßberg/Harz mine water tracer test shows, those three zones in a mine act very similarly to the real karst features.

In most mines, especially in the near surface locations, a fractured aquifer exists. That fracture network is usually a prominent region of infiltration and has intensively been studied by the works about underground radioactive waste disposals (e.g. Galloway and Erickson 1985; Abelin and Birgersson 1985; Sawada et al. 2000). Sometimes, this fractured zone is the source of catastrophic mine water intrusions and has also been studied to prevent inundation from this region of a mine (e.g. Singh and Atkins 1985; Singh et al. 1986; Grapes and Connelly 1998; Waterhouse et al. 2003). Though the importance of the fracture network for working or non-flooded mines is obvious, the role of fractures within the flooded part of a mine has not yet been studied in detail. From a hydraulic point of view their role should not be overestimated, as most of the transport occurs in the flooded voids. Yet, the tracer test in the aban-

done Königshütte/Harz mining field proved that fractures above the ground water level can be hydraulically well connected at high precipitation periods. A further fracture network can be found in the close vicinity of the mine workings due to the loosening of the rock due to the mining activity. Depending on the mining technology this fracture network can reach several centimetres to several meters into the rock matrix (see section 4.2). Though not important for the overall water flow, it is important for the diffusive transport of the metals into the mine water (see section 4.2).

Finally, every mine also consists of a porous aquifer. This is obvious from the rock matrix, which usually can be looked upon as a porous aquifer. But also backfilled areas or the goaf (collapse zones behind longwall mining operations) can be seen as porous aquifers with porosities commonly in the range of 0.55 to 0.6 (Paul Younger, pers. comm.).

Though mine aquifers have the characteristics of all of the three known aquifer types, mine water tracer tests usually prevail patterns which are typical for karst aquifers. Therefore, mine water tracer tests must be evaluated by using the methods for evaluating karst tracer tests or – in the future – the triple-porosity-model.

10.6.5 Evaluation of Breakthrough Curves

Breakthrough curves are a first means to evaluate the results of a tracer test (Fig. 81). Such curves plot the concentration c of each tracer against the duration t of the tracer test. From this graph, the following times can be deduced (Fig. 83):

t_0 :	Start of tracer test	
$t_{\max}$:	First tracer arrival	→ maximum distance velocity
t_{dom} :	Maximum tracer concentration	→ dominant distance velocity
$t_{\text{med}} = t_{50}$:	Median tracer concentration	→ median distance velocity
t_w :	Mean tracer concentration	→ real/mean distance velocity

whereas the velocity is calculated as follows:

$$v = \frac{x}{t} \quad (80)$$

The longitudinal dispersion coefficient D_L (Fig. 84) is estimated by using the times at which 16%, 50% and 84% of the tracers were detected (mean value and twice the standard deviation of a Gauss normal distribution). The equation for this is:

$$D_L = \frac{v^2 \times (t_{84} - t_{16})^2}{8 \times t_{50}} \quad (81)$$

As described earlier, the complete dispersivity consists of the three components: lateral, transversal and vertical dispersivity. In the case of pore or karst aquifers, the predominant dispersivity is the longitudinal one. According to different authors, the transversal dispersivity is between $\frac{1}{10}^{\text{th}}$ and $\frac{1}{5}^{\text{th}}$ (Käb 2004) or 1—2 orders of magnitude smaller (Kinzelbach and Rausch 1995) than the longitudinal dispersivity, whereas $D_T \approx 0.1 \times D_L$ has been found to be a well suitable mean approximation:

$$D_T \approx (0.01—0.2) \times D_L \quad (82)$$

By using computer codes or spreadsheet calculations these equations can easily be used to calculate the characteristic tracer test parameters. A simple to be used MS Windows code is TRACI98 which can be downloaded from the Borntraeger web page (Käb 1998, 2004) or the

more sophisticated programme QTRACER which is available from the U.S. Environmental Protection Agency's web page (Field 2002).

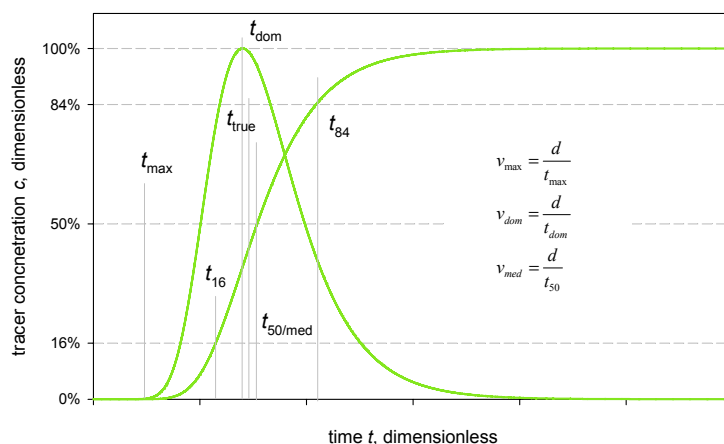


Fig. 83: Velocities that can be derived from a breakthrough curve. t_{16} and t_{84} are needed to calculate the longitudinal dispersion coefficient D_L . t_{true} is exactly in the middle of t_{dom} and t_{med} .

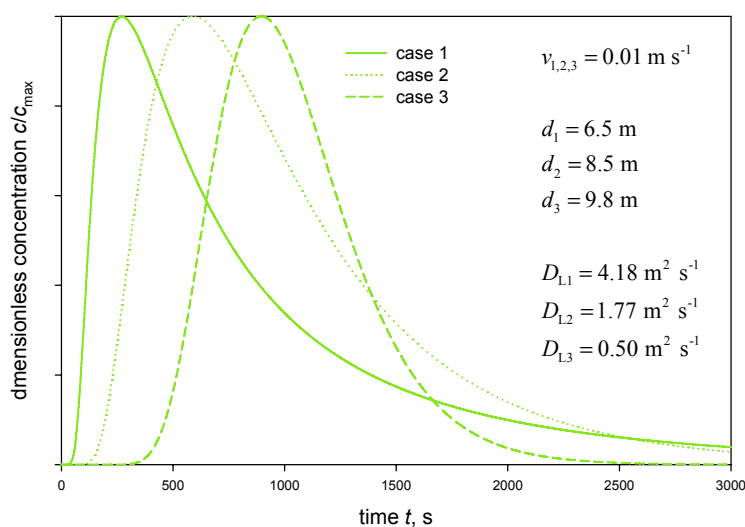


Fig. 84: Graphical representation of different longitudinal dispersion coefficients D_L . v : mean velocity, d : dispersivity.

10.7 Quality Control and Data Storage

10.7.1 Quality Assurance – Quality Control

Accurate and reliable results as well as the documentation of data acquisition during field and laboratory work are essential to conduct treatment or remediation measures in water management of mines, which in some cases might take several decades. Sampling and analysing costs a lot of money and consumes a lot of time during a project (Kellner et al. 1998). Very often the responsible project leader has to rely on external experts and does not take the samples personally or has them analysed under his supervision. Furthermore, it might well be that dur-

ing a project laboratories or staff changes but the data quality must be insured at all time. Therefore, quality assurance (QA) and quality control (QC) must be given special consideration if the variation of the analytical results due to false sampling or analytical procedures shall be excluded. They could be the result of a data quality objective (DQO) process (U.S. Environmental Protection Agency 1994), the experience of the project leader, existing QA/QC plans, or the result of expert groups decisions. In the USA, the Environmental Protection Agencies (EPA) employs Quality Assurance Managers, Quality Assurance Officers, or Quality Assurance Representatives which can be contacted if help is needed to set up QA/QC plans. Furthermore, several documents are available to obtain established procedures (U.S. Environmental Protection Agency 1994; EPA QA/R-xx and QA/G-xx series documents).

While QA measures refer to the whole project and ensure that an effective QC is in force, QC guarantees that a consistent quality of the data and the monitoring process is achieved. Most laboratories have their own QA/QC plans (compare Kellner et al. 1998); therefore, there is usually no need to include the laboratories in the QA/QC plans of the project. Nevertheless, the responsible project co-ordinator must record under which QA/QC procedures the relevant laboratories work and if uncertain about the results of a certain laboratory standard samples should be send to the laboratory in question (e.g. DIN EN ISO 8403 ("Qualitätsmanagement"), DIN EN ISO 8402, 1995-08, or other relevant national regulations). Calow (1991) listed potential and common laboratory errors of which only "chemical and physical interference" is non-human induced:

- incorrect identification of samples
- contamination
- improper, or inappropriate sample preparation
- inaccuracy of sample weights, or volumes
- improper, or inappropriate sample dissolution/treatment
- chemical and physical interference
- improper, or inappropriate instrumentation/inaccurate measurement
- calculation errors
- incorrect data handling/reporting

QA/QC measures are time consuming and they add costs to projects. In general, the higher the degree of confidence shall be, the more costs have to be added to QA/QC measures. Considered that false data interpretation could result in wrong decisions which might then result in unnecessary remediation measures, those expenditures should be spent. Saving money by ignoring the simplest principles of quality assurance and control very often results in long discussions about how a sample was taken, where it was taken, who took it in the simplest cases and in mal functioning remediation measures in the worst case. A thorough QA/QC plan will also be the basis for a trusting co-operation with the regulators and project reviewers (Downing and Mills 1998). A QA/QC program must guarantee the following objectives:

- document the procedures and methods of sample collection, preparation and analysis
- provide assurance about reliability of analyses using replicate samples, cross-laboratory checks and relevant reference standards
- provide assurance about the precision and accuracy from duplicate samples
- provide assurance about the accuracy from using recognised reference standards
- provide reliable information regarding the interpretation of data about the behaviour of the mine water during time (longevity of mine water discharge)
- provide a chain of custody of samples

Field sampling is the first step where mistakes could be done and there is no possibility of removing the error or even determine such an inappropriate sample in any of the later stages. Sampling and monitoring procedures must always be precisely described in field or laboratory books. Those books must have properly bound sheets (adhesive binding should be avoided), numbered pages, and no pages must be missing. Also, the removal of single sheets is against proper QC methods. Within the book, only field or laboratory observations and numbers should be written. In case, a calculation is conducted, this part should clearly be noted as SC (“side calculation”) or NR (“Nebenrechnung”). Erasing with a rubber is never allowed and no parts must be made unrecognizable. Wrong data should be “crossed” out with one single line to guarantee that the wrong number can still be read. This guarantees that the potential reason for the faulty data can be reconstructed (e.g. 7.45 instead of 7.54; or 27 °C instead of 27 °F). Besides the measured parameters, “soft” parameters have to be written down, too. Those include observations that seem to be “un-normal”, e.g. animal interference, unusually high or low turbidity, ice covers, or any other observation that might be of interest for co-workers. In the future, electronic field books (e.g. PDAs) connected to GPS/GLONASS/GALILEO-systems will substitute the traditional way of data recording (e.g. Ackman 2003). A QA/QC plan must ensure that this type of data storage is also covered.

Sampling points and samples must be numbered precisely throughout the whole project. It is advisable to agree to the same numbering system by all researchers and authorities involved, though such systems seem to be complicated (e.g. Mueller et al. 1997). Each sample must be labelled with the name of the project, sampling site, sampling date, responsible research institution, and person who took the sample. Sample numbers containing a three letter project code, the date, and the sampling site provide a means to exclude mistakes during sample preparation (e.g. FDR-1202-MP1: Project code-short date-sampling point). Optionally, the laboratory provides bottles labelled with barcodes. In those cases, the associated barcode number has to be recorded in the field book or PDA.

In a well based QA/QC protocol, the proper analytical methods and equipments for the catchment study are laid down as well. Errors in any project usually arise from the selection of wrong equipments or calibration methods. Chemicals or standard solutions used for equipment calibration or during analytical procedures must be fresh and the lifetime of the chemicals must be checked regularly. From time to time, blanks should be included in the analytical process to check for contamination or overlapping. Field equipment can be controlled by measuring standard solutions from time to time during the day.

A first quality control is often very easy and can separate “real” outliers – mostly they are typos instead of wrong data. Finding such “simple” errors in data bases of companies that got a lot of money for their work is really annoying – and how easily such errors could be avoided! One prominent Eastern German remediation agency paid by the tax payer used a spreadsheet for calculating the flow of a stream and for six years no-one was wondering about extreme flow situations at special water heights in this stream. Those extreme flow situations were due to just one typo in this calibration table! If a quick scatter plot would have produced that typo could have been eliminated at its first appearance (Fig. 85).

Quality control thus includes a quick review of the data given to the next instance and also looking through the data received from the previous instance. Many errors could be avoided if at least those steps would be conducted by every person involved in the data gathering and storing chain. Some simple test must not have a “real” scientific background they shall just identify errors or outliers. Scatter plots, bar plots, or time series should always be used to identify outliers (Fig. 85). Furthermore, in every step it is important that the data gathered can be traced back at all times in a “chain of custody” which shall answer the following questions:

- When, where, and by whom has the sample been taken?
- How, where, and how long has the sample been stored?
- How has the sample been treated and analysed?
- Who handled the sample where and at which time?
- Where have the potential left-overs been stored?

Finally, “it is rather unfortunate that QA/QC reporting is somewhat boring and not readily appreciated by many, however it is a necessary part of data analysis” (Downing and Mills 1998).

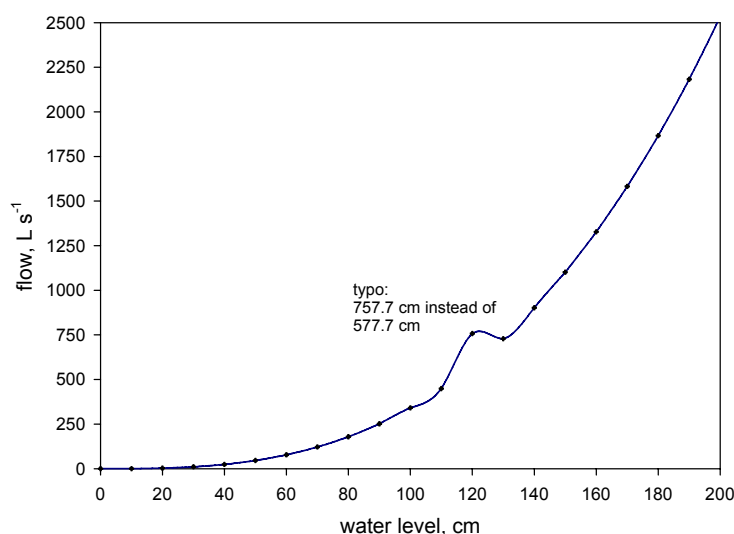


Fig. 85: Example of a scatter plot which would have revealed a typo in a flow/water level calibration table. The table with the wrong data was used more than six years by a huge German remediation agency without anybody recognising this typo at 120 cm water level.

10.7.2 Data Storage and Management

Before a project begins, the project leader has to make sure that a proper data base and data retrieval system is set up and used by all persons, companies and laboratories working on the project. Many tools are available to assist the project leader in this work, but the data structure and the type of data that shall be stored must be compiled by him, his colleagues or a working group. For large projects professional support is available. In Germany Pangaea, hosted by the Alfred-Wegner-Institute in Bremerhaven, can be used for the management and the storage of geo-projects. They guarantee the persistent storage of the data and they also provide support in setting up the sampling scheme as well as the data structure. In the case of a large and long lasting project, data storage can be a major task in the project and a certain amount of money should be allocated for this task.

Data storage and retrieval must be given special consideration, because storage systems change quickly and the half life of an electronic storage technology is in the range of several years – compared to paper storage with half lifes of many decades to centuries. This fact is true for hardware as well as for software. First widely available computer systems used 9" disks, which can only rarely be read with modern computer systems today and software produced 10 years ago might have produced data sets which do not produce more than hiero-

glyphs nowadays. Yet, data storage is not all. Data without a consistent description of the data – if possible in metadata sets – is essential if the data shall be stored for a long time.

Though professional archiving technologies are available that guarantee regular upgrade of the datasets, they are rarely used in mine water research. A discussion of such technologies is far beyond the scope of this report, and as it is very costly, SMEs or authorities usually are not connected to such systems. Therefore, if long life storage shall be achieved, printed reports, tables, and illustrations are the first choice of data storage – from the view of persistency Hethitian clay tablets and Egyptian hieroglyphs cannot be beaten – yet their storage and handling is rather strenuous and not the means to be used nowadays. Also, ASCII tables should be produced and stored on floppy-discs or CD-ROMs at the end of each project, because those systems are widely used compared to other systems which guarantees a higher half-life than other electronic storage systems (e.g. streamer tapes, video-cassettes, hard disks). Furthermore, if speaking about proper data storage: a spreadsheet is not a database! It can be used to make quick calculations or draft graphs, but they cannot substitute a real database system – though the reality of consultant work seems to be like this.

Also of importance is data security (data backup, data encryption). Many small companies or scientists do not backup their data on a regular basis, because they often think “what shall happen”. Fact is that all data storage systems could fail to work at any time, no matter which data storage system was used (even Egyptian steles have been reshaped and recycled). Therefore, data backup is an integral part of any mine water and river catchment study and should be included in the QA/QC protocols. Proper data backup has to be conducted daily, with full data backup on a weekly basis. Every month, a copy of a full backup shall be securely stored at a place outside the office. All those actions seem to be annoying and time consuming, but from experience, data retrieval after a system crash could take several days and sometimes, data are lost forever with unforeseeable consequences to the project or the company.

During writing of a report or after changing a database it is good practice to add a two or three digit number to the report’s file name (e.g. MINEWATER_01.DOC). This number will then be increased each time the report is stored to the storage system (e.g. MINEWATER_02.DOC). In case a file is corrupted, the previous version can be used instead. At the end of the project, the last file can be renamed (e.g. MINEWATER.DOC) and all the old versions be deleted (professional desktop publishing systems, such as QuarkXPress™, use a similar system for data security).

Yet, if river catchment projects involve many SMEs, research organizations and authorities, a common data storage procedure must be agreed *before* the project starts. Internet or intranet based groupware is commonly used by larger projects. All data should be stored with the corresponding coordinates (keep in mind that GPS/GLONASS/GALILEO data must be recorded with the reference system and the geoid used!), to be usable in GIS-systems. There are several countries or authorities that provide data storage systems, in which case those systems should be used to guarantee data exchange between all parties involved. Furthermore, as spreadsheets are no databases, data retrieval therefore is restricted or impossible. In a large project, spreadsheets should only be used for quick reference and then should be organised to provide as much information as possible in as few as possible spreadsheets. A good database system, though its use is more complicated, should always be given priority over spreadsheets (although, of course, databases are organised in spreadsheets).

11 Mine Water Treatment and Ground Water Protection

11.1 Introduction

This book covers most of the relevant aspects of mine water management during or after a mine's closure. Because mine water treatment is an essential aspect of mine closure this chapter will discuss the basic aspects of mine water treatment. After you have read this chapter you should be able to understand why mine water treatment is necessary, which different mine water treatment options are possible, and how the basics of those processes work. Active treatment is a standard procedure which – at least in principle – should be known by most mine water researchers. Yet, this is not necessarily the case for passive treatment options. Furthermore, the tracer tests described earlier in this book also aim at understanding how a mine, before flooding, could be modified for getting lower pollutant discharges, some basic passive treatment options are described in more detail than the active ones.

Polluted mine water emanates primarily from disulphide oxidation which releases acidity and initiates the dissolution of minerals in the mother rocks of the deposit (chapter 4). Thus, trace elements may be enriched in the mine water and contribute to the toxicity of the discharged water (Smith and Huyck 1999). Depending on the buffer capacity of the mother rock, the mine water might be acidic or alkaline. One of the main characteristics of polluted surface streams are yellow to red precipitates: iron oxihydroxides, commonly referred to as ochre, or yellow boy. Their chemical and mineralogical composition is of complex nature and the term iron oxihydroxides just describes what in reality is a list of different mineral species (Bigham 1994; Schwertmann et al. 1995; Fitzpatrick and Self 1997). Another, not as obvious effect as ochreous mine water emanating from a mine portal, is the contamination of ground water by polluted mine water. Usually, those cases are only of interest when aquifers used as drinking water reservoirs are directly affected by the mine water. So far, the publications referring to polluted ground water bodies are underrepresented in the international literature. Some of the issues related to ground water will be addressed in the last section of this chapter.

Polluted mine water is one of the most threatening environmental hazards known. Areas affected by such waters are usually damaged for many decades, if not centuries. Because the treatment of polluted mine water is usually expensive, active treatment can mainly be found in heavily populated areas, at working mines, or where governmental money was made available for treatment purposes. With respect to the European Water Framework Directive all river catchments must have a good ecological and chemical status by the end of the year 2015. But what, for example, shall be done in such extreme environments as the Spanish Rio Tinto area which has been heavily polluted by mining activities during the past millennia and now hosts about 1300 different forms or microorganisms and a macrophyte flora which is unique and well adapted to that extremely acidic environment (Salkield 1987; Ariza 1998; Leblanc et al. 2000)?

Since 2002, three comprehensive books about mine treatment were published (Brown et al. 2002; Younger et al. 2002; Lottermoser 2003) and therefore, this section will only deal with some fundamentals of mine water treatment. Furthermore, many research groups published papers on single treatment options and describe in detail how they could be implemented under special climatic conditions.

Mine Water treatment is an essential task for all mines where polluted mine drainage exists. Pollution cannot be seen as an absolute term. Legally non-polluted mine water at one site might be heavily polluted under the conditions of another mine with more stringent legal requirements. Many countries have limits or regulations which are based on geological background but in most countries chemical discharge limits are set up site specifically. Some

countries also have defined limits which make life easy for the administration but could cause severe problems for mine operators that have to treat their mine water to values cleaner than the background. Yet, according to the European Water Framework Directive, future limits must be based on a watershed perspective but such approaches are not compulsory before 2012 (European Commission 2000; Kroll et al. 2002a; ERMITE Consortium et al. 2004). Though this approach is common to several EU member states, such as the United Kingdom or France, most of the other member states have to implement a new perspective of water laws (e.g. Germany, Austria; Bongaerts 2002; Timmermann and Langaas 2004).

Though for most people mine water treatment is the removal or reduction of chemical constituents, mine water treatment includes physical treatment as well. The first mine water treatment schemes consisted only of physical treatment to remove sediment or coarse material from the mine water (Kegel 1950). The standard methods are settling ponds, oil/water separators, as well as lamella clarifiers and are often supported by flocculants and coagulants (Skousen et al. 2000). Though flocculation and coagulation can be seen as physical separation processes of water and the pollutants, the processes leading to flocculation and coagulation are chemically based surface reactions between the coagulant or flocculant and the metal oxihydroxides (Stumm and Morgan 1996).

If not otherwise specified, the following sections are based on Brown et al. (2002), Younger et al. (2002), and Lottermoser (2003). The latest development in active treatment was the development of the innovative HDS method (Kostenbader and Haines 1970) and currently, promising investigations are under way to remove sulphate from mine water by a coupled chemical-biological process (e.g. Eloff et al. 2003; Howell 2004; Maree et al. 2004a, 2004b, 2004c, 2004d), though a review of sulphate removal options has already been given earlier by Howell et al. (1997). To help the researcher or the engineer in deciding which treatment option would be possible, several software packages are available, among those are AMDTreat (Office of Surface Mining Reclamation and Enforcement, Pennsylvania Department of Environmental Protection, West Virginia Department of Environmental Protection) or AMDSelect (HERO Research Team; McKenzie 2005; Kruse and Younger 2005).

In the passive treatment field, many different methods are under investigation in all relevant research institutions in all continents. Regrettably, there is a tendency to negate the positive results of other research institutions as can be seen from the literature cited by the different authors – language barriers being only one problem. Nearly every year a new or adopted method is published. This shows that in contrast to conventional treatment, passive treatment is still under development and in a state of flux. So far, only passive treatment methods for net-alkaline, ferruginous waters can be considered as fairly settled (Paul Younger, pers. comm.).

Irrespective of which mine water treatment option is chosen, it must always be guaranteed that the prescriptive limits set by the authorities are met at all times. This is usually not a problem for active treatment plants as the discharge and pH is continuously monitored. Depending on the readings the amount of chemicals to be added to the polluted mine water is operated. In the case of passive treatment plants the situation is different. No control unit is available that would speed up microbial activity where necessary, increase solar input, or reduce the flow velocity to increase retention times. Regulators therefore should use slightly different approaches when approving passive treatment plants. Beside a passive treatment concentration limit for the parameters that have to be met, a load limit should be applied as well. Examples for such a procedure are the limits to uranium and radium at the abandoned Niederschlema/Alberoda/Germany uranium mine. In addition to the radium concentration of 800 mBq L⁻¹ also a mass load based annual limit of 2 GBq and for uranium 2.4 mg L⁻¹ and 20 GBq were set up by the authorities (values for 1991–1993; Wismut GmbH 1994).

Mine Water treatment, in this context, means the treatment of polluted mine water discharges from the mine. Yet, as has been described in section 11.6, mine operators should try to invent “*in-situ*” or “on-site” techniques to minimise the amount of polluted water. In most instances it is cheaper to discharge water than treating it for a long time. It has already been shown that the prevention of convective loops in flooded underground mines lowers the maximum concentration of pollutants and that by using transparent mine closure techniques, polluted mine water can be either kept in the mine or treated *in-situ* (e.g. Wolkersdorfer 1996).

11.2 Physical Treatment of Inert Solids and Oil

Nearly all mine waters contain considerable amounts of suspended matter in different grain sizes: clay, silt, sand, and gravel. Their transportation and sedimentation in the flowing water is a function of the flow velocity, in which the sedimentation is expressed in the Stokes equation and the transport according to the investigations of Hjulström and Sundborg (Hjulström 1935; Sundborg 1956; Selley 2000). Quickly flowing mine water can therefore transport coarser particles than slowly flowing water. According to Hjulström (1935) a 1 mm particle will be transported by water with a velocity of 4 m min^{-1} and a 0.1 mm particle by a velocity of 0.6 m min^{-1} . Therefore, if the mean velocity at the point of discharge is known, the maximum particle size can be estimated and the proper separation technique be chosen (Fig. 86).

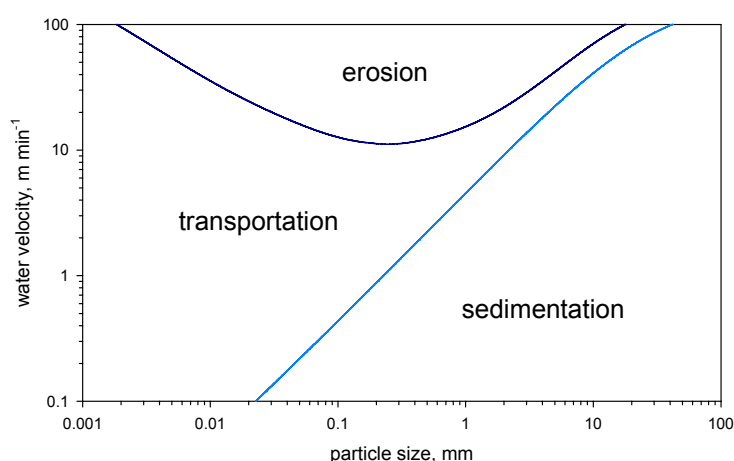


Fig. 86: Hjulström diagram representing the areas of sediment erosion, transportation, and sedimentation depending on the water's flow velocity (Hjulström 1935). The diagram is simplified for the mean particle sizes and velocities expected in the mine water field.

None of the lines is a fixed line, as the transportation/erosion or transportation/sedimentation properties of a single sediment particle do not only depend on the velocity of the water but on the particles charge, interaction with other particles or laminar/turbulent flow conditions. Especially the original transportation/erosion line has a broad transition zone.

Before or after the treatment of the chemical pollutants in the mine water the suspended solids, colloids, or floating material have to be removed from the water. This is an important task in all water treatment schemes wherever they are working and therefore those methods are standard methods that were established in the waste water treatment field and adapted to the special requirements for mine waters (Kegel 1950; Droste 1997).

Obviously, the most common equipments to reduce solid particles from mine water are settling ponds. They are only suitable for coarser particles which can simply be separated by gravity driven sedimentation. Colloidal particles, usually, cannot be separated by settling ponds. They require flocculants or coagulants so that they can form larger particles which

then can settle in a sedimentation pond. Their size depends on the design flow rate Q_d and the average size of the suspended solids. Linsley et al. (1988) and Shaw (1994) give details of how to calculate the design flow rate for treatment facilities. Younger et al. (2002), assuming average sizes of $4\text{ }\mu\text{m}$, and a retention time of 3–4 hours suggest the following sizing equation for solid settlement without the addition of flocculants:

$$A_{\text{SP}} = f_{\text{SP}} \times Q_d \quad (83)$$

with A_{SP} : area of settling pond, m^2
 f_{SP} : sizing factor for settling pond, $1 \cdot 10^5\text{ s m}^{-1}$
 Q_d : design flow rate, $\text{m}^3\text{ s}^{-1}$

Kegel (1950) suggests to use sedimentation ponds with a length of 15–20 m, a depth of 1.5–2.5 m and a width which can be calculated from the area calculation in equation 83.

An advantage over settling ponds are lamella clarifiers (plate settlers) or tube settlers. Lamella clarifiers consist of a set of inclined PVC panels which are flown through from the bottom to the top of the clarifier (Fig. 87). That way, the flow velocity as well as the Reynold's number are decreased and a turbulent re-suspension of the sludge will thus be prevented. The resultant movement vector of the particles' flow directs them to the lower side of a single lamella from where they slide down into a collecting chamber (Droste 1997). Plate or tube settlers are space saving and are either pre-manufactured or assembled according to the special characteristics of the mine water. At the Horden/UK mine water treatment plant three lamella clarifiers can potentially treat $180\text{ m}^3\text{ h}^{-1}$ each. In a normal installation a 4–5 m^2 lamella clarifier, depending on the particle size, replaces a settlement pond or lagoon of 100–300 m^2 (Siltbuster/Unipure Europe Ltd. Product Information). In an investigation about lamella clarifiers for rainwater treatment it could be shown that lamella clarifiers significantly increase the treatment performance. By addition of flocculants the treatment performance could be further increased (Kraut and Bondareva 2000). It can be concluded that the latter results can be transferred to mine water as well.



Fig. 87: Lamella clarifier at the Horden/United Kingdom HDS mine water treatment plant (photograph courtesy Andrea Berger).

Sometimes, mine water can also contain considerable amounts of hydrocarbons, namely petrol or oil – not only in oil shale mines (e.g. Rätsep and Liblik 1998; Tiwary 2001). Oil can interfere with the treatment process. In those cases where significant amounts of oil are in the

mine water, oil/water separators (interceptors) must be used to separate the floating oil from the mine water. If the hydrocarbons, for example phenols, are water soluble, they cannot be treated in this way but have to be stripped off by aeration. It is known from many mines that the hydrocarbon concentration in the discharged mine water rapidly decreases after mining ceased (Rätsep and Liblik 1998). Therefore, some time after mine closure oil/water separators are usually not needed any longer. Oil separators must be installed in a way that the hydrocarbons cannot explode. Staff cleaning the oil separators must be advised that the residues can be inflammable and must be handled with care. Furthermore, the type of separator has to be chosen based on the water quality requirements set up by the relevant authorities and in accordance with the valid directives (Environment Agency 1997).

11.3 Active Treatment

Active treatment plants aim at removing the pollutants in the mine water by either chemical processes or by physical processes involving filters or membranes. In the case of distillation or solvent extraction, which are not used for large scale mine water treatment, other physical processes are involved. Active treatment means that by human action either energy or chemicals are added to keep the treatment process running. Consequently, the term “active treatment” is not used if e.g. mine water is pumped from a mine into a constructed wetland.

In most active treatment plants, chemicals adjust the pH or E_H value of the water to reduce the solubility of the metals polluting the water. If the mine water is acidic, which is the most problematic case, reactants are added that neutralise the water and increase the pH to a value that guarantees that the relevant metals precipitate. Some of the commonly used chemicals are caustic lime, sodium-hydroxide, and limestone (Skousen et al. 2000). Barium chloride is added to the mine water to reduce the sulphate content of mine waters (Bosman et al. 1990; Adlem et al. 1994; Maree et al. 2004a) and Barium sulphate to eliminate radium (Skeaff and Campbell 1980; Georgescu et al. 1995; Stoica et al. 1998). Those chemicals raise the pH of the mine water and simultaneously decrease the solubility of the target pollutants. Thus, precipitates and co-precipitates form, which settle from the solution after adding flocculants and/or coagulants (Georgaki et al. 2004) in settlement lagoons, lamella clarifiers, or radial flow clarifiers (Fig. 88). Finally, the precipitates are dried and deposited off. Depending on the toxicity of the dried precipitates they are either backfilled in existing underground mines (e.g. Straßberg/Harz Mountains, “Grube Einheit” Elbingerode/Harz Mountains) or stored in toxic waste deposits. Two types of such conventional active treatment processes can be distinguished: low density sludge (LDS) with several percents of solids and high density sludge (HDS) with a 5–10fold higher sludge density compared to LDS plants.

Traditionally, mine water treatment is a chemical-physical process. The first chemical water treatment systems were invented by R. A. Henry, a Belgian, and through the German Niersverband in the early 1930s (Kegel 1950). Henry developed a method to treat slurry water from coal-washing plants by adding a flocculant (he used potato starch). He further noticed that alkaline conditions with an optimum at pH 11 and frozen potato starch significantly increase the settlement speed of the particles. The first descriptions to use the Henry technique for water treatment in mining operations were published in 1934 (Anonymous 1934; Goette 1934) and the first full scale water treatment plant with the Birtley-Henry treatment process went into operation at the Wallsend Rising Sun pit in Newcastle-upon-Tyne in 1935 (Anonymous 1935). In all cases the Henry process was used to treat water from a coal cleaning process with 265 mg of quick lime and 6.5 mg of frozen starch per kilogram of slurry water.

Nowadays, such conventional LDS treatment systems, simplified, consist of a reactor where the water reacts with the chemicals, a mixing installation, and a settlement pond. As long as a mine is in operation or somebody is willing to pay the treatment costs, such conventional

treatment systems reliably treat the water. Yet, because the reaction kinetics are slow, CaO or other added alkali is sometimes wasted and can be found in the sludge. During the course of the last two decades, operators and regulators were searching for alternative methods of water treatment. These alternative methods should also guarantee cheap treatment options which need less to no maintenance and would only work with naturally available energy.



Fig. 88: Radial flow clarifier of an active mine water treatment plant at As Pontes lignite mine (Spain).

In contrast to the LDS process, the innovative HDS process recirculates part of the sludge within the system. Thus, crystallisation “embryos” (or “seeds”) are invented into the systems at the point where the mine water is mixed with the alkaline material and the recirculated sludge and consequently, the colloids form quicker and reach higher solids/water ratios. The method was first investigated by Kostenbader and Haines (1970) at the Bethlem mines of the Bethlem’s Ellsworth and Cambria Divisions/USA. About $34 \cdot 10^3 \text{ m}^3$ of mine water were discharged and conventional treatment did not fulfil the state limits nor was there enough storing capacity for conventional treatment sludge. The aim of the investigation was to improve the settling characteristics of the sludges and the concentration of the solids. Because of their investigations and the recirculation of the sludge, the authors were able to increase the solid contents from about 1% to 30—40% at best.

Other large scale mine water treatment options used are electro-dialyses, membrane filtering, or membrane filtering combined with distillation. Frequently, where such treatment options were given preference, economic or environmental pressures were the basis for the decision towards membrane or dialyse techniques. Polish coal mine waters are often highly saline with chloride and sulphate concentrations up to 42 g L^{-1} and therefore cannot be directly discharged into the receiving streams (Gisman and Szczypa 1982; Turek and Gonet 1996). One of the first Polish mine water treatment plants treating saline mine water was constructed at the Dębieńsko coal mine in 1975 (Motyka and Skibinski 1982). Today, this plant produces process water, drinking water, and solid salts for the chemical industry, employing membrane and electro-dialyses techniques (Turek et al. 1995; Turek and Gonet 1996).

Over-treatment of mine water is seldom described in the literature and most authorities and legislators are more concerned of low metal and semi-metal concentrations than of potential hazards as a cause of over-treated mine water. Over-treated mine water has a two-fold meaning: due to extremely low legislative pollutant limits or to malfunctioning active treatment plants an excess amount of alkaline chemicals is used. In both cases this results in a high alkalinity of the receiving stream which might pose a new threat to the receiving water course and

the catchment area. Because many treatment plants operate in acidic environments over-saturation is usually of no concern, because the acidic stream is then treated as well. Yet, as in the case of the Straßberg/Harz treatment plant, an over-treatment can cause a severe threat to well buffered circum-neutral fish water streams. Miller et al. (2005) describe a treatment plant in West Virginia/USA, in which the manganese concentration of the water had to be lowered to 1 mg L^{-1} and the pH consequently rose to 8.8–9. Though the manganese concentrations were kept low, the high pH of the water was not suitable for trouts. On the other hand, over-treatment can be advantageous when the receiving stream has a low buffer potential and the surrounding area has also been affected by mining and releases acidic mine water (West Virginia Division of Environmental Protection – Office of Mining and Reclamation 2001).

Bogner and Doehler (1984) described the formation of ettringite ($\text{Ca}_6\text{Al}_2[\text{OH}_4|\text{SO}_4]_3 \cdot 24 \text{ H}_2\text{O}$), an end member of the ettringit-metavoltin group, which they assumed is due to over-treatment of mine water. Drainage water of a coal mine near Kingwood/West Virginia with a pH of 2 was treated by sodium hydroxide (NaOH) and was drained into a $2,500 \text{ m}^2$ settling pond. Within that pond, pH ranged from 4 to 11 because of partly inadequate treatment maintenance and rainfall events. Most of the precipitates in the sediment of the pond were amorphous oxihydroxides and, to a lesser degree, gypsum, quartz, kaolinite, calcite, and dolomite. However, ettringite was found in the north-western part of the pond where the sediment and water pH as well as water Ca and sediment SO_4 -concentrations were found to be at a maximum. Ettringite formation at high pH values around 11 is a common phenomenon within fly ashes and is subject to the binding of toxic elements, such as Cd, As, and Cr (Loop 2004). It is also a major process in the SAVMIN and CESR treatment processes (Lorax Environmental 2003). Because the ettringite covers the substrate in the receiving stream and forms an insoluble hard layer, biological activity will be minimised, or even excluded. Therefore, ettringite formation can be seen as an indicator of an over-treatment of mine water rich in sulphate or Al, because in most circumstances pH values under 11 are enough to successfully treat iron rich mine waters. If necessary, the treated water has to be dosed with acid to lower the pH below pH 8.5 after the sulphate has been removed as ettringite.

11.4 Passive Treatment

11.4.1 Introduction

Several hundreds of passive treatment systems were constructed within the last two decades (Wolkersdorfer and Younger 2002; PIRAMID Consortium 2003), and a recent evaluation of 83 of such systems proved that most of those systems work well (Ziemkiewicz et al. 2003). Their functioning relies on natural chemical and biological reactions which occur in the constructed or maintained treatment system without adding nutrients other than compost or energy into the system (systems where nutrients or energy are repeatedly added after construction must be called enhanced passive treatment systems). Furthermore, the operational and maintenance costs for the systems are comparatively low, though the investment costs in developed countries are usually higher than for active systems. This fact is due to the larger land use of passive systems. Also passive prevention techniques, such as dry or wet covers or reducing gas mixtures fall under this category of treatment system (Elander et al. 1998; Skousen et al. 2000; Taylor and Waring 2001; Integrated Geotechnical Engineering Services Specialists in Unsaturated Zone Hydrology 2003). For the PIRAMID consortium (PIRAMID Consortium 2003), passive remediation is:

Passive *in-situ* remediation (PIR) signifies an engineering intervention which prevents, diminishes and/or treats polluted waters at source, using only naturally available energy sources (such as topographical gradient, microbial metabolic energy, photosyn-

thesis and chemical energy), and which requires only infrequent (albeit regular) maintenance to operate successfully over its design life.

Some of those systems are sometimes summarised under the term phytoremediation. Always, passive mine water treatment requires an intensive understanding of the mine water chemistry is imperative as will be shown in a later section. As has been argued by Kalin (2004), who takes up the cudgels for a more intensive understanding for bacteriological processes in water treatment, “nature’s repair mechanisms may be slow but they’re thorough. We must find ways to assist and expedite them”.

Possibly the first reported scientific investigations of passive water treatment options were conducted by the German researcher Käthe Seidel in the 1960s (Seidel 1966). She investigated 10 water plants with respect to their capability to remove contaminants from the water and concluded that only *Schoenoplectus lacustris* would be a potential candidate for further investigations (Tab. 20). Especially the ability to remove aromatic substances, particularly phenols, was an unexpected result at that time. One of the reasons for *S. lacustris* to influence the water quality in such a positive way is the ability of the plant to supply the water with oxygen. Because of her investigations a first pilot treatment plant was constructed in Urach/Germany which treated sewage waters of different sources. It could be observed that whichever pH the sewage water had (between pH 4.5 and 12) the pH of the system was buffered to a pH value between 6.5 and 8.

Tab. 20: Metal accumulation of water plants in wetlands. Only the “above ground” parts of the plants were analysed. Scientific names adapted to today’s names. Metal concentrations in g m⁻². (Seidel 1966)

Scientific name	Common name	Fe	Cu	Zn	Mn
<i>Schoenoplectus lacustris</i>	Common Club-rush	2.62	16.13	168.00	4,032.00
<i>Carex elata</i>	Tufted sedge	10.34	15.23	171.36	2638.4
<i>Iris pseudacorus</i>	Yellow flag	3.22	14.14	124.0	947.36
<i>Typha angustifolia</i>	Small reed mace	1.58	6.77	62.92	1,121.76
<i>Glyceria maxima</i>	Reed manna grass	2.70	11.65	151.84	1,218.88
<i>Phragmites australis</i>	Common reed	4.12	18.82	165.76	743.68
<i>Acorus calamus</i>	Sweet flag	1.55	6.56	60.8	612.8
<i>Sparganium erectum</i>	Bur reed	4.61	7.17	97.28	773.12
<i>Myosotis palustris</i>	Water forget-me-not	3.26	3.90	33.28	640.0
<i>Mentha aquatica</i>	Water mint	5.21	14.28	131.04	640.1

Based on the outstanding work of Seidel (1966), the Othfresen/Germany full scale passive sewage treatment plant was constructed in 1974 and is still in operation. The treatment method was called the “root-zone-method” (e.g. Kickuth 1977; Bucksteeg 1986; Ebeling 1986; Brix 1987a, 1987b; and section 11.4.4). Twenty years later, between the 1980s and 1990s, the first publications about passive mine water treatment for coal mine drainage appeared (Kleinmann 1990; Cohen and Staub 1992; Wildeman et al. 1993; Hedin et al. 1994a). In the meantime several hundreds of investigations were conducted and pilot and full scale schemes using this treatment option were constructed, mainly in the Anglo-Saxon world. A well described case study of a constructed wetland for mine water treatment was Idaho Springs’ Big Five Tunnel, which started operation in 1987 (Wildeman et al. 1993).

Mine drainage from abandoned underground mines can be passively treated in several ways (Tab. 21). According to Younger et al. (2002) those different treatment technologies can be categorised into Inorganic Media Passive Systems (IMP), Wetland-Type Passive Systems (WTP), and Subsurface-Flow Bacterial Sulphate Reducing Systems (SFB). Even the en-

hanced passive systems, such as systems using alcohol to support microbial activity, can be categorised hereunder (e.g. Mascher et al. 2004). A fourth group, not discussed in the above mentioned publications, is phytoremediation (e.g. Chaney et al. 2000). This term is used for all treatment types using plants as a means for pollutant reduction, consequently some wetland type passive systems fall under this category as well. Because the term comes from the treatment of soils it is – for good reasons – usually avoided by mine water researches and is discussed here only for completeness. The limitations and shortfalls of that technology for the de-contamination of highly polluted soils and waters have been known since a long time (e.g. Ernst 1996, 2004).

Tab. 21: Categories of Passive Mine Water Treatment (adapted after Younger et al. [2002] and other sources).

Inorganic Media Passive Systems	Wetland-Type Passive Systems	Subsurface-Flow Bacterial Sulphate Reducing Systems	Phytoremediation
Anoxic limestone drains	Aerobic wetlands	<i>In-situ</i> permeable reactive barriers for ground water	Aerobic wetlands
Oxic limestone drains	Compost wetlands	<i>In-situ</i> permeable reactive barriers for mine water	Compost wetlands
Closed system Zn removal reactors	Reducing and alkalinity producing systems (RAPS)		Algal mats
Siderite calcite reactors			deep-rooted hybrid poplars
SCOOFI reactors Pyrolusite process reactors			bioreactors rhizofiltration, phytoextraction, phytostabilisation, phytofiltration

Other than mine water from underground mines, the passive treatment of acidic or basic mining lakes is still a uncertain task. Even after years of research, especially on the different aspects in the Eastern German lignite mining areas (e.g. Peukert et al. 1982; Kaden 1997; Friese et al. 1998; Herzsprung et al. 1998; Rolland et al. 2001; Werner et al. 2001; Morgenstern et al. 2001; Karakas et al. 2003; Uhlmann et al. 2004; Meier et al. 2004), where enormous problems with acidic mining lakes exist, no general solution of how to remediate an acidic mining lake by passive treatment options can be given. Several methodologies – not only in Germany – were used for the *in-situ* treatment of acid mine lakes among them suspension of fly ash, limestone, the application of soda or dolomite powder, biological sulphate reduction, and large scale water management (Geller et al. 1998; Castro and Moore 2000; Grünwald 2001; Howell 2002; Koschorreck et al. 2002; Loop et al. 2003). Yet, many of those options are only temporal, have to be repeated on a regular basis and – strictly speaking – cannot be called passive treatment options. As has been described in section 5.5.2 open pit lakes are not the focus of this publication and shall therefore be excluded from in depth studies of their remediation potentials.

Until today, authorities, researchers, and mining companies sometimes believe that the different passive treatment systems are rendered interchangeable. Furthermore, they think that whatever mine water drains out of an adit or shaft whichever system can be used. Yet, and many examples prove that assertion, such an interchange is not possible! If an iron rich mine water of high acidity is treated in an aerobic wetland, the iron might be precipitated to a certain degree, but the pH of the water decreases due to the hydrolysis of the iron ion. This fact

can easily be seen from the governing equation (Stumm and Morgan 1996; compare page 30 for full reactions):



Though the iron is removed from the mine water, the pH decreases. There are even mine waters, where the iron does not precipitate to the expected degree when treated in an aerobic wetland. In such cases, money and land has or would have been wasted and – unfortunately – exactly those examples are regularly used by the opponents of passive treatment systems to show that passive treatment is not feasible in the real world of mine water treatment.

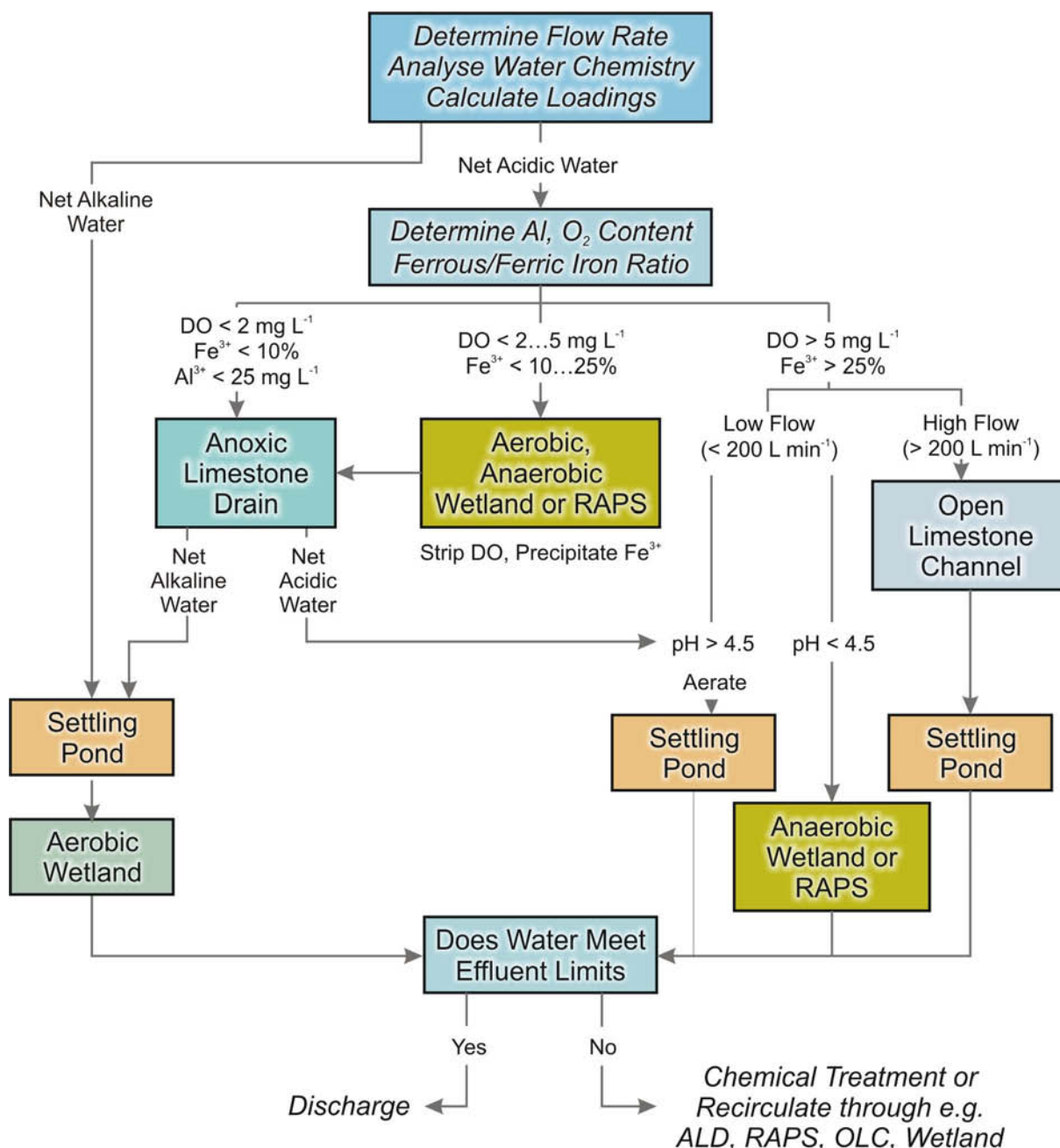


Fig. 89: Flow chart for selecting the most appropriate passive mine water treatment technology depending on the mine water parameters (DO: dissolved oxygen, RAPS: reducing and alkalinity producing system, ALD: anoxic limestone drain, OLC: open limestone channel; modified after Hedin et al. 1994a; and Skousen et al. 1998, 2000; a more sophisticated flow chart is given in PIRAMID Consortium 2003).

One of the key question is, how long such passive systems work until the substrate has to be changed or the precipitates have to be removed. So far, there is a lack of investigations in this direction, but the compost substrate of the anaerobic Quaking Houses treatment scheme seems to be used after approximately 10 years of work. For other wetland based systems a life time of about 20—30 years can be expected before the substrate and the precipitates have to be removed.

Because every mine water is as special as the mine from which it emanates, it needs special and careful investigation before a passive treatment plant is designed. Neglecting this prerequisite for passive mine water treatment might be one of the reasons that some systems do not work as expected (Johnson and Hallberg 2002). After at least a full hydrological cycle, the published flow charts for passive mine water treatment must be used and according to the type of water the most advantageous treatment option be chosen. Such flow charts were published and explained by Wildeman et al. (1993), Hedin et al. (1994a), Mc Cleary and Kepler (1994), Skousen et al. (1998, 2000), PIRAMID Consortium (2003), and Ziemkiewicz et al. (2003) and are summarised in a simplified form in Fig. 89.

11.4.2 Open Limestone Channels

Open limestone channels are rather simple constructions for treating mine water. Currently, two constructive methods are used: ditches filled with limestone through which the drainage water flows or open channels or streams in which limestone blocks of reasonable size are positioned. In both cases the mine water is allowed to flow freely through or over the limestone, thus raising the pH and adding alkalinity into the acidic mine water. To prevent the metal precipitates from coating the limestone, the water velocity must be high enough to keep the precipitates in suspension and to transport them away from the limestone blocks. For a long term performance of the system, large quantities of limestone are needed and the treatment efficiency must be monitored on a regularly basis.

Yet, if the flow velocity is too high and the amount of limestone used is too low, the contact time will be too short and the treatment effectiveness will not be too good. Furthermore, there are special mine waters which cannot be treated with open limestone channels because their chemical composition is too complicated (e.g. Gernrode/Harz; Simon 2003). At the Mina de Campanema/Minas Gerais/Brazil a several decametre long open limestone channel has been constructed to treat the acidic mine water after flowing through a large settling pond. As can be seen from Tab. 22, iron precipitates during the flow through the channel, but the treatment efficiency concerning the acidity of the mine water can be seen as zero, as no alkalinity is added to the mine water nor is the pH increased. The lowering of the pH from 6.4 to 6.2 is

Tab. 22: On-site parameters of the Mina de Campanema/Minas Gerais settling pond and open limestone channel. Inflow 1 is not influenced by the mine, while inflow 2 comes from the mine site. The water then flows from the outflow of the pond through the open limestone channel.

	Inflow 1	Inflow 2	pond	Outflow pond	Outflow limestone channel
pH	5.0	5.5	7.3	6.4	6.2
cond, $\mu\text{S cm}^{-1}$	8.8	19.5	9.3	55.1	20.3
TDS, ppm	5.6	12.4	5.9	35.4	12.8
redox, mV	502	451	432	250	327
temp, $^{\circ}\text{C}$	25.6	21.9	25.1	24.2	24.4
Fe_{tot} , mg L^{-1}	0.22	—	—	6.53	0.72
Fe^{2+} , mg L^{-1}	0.10	—	—	2.92 (45%)	0.61 (85%)
Fe^{3+} , mg L^{-1}	0.12	—	—	3.61 (55%)	0.11 (15%)
alkalinity, mmol L^{-1}	—	—	—	0.10	0.04
acidity, mmol L^{-1}	—	—	—	0.18	0.10

due to iron hydrolysis which produces three moles of protons per mol of iron.

11.4.3 Anoxic Limestone Drain (ALD)

Anoxic limestone drains are constructed and buried channels filled with high quality limestone (Skousen 1991; Hedin et al. 1994b). Thus, the mine water flowing through the channel is prevented from getting oxidised which would subsequently lead to the precipitation of metal oxides and a coating of the reactive limestone. It is essential that the mine water flowing through the constructed channel has negligible oxygen, Fe^{2+} , and Al contents. Oxidised mine water shall never be allowed to flow through the anoxic limestone drain, as the limestone will be coated within a relatively short time. Such a situation exists at the Lehesten/Thuringia passive treatment plant, which, consequently failed to work after some years (the plant is described in Janneck and Krüger 1999; Rothenhöfer et al. 2000). Within the covered limestone bed, the mine water reacts with the carbonate to generate alkalinity and to consume acidity. Usually, anoxic limestone drains are a pre-treatment for acidic and anoxic mine water to rise the pH before it enters an aerobic constructed wetland. They are also used in combination with RAPS-systems at highly contaminated mine water sites (Barton and Karathanasis 1999).

Their lifetime is based on the amount and quality of the limestone and is given to range between 15 and 20 years. Mine waters with high aluminium concentrations and bad maintenance can cause such systems to fail even after less than a year (Robbins et al. 1999; Watzlaf et al. 2000; Ziemkiewicz et al. 2003). If coating of the limestone is expected, such systems can be equipped with a flushing mechanism which enhances the lifetime of an ALD (Vinci and Schmidt 2001).

No oxygen measurements are reported for the first German passive treatment plant for acid, Al-rich mine water which started operation in 1997 (in Lehesten/Thuringia). According to Janneck and Krüger (1999) the pH of the treated water ranged between 4 and 5, the Al between 8 and 11 mg L^{-1} , Mn between 2 and 4. Fe, acidity, and oxygen contents have been reported during a workshop (*Wasserbehandlung und Rückstandsentsorgung – Konventionelle und innovative Lösungen*, Chemnitz/Germany, September 24th to 26th 1997) but they are missing in the official publications. Fe ranged between 0.1 and 10 mg L^{-1} , acidity ($K_{\text{B8.2}}$) between 1.5 and 2 mmol L^{-1} (75–100 $\text{mg L}^{-1} \text{CaCO}_3$) and the oxygen saturation was $\approx 100\%$. The pilot plant is constructed at a rivulet downstream an acid releasing waste rock pile (Janneck and Krüger 1999; Rothenhöfer et al. 2000) and therefore the water flowing into the ALD is saturated with oxygen. After a period of good treatment results the system failed to treat the water because the limestone in the ALD was coated with Fe- and Al-precipitates and the flushing mechanism failed to work.

11.4.4 Aerobic Constructed Wetlands

Passive treatment of polluted water with constructed wetlands is known since 1974, when the first of such systems was installed in Germany (Brix 1987a, 1987b). This system has been initiated by Reinhold Kickuth for treating domestic sewage from the town of Othfresen between Goslar and Salzgitter in Lower Saxony (Kickuth 1977) and is based on the outstanding research of Käthe Seidel (Seidel 1966). Since then the system operated successfully and treats domestic waste water for a population equivalent of 5,100 people as well as industrial water from a sausage factory. In the meantime more than 600 systems of that kind were installed, among them at British Steel, British Aerospace, Huntsman Corporation, Zurich Airport; Berlin's Schönefeld Airport, Daimler Benz, Cerestar Corporation and an Exhibition at the 2000 World Expo in Hannover, Germany (Abydoz Environmental, Canada, pers. comm.). An outline of the processes involved in wetlands has been given by Walton-Day (1999).

Aerobic wetlands, or reed beds, are constructed or natural ponds planted with typical wetland plants such as common reed (*Phragmites australis* (CAV.) TRIN. EX STEUD.; U.S.: giant reed, German: Schilf), Reedmace (*Typha latifolia* L.; U.S.: cattail, German: Rohrkolben), common rush (*Juncus effusus* L.; U.S.: soft rush, German: Binse), or others. Though those plants take up metals and contribute significantly to the polishing process (Batty et al. 2002; Batty and Younger 2002, 2003), they are not meant to treat the mine water by up taking the pollutants and being harvested thereafter. Primarily, those plants reduce the velocity of the water flowing through the system and prevent shortcuts, thus leaving enough time for the metal-microorganism-oxide-reactions (Brix 2003). Consequently the plants should not be planted in straight lines as has been the case in the Germany Pöhla/Saxony pilot scale scheme. Metals are precipitated by microbial catalysed oxidation as oxides and hydroxides and co-precipitation. For an effective operation of the wetland the mine water must be net alkaline, which means that the mine water's alkalinity exceeds its acidity ($K_{A4.3} \geq K_{B8.2}$). Mine waters with a pH lower than approximately 5.5 shall never be treated with an aerobic wetland as this leads to even lower pH-values. To increase the wetland efficiency, the CO_2 of the mine water should be stripped off and the O_2 -content increased. This can be done by cascades, riffles, or falls before the mine water enters the wetland which further increases the wetland efficiency (Novak 1994; PIRAMID Consortium 2003).

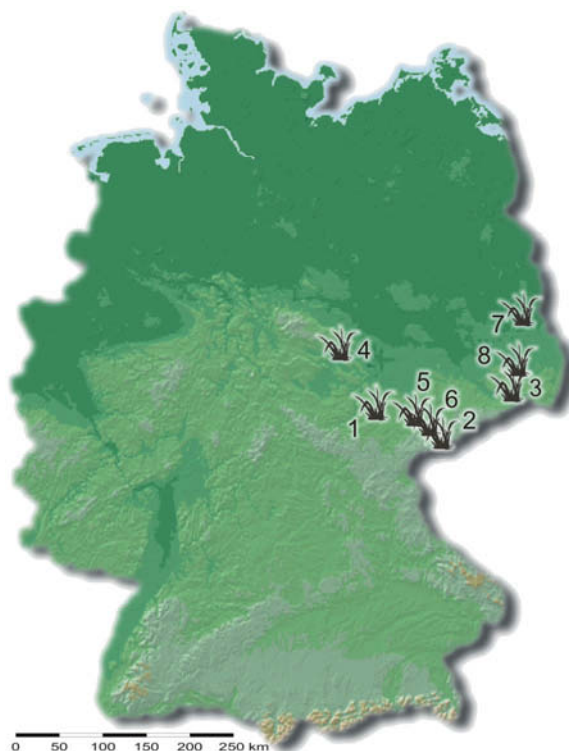


Fig. 90: Map with the sites of constructed or natural wetlands investigated/used for mine water remediation in Germany (end of 2005). References and numbers see Tab. 23.

A slightly different wetland treatment option is employed in free floating macrophyte and algal systems as they are used in Canada. In those systems, the plants do not grow within the wetland substrate, but they freely float on the water's surface. Pilot scale investigation as well as full scale treatment plants proved that those systems can be effectively used to treat polluted mine water (e.g. Kalin et al. 1994; Fyson et al. 1997).

Though aerobic wetlands are very common in the world and the technology emanated from Germany, this country has only a few of such wetlands under construction or in operation (Tab. 23; Fig. 90). Over and above, the knowledge gathered in the international community

was mainly used from the literature without effectively co-operating with the already gathered experiences.

Aerobic constructed wetlands are very effective in treating iron-rich mine waters, if the pre-requisites summarised in Fig. 89 are met. Currently, manganese, which very often accompanies high iron concentrations in the mine water, can only be treated in such systems when special conditions apply. In the case of manganese, the water's pH must be circum-neutral to basic (Walton-Day 1999) and at the outlet of the wetland, the pH should be increased through an open limestone channel (Bob Kleinmann, pers. comm.). With respect to the sludge precipitated in aerobic and anaerobic wetlands their density is usually about 10 times higher than for sludges from active treatment operations (Dempsey and Jeon 2001).

Tab. 23: List of German constructed wetlands. Except Winkler (2001) the authors ignore the pioneering European work of Younger. N.N.: not known.

Site	Site Name, mine type	Date Completed	Mine water type	Passive system type [area, m ²]	Typical influent	Typical effluent
1	Lehesten Thuringia shale mining	Start: Dez. 1997 (out of work at the moment), Janneck and Krüger (1999); Rothenhöfer et al. (2000)	Net acidic Al-rich	ALD, settling pond, aerobic cell [130]	pH $\approx$ 4.5 13.9 mg L ⁻¹ Al 3.43 mg L ⁻¹ Mn 1.24 mg L ⁻¹ Zn	pH = 7—8 0.035 mg L ⁻¹ Al 0.113 mg L ⁻¹ Mn 0.042 mg L ⁻¹ Zn
2	Pöhlitz Erzgebirge uranium	Start: July 1998; N.N. Kießig et al. (2004)	drift and underground mine	Aerobic and anaerobic pilot system; build in a former concrete settling pond [300]	pH $\approx$ 6.9 0.09—0.15 mg L ⁻¹ U 2—3 mg L ⁻¹ As 4.2—5.3 Bq L ⁻¹ Ra 9—11 mg L ⁻¹ Fe 0.6—1.1 mg L ⁻¹ Mn 5 mg L ⁻¹ SO ₄ ²⁻	0.1 mg L ⁻¹ U 0.34 mg L ⁻¹ As 1.3 Bq L ⁻¹ Ra 0.3 mg L ⁻¹ Fe 0.3 mg L ⁻¹ Mn
3	Helmsdorf uranium tailing	N.N.	Acidic drainage water from tailings dam	reconstructed natural wetland [N.N.]	N.N.	N.N.
4	Straßberg Harz flourite	Investigated; Winkler (2001)	Slightly net acidic underground mine	Aerobic and anaerobic pilot system; settling pond, open channel, compost wetland, 2 aerobic cells [1,700]	pH $\approx$ 6.5 $\approx$ 25 mg L ⁻¹ Fe $\approx$ 8 mg L ⁻¹ Mn $\approx$ 8 mg L ⁻¹ F	N.N.
5	Paitzdorf Saxony uranium	Under construction; Kießig and Hermann (2000)	N.N.	Anaerobic pilot system [N.N.]	N.N.	N.N.
6	Schlema uranium	Experimental; Kunze and Küchler (2003)	N.N.	N.N., possibly aerobic and anaerobic wetland [N.N.]	N.N.	N.N.
7	Lausitz (Laasower Fließ) lignite	Experimental; Pietsch and Schötz (2004)	Net alkaline	Aerobic, anaerobic wetlands [small]	pH $\approx$ 7 $\approx$ 23 mg L ⁻¹ Fe $\approx$ 1 mg L ⁻¹ Mn $\approx$ 500 mg L ⁻¹ SO ₄ ²⁻	pH $\approx$ 6,4 $\approx$ 1 mg L ⁻¹ Fe $\approx$ 0.5 mg L ⁻¹ Mn $\approx$ 570 mg L ⁻¹ SO ₄ ²⁻
8	Lausitz (Grünwalde RL 78) lignite	Experimental; Pietsch and Schötz (2004)	acidic	Modified natural aerobic wetland [100]	pH $\approx$ 3 $\approx$ 70 mg L ⁻¹ Fe $\approx$ 1200 mg L ⁻¹ SO ₄ ²⁻	pH $\approx$ 5 $\approx$ 10 mg L ⁻¹ Fe $\approx$ 300 mg L ⁻¹ SO ₄ ²⁻

11.4.5 Compost (Anaerobic) Constructed Wetlands

Compost wetlands (sometimes called anaerobic wetlands or sulphate reducing bioreactors) comprise of a layer of organic material which should be 0.3—0.6 m thick and have the same type of wetland plants as described for aerobic wetlands. In the first instance, compost and aerobic wetlands look very similar to each other, but their functioning is fundamentally different. While aerobic wetlands are surface flow wetland types, where the water flows above the substrate, compost wetlands are horizontally flow through systems, where the water flows through the compost layer.

Microorganisms in the substrate and the reducing conditions therein reduce the ferric iron to ferrous iron and reduce the sulphate to sulphide. Furthermore, as the protons in the mine water are consumed by the decomposition of the organic material acidity is removed from the mine water.

Though the first compost wetlands were constructed with spent mushroom compost (Hedin et al. 1994a) and a lot of research has been conducted with using type of organic material (e.g. Stark et al. 1994, 1996; Manyin et al. 1997; Nairn and Mercer 2000; O'Sullivan et al. 2004), all kinds of organics which release alkalinity into the mine water are suitable for compost wetlands (PIRAMID Consortium 2003), thereunder:

- spent mushroom compost (e.g. Hedin et al. 1994a)
- horse manure and straw (e.g. Younger 1997)
- cow manure and straw (e.g. Cohen and Staub 1992)
- composted municipal waste (e.g. Jarvis and Younger 1999)
- composted conifer bark mulch (e.g. Younger 1998)
- sewage sludge cake (e.g. Laine and Jarvis 2003)
- paper waste pulp (e.g. Laine and Jarvis 2003)

Whichever carbon source is used, the governing reactions within the substrate are equal to each other, though the first three in the list proved to be better than the latter ones.

One important factor of a compost wetland is its permeability. If the permeability of the substrate is too low, the mine water will not flow through the substrate but overflows it. If that happens the microbial catalysed reducing reactions will not occur and the treatment efficiency will decrease. Studies about the hydraulic conductivities were conducted by Bolis et al. (1992). Their results for manure in different conditions ranged between 10^{-6} and 10^{-2} cm s⁻¹, which is also important to know when constructing RAPS systems. Another important issue is the water depth. If it exceeds approximately 15 cm, little exchange between the water and the substrate occurs, thus reducing the treatment performance (Paul Younger, pers. comm.). Gusek (2002) and Gusek and Wildeman (2002) describe several successful aerobic and aerobic systems and the advantages of those systems even in cold climates. Furthermore, he remarks that “a phased design program of laboratory, bench, and pilot scale testing has been shown to increase the likelihood of a successful design”.

11.4.6 RAPS (Reducing and Alkalinity Producing Systems)

A RAPS (reducing and alkalinity producing systems, originally called SAPS – successive alkalinity producing systems, also called vertical flow ponds, vertical flow systems) combines the advantages of an anoxic limestone drain (ALD) with those of a compost wetland (Kepler and Mc Cleary 1994; Hedin et al. 1994a). In contrast to classical wetland type systems, the two reactive layers are constructed on top of each other and the water flows vertically through the system. Therefore, they are also referred to as vertical-flow wetlands (Nairn and Mercer

2000). According to theory, the acidic mine water first flows through the compost part of the RAPS system. In this layer, the sulphate is biologically reduced to sulphide, ferric iron to ferrous iron and pyrite forms. Then, the mine water, now being in anoxic conditions, flows through the anoxic layer of limestone and becomes alkaline. For mine waters with elevated Al-concentrations it is advisable to construct an 'aluminiumator', which can be used to flush the RAPS and remove, to a certain degree, Al-precipitates (Watzlaf et al. 2002). Currently, about 100 RAPS-systems are installed worldwide and treat acidic mine water of different quality (Edwards et al. 1997; Younger et al. 2002; Brown et al. 2002; Wolkersdorfer and Younger 2002; Rose and Dietz 2002; Rose 2004; Thomas et al. 2004).

Because the chemical reactions within the limestone layer of a RAPS are similar to those of an ALD, the amount of limestone that is needed in a RAPS is calculated in the same way as for an ALD. Due to hydraulic reasons the thickness of the limestone as well as the compost layer should be around 0.5 m each. Details about the design practise can be found in the relevant literature about passive treatment options.

During the installation of the Bowden Close pilot RAPS system (Younger 2002c; Younger et al. 2004) it happened that the limestone and the organic material were mixed with each other (Paul Younger, pers. comm.). Though this fact was thought to act negatively on the performance of the RAPS it could be shown that the system performance was better than expected. Because, especially when large machinery is used, mixing of limestone and organic material cannot be excluded, there is no certain need to keep both substrates separated. Thomas and Romanek (2002) were able to show that such a mixing can be advantageous for some types of mine water. In the Gernrode/Germany experimental RAPS the limestone drain was only used to protect the inflowing drainage pipes from being blocked by precipitates, but above this protective limestone layer, the manure and limestone were evenly mixed with each other (Simon 2003; Hasche and Wolkersdorfer 2004). When constructing a RAPS it is essential to avoid any shortcuts between the inflow of the mine water and the outflow as this might result in a malfunctioning of the system as well as metal precipitation on the limestone surfaces. Furthermore, a sufficient head is necessary, as the water has to flow through a low hydraulic conductivity layer of compost. If the head is not large enough, as in the Pelenna II (Garth Tonmawr) system, then the water overflows the RAPS and the treatment efficiency equals zero (Rees and Connelly 2003; Fig. 91).



Fig. 91: Mine water overflowing the Pelenna II/Wales system. The hydraulic head is too low to drive the water through the compost (March 24th 2000).

A recent innovation was the construction of a RAPS in Deerplay/Lancashire/UK where the two reacting components, limestone and organic material, are neither built over each other nor mixed with each other, but constructed side by side. First, the acidic, iron and aluminium-rich mine water flows through organic substrate and then into a limestone bed, which is separated from the organic substrate by an impermeable barrier. To prevent oxygen from coming into the system, the organic reactor is isolated from the atmosphere by an impermeable cover (Jarvis and England 2002). An interesting result is the fact that the nitrogen concentration, which comes from the organic matter, decreases relatively quickly after constructing the RAPS. After a first peak at 10—20 mg L⁻¹ NH₃-N the concentration came down to 2—4 mg L⁻¹ NH₃-N in 2 month only and thereafter decreased to about 2 mg L⁻¹. Compared to the influent concentration this is roughly twice the concentration of the influent mine water (Fig. 92).

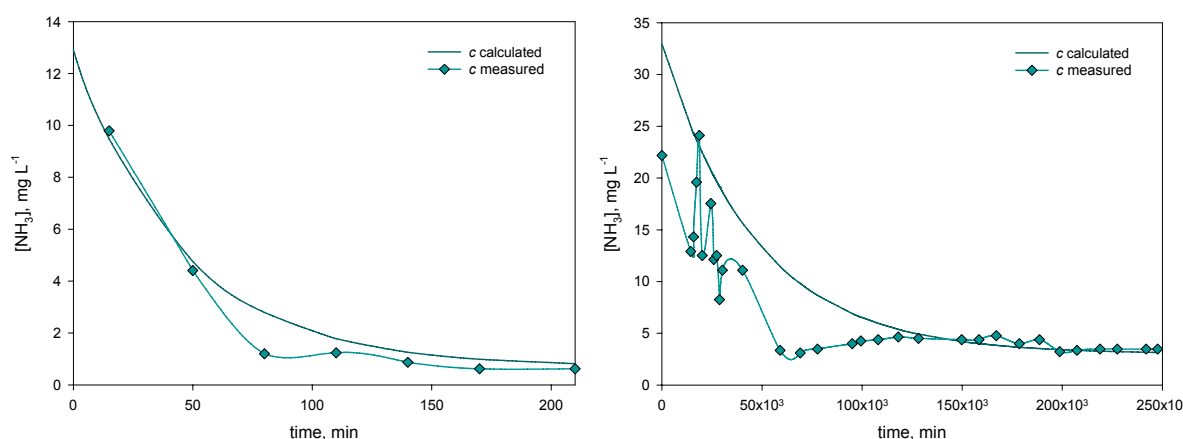


Fig. 92: Comparison of measured and modelled nitrate concentrations in a laboratory experiment (left) and the Deerplay/Lancashire/UK (right) full scale RAPS (Deerplay data from Jarvis and England 2002; modelled data from Simon 2003).

As the nitrate concentrations are always of concern for regulators and authorities when speaking about RAPS or compost wetlands, the reduction of nitrate has been studied experimentally and compared with real case values (Simon 2003; Hasche and Wolkersdorfer 2004). It could be shown that the nitrate concentration mainly depends on the volume of the compost V , an initial nitrate concentration factor f_1 , the background concentration c_{i0} in the mine water and the time t :

$$c_t = \left(f_1 - 80 \times \frac{Q}{V} \right) \times e^{-\frac{t}{b}} + c_{i0} \quad (85)$$

with c_t : nitrate concentration at time t , mg L⁻¹
 c_{i0} : background nitrate concentration of mine water, mg L⁻¹
 f_1 : initial nitrate concentration factor, mg L⁻¹ (usually 20—40 mg L⁻¹)
 Q : measured mine water discharge, L min⁻¹
 V : bulk volume of compost, L
 t : time, min
 b : $100 \times \left(\log \frac{Q}{V} - 0.2 \right) \times \begin{cases} (\log V)^3 & V > 1 \\ 1 & V \leq 1 \end{cases}$

Only one of those parameters, namely f_1 , has to be evaluated in a laboratory test, though the other parameters can be either measured in the field or are the result of the RAPS design. To

evaluate this initial nitrate concentration factor, the organic material has to be mixed with the mine water at a ratio of 3:5 and the nitrate concentration in the mine water be measured. The factor f_1 is then calculated by multiplying this concentration by 0.7 and rounding the result to the nearest mg L^{-1} . For the laboratory experiments f_1 was calculated as 33, while for the Deerplay/Lancashire/UK f_1 must have been around 22. The reason for this difference is to be seen in the maturation of the organic material. For the laboratory tests relatively immatured horse manure had to be used, as the supplier did not have matured manure available. It might be assumed that for the Deerplay RAPS more matured manure was used and thus the initial nitrate concentration factor was lower than in the laboratory experiments. Consequently, it is now possible to simply show the regulators after which time the nitrate concentrations in the mine water will come down to nearly the natural background values.

11.4.7 Reactive Barriers

Reactive “barriers” are subsurface engineering constructions within the flow path of the polluted mine water. The material within the barrier, which must be permeable, reacts with the mine water pollutants and decreases their concentration. Different materials, such as zero-valent iron, activated carbon, zeolites, iron oxides/oxyhydroxides, phosphates, or clay minerals were used so far. Usually they are constructed downstream of tailings or waste rock dumps but due to several failures of those systems in the past their practical application is not widespread so far. Yet, several investigations throughout the world give rise to the hope that such systems could be used more often than today (e.g. Benner et al. 1997; Younger 2000c; Schneider et al. 2002; Bolzicco et al. 2003; Younger and Elliott 2003; Younger et al. 2003; Jenk et al. 2003; Simon et al. 2003; Gibert et al. 2004; Roehl 2004; Munro et al. 2004).

So far, no reactive barriers were constructed within the downstream area of a polluted, flooded underground mine, though at least the construction of vertical subsurface limestone drains should be a promising option for large scale ground water treatment. “Reactive barriers” within a flooded underground mine will be described in the chapter about *in-situ* and on-site treatment options (section 11.6).

11.4.8 Phytoremediation

Phytoremediation is a commonly used term for all types of treatment technologies that include plants and the uptake of the pollutants into the plant (Cunningham et al. 1997; Adams et al. 2000). Enhancement techniques for the rhizosphere are becoming more and more prominent and future investigations have to make sure that the plant–rhizomes interaction is better understood to improve such enhanced biodegradation technologies. Originally the term comes from the remediation of contaminated soils where metal tolerant plants are used to accumulate pollutants in the plant material (e.g. PHYTO-DEC 5th EU framework project EVK1-1999-00024, EVK1-2002-00552) and some promising organisms show extreme accumulation rates (Bryan and Langston 1992). During the last decade phytoremediation has been extended to treat polluted mine waters. Since then the term can also be found in conjunction with mine water treatment. Yet, the technology is still as far away from large scale implementations as it was 10 years ago, when Wilfried Ernst wrote: “there is still a long way to go from the potential small-scale to a realistic large scale approach” (Ernst 1996). Especially for mine waters, where large amounts of metals have to be removed from the water, phytoremediation *sensu strictu* cannot be used as a successful technology – yet, because the methodology has large potentials for remote areas, research must go ahead to find optimised plant-root-bacteria communities which foster the uptake of potentially poisonous elements from the mine water. Some promising results of that kind of research and for elements with lower concentrations in the mine water were studied recently (Merten et al. 2004). For all cases where phytoremediation is also used as a method to stabilise soils and to minimise soil erosion phytoremediation

still is the method of choice (e.g. Cai et al. 1990; Baker et al. 1991; Chaney et al. 2000). Furthermore, it is an excellent method to increase the biodiversity of abandoned mine sites (Batty 2005).

Currently, the following three groups of phytoremediation are distinguished (only papers relevant to mine sites are listed):

- *phytoextraction*: (semi-)metal tolerant plants are grown and harvested with the intention to remove the accumulated elements from the soil or water (e.g. Chaney et al. 2000; Karathanasis and Johnson 2003)
- *rhizofiltration or phytofiltration*: (semi-)metals are adsorbed on the surfaces of roots or the whole plant. To remove the contaminants from the environment, the plants are harvested and deposited off (e.g. Dushenkov et al. 1997)
- *phytostabilisation*: (semi-)metal tolerant plants are grown on polluted sites to stabilise the pollutants in the soil and to minimise soil erosion at the contaminated site. Usually, the plants are not harvested (e.g. Stoltz and Greger 2002; Bleeker et al. 2003; Chan et al. 2003; Wong 2003).

The disadvantage of the above listed phytoremediation methods is that the clean-up of contaminated sites might take hundreds if not thousands of years and that annual harvesting as well as the disposal of the contaminated plants or – after their burning – ashes is imperative.

Because phytostabilisation reduces the amount of erosion on tailings, waste rock dumps, or decommissioned mine sites, this methodology is very promising for the fixation of pollutants within the soil. Thus, the pollutant load into the receiving water course and the subsequent watershed can be significantly reduced. If it is possible to enhance the metal tolerance of the plants, by manipulating the microorganisms living in symbiosis with the plants acid draining from the dumps into the ground water might be positively controlled. Such research, for example, has been conducted by Quinn et al. (2001). They planted 420 poplars with roots reaching into the contaminated aquifer at a depth of 8–9 m and a further 389 willows and poplars for soil and shallow ground water remediation.

Trees, depending on their age and type, can evaporate between 30–1,500 L per day, thus they take in a large amount of pollutants and can lower the ground water table (Adams et al. 2000). Consequently, the dissolution of metals from the upper soil horizons can be minimised and the transfer chain from the soil to the ground water can be stopped or at least hindered.

11.5 Natural Attenuation

Natural attenuation is the next step after active and passive treatment: however it is *no* treatment. Indeed, natural attenuation is – slightly exaggerated – “do nothing” and monitor as well as understand the processes occurring naturally in the environment. Only in cases where a severe negative impact on the anthroposphere of humans might be expected, natural attenuation will shift to enhanced natural attenuation or to one of the above mentioned treatment processes.

Natural attenuation comprises mainly of the following processes (Höglund et al. 2004):

- abiotic
- physical
- biotic

Some typical abiotic processes are the precipitation of iron or aluminium minerals because of chemical reactions. Very often, the precipitation of minerals such as jarosite, schwertmannite, ferrihydrite, siderite, and pyrite is due to changing physico-chemical parameters. Physical

processes might be the filtration of colloids or their settling during the course of the stream, while biotic processes might be the fixation on roots, branches or by microbiological activities.

Because natural attenuation just happens independently of human interference, there seems to be no need to discuss natural attenuation concerning mine water issues. Yet, cases of natural attenuation all over the world clearly show that this “treatment” option can be hampered by human activities or in the worst case consultants who want to remediate a polluted mine site destruct the basis for natural attenuation of mine water. At the German Gernrode/Harz Mountains fluorspar mine, polluted mine water discharges from an abandoned adit. A consultant straightened the drainage gully and thus prevented the water from flowing over an already established, natural cascade into a settling pond. The mine water is now directly diverted into a mountain stream without taking advantage of the already established natural attenuation potential of the cascade and the natural settling pond downstream of the cascade. An example for natural attenuation is the Buttonwood Tunnel discharge in the Eastern Pennsylvanian Wyoming Basin/USA. This drainage adit was regularly monitored between 1968 and 1980 (Ladwig et al. 1984). As can be seen, the mean sulphate concentration decreased from 2,130 mg L^{-1} in 1968 to 982 mg L^{-1} in 1980 and also the minimum and maximum values decreased continuously (Fig. 93).

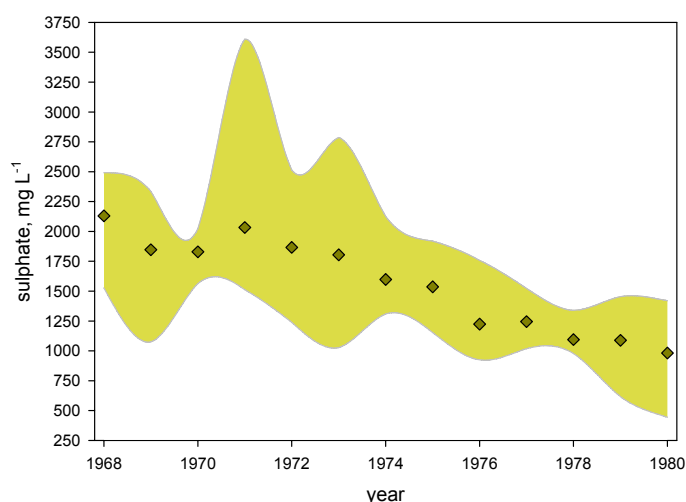


Fig. 93: Natural attenuation at the Buttonwood Tunnel, one of the main discharge adits of the Wyoming Basin in Eastern Pennsylvania/USA. The mean sulphate concentrations (diamonds) as well as the maximum and minimum values (shaded area) decreased continuously between 1968 and 1980 (modified from Ladwig et al. 1984).

Natural attenuation of mine water occurs as long as people were mining raw materials. It is not a new invention or a new technology introduced into mine water research, it is just a name for a process that occurs even without human interference. All the organisms mentioned in the previous chapter are the activists in this type of treatment. Beginning from some of the small extremophiles up to the large macrophytes providing nutrients or surfaces for microorganisms, as well as filter mechanisms for suspended solids or obstacles in the mine water flow play an important role in natural attenuation – and ironically most of the sedimentary raw material deposits and as far as we know many other metal deposits are a result of “natural attenuation” (Pohl 2005). Possibly the first mine water related paper using the expression “natural attenuation” appeared in 1994 (Kwong and Van Stempvoort 1994). They describe the natural reduction of Cu, Zn, and sulphate in acid rock drainage downstream of an open pit in central Vancouver Island/Canada by a natural wetland system. This was clearly not the first

work on “natural attenuation” – they were just the first naming that type of remediation in a mining impacted environment. One of the publications describing natural attenuation was Barnes and Clarke (1964): “It is well known that the acid streams recover in the course of travel as the iron precipitates and the acid is diluted and neutralised by contact with alkaline minerals in the streambed and with diluting waters”.

From observation it is known that nature is able to take care of itself and that natural attenuation has a large potential to treat mine water (e.g. Wolkersdorfer et al. 2003; Höglund et al. 2004). If the processes involved are understood it is possible to help nature to take care of itself or to avoid any interference which might prevent natural attenuation. At the abandoned Straßberg/Harz/Germany fluorspar mine, part of the drainage water is flowing through a natural wetland and the mine water chemistry is improved while it flows through the system (Neef 2004). The natural wetland did not exist before the mining at this site started in the 18th century. It is clearly the result of mine water seeping through leaky settling ponds, which were constructed to treat the pumped mine water. In the course of time an outstanding flora with wild orchids and wetland plants established and several decades ago the mining impacted site was declared a nature reserve called “Eisenwäsche”, which – ironically – means “iron wash” (Fig. 94). Due to this fact it was not possible to divert part of the mine water through this wetland as the authorities feared an irreversible impairment of the flora through the treatment of the iron enriched mine water.



Fig. 94: Natural attenuation occurring at the Straßberg/Harz mine site and the *Eisenwäsche* (“Iron wash”) natural wetland. Flocculants of iron hydroxide and a colourful film of iron-hydroxide/bacteria aggregates can be seen in and on the water.

11.6 *In-Situ* and At-Source Treatment Methods

In-situ and at-source treatment means that the source of contamination is treated essentially at its origin: within the flooded mine or the acid producing rock (Pearce and Ries 1982). It is a common method for preventing or abating the generation of acid rock drainage in overburden material, but has not found wide application in the treatment of the mine water of flooded underground mines. Theoretically, nearly every acidic mine water could be converted to alkaline water if only enough alkalinity would be added to the drainage system. Yet, past experiences demonstrate that the situation is not that simple and further investigations have to be carried out.

Before the application of alkaline material, an acid-base accounting must be conducted. Those calculations will result in a net neutralization potential (NNP) and when it is positive, this *in-*

situ method can be used with success. In addition, the most suitable place within the mine has to be found, preferably based on the results of mine water tracer tests, to guarantee a good mixing between the alkaline material and the acidic mine water. While the acidity load is positively influenced this must not be the case for the metal loads and the sulphate concentration. Detailed discussions of different *in-situ* methods and alkaline addition can be found in The Pennsylvania Department of Environmental Protection (1998), Skousen et al. (1998), Prediction Workgroup of the Acid Drainage Technology Initiative (2000), and U.S. Environmental Protection Agency et al. (2001). Note that the “Alkaline Addition” sections in U.S. Environmental Protection Agency et al. (2001) are nearly identical with Smith and Brady (1998).

Ideally, an *in-situ* or at-source treatment option would prevent the formation of acid mine drainage. Such techniques – besides flooding the mine – are known, and comprise of

- surface inactivation by capsulation or coating (Evangelou 1995; Fytas and Bousquet 2002),
- reducing the hydraulic permeability of pollutant bearing rocks (Ziegenbalg 1999; Waring and Taylor 1999; Ziegenbalg 2005),
- injection of chemically reducing gas mixtures (Taylor and Waring 2001; Houston et al. 2005),
- reactive material (“reactive barriers”),
- alkaline addition.

Capsulation and coating can be conducted with a range of substances, thereunder Fe-phosphate, silica, sodium silicates, or “simple” but effective clay caps (Brandberg et al. 1993; MEND [Mine Environment Neutral Drainage Program] 1994; Evangelou 1995; Roy and Worral 1999; Fytas and Bousquet 2002; Carlsson 2004). As the “oxidation proof silicate surface coating on iron sulfides” method is patented (U.S. Patent 5,494,703), it cannot be freely used! In addition, a proof of their successful usage in flooded underground mines is still pending.

Addition of alkaline material is most commonly used for reducing or preventing acid drainage generation from spoil or waste rock heaps (Prediction Workgroup of the Acid Drainage Technology Initiative 2000). Alkaline material that has so far be used to hamper the generation of acid mine drainage includes limestone, dolostone, coal combustion residues such as fly ash, caustic soda, and active treatment sludges (U.S. Environmental Protection Agency et al. 2001). By far the most commonly applied material is limestone and dolostone because they are widely available und usually of low cost (Prediction Workgroup of the Acid Drainage Technology Initiative 2000). If enough alkaline material is added, the acidity removal will be sustainable and as a general rule, those remediation projects where more alkalinity than previously calculated was added, positive results were obtained (Perry et al. 1997; Smith and Brady 1998). Alkaline material can also be deposited at potentially acid producing areas within the abandoned mine before flooding. Yet, the localisation of such zones might become problematic if those mine parts are not any longer maintained and therefore cannot be examined. Furthermore, compared to waste rock piles or tailings, the amount of acid producing material contributing to the acidity can only be estimated to a certain degree. Often, it will probably surpass the storage capacity of the underground space for the deposition of alkaline material. Yet, it should be reconsidered, if a controlled backfill with alkaline material before flooding would be effective to prevent the first flush or to reduce its extend. A cost comparison for every single case might be a feasible way to find an environmentally sound solution.

In the case of the abandoned Broken Aro coal mine near Coshocton Ohio/USA, a permeable mine seal with fixated flue gas desulphurization sludge was used. This sludge comprises of

35% calcium oxide, 21% sulphur dioxide, 10% silica and other oxides, thus containing a lot of potential alkalinity. In fact, the mine water quality significantly improved after the final installation of the plug and the receiving water course received up to 98% less contaminant loads at the time of construction (Rudisell et al. 2001). In the meantime, the site has to be reclaimed again, because the measures taken are not satisfactory enough to guarantee an ecological good quality of the Flint run watershed. In principle, such mine seals can be looked upon as reactive barriers. Investigations about reactive mine seals were also conducted by Penrose (1974). He investigated different limestone types and sizes for the optimization of a permeable plug mine seal and concluded that the acid discharge can be reduced if such “reactive” mine seals are used.

Technically feasible would be to pump the mine water out of the mine, adding alkalinity to the water stream and pumping the water-alkalinity mixture back into the mine. Because low density sludge from mine water treatment operations contains large amounts of alkalinity, this sludge can also be pumped back into the flooded mine workings. Experimental and modelled results clearly showed an increase in pH and, after an initial mobilization, a demobilization of iron and aluminium (Hansen 1998; Klinger et al. 2000a). The analytical data of the Straßberg/Germany and *Grube Einheit* Elbingerode/Germany mines clearly prove that the re-injection of the sludge does not pose negative environmental consequences to the receiving water courses. In contrast, the alkalinity of the mine water rose and thus the treatment gain increased. If the stratification in the mine water bodies of those two mines favours this effect was not studied in detail so far. A similar approach is conducted at the Spreetal Nordost/Germany open pit lake which receives the LDS sludges from the *Schwarze Pumpe* active treatment plant (see further ahead in this chapter).

One of the most obvious *in-situ* treatment options is adding alkalinity in the to be flooded mine, the waste rock, or overburden. The most common alkaline materials used are limestone, dolostone, or combustion ash, but other, alkalinity rich material has been used as well (Skousen et al. 1998). Beside the three German mines where alkaline water treatment sludge is pumped back into the underground mine or acid pit lake, only a few other cases are known where alkaline material is used to improve the water quality of a flooded underground mine. In 1984 phosphate rock dust (apatite), lime slurry, and NaOH were pumped into the sealed Beth Energy's underground Mine № 105W near Buckhannon in West Virginia/USA. In total, 120 t of lime and 568 m³ of NaOH were injected between September 1984 and June 1985. This treatment resulted in a decrease of the pollutant loads after the time of seal installation. Yet, those concentrations were still lower than pre-sealing conditions. Compared to conventional treatment costs, estimated to be about \$4.7 million (U.S.), the *in-situ* method had total costs of only \$1.3 million (U.S.). Because of that treatment the discharge pH had risen from 3.9 to 7.0 and has remained above 7.0 up to January 1986. Iron concentrations decreased from 133 mg L⁻¹ to ± 1 mg L⁻¹, sulphates from 2,500 to 1,000 mg L⁻¹, calcium from 385 to 120 mg L⁻¹, alkalinity from zero to 350 mg L⁻¹ CaCO₃, and acidity from 470 to 20 mg L⁻¹ CaCO₃ (Hause and Willison 1986). Even in 1989 the mine water quality, in regards of pH and acidity stayed at an acceptable level, although iron concentrations increased to 15 mg L⁻¹ (Aljoe and Hawkins 1993). Nothing, so far, has been published about the today situation.

Another mine, where *in-situ* alkalinity injection was applied to is the Sherman Helms № 2 mine in West Virginia/USA (Burnett et al. 1995). This mine comprises of a 115 m³ mine pool with an average flow of 4 L min⁻¹. Fluidised bed combustion by-products were used to improve the mine water quality and to prevent surface subsidence. Using a supersonic pneumatic stowing technique, a total of about 400 t of ash were placed into 4 boreholes. Yet, the water quality was not affected by the fly ash during that first investigation, as the fly ash did not get into contact with the mine pool; possible due to a misinterpretation of the hydrodynamics of the mine water. Meanwhile more effective injection techniques were established and the acid-

ity dropped by 83% (NRCCE – National Research Center for Coal and Energy – News: February 2nd 2005).

The Keystone State Park site in Pennsylvania/USA experienced a thorough hydrogeological investigation before alkaline injection. Keystone State Park hosts the abandoned Salem № 2 coal mine which was closed in 1954. A tracer test was conducted but no reliable data could be obtained from the first test, as no tracer arrived at the sampling points (Aljoe and Hawkins 1993). A second test was conducted at a later stage (Aljoe and Hawkins 1994; the mine's name is not reported in this paper, but from the drawings it is obvious that this mine is Salem № 2 as well) which gained positive results. In 1990 1,890 kg of NaOH (as a 25% solution) were injected into two boreholes of the Salem № 2 mine, which were thought to be enough to treat the whole mine water within the mine. Yet, 2 months after alkaline injection only a minor change of the sodium concentration occurred. Calculations of the recovery rate showed that only about 5–6% of the injected sodium mass passed through the drainage portal and the rest stayed within the mine. Also the chemical analyses of the mine water concerning acidity, total iron, manganese, and aluminium proved that no neutralization reactions occurred after alkaline addition. At the time of the publication in 1994 no reasons for this behaviour were available.

Even more hydrogeological and biological investigations were conducted at the Red Oak coal mine near Red Oak in Oklahoma/USA before the injection of alkaline fluidised bed ash (Canty and Everett 1998, 1999). An extensive tracer test, historical, and GIS-based site investigations were conducted to define the potential flow paths and the hydrochemical development of the acidic mine water within the flooded mine. From old mine maps a mine volume of $1.6 \cdot 10^5 \text{ m}^3$ was calculated with the mine pool having a volume of $1.1 \cdot 10^5 \text{ m}^3$. During the hydrogeological monitoring phase a mean pH of 4.4, an electrical conductivity of 1.2 mS cm^{-1} , and an acidity of $475 \text{ mg L}^{-1} \text{ CaCO}_3$ was observed. Finally, 418 t of fluidised bed ash could be injected into the abandoned mine voids through wells by high pressure pumps with $1.7\text{--}2.1 \cdot 10^3 \text{ kPa}$ at an average rate of $84 \text{ m}^3 \text{ h}^{-1}$ ($110 \text{ yd}^3 \text{ h}^{-1}$ in the original report were wrongly given as $28 \text{ m}^3 \text{ h}^{-1}$). After alkaline injection, the pH and alkalinity increased, and the concentration of Al decreased significantly, whereas most of the other parameters did not show significant changes. They concluded that only some of the goals were reached but with an increased alkaline addition an improved water quality in the receiving stream could have been reached. Based on their geochemical calculations an access amount of 2,000 t of alkaline fluidised bed ash would have been necessary.

A more sophisticated *in-situ* project was conducted at the Tide Mine in Pennsylvania/USA. The project consisted of three phases with different *in-situ* treatment options involved (Houston et al. 2005):

- addition of sodium hydroxide, a bromide tracer, and an organic carbon source into a recirculation scheme,
- carbon dioxide gas injection into the mine pool to suppress the generation of acid mine drainage
- passive alcohol addition to the mine pool to sustain sulphate reduction process

In fact, the *in-situ* method proclaimed by the demonstration study was neither an *in-situ* method nor was it a passive method, nor does the injection of carbon dioxide on the surface of the mine water suppress the generation of acid mine drainage. The chemicals were injected into the mine water on-site and recirculated into the mine pool, addition of alcohol into the mine water is definitely an active process, and the covering of the mine water with carbon dioxide does not stop the formation of acid mine drainage at the point of its generation: above the mine water table, in the disulphide bearing strata. Though the first results of the study

seem to be promising, the time period after the treatment is just too short to draw final conclusions and it seems as if the treatment conducted is not self-sustainable, as can be seen from the last measurements they conducted.

Fly ash treatment can also be applied to mining lakes. Especially in the Lausitz/Germany lignite mining area, several of those “in-lake” techniques were applied to acidic mining lakes with varying success (Benthaus and Nitsche 2005; Scholz 2005). Recently, investigations started to use CO₂, coal combustion ash and LDS slime to produce alkalinity and thus treat acidic mining lakes (Merkel 2005). Yet, the addition of alkalinity without simultaneous reduction of sulphide results in lesser success than the coupled acidity-sulphate reduction inhibited by bacterial sulphate reduction (Castro and Moore 2000; Koschorreck et al. 2002; Meier et al. 2004).

Remining very often incorporates the addition of alkaline material, either in the overburden, the mine backfill, or the daylighted mine environment (Hawkins 1995). By using acid-base accounting tests the necessary amount of alkaline material can be calculated (Skousen et al. 2002). If done properly, those techniques successfully reduce or avoid the formation of acid mine drainage.

As can be seen from the results of the Aljoe and Hawkins (1993) and Burnett et al. (1995) investigations, a thorough knowledge of the hydraulic situation of a flooded underground mine is crucial for choosing the correct locations of potential treatment methods. Several of the treatments described by them failed because the mine water did not come into contact with the injected alkaline materials. Therefore, mine water tracer tests should be conducted before injecting alkaline or reactive material into a flooded mine. The results of such a test can then be used to locate the correct injection sites for such a treatment approach.

Some experiments with *in-situ* treatment options were conducted within the Königstein/Germany uranium mine and other mine sites in Saxony. Those methods included the injection of chemicals into the uranium bearing sandstone to reduce the hydraulic conductivity of the rock matrix. Furthermore, water was led through large tanks filled with different types of reactive materials such as resins or wood chips. A very common material is zero-valent iron, which has very good treatment efficiencies and was used in the form of scrap iron in those tanks. Yet, the chemical reactions involved release hydrogen gas which has to be removed if underground explosions should be avoided. Another *in-situ* treatment option conducted in the Königstein mine was the injection of Ba(OH)₂·8H₂O, Na₂SO₃, water glass, and a precipitation inhibitor into the rock matrix. Thus, the formation of acid mine drainage in some selected parts of the mine shall be prevented (Jenk et al. 2005). Yet, besides large scale experiments none of those promising *in-situ* treatment options were used as a final solution for mine water treatment at the abandoned Königstein uranium mine; one of the reasons being possibly the large amount of mine water that would have to be treated – even after the partial “passivation” of some leached blocks (selected references are: Klinger et al. 2000b, 2002; Ziegenbalg 2005). An extraordinary simple *in-situ*/on-site treatment option was suggested by Merkel (2002). Based on geochemical calculations he concluded that mixing the mine water with unpolluted water could be a cheap measure to treat the large amount of acid mine water expected from that abandoned mine. It seems as if some people listened carefully some years ago and that even the mine operator found advantages about that “treatment” option (Zeißler et al. 2005). Yet, this measure does not reduce pollutant loads and is therefore not advised. Schneider et al. (2001) conducted small scale field experiments with promising results at two abandoned uranium mines. They used glass columns at those sites, which were filled with zero valent iron, iron and lignite, lignite, and waterworks sludge. Yet, the large scale application still has to prove that *in-situ* method.

Red mud, a residue from the aluminium production, and fly ash as a means for *in-situ* treatment was investigated for the abandoned Freiberg/Germany silver-lead-zinc mines. It was suggested to fill parts of the Reiche Zeche shaft (Fig. 15) with a reactive material to get a small reactive barrier within the shaft. Because a large amount of the polluted water flows through that shaft, it was expected that the mine water could be treated *in-situ* by flowing through that reactive barrier (Zoumis et al. 1999, 2000; Zoumis and Calmano 2000). Yet, what they disregarded is the reduction of the hydraulic conductivity of the shaft by installing a relatively impermeable reactive barrier. Consequently, the mine water would not any longer drain through the Reiche Zeche shaft but through one of the other shafts. In Zoumis (2003), finally, the author concluded that the barrier is unpractical for a large scale application because either the break through time is too short or the permeability of the barrier is too low.

A last case, yet being a flooded open pit lignite mine, is the abandoned Spreetal Nordost/Germany mining lake. This lake receives the residues of a mine water treatment plant at the *Schwarze Pumpe* industrial site, at a current annual rate of $2 \cdot 10^6 \text{ m}^3$. Because the low density sludge of this treatment plant contains large amounts of none reacted lime (CaO), the lakes pH currently ranges between 4 and 6, yet the buffer capacity in this lake is not high enough to keep this pH level stable after the sludge disposal would fall away (Wolfgang Rabe, pers. comm.). A research project at the TU Bergakademie Freiberg currently investigates the possibilities for a sustainable pH stabilisation in this abandoned mining lake (Merkel et al. 2005a; Merkel 2005).

11.7 Use of Mine Water

This chapter describes using treated mine water as well as untreated mine water during and after the mine's operation. Section 5.2, dealing with a similar subject, describes the use of post-mining areas on a more general term, though the two chapters partly overlap. A common post-mining use of abandoned mines, especially underground mines, are visitor mines. Their number goes in the thousands and because no special considerations – unless safety procedures – are necessary, they will not be considered here – which does not mean that they are economically unimportant as shown by the Schwaz/Austria visitor mine with more than 100,000 visitors annually (Palme and Ingenhaeff-Berenkamp 1993; Palme et al. 2002).

Usage of mine water mainly depends on its chemical and physical quality. If the quality criteria meet drinking water standards, mine water can be used as drinking water (Burbey et al. 2000; Teaf et al. 2006), and several hundred smaller and larger communities depend on such water supplies. It is advisory that regular monitoring is installed, to make sure that any changes in mine water are recognised at an early stage. One of the larger cities which has drinking water reservoirs in mines is Freiburg/Breisgau/Germany (Wirsing 1995). They are used if not enough of the normal drinking water is available. In total, several 100 mine water adits are used for drinking water purposes in Germany, among them for Bayreuth, Selbitz, Kassel, Bensheim (used as a back up system for emergency cases), Obernkirchen, Plettenberg (Thomas Krassmann, pers. comm.). More detailed descriptions of mine water for drinking water proposes in the Rhenish Schiefergebirge gives Stengel-Rutkowski (1993, 2002). He lists at least 49 abandoned mines of which the drainage water is used for drinking water purposes. Those mines are iron stone, coal, shale, and other metal mines with discharges between 1 and more than 80 L s^{-1} . Especially in Stengel-Rutkowski (2002), where the mines in the Taunus region are discussed, potential problems with drinking water supplies from underground mines are described: high arsenic (up to $1,4 \text{ mg L}^{-1}$) and iron contents (up to 18 mg L^{-1}) in a polymetallic underground mine and too low inflows in relatively dry periods. None of those problems occurred to that degree in the Lahn-Dill area. In the German Harz Mountains, the Teichtalstollen adit near Königshütte is used as a secondary drinking water supply, but a tracer test revealed a good hydraulic connection between a nearby abandoned

day shaft (Ahrendfeldschacht) and the drinking water gallery (Wolkersdorfer and Hasche 2003; Ohly 2005). Because the shaft is situated in farm land which is used for sheep grazing, it cannot be excluded that the drinking water might be affected by bacteria or other pollutants.

In the case of the St. Aidan's opencast coal site Leeds/United Kingdom polluted mine water was treated by a reverse osmosis plant for drinking water supply (Wilson and Brown 1997). It was the largest plant of that type treating a daily amount of about 20,000 m³ of mine water and was planned to work for only one year (Paul Younger, pers. comm.). Experiments for another mine water treatment plant with reverse osmosis were conducted in Poland at the Pokój coal mine with a daily treatment capacity of 2,500 m³ (Magdziorz and Sewerynski 2000). Huge amounts of mine water were also used in the Eastern German lignite mining areas. With respect to steadily increasing drinking water demands for East Germany Luckner and Hummel (1982) discussed quantitative and qualitative measures to guarantee the supply with drinking water in the area.

Underground mines are commonly used as spas, especially for people with diseases of the respiratory tract. One of the best known spas of that kind before the Second World War was Bad Schlema, a radon spa which was regularly visited by well-known personalities of that time (Böttcher et al. 1991). Not until a decade ago, and after nearly 50 years of uranium mining, this area became a well known health resort again. Yet, Bad Schlema does not use the mine water but the radon in the mine air as a remedy. Other German spas using mine water are Bad Suderode/Harz Mountains, Feengrotten Saalfeld/Thuringia, or Alexisbad/Harz Mountains.

The mine water from an adit at the *Bad in der Au* near Brixlegg/Tyrol was once used as a spa, but today the Ca-SO₄-water is bottled and sold as mineral water. More important is the Franziskabad in Bad Häring/Tyrol, which uses the sulphate rich mine water of an abandoned hard coal mine. The current spa water comes from an exploration bore hole which was sunk in 1951 to explore the coal and copper reserves (Angerer and Lechleitner 1893; Lechner 2003). Another spa, already known since the 17th century, was the Silbertaler Bädli in Vorarlberg/Austria, flowing out of an abandoned medieval copper mine and closed in 1905. There, the water was an iron-copper sulphate water (Vogt 2001).

Yet, one of the most prominent European spas associated with underground mining is Bad Gastein in Austria. Its 19 thermal springs, known since the 14th century, have already been described by Paracelsus and the list of well known persons searching for health is long, thereunder kings, politicians and Pope Benedict XVI. Documented gold mining in the area dates back to early medieval times (Ludwig and Gruber 1987) and the most recent mining activities were conducted in the 1940s. The latest development was the *Gasteiner Heilstollen* (formerly *Paselstollen*), a radon health gallery, in Böckstein near Bad Gastein. During the Second World War, the deposits of the Hohe Tauern were exploited and it was found that in one part of the *Paselstollen* adit elevated rock temperatures occurred and mine workers reported about positive health effects while they worked in the mine. After the War, the *Paselstollen* was investigated and it was found that it can be used for recreational purposes and since the 1950s it has been used and added positively to the already known thermal and mineral waters of the Gastein spa area. Today, the *Gasteiner Heilstollen* can be used by up to 600 people and the air temperature ranges between 37 °C and 41 °C. Alike Schlema, radon is the reason for people coming to the Gasteiner Heilstollen, but also the water from the mine and the famous thermal springs are used (Hayek and Wierer 1951; Exner 1952; Ludwig and Gruber 1987; Zötl 1995; Sohrabi 1998; Zötl 2001; Franz Xaver Rieser, pers. comm.).

Mine water is relatively often used to drive turbines for the mine itself (Döring 1993) or for the surrounding communities (e.g. Max-Braun-Adit, Biberwier/Tirol; Wolkersdorfer 1991). The latter one, though a very small engine with less than 50 kW, is still in use and was

planned at the end of the 19th century, and thus belongs to one of the earliest turbines that were used for producing electricity. Two of the big German installations, one in Clausthal and another in the Freiberg mining area were abandoned in the late second half of the 20th century (Döring 1993; Galinsky et al. 2001).

The usage of treated or untreated mine water for recreational purposes is manifold, and only some outstanding cases shall be described here. The Passagem de Mariana Gold mine, Minas Gerais/Brazil once was a very productive gold mine but after the death of the last owner it was sold and since then it is used as a visitor mine for divers and as an underground pool. As in many gold mines of Minas Gerais, the arsenic content is very high and reaches 1.7 mg L^{-1} (Williams 2001) – which apparently does not deter people from swimming in the water (Fig. 95).



Fig. 95: Pupils swimming in the arsenic contaminated mine water pool of the Passagem de Mariana/Brazil Gold mine.

Another use of mine water is for power plant cooling and power plant process water. As long as the mine water is of good quality and does not interfere with the technical installations of the cooling system, it can be used for that purpose. Veil et al. (2003) investigated mine water in Pennsylvania and West Virginia/USA, where more than 120,000 abandoned mines and consequently a large number of flooded mine pools are available. In this area six power plants are already using mine water for cooling and process purposes. Among them is the Tremont 31-MW block having started to use the treated Lyken mine discharges in 1986. They also report about the proposed Maidsville 600-MW coalfired power plant of GenPower LLC (known as Longview Power Plant and intended for commercial operation in mid 2007). GenPower is going to use $1,600 \text{ m}^3 \text{ h}^{-1}$ of treated mine water from the Shannopin mine as a source of cooling water, whereas the treatment will be conducted by a public/private partnership of Pennsylvania Department of Community and Economic Development, the Pennsylvania Infrastructure Investment Authority, AMD Reclamation Inc., and Dana Mining. Treated open pit water is used as cooling and process water at the German Jänschwalde, Boxberg, and *Schwarze Pumpe* coalfired power plants. In 1999 $21 \cdot 10^6 \text{ m}^3$ of treated mine water were used for that purpose (Arnold and Baumgarten 1996; Lausitzer Braunkohle Aktiengesellschaft 2000). At the South African Tutuka Power Station, Eskom mine water that is treated by a spiral reverse osmosis plant is used as cooling water (Buhrmann et al. 1999). Thus, the company was able to solve their water problem and the environmental problems of a mine water discharge.

In some areas with a shortage of irrigation water, treated and untreated mine water is used for irrigation purposes (e.g. Du Plessis 1983; Voznjuk and Gorshkov 1983; Annandale 1998; An-

nandale et al. 1999, 2001; Jovanovic et al. 2001; van Zyl et al. 2001; Annandale et al. 2002; Jiries et al. 2004). Most of those studies were conducted in South Africa due to the climatic situation in this area of the world. Mine water use for aquicultural crops, e.g. sugar beans and wheat, is obvious but the long term effects on the ground or surface water have not been studied in detail. Soil balance models and weather data generators were used to model the long term impacts of mine water on the soil functioning. Jovanovic et al. (2001) describe the modelled precipitation of gypsum in the soil profile after an eleven year irrigation cycle, and therefore, the desalination procedures necessary for irrigation water of that type are therefore becoming more important (van Zyl et al. 2001). In another investigation Annandale et al. (1999) modelled a 30-year irrigation cycle where they confirmed the gypsum precipitation in the soil and thus stated that the soil is an effective sink for the salts in the mine water with a low potential of remobilisation. As a high amount of the precipitates is immobile they conclude that using mine water for irrigation “should be an effective and economical means of utilising large quantities of gypsiferous water without causing irreparable damage to soil resources”. Though this is correct for the chemical effects of the precipitates it is not clear what long term effect the reduction of the soils hydraulic conductivity due to the blockage of the soil pores might have.

Mine water has also been used for heating and cooling purposes. In the Henderson Molybdenum Mine, one of the world’s largest molybdenum mines, the ventilation air was heated with heat exchangers from mine water (Jensen 1983). The Freiberg/Saxony Hospital uses mine air from a 2.7 km long abandoned mine gallery within their ventilation system. This method was chosen as the temperature of the mine air is constant throughout the whole year and has been proven to be microbiologically clean and dust free. Though mine water is not directly connected to the Freiberg ventilation system, the energy needed to evaporate the mine water in the gallery is one reason for the cool temperature of the mine air (Weißenborn and Benthin 2003).

Another use of mine water is the recovery of metals from the highly enriched acidic waters. One of the first reports about metal recovery from mine water is given by Lai (1980). In his work, copper was extracted from synthetic mine water through a liquid membrane. Wright et al. (1995) also describe a successful on-site experiment with unsupported liquid emulsion membranes for the extraction of copper in an Arizona/USA copper mine. In all experimental set-ups an extraction rate of more than 90% could be achieved. The Mynydd Parys mine site in Anglesey/Wales (compare section 5.7.4) is another mine where metals were extracted from the mine water (Boult 1996). A similar experiment with liquid emulsion membranes was conducted in Chile and resulted in similar recovery rates (Valenzuela et al. 1999). This experimental situation was modelled by Valenzuela et al. (2002) with a good fit between the experimental results and the numerical model. REEs were extracted through a combined extraction/electrostatic pseudo liquid membrane process by Yang et al. (1999). All those before named studies were only on laboratory or on the experimental field scale. Different from this is the situation at the Hungarian Mecsekérc uranium (Pécs/Hungary) mine water treatment plant. There, mine water polluted with radionuclides is treated and through a membrane process uranium is recovered from the treated mine water. In a second step uranium oxide is produced and sold to the world market (Benkovics et al. 1997; Gombkötö et al. 2002; Csövári et al. 2004).

11.8 Ground Water Protection

One of the most obvious characteristics of polluted mine water are ochreous precipitates emanating from mine portals. Their worldwide number goes into the hundred thousands and even the inexperienced individual might notice that “something is wrong” when he or she passes by such a surface stream or pit lake. Yet, the real challenge for the future is the protection of

ground water and aquifers from polluted mine water. Is the degree of surface stream pollution by polluted mine water yet unclear, this is even more true for aquifers and even drinking water reservoirs. Though several publications about mine water impacted ground water reservoirs appeared during the last decades, a full coverage of ground water protection in the vicinity of abandoned or working mines is not in sight.

Nearly every mining operation impacts the natural ground water system by either lowering the ground water table, polluting the ground water with chemical compounds or suspended solids, changing the ground water flow path or by just changing the natural flow rates. Because the treatment of aquifers polluted with acidity, alkalinity or metals is not as simple as the treatment of water discharging from shafts or mine portals, the protection of the aquifer has to be given special consideration. Unfortunately, very often the treatment of polluted ground water is in some way technically and economically unaffordable. On the other hand, due to the low flow velocities of ground water in low permeable rocks, natural attenuation plays an important role if aquifers are affected by polluted mine water. Yet, in the case of karst aquifers or fissured aquifers the situation might become problematic, as the natural attenuation processes are too slow or not effective enough.

Because the potential pollution of ground water by mining activities is obvious, many procedures exist to predict that contamination. Both, geochemical modelling and reactive transport codes are used to predict the degree of impact on the aquifers and most of the models used are sophisticated and gain reasonable good results (see chapter 8). It is therefore interesting to see that the potential solutions to protect the ground water from mining induced pollution are limited and only seldom used. Even the authors in Teaf et al. (2006) restrict themselves to list potential dangers and potential treatment options but do not give case studies where ground water was treated or protected from underground mine drainage.

Several options would be possible to prevent the pollution of aquifers around a mine or to treat polluted ground water:

- *in-situ* treatment of mine water
- permeable reactive barriers
- pump-and-treat options
- encapsulation

None of these options has received wide-spread application in the mining industry or in the remediation of abandoned mine sites, though some case studies are of course known. *In-situ* treatment options have already been described in section 11.6. One of the more frequently used options are reactive barriers, probably the first one installed at a mining location being the Nickel Rim mine site near Sudbury, Ontario/Canada (Benner et al. 1997). In Europe at least three reactive barriers were installed at mining locations: at the Aznalcóllar/Spain site (Bolzicco et al. 2003; Gibert et al. 2004), the Pécs/Hungary uranium mine (Gombkötö et al. 2002; Simon et al. 2003; Csövári et al. 2004), and the Shilbottle/United Kingdom mine spoil site (Younger 2002a; Younger and Elliott 2003). An experimental set-up with good treatment results was installed at the Königstein/Germany uranium mine (Klinger et al. 2000b, 2002; Jenk et al. 2003). The long term reliability of the systems still has to be validated and many researches describe problems with the decreasing permeability of the system after it became operational. Further investigations must ensure that those systems should at least work as long as the peak of the first flush lasts. Classical pump-and-treat measures for ground water polluted by mine water, as they are used in ground water remediation, are very uncommon in the mining field, though it is the standard procedure for many flooded abandoned mines. Encapsulation, though discussed for some mine sites, is mainly used for mine tailings or waste rock but not for large scale mine sites.

In the United Kingdom, potential negative impacts on the Magnesian Limestone Aquifer in County Durham, or the Sherwood Sandstone Aquifer have been reported (Younger and Adams 1999; Dumpleton et al. 2001). To prevent the pollution of the Basal Permian Sands aquifer a pump-and-treat operation was installed near Horden/UK (Coulton et al. 2004). One of the best known conflicts between save mine closure and the interest of protecting an aquifer used for drinking water purposes is the flooding of the German Königstein uranium mine, as the flooding might impact an aquifer which is used to supply the cities of Pirna and Dresden (Biehler and Falck 1999; Merkel 2002; Schreyer et al. 2002). Some potential remediation measures for the Königstein mine are described in section 11.6. Another, large scale, ground water pollution area is the German Lausitz lignite mining area with pH-values as low as pH 3 (Nixdorf 1999; Grünewald 2001; Rolland et al. 2001; Nixdorf and Deneke 2004; Fyson and Nixdorf 2005). Based on the assumption that all the ground water in the area is to some degree polluted a volume of $2 \cdot 10^{10} \text{ m}^3$ of ground water is affected. Even after more than 15 years of research and applying an unknown number of remediation techniques, the situation did not significantly improve from a ground water point of view.

One key procedure to protect ground water from polluted mine water is the sealing of waste disposal or tailings sites (see section 5.4.2). Covering such facilities results in a significant reduction of pollutant loads and therefore can be seen as an imperative measure to protect ground water reservoirs (e.g. Sheibach et al. 1982; Brandberg et al. 1993; MEND [Mine Environment Neutral Drainage Program] 1994; Cabral et al. 1997; Elander et al. 1998; Waugh et al. 2001; Carlsson 2004). Modern tailings dams, such as the Río Narcea El Valle and Carlés gold mines and the Aguablanca nickel mine (Asturias/Spain), are underlain by a geomembrane to prevent seepage water polluting the ground water. Such geomembranes, which can also be used for tailings impoundment, consist typically of HDPE or PVC. Another option to prevent or even minimise the impacts of tailings or waste rock material is to add alkaline material (Wisotzky 2001).

This is a good place to discuss the negative impacts of abandoned ISL (*in-situ* leaching or solution mining) mines on the ground water quality. Solution mining for uranium, copper, or sulphur significantly changes the aquifer because the acid, basic, or hot solution as well as the pumping alters the host rocks and mobilises metals and arsenic which might pollute nearby aquifers (U.S. Environmental Protection Agency 2000). Yet, there is usually not much open space to be flooded and therefore ISL mines are not really covered by the relevant subjects in this book. Two of the exceptions would be the Königstein/Germany and the White Pine/USA *in-situ* leaching mines, the first one for uranium, the other one for copper. Both mines started as a conventional underground mine and were converted to *in-situ* leaching mines thereafter and have already been discussed in earlier chapters or in the case studies hereafter (e.g. Lewis et al. 1997; Jakubick et al. 2002). A well investigated ISL site is the Stráž/Czech Republic uranium deposit that was mined from 1967—1996 and is intensively discussed in the literature about this mine site (e.g. Frydrych et al. 2000; Novák 2001; Smetana et al. 2002). Because of the experiences gained during the abandonment of those three ISL mines, modern mines of that type are planned and operated in a more environmentally save manner (Kalka et al. 2005).

In conclusion, the protection of ground water from flooded mine sites is to date not a common procedure as the engineering methods are not fully developed yet or the costs of preventing polluted mine water from impacting the ground water are not economically feasible. However, the protection of ground water from polluted seepage emanating from spoil heaps, waste rock piles or tailings dams is common procedure and used wherever aquifers have to be protected from negative influences.

12 Case Studies about Flooded Underground Mines

12.1 Introduction

In the following chapter 13 case studies for flooded underground mines will be described in detail (Fig. 96). The first six examples are focused on mines with mine water stratification, where no tracer test has been conducted so far, whereas another six examples are flooded underground mines, where extensive tracer tests were carried out to evaluate the hydrodynamic regime of the mine. The last case study is focused on salt mines. All known and published investigations in such types of mines are described and discussed in this chapter. How the knowledge about the hydrodynamics of mine water and potential stratification might be used as a remediation measure can be seen at the end of section 12.9.4.



Fig. 96: Worldwide location of the flooded underground mines discussed in this chapter.

Down hole probes for physico-chemical measurements in flooded mine shafts or wells connected to the mines usually measure pressure, temperature, electrical conductivity (or resistivity; due to space restrictions some of the following figures are labelled with ‘conductivity’ instead of ‘electrical conductivity’), pH, redox potential (E_H) and oxygen. While the measurement of pressure, temperature, or electrical conductivity, even in great depths, is less problematic, the electro-chemical probes for pH, redox potential, and oxygen are restricted in their application (Schindler et al. (1995), though the modern LDO oxygen sensors can also be used under higher pressure conditions (Körtzinger et al. 2005). This results from the mechanical arrangement of the probes, which are constructed of glass parts and membranes being exposed to the depth or salinity dependent pressures. For the Hope/Germany investigations, mis-measurements due to pressure-effects of the probes were excluded by using a specially constructed down hole probe for pressure free measurement of all the relevant parameters within the water (GSF – Gesellschaft für Strahlen- und Umweltforschung 1986). Unfortunately, this probe has a diameter, which usually excludes the probe from being used in small wells or shafts with damaged pit frames, but *in-situ* probes with smaller diameter are available in the meantime (Schindler et al. 1995). Due to the above described limitations of the probes, some measured parameters, especially pH or the redox potential must be interpreted with precaution. Another effect which could be observed in the Niederschlema/Alberoda uranium mine is the overlap of two potentials: the self potential of the rocks and the redox potential of the mine water. In some cases, the differences between the redox potential measured in the shaft and at the surface was as high as 200 mV and a successful calibration of the probe therefore

failed. Usually, the selfpotential method is used for geophysical investigations of ore deposits (Wöbking 1982; Giese et al. 1994; Corwin 1997).

Fluctuations of the electrode signals, due to random signals or noise, are common in the situations described here. Such random signals or noise is typical for signal detection and information transmission and can be due to mechanical or electrical noise (Carlson et al. 2002). In case they are ergodic, they usually do not cause problems for the measurement of physical parameters (Fig. 39). Yet, such fluctuations can also be real fluctuations, as in turbulent flow situations, or due to a malfunction of the electrode (Fig. 97). In some cases, both effects might be in the same range, but usually, extreme signal fluctuations can be seen as electrode dependent drifts or noise – whatever the reasons might be. Furthermore, systematic or random errors of the probe's signals must be considered. Systematic errors during down hole measurements usually shift the measurement to higher or lower values, while random errors are peaks in the curve.

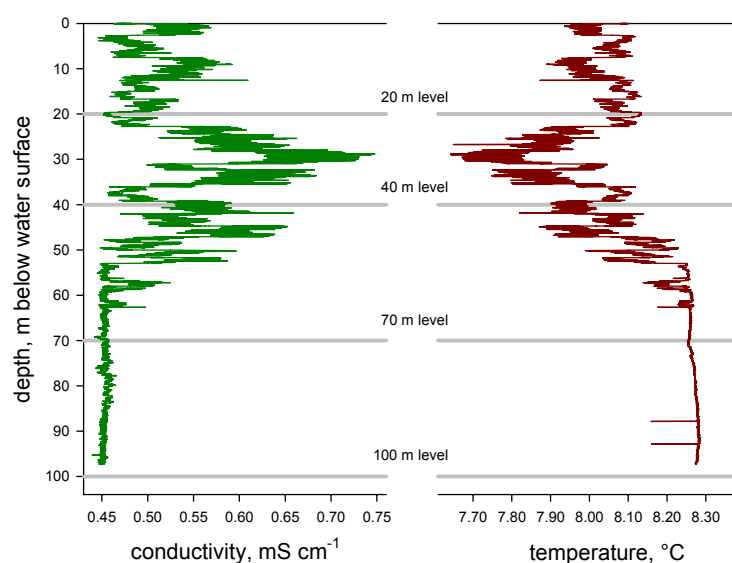


Fig. 97: Temperature and electrical conductivity depth profiles with a malfunctioning downhole probe. Several reasons are possible and they cannot always be deduced from the measurement. The probe came back to normal operation at a depth of about 63 m.

Although the effects considered before must be kept in mind, they will not be further discussed here. Readers interested in these effects and what kind of filter algorithms deduced from communication systems can be used are referred to the literature (e.g. Carlson et al. 2002). In the following chapters several examples of stratified and unstratified flooded mine shafts will be described using a similar style for each case study. It will not be possible to give all the geological, hydrogeological and mining relevant data for each of the cases reported, either because they are not available or due to a lack of space. However, the relevant literature for each mine is given in the respective introduction or can be found in the literature cited.

Hitherto, numerous mines are known where stratification of the water body occurs. Most of the papers published describe the stratification effect, without trying to explain the physical backgrounds involved nor did they try to classify the different stratification cases observed so far.

A very similar situation can be found for publications about mine water tracer tests. Though described in several publications, only one publication so far summarised the results of different mine water tracer tests and draws some first conclusions out of those tests (Wolkersdorfer 2002b).

12.2 Northern Anthracite Fields, Pennsylvania, USA

12.2.1 Description of Location

Mining in the Pennsylvanian Anthracite Fields/USA started in 1807 and ceased in the mid 1970s. During that time, about $5 \cdot 10^9$ t of anthracite were mined in four mining fields of which the Northern Anthracite Field covers an area of 456 km². Of course, an extensive dewatering scheme was installed during operation to dewater the mines, which were as deep as 450 meters below sea level. Even then, the mine water was highly acidic (Felegy et al. 1948), causing damage to the pumping machinery. The Henry colliery, for example, pumped some 28 m³ min⁻¹ in 1950 and discharged the untreated water to surface streams. After the mines were abandoned, the pumping stopped and the flooded area now covers approximately 175 km². More than 95% of the open voids were flooded in 1972 except the Wanamie colliery, which was the last working colliery by then (Stuart and Simpson 1961; Erickson et al. 1982; Ladwig et al. 1984).

While the water table rose, the mine water became enriched in contaminants and created a strongly acidic mine drainage discharging through several point- and non-point-discharges. In addition, the rising mine water table caused flooding of homes and surface subsidence (Erickson et al. 1982). Four drainage adits (“outfall discharges”) are used to drain the mine water of the Wyoming Basin, reaching a maximal annual mean flow of $63 \cdot 10^6$ m³ of highly contaminated mine water. Due to the long mining history in the area, with many interconnections between the mines, goaf, or collapsed voids the flow system is of high intricacy, with a general flow direction towards the drainage adits. Though the mine water quality steadily improved the contaminant levels were still extremely high in the 1980s. pH ranged between 5 and 6, sulphate between 1 and 2 g L⁻¹ and the acidity between 100 and 400 mg L⁻¹ CaCO₃ with a strong trend to change to alkaline conditions (93% were already alkaline in 1984). From the Pennsylvanian Anthracite fields the first report of stratification in flooded underground mines was published (Stuart and Simpson 1961; Fig. 98).

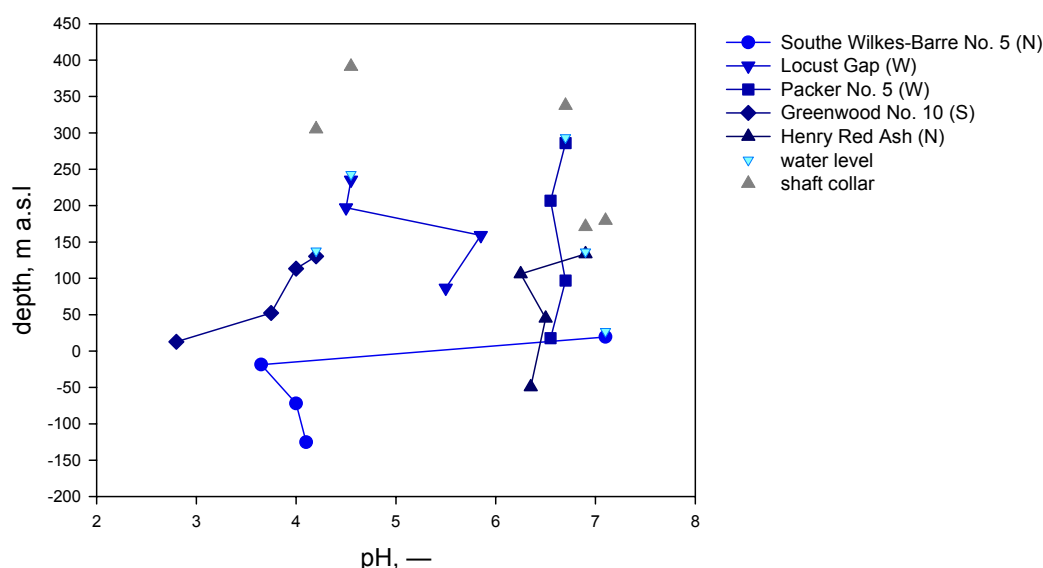


Fig. 98: First known, depth dependent pH-measurements in the Pennsylvanian Northern Anthracite field (modified after Stuart and Simpson 1961).

Geologically and structurally, the Northern Anthracite field is a crescent-shaped syncline striking in a north-eastern direction. It comprises many discontinuous folds, flexures and

faults, of which a major saddle structure divides the basin into the south-western Wyoming and the north-eastern Lackawanna Basins. The host rocks of the anthracite are Upper Carboniferous (Pennsylvanian) conglomerates, sandstones and shales interbedded with 18 workable coal seams and a total thickness of 610 m. They are underlain by Devonian sandstones as well as conglomerates and partly overlain by Pleistocene till (Erickson et al. 1982; Ladwig et al. 1984).

12.2.2 Appearance of Stratification

Between 1961 and 1984 twenty shafts, eight of them more than once, were studied by Stuart and Simpson (1961), Barnes et al. (1964), Erickson et al. (1982), and Ladwig et al. (1984) (Tab. 24). Stuart and Simpson (1961) measured pH in flooded shafts, realising that in nearly all of them, water with higher pH overlaid mine water with low pH, indicating a clear stratification with two major mine water bodies in the pools (Fig. 98). Comparing their results with those of Erickson et al. (1982) and Ladwig et al. (1984) measured 20 years later indicates that the stratification was stable between the 1960s and 1980s whereas the mine water quality in the flooded mine workings notably improved (Fig. 99).

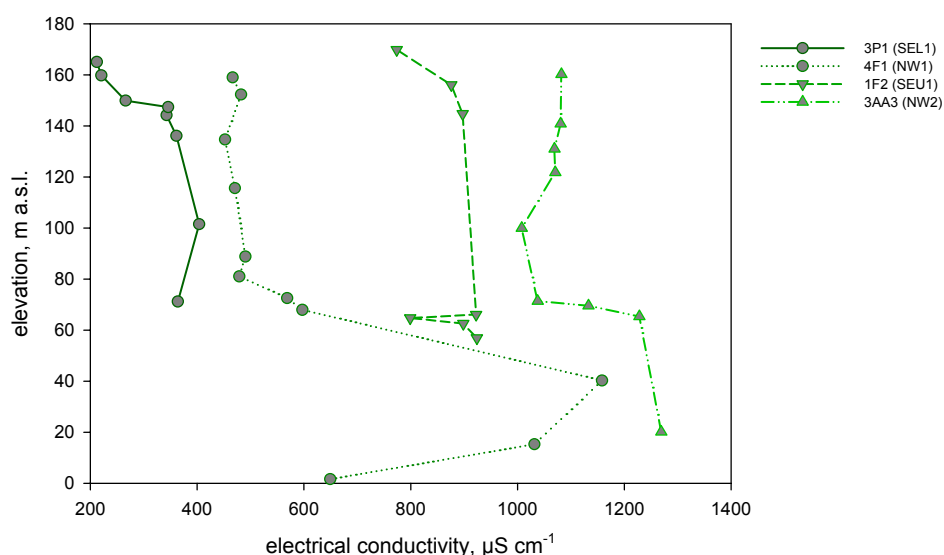


Fig. 99: Depth–conductivity plots of four shafts in the Northern Anthracite/USA mining field (modified after Erickson et al. 1982).

Five of the nine shafts investigated by Ladwig et al. (1984) showed a clear stratification indicated by abrupt changes in the parameters measured (Fig. 100). It can also be seen that the stratification change appears in a depth of about 20–100 m a.s.l. (meters above sea level) even though the distance between the shafts is 15 km (from NW1 to SEU1, Fig. 99 to Fig. 101). The electrical conductivity tends to increase in east–west-direction from about 400 $\mu\text{S cm}^{-1}$ to 1,000 $\mu\text{S cm}^{-1}$ in the shallow layer and 1,000–1,400 $\mu\text{S cm}^{-1}$ in the deeper one. Whilst the shallow layers seem to be more separated from colliery to colliery, the deep ones are better connected because of their similar physico-chemical behaviour. According to Ladwig et al. (1984) “the deep zone reflects the poor water quality in a more stagnant subsystem”. If that would be true, the variations between the measurements in a single shaft should be very small, those between the different shafts be significant large, and there should be no fluctuations during one down hole measurement. In fact, the results as indicated by some of the physico-chemical measurements and chemical analyses imply that turbulent flow in the deeper layers is likely to occur.

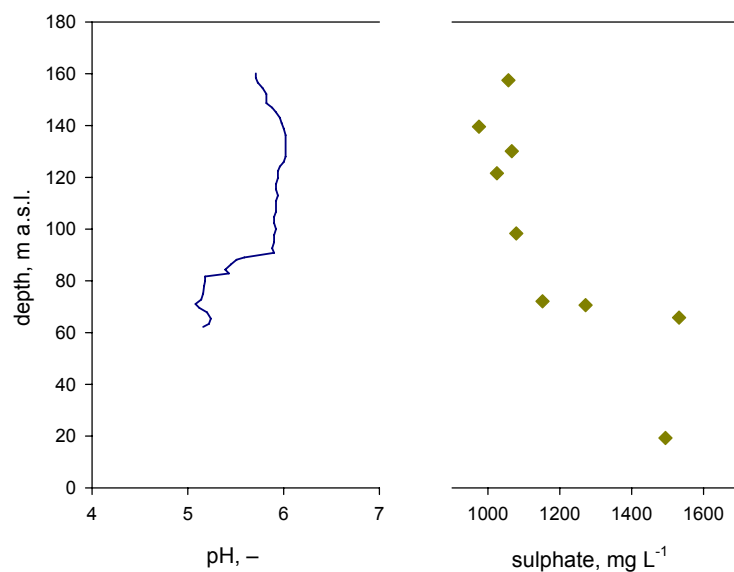


Fig. 100: Depth pH and sulphate plots of the Gaylord Colliery NW2 shaft on June 26th 1981 (modified after Ladwig et al. 1984).

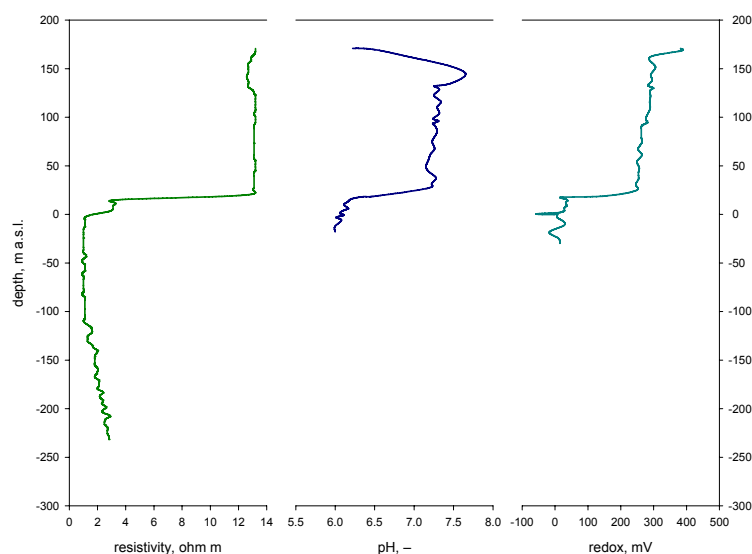


Fig. 101: Depth resistivity, pH and redox plots of the Truesdale-Bliss Colliery SEU2 shaft on November 17th—18th 1981 (modified after Ladwig et al. 1984).

Erickson et al. (1982) conclude: “All the parameters monitored in the preliminary field studies indicate that most sharp changes in water quality occur near the elevation of mined-out seams intercepted by the shafts. Consequently, such voids must be significant flow conduits for mine pool circulation. Smoother depth profiles would be expected if the mine pool complex was well mixed or if flow was slow enough to allow chemical equilibrium”.

Tab. 24: Shafts where mine water stratification was studied in the four Pennsylvanian Anthracite fields. Each line corresponds to one single shaft, which were named differently by each of the authors. κ : electrical conductivity; res: resistivity; T: temperature. References: 1: Stuart and Simpson (1961); 2: Erickson et al. (1982); 3: Ladwig et al. (1984).

Mining field	Shaft	Colliery	Parameters	Reference
North-west complex	NW1, 4F1, Red Ash	Henry; Fig. 99	κ , pH, E_H	1, 2, 3
North-west complex	NW2, 3AA3	Gaylord; Fig. 99, Fig. 100	κ , pH	2, 3
North-west complex	NW3, 5R4	№ 6	κ , pH, E_H	2, 3
North-west complex	NW4, 3HH3	Kingston	κ , pH, E_H , res	2, 3
North-west complex	6G2	Nottingham	κ	2
North-west complex	Red Ash	Exeter	pH	1
North-west complex	№ 1	Schooley	pH	1
South-east Upper Complex	№ 2	№ 7	pH	1
South-east Upper Complex	SEU1, 1F2	Bliss; Fig. 99	κ , res, T, E_H	2, 3
South-east Upper Complex	SEU2	Truesdale; Fig. 101	pH, E_H , res	3
South-east lower Complex	SEL1, 3P1	Hollenback; Fig. 99	κ , pH, E_H , res	2,3
South-east lower Complex	SEL2, 4H2	Peach Orchard	κ , pH, E_H	2,3
South-east lower Complex	SEL3, 2F1	Loomis	κ , pH, E_H , res	2,3
South-east lower Complex	Clear-Spring	Clear-Spring	pH	1
South-east lower Complex	№ 5	South Wilkes-Barre	pH	1
Eastern Middle	Hazeltan	Hazeltan	pH	1
Western Middle	Locust Gap	Locust Gap	pH	1
Western Middle	Reliance	Reliance	pH	1
Western Middle	№ 5	Packer № 5	pH	1
Southern	№ 10	Greenwood	pH	1

12.3 Picher Mining District, Oklahoma, USA

12.3.1 Description of Location

Located in north-eastern Oklahoma/USA, the Picher Mining District was one of the world's largest lead-zinc mining fields, covering an area of 186 km² (the studied area covers about 40 km²) and originally containing ore reserves of $1.22 \cdot 10^8$ m³. Mining of galena and sphalerite started between 1891 and 1901 and ceased in 1976 with a peak period between 1915 and 1920, when 230 mines were under operation. During the Second World War, 63 major pumping stations pumped a maximum of 60 m³ min⁻¹ of mine water out of the mines, decreasing to 24 m³ min⁻¹ after the war. The deepest mines in the Picher Mining District reached a depth of about 200 m below surface and the total volume of the flooded voids sums up to $4 \cdot 10^7$ m³. Due to the room and pillar mining technique, pyrite- and marcasite-bearing waste material was deposited in the mined-out areas, thus giving rise to highly contaminated acid mine water with pH values of 4 to 5. Furthermore, the mine water is enriched in zinc and iron, reaching up to 150 mg L⁻¹ each. In 1979 the water levels reached the surface finally discharging into the river Tar Creek (Sheibach et al. 1982; Spruill 1983, 1987; Parkhurst 1987, 1988; Suman et al. 2000).

The oldest rocks in the Picher Mining District are Ordovician dolostones with interbedded sandstone layers which are in places overlain by Devonian shales. Above the shales and

dolostones follows the ore bearing lower Carboniferous (Mississippian) Boone Formation with a thickness of 110 to 120 meters. This formation consists of greyish, karstified limestones and cherts and yields a good drinking water reservoir outside the mining area. Above the Mississippian rocks mainly shales and sandstones of Pennsylvanian age can be found. Bedrocks in the area are faulted and folded and slightly dip to the north-west with angles around 0.2° . Sphalerite and galena were the economic minerals mined in the area. Above that, chalcopryite, enargite, pyrite, and marcasite were found in uneconomic quantities. Common gangue minerals were jasperoid, dolomite, calcite, quartz, or barite. Besides that secondary minerals such as smithonite, calamine, anglesite, cerussite, malachite, epsomite, or gypsum were found (McKnight and Fischer 1970; Tar Creek Superfund Task Force Water Quality Subcommittee 2000a). Genetically the mineralisation in the Picher Mining District belongs to the Mississippi-Valley-Type (MVT) deposits.

In January 2000 the Tar Creek Superfund Task Force was appointed in the Picher Mining District to “[develop] a comprehensive remediation plan for the ill-fated area of north-eastern Ottawa Country” (Griffin et al. 2000). During its eight months of work, the Task Force, consisting of eight subcommittees, developed a remediation plan for the whole area. In addition, the Mine Shaft Subcommittee identified more than 1,300 shafts in north-eastern Oklahoma and assumed that possibly up to 2,600 abandoned shafts exist in the whole Tri-State mining district. They recommended an extensive 5-year shaft and hole closure programme, to plug all known shafts in the area (Suman et al. 2000).

12.3.2 *Appearance of Stratification*

In 1978 Playton and Davis (1978) and Playton et al. (1978) published the first results of their measurements in abandoned mine shafts of the Picher Mining District and two years later they published a comprehensive report about their findings (Playton et al. 1980). A total of seven mine shafts were sampled by physico-chemical parameters at different depth intervals, taking up to 8 water samples. Most of the sampled shafts showed a clear stratification identified easily by the specific electrical conductivity and the pH, and given the extent of the mine field it can be concluded that it is the largest stratified mine water body known so far. Two water layers could be observed: an upper one characterised by lower electrical conductivities ($0.7\text{--}1.9\text{ mS cm}^{-1}$) and higher, circum-neutral pH, and a lower one with higher electrical conductivities ($3\text{--}4\text{ mS cm}^{-1}$) and acid pH. In most shafts, the boundary between the two mine water bodies could be observed between a depth of 185 and 195 m a.s.l., those shafts being above that height, not showing a clear stratification. Playton et al. (1980) realised that no clear spatial variation of the physico-chemical data could be observed. Furthermore, they discuss the fact that cooler water overlies warmer water and conclude that “an unstable thermal-stratification condition apparently exists” which is masked by the higher density of the higher mineralised water. Yet, they forget that the stability of a stratified water body is a result of the water density as a function of the water temperature and its mineralisation. Applying the UNESCO polynomial (Fofonoff and Millard Jr. 1983), the density of the upper layer at 14°C and 1 mS cm^{-1} is around 999.7 kg m^{-3} and the density of the lower layer at 15°C and 4 mS cm^{-1} is around 1001.2 kg m^{-3} (see also Fig. 41). No explanation is given for the stratification of the mine water body.

Sheibach et al. (1982) also reported that lower mineralised mine water was overlying a mine water body with a higher mineralization and lower pH values, whereas the separation of the two water bodies was at a depth of about 50 m (Fig. 102). In the three shafts, where they took samples (Kenoyer, Consol 2, and Lawyer shafts) the total dissolved solids ranged roughly between $1\text{ and }3\text{ g L}^{-1}$ in the upper and $3.5\text{ and }5\text{ g L}^{-1}$ in the lower mine water pool. In the same shafts, the pH was circum-neutral in the upper and acidic (pH 4–5) in the lower parts.

Without further investigations, they concluded that the stratification might have been a result of acid producing waste rock left in the deeper parts of the mines.

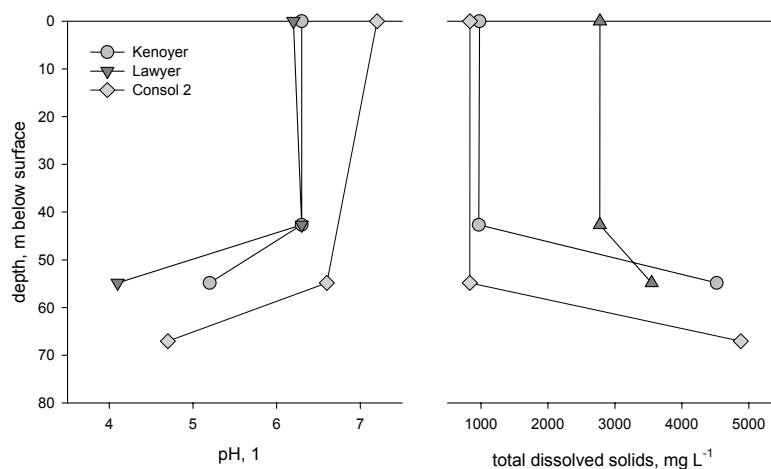


Fig. 102: Depth dependent pH and TDS measurements in three shafts of the Picher mining field (modified after Sheibach et al. 1982).

More than 20 years later, DeHay (2003) and DeHay et al. (2004) reinvestigated the Picher Mining District. Though the electrical conductivity has decreased from a range of 3.5—4.5 mS cm⁻¹, to 2.5 mS cm⁻¹, and the pH increased from 5.5—6 to over 6, the stratification patterns in the mining field were similar to those of the late 1970s and early 1980s. The boundary between the lower and the upper mine water bodies is approximately at 185—202 m a.s.l. and is indicated by a significant jump in the electrical conductivities and in the concentrations of the water constituents. Age determinations in the shafts with stratification showed that the lower mine water body has a significant higher mean residence time than the upper one (1980s in the lower, 1990s to early 21st century in the upper layer, with some exceptions from this general rule). Though the water is flowing through the mine pool, the stratifications stayed very stable during the past 20 years. For a more detailed description not enough data are presented by the authors.

12.3.3 Conclusions

Based on the data presented by the several investigators between the 1980s and 2004, the stratification in the Picher Mining District remained very stable over a duration of nearly 25 years with physico-chemical parameters of the upper and lower mine water body being very similar in all the investigated cases. This shows, on the one hand that the hydrodynamic connection between the upper and lower mine water bodies is extremely poor and no convective flow between those two is taking part. On the other hand, the similar physico-chemical parameters in each of the water bodies leads to the conclusion that they are hydraulically well connected, as can also be deduced from the mean residence times of the mine water. Furthermore, the measurements in the shafts show that the flow is dominated by convective flow and not by diffusive flow, because the electrical conductivities and temperatures in the different depths are very similar in each single shaft. One exception is the Royal shaft, which has – from top to the bottom of the upper layer – an increasing electrical conductivity from 1.2 to 8.5 mS cm⁻¹. This fact can be interpreted as a diffusion controlled flow in this shaft and the age determination of the water in fact differs from the other shafts investigated: it is about 35 years old.

So far, no tracer test has been reported from the Picher Mining District. It would be interesting to see if the water age determinations could be validated by the results of a tracer test or not. Furthermore, such a test could be used to characterise the type of stratification in the Picher Mining District.

12.4 North Rhine-Westphalian Coal and Ore Mines, Germany

12.4.1 Description of Locations

Coldewey et al. (1999) investigated 19 shafts and wells in the German Ruhr (Niederrhein-Westfalen: Lower Rhine-Westphalia) and the Aachen coal field as well as the Sauerland ore mining areas (Tab. 25). Besides the physico-chemical parameters measured with down hole probes (temperature, electrical conductivity, pH, E_H , and pressure) they took water samples from discrete depths in the shafts that were analysed in the laboratory. Furthermore, they conducted a laboratory experiment with which they wanted to investigate and understand some of the stratification patterns in the flooded shafts. The following descriptions are only based on the work of Coldewey et al. (1999) if explicitly cited. What is presented here is a personal interpretation of their raw data, which in parts deviates from their findings.

All three mining areas differ geologically, although all of them belong to the same geological system: the rhenohercynian zone of the middle European Variscan orogeny. While the comparably small Aachen coal mining area is part of the Ardenne mountain chain, the Sauerland, easterly from it, is part of the Rhenish Schiefergebirge and the Ruhr area, north of the Rhenish Schiefergebirge, is in the molasse trough of the Rhenish Schiefergebirge. Devonian and Carboniferous rocks are dominating and due to the excellent surface outcrops and underground profiles, many locations became the *locus typicus* for stratigraphic units (e.g. Siegenium, Emisium, Eifelium, Westfalium, or Namurium). All investigated Aachen shafts are within the Wurm syncline, the easternmost part of the Namur syncline, filled with Carboniferous sandy, silty, and clayey deposits of Namurian C to Westfalian B ($\pm$ lower Pennsylvanian) age. This syncline runs into a south-west—north-east-direction, dips with less than 20° into southerly directions, and because of faulting the syncline is divided into several blocks, which intersect the coal seams. Of minor importance are the overlaying deposits of the Buntsandstein and Cretaceous ages. During the Tertiary and the Quaternary, the Carboniferous rocks were overlain by 200—300 m thick sediments. East of the Ardenne mountains, and also east of the river Rhine is the Rhenish Schiefergebirge, with several thousand smaller and larger lead, zinc, and pyrite ore deposits and a large number of small, but long west-south-west—east-north-east striking synclines and anticlines as well as faults. The oldest rocks are of Silurian age and the youngest belong to the Westfalium C. In the closer vicinity of the Meggen pyrite-zinc mine, which hosts the Sicilia shaft, the rocks are of middle Devonian age and consist of sandy to clayey sediments, overlain by upper Devonian schists and limestones, the overall thickness reaching 1,000—1,600 m. By far the largest German coal mining district is the Ruhr area north of the Rhenish Schiefergebirge, which hosts about 100 coal seams of different productivities and thicknesses. A characteristic of the Ruhr area are its up to 100 km long west-south-west—east-north-east striking synclines and anticlines, which are intensively internal folded and faulted. Those tectonic structures caused severe problems for the miners, as they had to extract the coal also from steep seams and the following of the coal seams sometimes was difficult for them (Walter et al. 1992; Schöenberg and Neugebauer 1997).

From the three mining localities investigated, the Aachen area and the Meggen mine are abandoned, whereas the Ruhr coal mining area is still in operation. Both, Aachen and Meggen were abandoned in 1992 and were flooded since then (Heide and Hasse 1997, 1998; Heitfeld et al. 2002), whilst the Ruhr coal mining area has only been flooded in the parts which are not directly connected to the still working mines in the northern part of the Ruhr coal mining area.

Tab. 25: Shafts and wells investigated by Coldewey et al. (1999). κ : electrical conductivity; T: temperature.

Mining Area	Shafts, wells	Parameters
Aachen	Eduard-Schacht, Emil Mayrisch 1, von-Goerschen-Schacht, Willem II	κ , T
Ruhr	Shafts: Friedlicher Nachbar, Friedrich, Hermann 1, Hermann 2, Radbod 7, Wehofen 1; Wells: B4B, Landwehr 1, Landwehr 2, Schacht 7, Stuckenbusch 1, Stuckenbusch 2, Millinger M2	κ , T, pH, E_H
Sauerland	Sicilia Schacht	κ , T

12.4.2 Appearance of Stratification

In total, 24 continuous and discontinuous measurements were conducted in mine shafts and 23 measurements in wells. At a first glance, it seems as if no general conclusions can be drawn, because the results of the measurements are varying from shaft to shaft and from well to well.

Yet, there are some characteristic similarities in the depth dependent parameters:

- zones with constant physico-chemical parameters
- zones with gradually changing parameters
- sudden changes in the parameters
- transition zones.

The electrical conductivity and temperature, usually, but not always, increase with depth and sudden changes of parameters usually occur at the shaft's onsetting stations (Fig. 103), which can nicely be seen at the Meggen Sicilia shaft (Fig. 104). Those sudden changes are very constant in their depth and even measurements in different years are relatively consistent. The geothermal gradient does not seem to have an important influence on the temperature-depth profiles of the shafts, but does in the small diameter wells. Because the ground water in the Ruhr coal mining area is usually of high salinity the electrical conductivity in the shafts and wells can reach several milli-Siemens of up to 189 mS cm^{-1} . Because the deeper mine zones are commonly overlain by ground water of lower conductivities, the density differences are very large and the stratification therefore constant for a long time period. Though the transition zones are commonly small between different water bodies, there can also be transition zones with thicknesses of up to 20 m, which seem to be more typical for wells than for shafts (the Landwehr 1 shaft shows such a 20 m transition zone). Because the flow in a shaft is better established than in a well, this can be seen as the reason for this characteristic. Some shafts show a stepwise change of the salinity or temperature (e.g. Hermann 1, Sicilia, Fig. 104), which can at least for the Hermann 1 shaft be due to double-diffusive convection (see section 7.2.1 for explanations). Yet, the data available from the report are not enough to verify this hypothesis. In the Sicilia shaft the steps are clearly linked to the on-setting station and might therefore reflect different flow cells or the inflow of water with different physico-chemical parameters.

All the 23 measurements in the seven wells are of limited meaningfulness for an interpretation of stratification patterns. In wells, the water will not be affected by free convective flow, because of the small diameter to length ratio in which the Rayleigh number Ra might be smaller than the critical Rayleigh number Ra_{kr} . This is true for both, cased and uncased wells. Therefore, it cannot be excluded that the temperature and electrical conductivity patterns observed in the B4B, Landwehr 1, Landwehr 2, Schacht 7, Stuckenbusch 1, Stuckenbusch 2, and Millinger M2 wells are partly due to the hydrodynamics occurring in small diameter pipes

with stagnant water (see also Ronneburg uranium mine). At least some of them clearly show the characteristic depth profiles for diffusion dominated transport.

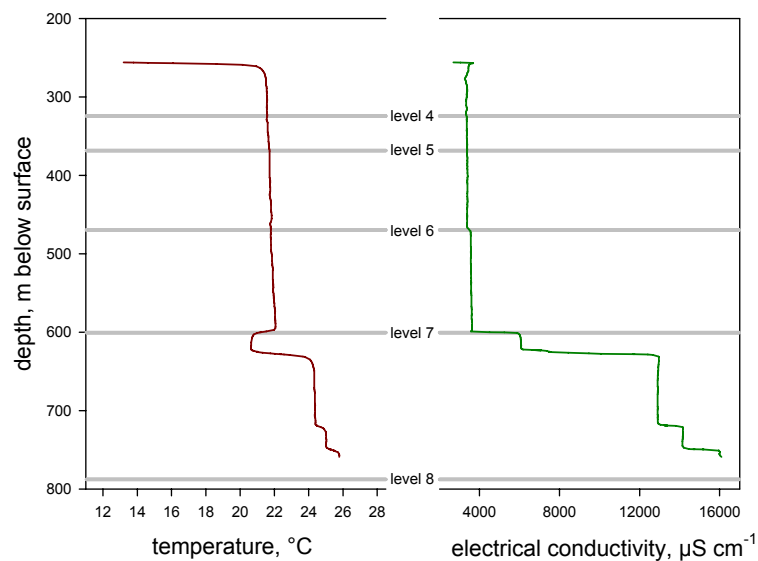


Fig. 103: Temperature and electrical conductivity measurements in the *Friedliche Nachbar* shaft (modified after Coldewey et al. 1999). Horizontal lines indicate the levels in the mine.

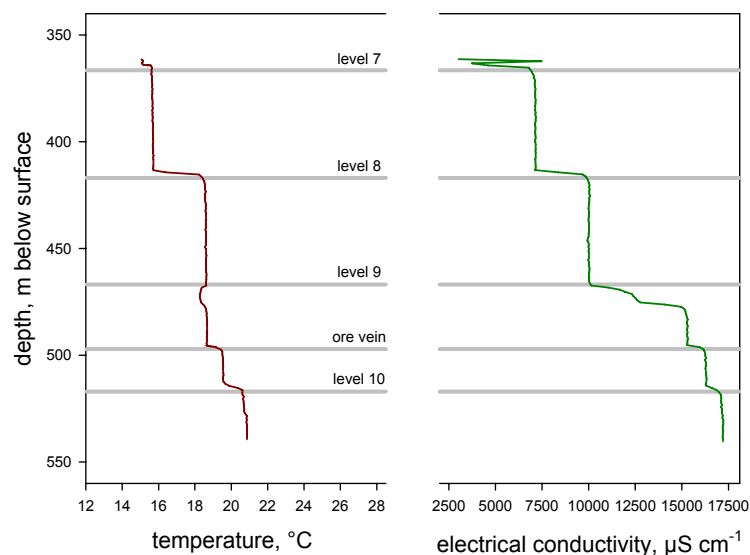


Fig. 104: Temperature and electrical conductivity measurements in the Sicilia shaft of the abandoned Meggen/Germany mine (modified after Coldewey et al. 1999). Horizontal lines indicate the levels in the mine.

12.4.3 Laboratory Experiment

With a laboratory experiment Coldewey et al. (1999) wanted to investigate and understand some of the stratification patterns in the flooded shafts. Therefore, they built a “laboratory mine” with a 0.5 m diameter and 0.8 m height.

Because no similarity calculations were conducted and no Rayleigh numbers were used to downscale the real world situation the laboratory experiment is of restricted use for explaining the stratification or hydrodynamic situation of a flooded underground mine. The temperature and salinity distribution during the 14 months of the test clearly proves that the flow within the system was – as could be expected from the experimental set-up – controlled by diffusive flow. Their interpretation of an upward/downward flow in the shafts is – due to the scaling of the system – physically not comprehensible and their temperature and salinity logs clearly show that the difference between the two shafts is negligible.

Overall, the results of the experimental mine show more similarities with the measurements from the wells than with those of the shafts. Therefore, there is no urgent need to discuss more of the results of this experimental mine.

12.4.4 Conclusions

Clearly, it can be concluded that the final temperature–salinity profiles of a shaft or a well are unpredictable. There is a high likelihood that steps or transition zones are within the reach of onsetting stations but which onsetting station results in such steps seems to be unpredictable. Interestingly, once stratification has established it appears to be rather constant even for time scales of several years. Stratification in shafts is characterised by small transition zones between different mine water bodies, where wells have usually larger transition zones. Another interesting feature is the fact that – though the depth of the different mine water bodies seems to be rather constant – this is not true for the electrical conductivities within the different layers. There seems to be a common trend that electrical conductivities decrease with time, such as in the Willem II and Friedrich and in the lower parts of the Sicilia as well as the Glückauf-segen 3 shafts. Those differences prove that the water is not in a steady state condition.

12.5 Ronneburg Uranium Mine, Thuringia, Germany

12.5.1 Description of Location

Ronneburg is located in Thuringia/Germany, 8 km south-east of Gera and 50 km west of Chemnitz. Uranium mining by the former Soviet-German company SDAG Wismut started there in 1951 and stopped in 1991. During that time, several interconnected underground mines with a total volume of $68 \cdot 10^6 \text{ m}^3$ of which $44 \cdot 10^6 \text{ m}^3$ are backfilled and a surface mine with a volume of $160 \cdot 10^6 \text{ m}^3$ of open space covering a total area of 60 km^2 were under operation. 22 mine levels between depths of 30 and 940 m below surface and interconnected by 40 shafts summed up to a length of 3,000 km. In 1998, flooding of the underground mines started following the installation of about 120 underground dams to separate areas with high pollution potentials from those with low ones. Only deep and isolated mine parts adding up to $0.8 \cdot 10^6 \text{ m}^3$ were already flooded between 1989 and 1997. Final seepage of the mine drainage water is expected to occur at several topographic low areas, especially in the Gessenbach valley (Unland et al. 2002). To control the flooding process and, in case of unexpected flooding situations, a 600 mm monitoring and pumping well and 20 monitoring wells were constructed, as all shafts were backfilled. Whilst the mine water in the southern fields needs treatment to remove uranium, radium, iron, arsenic, and potentially toxic metals, the mine water in the northern field probably will not need treatment. The end of flooding had been expected for 2003–2005 and treatment is required for another 20 years (Gatzweiler et al. 2002; Paul et al. 2002). At the time of the publication of this book the water has not discharged yet.

All of Ronneburg's uranium deposits (Schmirchau, Reust, Paitzdorf, Beerwalde, Drosen, Ditrachshütte, Lichtenberg, and Stolzenberg) are located within the north-west—south-east-

striking Nejdeck-Crimmitschau-fault-zone, which also hosts the important Joachimsthal/Bohemia and Aue/Saxony uranium deposits (Zentralinstitut für Physik der Erde 1989; Lange and Freyhoff 1991; Büder and Schuppan 1992; Meinel 1993). Typically, the strata-controlled deposits form lenses and stock works within Palaeozoic pyritic black shales, limestones, and diabases (Dahlkamp (1993) Type 16). Besides the Silurian Graptolithenschiefer, Ordovician shales and Silurian carbonates (“Ockerkalk”) are abundant and the rocks contain 0.5—7% sulphides, 0.5—9.5% organic carbon, and 40—60 ppm uranium. During the deposits life-time a total of 111 kt of UO_2 were produced (Szurowski 1968; Lange and Freyhoff 1991; Dahlkamp 1993).

Previously, an extensive approach to model flooding and post-flooding scenarios was undertaken at the Ronneburg mine site. Gatzweiler et al. (2002) put the insufficiency of the modelling results down to the “complex mine structure, size and nature of the influenced ground water regime and changing infiltration water quality due to ongoing remediation work at the surface”. As discussed in the theoretical part, the reasons for the failure of the model are mainly caused by the fact that such systems are chaotic systems (see section 7.1.2). Therefore, they cannot be numerically modelled far into the future because the accuracy of such models is quickly lost!

12.5.2 Appearance and Interpretation of Stratification

Since the beginning of depth dependent temperature and electrical conductivity measurements at the Ronneburg mine site, stratification could be observed in the uncased monitoring wells. Paul et al. (2002) give details of three measurements within the e-1260 observation well, located in the Ronneburg north-west mine field (March 2000, December 2000, March 2001). Different from the other case studies described here, the shafts and most of the mine’s voids are backfilled. Therefore, monitoring is conducted within especially drilled wells of comparatively small diameters. In those wells, the water will not be affected by free convective flow, because of the small diameter to length ratio (see section 12.4). Furthermore, it cannot be excluded that the effects observed are partly due to the hydrodynamics occurring in small diameter pipes. The figure published in Paul et al. (2002) is typical for standing water in wells and the conclusions drawn here and by the authors, therefore, must be interpreted with caution (Fig. 105).

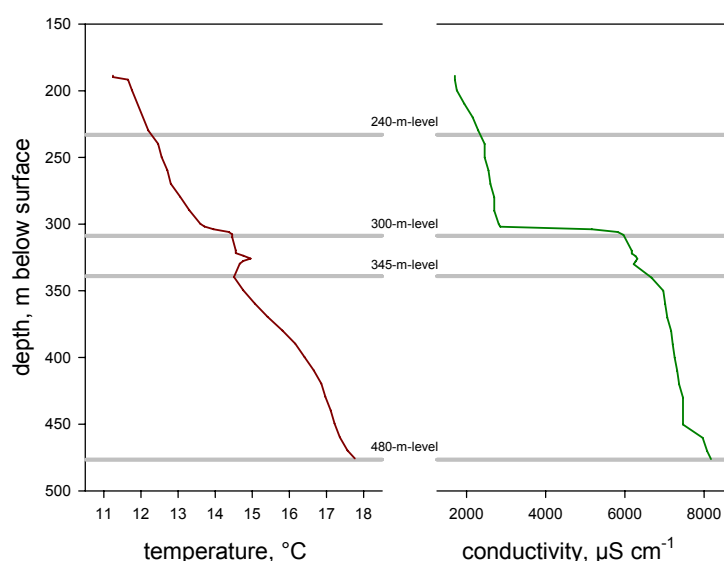


Fig. 105: Temperature and electrical conductivity plots of the Ronneburg/Germany north-west well e-1260 (modified from Paul et al. 2002).

Nevertheless, it becomes clear that there is a stratification in the Ronneburg north-west uranium mine with highly contaminated mine water (electrical conductivity above $4,000 \mu\text{S cm}^{-1}$) in the lower and a lesser contaminated mine water ($1,000\text{--}3,000 \mu\text{S cm}^{-1}$) in the upper parts of the well. Furthermore, there is no evidence for an overall convective flow, as the temperature profile mirrors the geothermal gradient very well. This absence of mixing would explain the differences in the chemical composition of the mine water in the various mine fields and also proves that dams and backfilling delay the development of large scale mixing. However, as the experimental investigations of Robillard et al. (1986) or the field observations of Le Carlier de Veslud et al. (1998) showed, porous media does not exclude long term thermal convection.

Some dynamic processes can be observed between levels 300 and 345 (the levels in Fig. 105 are taken from the original publication. Differences between the numbers given here and the figure are because mine levels usually are not horizontal and that the levels meet every shaft in different depths). Mine water with a 1 K higher temperature seems to enter the well at the level 300 and interacts with the water of the level 345. Certainly, there is a shortcut between both levels, causing a small ca. 45 m long convective flow cell, which is proved by the temperature log.

Summarised, the measurements in the Ronneburg north-west mine field show that there are two separate mine water bodies. No convective flow occurs within each of them and the temperature is due to the regional geothermal gradient. It can be assumed that the construction of dams delays the mass transport because they obstruct the establishment of a regional convective flow regime.

12.6 Frazer's Grove Mines, County Durham, United Kingdom

12.6.1 Description of Location

Frazer's Grove fluorspar mines are situated in the United Kingdom's North Pennines, 40 km south-west of Newcastle-upon-Tyne near the river Rookhope Burn in the Weardale District. Large scale mining in this area started in the 18th century and was stopped in 1999. Frazer's Hush underground mine was opened in the early 1970s and closed as the last working mine in the North Pennines. Similar to the Straßberg/Germany mine, the area around the Frazer's Hush mine was first worked for lead and zinc, later for barium and fluoride. All four mines, Frazer's Hush, Rake Level, Firestone and Grove rake workings, reached down as far as about 200 m below surface and worked on 5—9 interconnected main levels (Johnson and Younger 2000, 2003).

During the operation of the mine, a daily water make averaging $1.9 \cdot 10^3 \text{ m}^3$ was pumped, which mainly originated from the Carboniferous Great Limestone. Grove Rake's № 2 Shaft has a depth of 160 m and is connected to the 30, 40, 50, 60, 80, and 90 fathom (fm) levels (55, 73, 91, 120, 146, 165 m). Though the 40 fm level is not fully connected to the № 2 shaft (Paul Younger, pers. comm.), the temperature-depth plot indicates that "something" happens, at that level. Therefore, the 40 fm level is drawn in the figures Fig. 106 and Fig. 107. Unfortunately, the water level in Johnson and Younger (2002) figure 3 is not correctly drawn, causing some problems in understanding the hydraulic situation.

Geologically, the Frazer's Grove fluorspar mines are located in the Northern Pennine Ore-field, which covers an area of approximately $1,400 \text{ km}^2$. The mine itself is located in the Eastern Fluorine Zone. An east—west trending syncline, the Stainmore syncline, divides the Ore field into a southern (Askrigg block) and northern part (Alston block), the latter one hosting the Frazer's Grove mines. Host rocks there are Lower Carboniferous sandstones, limestones, and shales of the Great Limestone between the overlying Millstone Grit and the underlying

Weardale Granite. The mine worked the Greencleugh Vein which consisted of the following minerals: galena, fluorite, quartz, sphalerite, siderite, calcite, pyrite, marcasite, bismuthinite, and pyrrhotite. Genetically, the mine is similar to the Mississippi-Valley-Type deposits of the USA Tri-State district, such as the Picher Mining District described in section 12.3 (Taylor 1978; Dunham 1990 [1948]; Fisher and Greenbank 2000; Johnson and Younger 2002).

12.6.2 Appearance of Stratification

Between 1998 and 2002 Johnson and Younger (2002) intensively investigated Frazer's Grove mines, concentrating on the mine water hydrogeochemistry. They found the mine water to be slightly acidic with conductivities ranging between 500 and 3,200 $\mu\text{S cm}^{-1}$ with Zn concentrations up to 111 mg L^{-1} , iron concentrations up to 68 mg L^{-1} , and sulphate as high as 2,000 mg L^{-1} . Furthermore, in 1999, they noticed that the № 2 shaft was stratified, with pH of 3.8 at the lowest levels and 6.2 at the surface of the mine water. While the water's quality gradually improved, the stratification preserved, albeit the character changed.

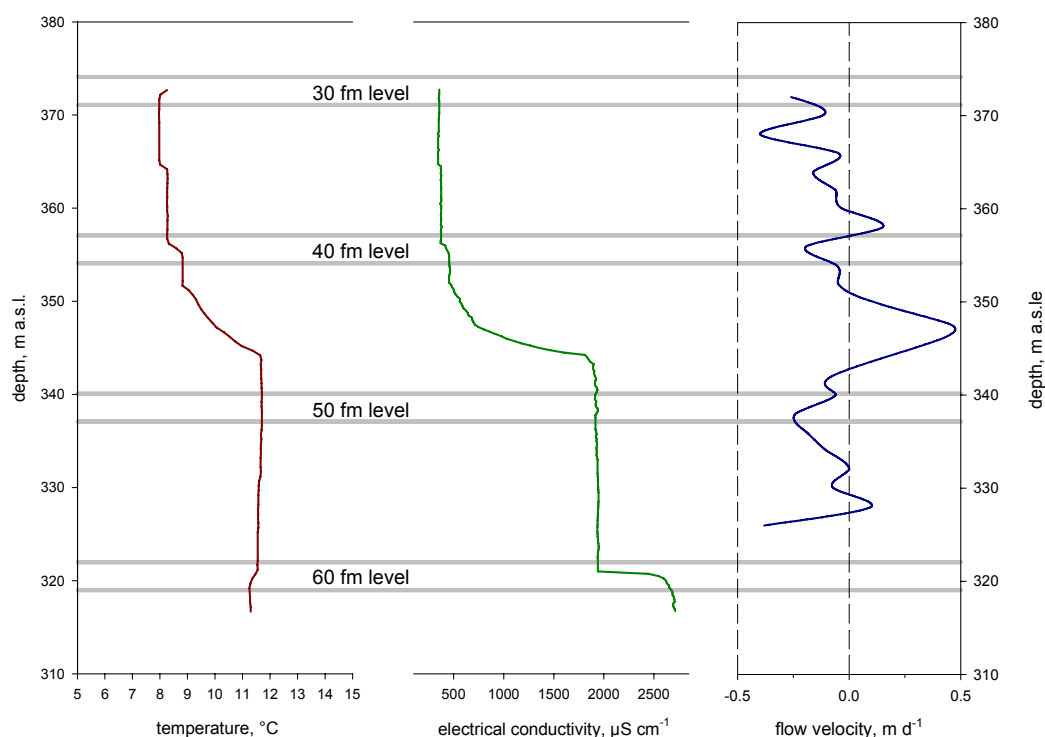


Fig. 106: June 2002 temperature, electrical conductivity, and flow velocity measurements in the Frazer's Grove № 2 shaft. All elevations of the four levels are taken from a graph and might vary within several meters. The height of the individual levels has been estimated from a photograph taken by Wilkinson (published in the Internet at MinDat.org) and the photographs in Younger (2003) to be approximately 2—3 m (modified and supplemented from Johnson and Younger 2002).

In June 2000 electrical conductivity, temperature and velocity in the № 2 shaft was measured with continuously measuring down hole probes (Fig. 106). Three water bodies could be observed by then, being separated from each other on the 50- and 60-fm-levels. On the top of the mine water body, the water quality was relatively good (electrical conductivity $\approx 500 \mu\text{S cm}^{-1}$) and in the deepest accessible parts, the water quality was relatively bad (electrical conductivity $\approx 2,600 \mu\text{S cm}^{-1}$) with a slight temperature inversion occurring there. In Fig. 107 the mine water's density and the temperature are shown. This figure clarifies that the

density of the mine water is steadily increasing from the mine water surface to the deepest measuring point reachable with the down hole probe.

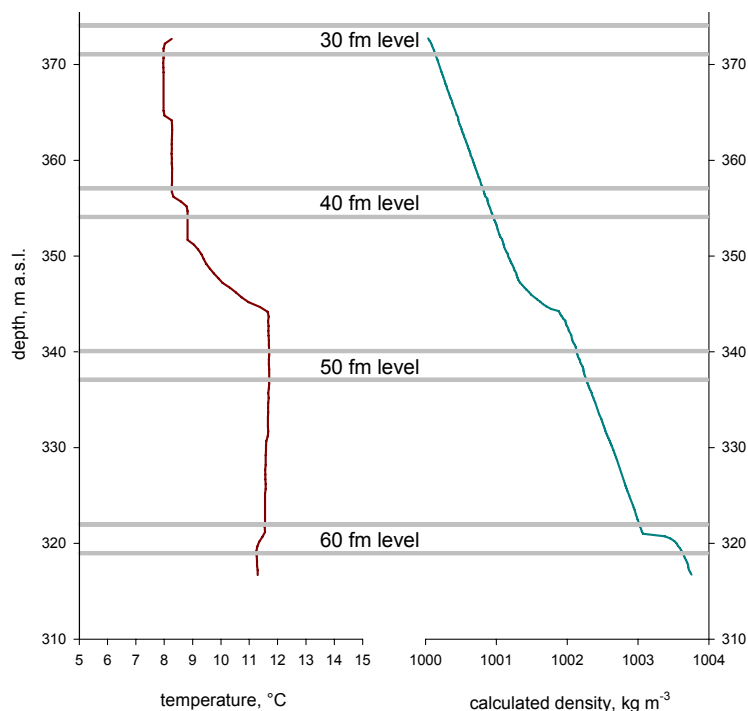


Fig. 107: June 2002 temperature measurement in the Frazer's Grove № 2 shaft. In addition the calculated density of the mine water is given (explanations as in Fig. 106; modified and supplemented from Johnson and Younger 2002; density calculated according to Fofonoff and Millard Jr. 1983).

12.6.3 Results and Interpretation

According to the velocity profile, Johnson and Younger (2000) interpret the results as follows: water is moving downward from the water surface to 350 m a.s.l. and there water enters the shaft from the Great Limestone, thus causing the water to flow upward. Yet, as Younger (2003) noted, the shaft is fully brick-lined and so it is unclear how water should enter the shaft from the aquifer through the bricks. Hydraulically, this interpretation is highly problematic, because the interpretation does not answer the question, where the downward moving water and where the water entering the shaft at 350 m a.s.l. does flow to. Furthermore, the whole shaft is brick-lined over its full depth and therefore water entering from this aquifer might not be expected. One of the reasons for their misinterpretation might be that they forgot to draw the 40 fm level in their publication (yet, correct in table 2 of Younger 2003). Furthermore, if water would enter the shaft from the Great limestone (to be more precisely: the 40 fm level), a forced flow would occur, and after a small mixing zone between the warmer water swelling upward and the cold water from the aquifer there should be a relatively homogenous temperature, electrical conductivity, and velocity profile. Considering the hydrodynamic relationships explained above, the situation can be explained easily:

- First, the velocity and the temperature profiles prove that there is a turbulent convective flow. No sign for a forced flow is given, as it is the case of the № 530 shaft of the Straßberg mine. Therefore, there are several convective cells and in each of the cells water is moving up- and downward depending on the time of measurement and the current occurrence of the convective cell.

- Second, most of the water discharging from the № 2 shaft seems to come from the 30-fm-level as there is no clear indication that water is up welling from the 50-fm or 40-fm levels to the 30-fm-level. If that would be the case the linear range between ca. 344 and 352 m a.s.l. could not be stable. On the other hand, the logs clearly show that there is convective flow between the 60-fm- and 50-fm-levels as the temperature and conductivity stay nearly stable.

12.7 Frances Colliery (East Fife Coal Field), Scotland, United Kingdom

12.7.1 Description of Location

In 1995 pumping stopped in the Scottish Frances and Michael coal mines. After the recovery of the mine water table to a certain level it was predicted to be necessary to start pumping again, as the mine water would be polluted and a contamination of the area can only be prevented by treating the mine water (Nuttall et al. 2002; Nuttall and Younger 2004). As mine water pollution ranks fifth in the causes of Scottish water pollution (Anonymous 1999a) the treatment of such types of water must given special consideration and therefore an intensive pumping test was carried out at the Francis Colliery site.

The Fife coal field, together with the Ayrshire, Sanquhar, Douglas, Central, and Midlothian coal fields (Wood et al. 1999), is part of the Midland Valley tectonic structure, an intra mountainous Old Red Sandstone basin filled with Ordovician to Devonian rocks. After the Old Red development came to an end, Carboniferous sediments were deposited, among them the thick Westphalian “Coal Measures” (Schönenberg and Neugebauer 1997). They were folded at the end of the Palaeozoic and within the 500 m thick sequence at the Frances Colliery, about 20 coal seams of workable thicknesses with up to 1.5 m width are reported (Nuttall et al. 2002; Nuttall and Younger 2004). Some of those sequences, namely the Dysart Main Seam contained up to 3% of pyrite, such being a potential source of acid drainage generation. Though initial pH values in the Frances Colliery were nearly neutral, they decreased after several days of pumping and reached values as low as have been reported from the active mining period (Nuttall and Younger 2004).

12.7.2 Appearance and Interpretation of Stratification

Stratification in the Frances mine was not known before the 1995 pumping test. Because the quality of the pumped water decreased rapidly during the six weeks pumping test it was assumed that this effect is a result of the stratified mine water body. At the start of the pumping test the electrical conductivity was 5.6 mS cm^{-1} , increasing to 25.6 mS cm^{-1} after day one and reached a value of 26.5 mS cm^{-1} at the end of the test (Nuttall et al. 2002; Nuttall and Younger 2004). At the same time the iron concentration increased from 7 mg L^{-1} to 597 mg L^{-1} , thus exceeding the pre-flooding iron concentrations of $7\text{—}41 \text{ mg L}^{-1}$. Several months after cessation of the pumping test in 1995 and at two later occasions in 1997 and 1998 another set of vertical samplings were conducted in the Frances Shaft. It could clearly be shown that the mine water was stratified with two layers being separated at a depth of 260—270 meters below ground level with conductivities of about 3 mS cm^{-1} in the upper part and 20 mS cm^{-1} in the lower parts of the mine.

For the duration of the pumping test Nuttall et al. (2002) calculated Reynolds numbers of $Re = 190\text{—}350$ and concluded that the flow in the four interconnected adits, consequently, was turbulent. Whilst this might be true for the duration of the pumping test there is no evidence that during non-pumping times the flow in the adits must be turbulent as well.

From the data available it can be concluded that a major inflow from an adit of lower contaminated water between 260 and 270 meters below surface exists. This adit is hydraulically well connected with other adits at higher levels and within the part of the shaft above the stratification horizon laminar or turbulent convective mixing exists; otherwise the physico-chemical parameters of the water column above the stratification horizon would be different. It can clearly be seen that the lowest parts of the mine are not taking part in the convective flow of the higher parts of the mine and that the mine water quality in those deeper parts therefore is rather low.

12.8 Stahlberg Iron Mine, Siegerland, Germany

12.8.1 Description of Location

Müsen (part of Hilchenbach) locates the Stahlberg Iron Mine and is situated 14 km north-east of Siegen in the German Rothaar Mountains of the Variscan Rhenish Schiefergebirge. Another 16 km north of Müsen is the well known Meggen iron mine with the Sicilia shaft, another mine where mine water stratification with turbulent convective flow can be observed (Coldewey et al. 1999; Hasse 2000; see section 12.4).

Lusznat (1985) and Döring (1999) give a comprehensive description of the mining history and geological situation of the Stahlberg iron mine, which is dewatered by the *Kronprinz-Friedrich-Wilhelm-Erbstollen* drainage adit (1826—1875, 144-m-level). Today, the only accessible shaft is the № II shaft in the recreation area Stahlberg, with a depth of 670 m (-243.7 m a.s.l., 660-m-level) consisting of 14 sub-levels, beneath which are 12 flooded ones. On the 304-m-level, the № II shaft is connected to the 312 m deep Stahlberg shaft, which is also drained by the *Kronprinz-Friedrich-Wilhelm-Erbstollen* adit. Below the 304-m-level, the № II shaft was used for exploring the *Neues Mittel* ore body, which was mined in an overhead stoping technique.

Mining dates back to the year 1313 (open pit mining) and the mine was worked continuously until 1930. Host rock of the two main siderite ore bodies *Altes Mittel* und *Neues Mittel* (Fe: $\approx 37\%$; $\approx$ Mn: 8.12%) are the Lower Devonian Siegen and Müsen series consisting mainly of shales and schists. Where the siderite died away, quartz was the gangue material, accompanied by pyrite, copper pyrite, fahlore (0.98% Ag) and galena (0.06—0.07% Ag). During its more than 600 years of mining a total of $1.7 \cdot 10^6$ t of iron ores were produced (Lusznat 1985).

12.8.2 Appearance and Interpretation of Stratification

So far, only one measurement was conducted in the № II shaft of the abandoned Stahlberg Iron Mine. On November 12th 2000 a 500 m cable dipper with a resolution of 0.1 K was lowered to a depth of 500 m below the surface and a temperature measurement conducted at every 10 meters. Furthermore, two water samples were taken in 160 and 210 m below the surface (Fig. 108). Due to a malfunction of the on-site instruments for pH, redox, and electrical conductivity, no readings were conducted. On the water surface, which was 143 m below the shaft, the water temperature reached 12.5 °C and it increased to 13.8 °C at a depth of 500 m below surface, thus showing a temperature difference of 1.3 K. Between 410 and 500 m depth, no constant temperature measurements were obtained and the probe showed fluctuations between 13.8 and 13.9 °C, indicating turbulent flow conditions. No significant stratification could be seen from the chemical composition of the mine water, which is because the stratification between the two sampling points was not distinctive enough. Furthermore, the metal concentration of the mine water was reasonably low (Tab. 26).

Disregarding the small temperature changes within the depth profile, the mine water body can be distinguished into two parts: a part with a relatively stable temperature below the 304-m-level and another one with an upward decreasing temperature between the 304-m-level and the 144-m-level drainage adit. Considering the theoretical considerations and the experience from better examined shafts, turbulent convective flow seems to take part below the 304-m-level, whilst the upper part is dominated by a stagnant, diffusive flow situation, thus indicating a connection between the Stahlberg and № II shafts.

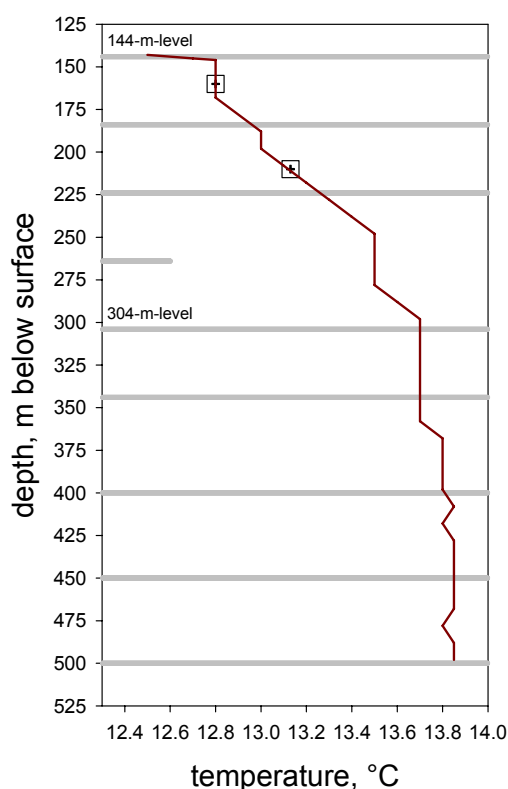


Fig. 108: Temperature—depth profile in the № II shaft of the abandoned Stahlberg/Germany mine (measurement on November 12th 2000). The gray horizontal lines indicate the depth of the different mine levels, the two squares the depth of the water samples. Resolution of the temperature probe: 0.1 K. Gray horizontal lines indicate the mine levels.

Tab. 26: Selected parameters of the Stahlberg № II shaft mine water. Depth in m below surface, Temp in °C; metals and main ions in mg L⁻¹.

Sample	Depth	Temp	Na	K	Ca	Mg	Li	Cl	NO ₃	SO ₄
MÜS-1211-160	160	12.8	4.4	2.2	45	46	< 0.05	9.6	< 1	64
MÜS-1211-210	210	13.1	4.4	2.2	45	46	< 0.05	9.5	< 1	61
Sample	Depth	Zn	Mn	Sr	Ba	Ni	Sb	Cu	As	Pb
MÜS-1211-160	160	1.12	0.27	0.13	0.05	0.03	0.01	0.01	0.005	0.004
MÜS-1211-210	210	0.94	0.29	0.14	0.06	0.02	0.01	0.01	0.005	0.004

12.9 Niederschlema/Alberoda, Saxony, Germany

12.9.1 Description of Location

The Niederschlema/Alberoda uranium mine and deposit is located 28 km south-west of Chemnitz/Saxony in a densely populated area of Germany. It has a depth of nearly 2,000 meters and consists of 50 main levels with more than 50 shafts connecting the different levels of the mine. The length of the mine works sums up to 4,150 km (Büder and Schuppan 1992) and the total volume to $36 \cdot 10^6 \text{ m}^3$. For 45 years the mine has been operated by the SDAG Wismut, a Soviet-East German Company. Due to economic, environmental and political problems of uranium mining and milling, in 1990 the Government of East Germany decided the closure of one of the largest gangue deposits for uranium in the world. In total, the mine produced about 81,000 t of uranium ore (Böttcher et al. 1991; Mühlstedt 1992; Bundesminister für Wirtschaft 1993; Barthel 1993).

Geologically, the mine is situated within the Lößnitz-Zwönitz-Syncline, a south-west—north-east striking fold zone north-west of the central Erzgebirge (Ore Mountains). The host rocks are of Ordovician, Silurian, and Devonian age, which are contact-metamorphosed in the vicinity of the post-kinematic granites of Aue, Kirchberg, Oberschlema, Auerhammer, Lauter, and Eibenstock. In conjunction with the granite intrusions, aplites, and kersantites intruded during the Stefanium and Autunium. Important for the emplacement of the mineralisation is the Gera—Aue—Joachimsthal thrust zone, which also hosts the famous Joachimsthal/Bohemia and Ronneburg/Thuringia uranium mineralisations. Typical rock types of the Niederschlema/Alberoda deposit are skarn, amphibolites, graphite-schists, meta-diabases, kersantite, quartzite, limestone, granite, marble and different types of schists. During mining, nearly 200 mineralised gangues were exploited (Janischewskij and Konstantinow 1962; Zentralinstitut für Physik der Erde 1989; Lange et al. 1991; Büder and Schuppan 1992; Stempok 1992; Meinel 1993). Important for the uranium mineralization and the hydrogeochemical development of the mine water are about 140 minerals (Schröder and Lipp 1990a, 1990b). Those minerals can be attributed to one or more mineralization cycles (called formations) of the Erzgebirge. Uranium and arsenic were mainly found in the U-quartz-carbonate-, quartz-polymetallic, and BiCoNiAg(U) formations (formation names after Baumann 1992). The most important uranium and arsenic minerals were uraninite (UO_2) and loellingite (FeAs_2).

In January 1991 the controlled flooding of the mine began, and will be completed in 2006. During the circulation of the mine water through the open mine works it becomes enriched in various elements, some of them being toxic to both humans and the environment. Therefore, the up to date remediation plans for the Niederschlema/Alberoda mine include treatment of the discharged water as soon as the water table reaches the Markus-Semmler dewatering adit (Gatzweiler and Mager 1993; Hagen and Lange 1995) within an active and a passive mine water treatment plant (Gerth et al. 2005).

At the beginning of the flooding only little was known about the hydrogeochemical, thermal, and hydrodynamic processes during the flooding. Hitherto numerous consultant works and a Ph.D. thesis have been undertaken on these subjects (Wolkersdorfer 1996), but the hydrodynamic conditions during the flooding and within the water body have only been outlined in general (Merkel and Helling 1995).

Meyer et al. (2002) state that “according to current knowledge, convective exchange will diminish and continue in mid- and long-term” without giving any references to their “current knowledge”. As discussed in the theoretical chapters above, “convective exchange” neither can “diminish” nor can it “increase”! Convective exchange either exists or it does not exist.

In the deepest parts of the mine rock temperatures of 70 °C were measured. It is not known if the water there currently has the same temperature, but in January 1991 a water temperature of 52 °C was observed at level -1800 (Günther Fröhlich, pers. comm.) and between June 1992 and December 1994 a maximal water temperature of 41 °C with an average of 36 °C was measured (Merkel and Helling 1995). Convective flow will therefore exist as long as the energy source exists and as long as no measures are undertaken to interfere with the convective cells to prevent convective exchange and mass transfer. A temperature of 33—35 °C for the end of the flooding process had already been predicted by Wolkersdorfer (1995).

12.9.2 Investigations conducted

Between 1992 and 1993 the author conducted discontinuous and continuous *in-situ* temperature and conductivity measurements (Wolkersdorfer 1996). Discontinuous temperature measurements were conducted with a 500 m downhole dipper at intervals between 1 cm and 10 m, depending on the degree of stratification known from each previous measurement. Continuous multi-parameter measurement were done with electronic probes from the login Gommern/Germany geophysical apparatus company. Furthermore, those probes were installed at predefined depths to investigate the course of the water rise and the change of the physico-chemical parameters during the rise of the water level. In addition, the mine operator conducted regular chemical analyses of the water according to the mine abandonment plan. Since the flooding process started the abandoned Niederschlema/Alberoda mine underwent an intensive chemical and physico-chemical monitoring, including the before mentioned measurements.

After 1993 the mine operator continued to monitor the mine flooding process, but the data was not available for discussion in this publication. A continuous downhole measurement in the № 371 shaft has been presented by Meyer and Schramm (2004) and is used to interpret today's hydrodynamic situation within the mine.

At the end of 1995 Wolkersdorfer 2001 conducted a multi-tracer test in the mine. This test evaluates the hypothesis that the water within the mine is well mixed because of convective flow within the well connected parts of the mine. Tracers were 4 differently coloured bulks of *Lycopodium clavatum* spores that were injected at different, pre-defined depths in 4 shafts.

12.9.3 Appearance of Stratification

During the active flooding period the water in the mine showed a clear density stratification with usually higher temperatures in the deeper parts of the mine and lower ones in the higher parts (with some exceptions). Most of the temperature logs of the *in-situ* shaft measurements showed relatively constant temperatures and electrical conductivities in the lower parts of the mine while the temperature and the electrical conductivity in the upper parts decreased with lower water depths. A continuous *in-situ* measurement reported by Meyer and Schramm (2004) showed that this situation seems to be stable even in 2004. Yet, the measuring location shaft № 371 is characterised by only 3 levels connected to the shaft: level -540, level -990, and level -1080. As Fig. 109 shows, the flooded parts below the level -540 (-227 m a.s.l.) show a relatively constant temperature profile, whilst above the level -540 the temperature steadily increases with lower water depths. This is a characteristic indication for a convective mixing in the lower and a diffusive flow in the upper parts of the shaft. If the shaft would be representative for the whole mine, the whole shaft should be a part of the overall flow regime of the mine. As the shaft is not connected to levels above level -540 the water above that level just stands in the shaft and diffusive flow dominates. It cannot be seen as an indication of an overall stratification in the abandoned Niederschlema/Alberoda mine.

Because of the repeated temperature measurements in the three shafts accessible at that time (mid 1990; up to 51 weekly to bi-weekly down-shaft measurements) the course of the stratification in three shafts could be followed. It could be seen that the stratification breaks down when the temperature difference between the upper and lower stratified part falls below a threshold value of about 5—6 K. Once the stratification breaks down, it does not develop again. The reasons for this behaviour are given in section 7.2.1.

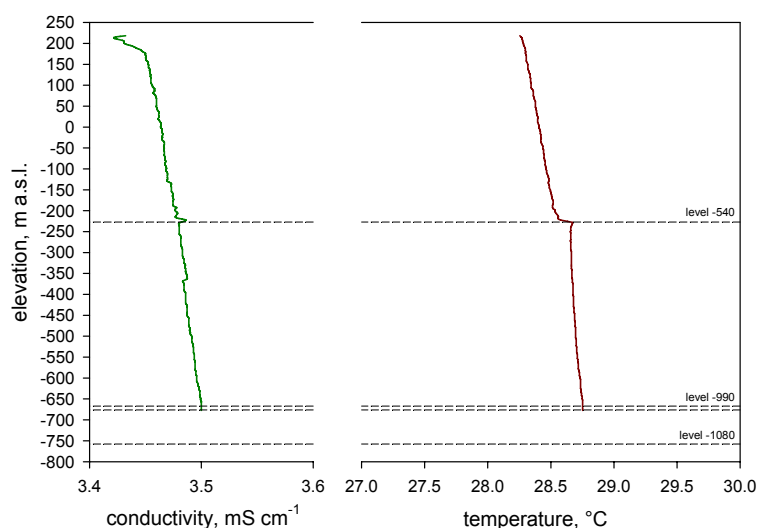


Fig. 109: Electrical conductivity and temperature measurement in the Niederschlema/Alberoda № 371 shaft on March 10th 2004 (modified after Meyer and Schramm (2004) and Jürgen Meyer, pers. comm.).

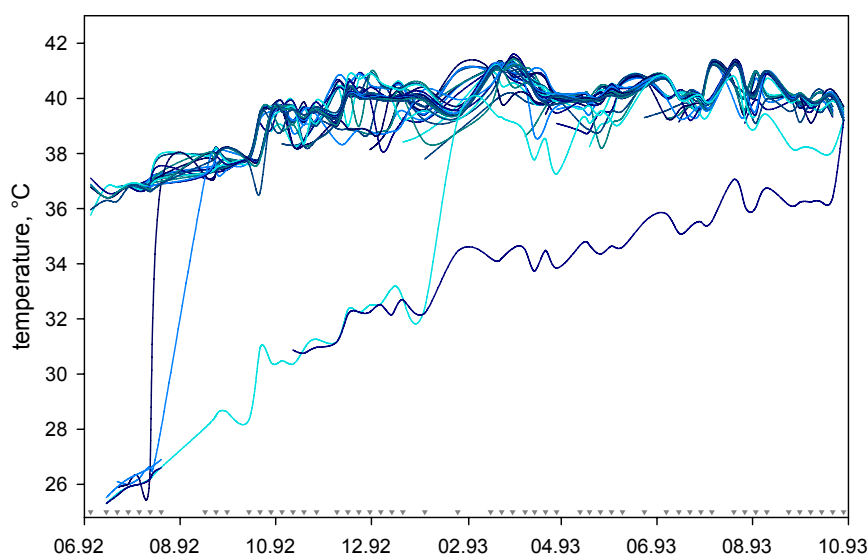


Fig. 110: Temperature–time course in the flooded № 366 II b shaft of the abandoned Niederschlema/Alberoda uranium mine showing a turbulent flow scenario in this shaft. The small triangles above the abscissa with the time scale indicate the individual measuring dates. Each line represents the temperature course in a predefined depth of the shaft (modified and amended after Wolkersdorfer 1996).

As the temperature showed significant fluctuations turbulent conditions in the observed shafts must be assumed. Furthermore, the measurements showed that the temperature differences within a single shaft ranged between several K and less than a K and independent from those differences stayed relatively stable (Fig. 110).

Because of the two tracer tests conducted (1992 and 1995) it became clear that large parts of the mine are extremely well hydraulically connected, because effective mean water velocities of up to 8 m min^{-1} were evaluated.

12.9.4 Results and Interpretation

As the measurements of Wolkersdorfer (1996) in a total of 7 shafts show, their hydrodynamic characteristics were very similar during the flooding process. Just one shaft showed differing conditions which could not readily be explained (shaft 372 b above level -1305; Fig. 111).

One of the most prominent similarities is that the changes in the physico-chemical parameters during the flooding process were similar in every shaft. This could especially be seen in the temperature progression of the shafts Nos 366 II b, 371 II b, and 372 b (Fig. 111), but also the electrical conductivity and pH-measurements back those similarities. All of those shafts had a stratification showing two shaft mine water bodies and a separate water body in the shaft's sump.

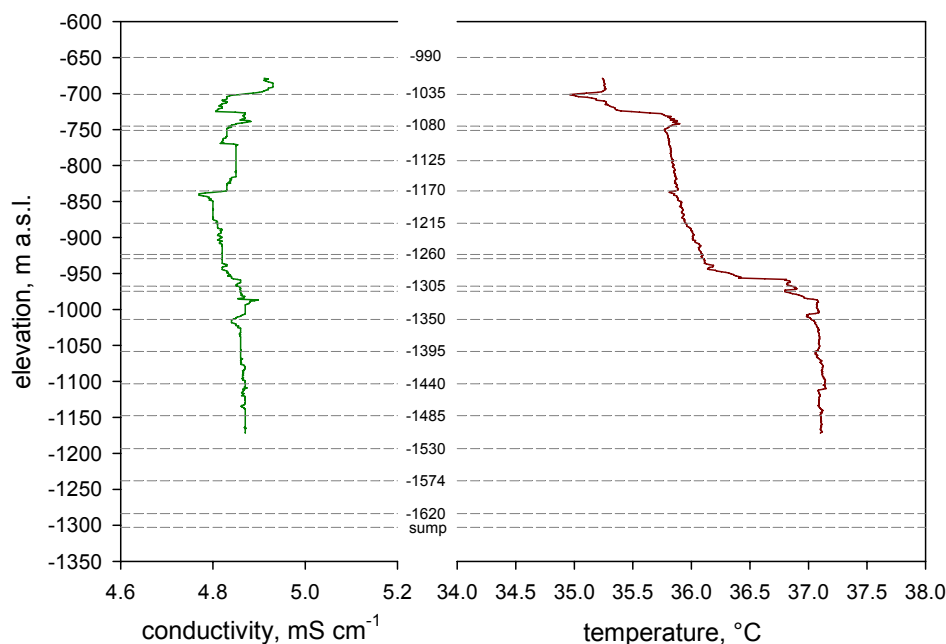


Fig. 111: Electrical conductivity and temperature measurement in the Niederschlema/Alberoda No. 372b shaft on March 15th/16th 1994.

The lower shaft water body ranges from the lowermost level in a shaft to the uppermost fully flooded level whilst the upper shaft water body consist of the water body above the last fully flooded level. Both water bodies have distinguishing properties: the lower shaft water body is characterised by similar physico-chemical properties throughout the whole water column, whereas the physico-chemical parameters in the upper shaft water body tend to have an increasing or decreasing trend up to the water surface, depending on the parameter. While the features of the lower shaft water body are typical for convective transport, the upper one indicates diffusive transport. The convective behaviour of the lower shaft water body is also supported by the oscillations that were measured by stationary temperature and electrical conduc-

tivity measurements and which can be seen as an indication of such a type of transport (Uerpmann 1980; Gebhart et al. 1988). Those features can be used to interpret the Niederschlema/Alberoda mine as an open thermosyphon system as they are described in Gebhart et al. (1988) and were studied by Bau and Torrance (1981a, 1981b).

Both in the shafts and the adits the rock temperature, resulting from geothermal processes, causes an interaction with the mine water. If the rock temperature is higher than the mine water temperature, the mine water is heated, while the water is cooled at the water table and where the rock temperature is below the mine water temperature. The warming of the mine water, usually in deeper parts of the mine, results in a lower density of the water and therefore, the water flows upward until it either reaches the water surface, a boundary layer, or a depth where the density of the up flowing water equals that of the water already being in that depth. Because of cooling at the water surface or the other water body, the water flows downward in the centre of the shaft thus inducing a natural convection, also called free convection (see section 7.1). Ideally, laminar convective flow cells would develop in the Niederschlema/Alberoda shafts and adits. Yet, the mine water velocity together with the numerous shaft installations (e.g. guide rails, air tubes, ladders, power supply lines, air ducts) and the surface roughnesses hinder such laminar flow regime. Therefore, it must be concluded that within the Niederschlema/Alberoda mine the following hydrodynamic situations exist:

- open loops (Fig. 112a)
- open loops with an isothermal reservoir (Fig. 112b)
- turbulent convective flow (Fig. 112d)
- areas with diffusive transport (Fig. 109)

During the flooding of the Niederschlema/Alberoda mine, and very likely also in other multi-shaft mines, the flooding process can be categorised into four stages. Those four stages are characterised by open loops, open loops with isothermal reservoirs and finally closed loops (Fig. 112). The flow in the matrix, due to the low hydraulic permeabilities, can be neglected here.

Stage I a

Seeping water collects in the sumps and mixes with the water already standing in the sumps. The water flowing into the shaft from the onsetting stations has usually a higher temperature than the water in the sump, as it comes from greater depths. Therefore, it does not mix with the water in the sump and forms a separate layer. If a closed loop has established, the water flowing through the onsetting station into the shaft can be seen as an entrance flow (Kranawettreiser 1989) and in conclusion no mixing between the sump and the shaft water body occurs.

Stage I b (Fig. 112a)

Water overflows the sumps and the first level and starts to rise in the shafts. According to the principles of communicating pipes, the water level establishes according to equation 31. The shafts and the first flooded level excluding the sump form an open loop with two reservoirs. In that system the flooded level is the heater (energy source) and the shafts and the water level are the coolers (energy sinks).

As has been summarised by Bau and Torrance (1981a) and Gebhart et al. (1988), the water flows up- and downward in the vertical workings driven by convective density driven flow. Even if the interconnected shafts have slightly different temperatures the overall flow regime is not significantly affected. For the Niederschlema/Alberoda mine this means that the water in the adits and shafts as well as the workings mixes with each other. The water in the sump

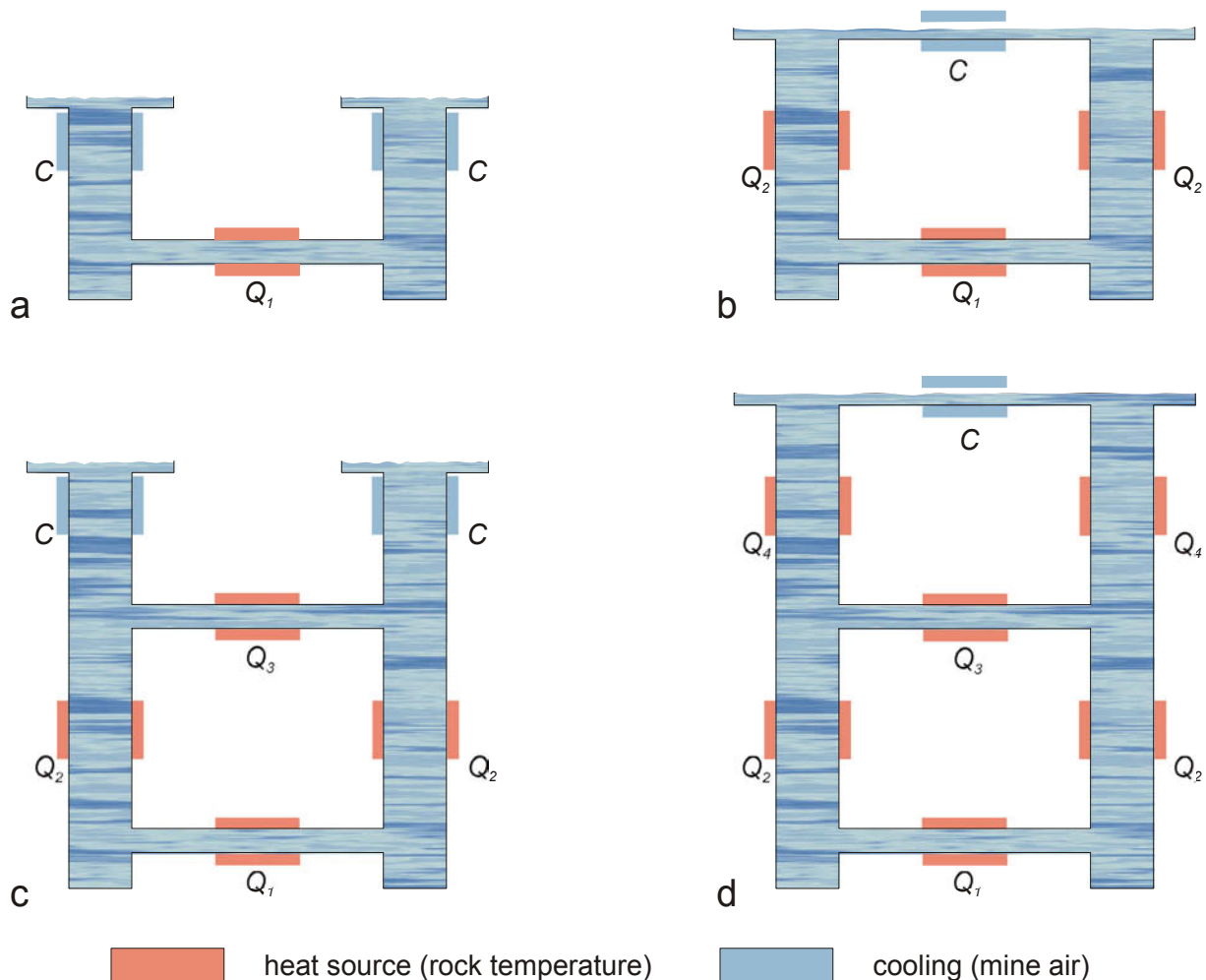


Fig. 112: Schematic representation of the flooding process in an abandoned underground mine as a combination of open and closed thermosyphons with free convection. a: water rebound in the shafts with the walls of the flooded adits as energy sources (Q_{1-4} : geothermal energy) and the shafts as energy sinks (C : cooling through mine air). Free convection in the shafts with two reservoirs; b: flooding of the next mine level. The walls of the flooded adit and the shaft are energy sources and the water surface in the partially flooded adit is the energy sink. Open fluid loop with one isothermal reservoir and free convection; c: further mine water rebound in the two shafts. A combination of a closed and an open fluid loop with two reservoirs, both with free convection; d: flooding of a mine level above a minimum of two flooded levels. Closed and open loops with one isothermal reservoir and free convection. Heat flux $Q_1 > Q_2 > Q_3 > Q_4$ (modified after Wolkersdorfer 1996).

only mixes with the rest of the water body if the density difference is lower than about 2 kg m^{-3} (Kranawettreiser 1989). The oscillation that can be expected by such a type of flow is well documented by the temperature measurements.

Stage II (Fig. 112b)

Starting with the flooding of the next level, the two reservoirs can form a single, near isothermal reservoir. Bau and Torrance (1981a) describe such a situation experimentally and theoretically. After a short initial phase, the water temperature at the flooded level reaches its maximum, whereas this initial phase can be seen as extremely short. Both the maximal temperature and the time until which the system reaches a stable mean temperature are a function

of the heating rate Q . Compared to stage I b no regular flow builds up, but more likely the flow changes its direction depending on the initial conditions. Only at the stationary flooding the temperature in all parts of the model mine is near homogenous. For a mine like Niederschlema/Alberoda such a system is unlikely to occur as the rock temperature does not completely cool down to air temperature.

A behaviour similar to that one described experimentally by Bau and Torrance (1981a) could be observed by a stationary measurement during which the over flooding of a level was observed. This measurement was conducted in August 1992 (Fig. 113).

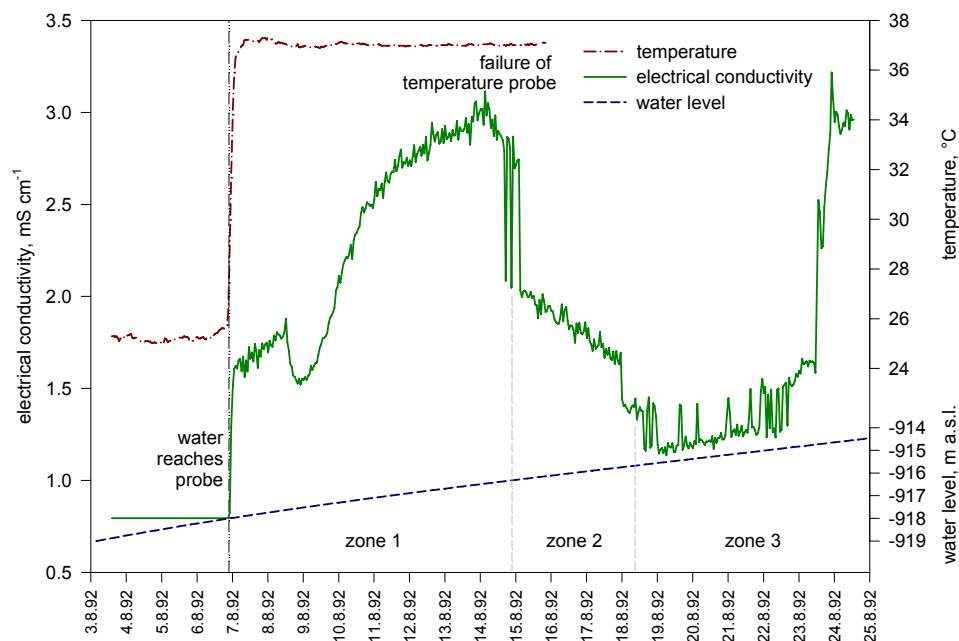


Fig. 113: Temperature and electrical conductivity measurements in the Nr. 366 II b shaft of the abandoned Niederschlema/Alberoda mine during the over flooding of the level - 1260 (Wolkersdorfer 1996).

Stage III (Fig. 112c)

In stage III the water further rises in the shafts connected to the second flooded level. Only the water below the on-setting stations is directly connected with each other, but not the water in the shafts. As could be shown by several measurements, the quasi-isolated water in the shafts cools relatively quickly and only after a longer period of time the water in the shaft gets the temperature of the already flooded parts below the on-setting station. This diffusive and convective temperature adaptation between inflowing water (ca. 20 °C) and mine water (ca. 40 °C) is hindered by different densities of the water as a function of their temperature and mineralisation. Also the closed flow field of the lower shaft water body and the geometry of the shaft hinder a complete mixing of the upper with the lower water body. Furthermore, in case there is an intermediate layer of thickness $z(v, g, \rho_1, \rho_2)$ between the upper and lower shaft water bodies, a direct interaction between those two layers is nearly impossible (Kranawettreiser 1989).

In a closed loop, temperature as well as velocity varies continuously. In some critical cases, depending on the border conditions, it is even possible that the flow field reverses (Bau and Torrance 1981a). Creveling et al. (1975) and Damerell and Schoenhals (1979) could show that the oscillating flow in closed loops is continuously repeated through time.

Stage IV (Fig. 112d)

Stage IV differs in so far from stage II as the connection between the shafts is possible over more than one level. The loop can incorporate two or three levels, depending on the available energy. Such an effect could be observed at the 1992 tracer test, where the tracer was injected into two shafts at different depths. After 5 weeks, a homogenous mixing over 9 levels (about 400 m vertical distance) could be observed. Considering the experiments by Bau and Torrance (1981b) with asymmetrically heated open convection loops also shafts with a colder water temperature than the overall temperature may be included in the fluid loop (in that case shaft 372 b: Fig. 111).

In summary, the following conclusions for the final flooding scenario with pumping below the Markus-Semmler-adit level might be drawn from the 4-stages flooding development:

- Flooding stops at stage III: upper shaft water body is not in convective contact with the rest of the mine water pool. Pumping in such a shaft causes contaminated water from lower parts to flow into the pumping shaft.
- Flooding stops at stage IV: the last flooded level is completely included in the overall fluid loop causing that pumped drainage water to be polluted for a long time being.

As none of those two scenarios results in a significant water quality improvement another solution is needed. This solution would be a vertical hermetic sealing of the mine below the last flooded level. When doing so, it is important to either interrupt or to prevent large convective flood loops which quickly transport the pollutants. Sealing just some of the on-setting stations is not a recommended measure as small dams quickly might break and therefore would not help to prevent the loop.

To conclude and to point out the impact of such convective loops, Bau and Torrance (1981b), who investigated the advantages of fluid loops in solar water heaters shall be cited here:

„Free convection loops provide a means for circulating fluid without the use of pumps“.

12.10 Felsendome Rabenstein, Saxony, Germany**12.10.1 Description of Location**

Rabenstein/Germany is a suburb of Chemnitz/Saxony and its former limestone mine is located at the western edge of the town. Since 1936 it serves as a popular visitor mine called *Felsendome Rabenstein* (Geißler et al. 1986).

Already early in medieval times open cut mines in the close vicinity of the Rabenstein underground mine extracted limestone (Riedel 1993). In 1834 the Rabenstein underground mine opened, and 40 years later the 35 m deep Maschinen Shaft was sunk, connecting the three lowermost levels with each other (15, 22, and 27 m below surface). At the end of the mine's lifetime four levels existed: the upper and lower level as well as the 1st and 2nd deep levels (level 3 and 4) extending over an area of 250 × 100 m with caverns ranging from 6–8 m height (Fig. 114). An underground drift adit in the *Grüne Grotte* (green grotto) and the Maschinen Shaft connected the two deep levels with the lower level. To dewater the mine, a 220 m long drainage adit to the Pleiß brook was constructed in 1855, which still dewateres the mine from the lower level with daily mean discharges between 3 and 40 m³ and a weighted average of 12 m³ d⁻¹. Extreme precipitation can induce total discharges of up to 120 m³ d⁻¹. During the active mining period the water from the 1st and 2nd deep level was pumped into the adit. In 1902, six years before the closure of the mine, the two deep levels were allowed to flood and since then the shaft is filled with a 15.8 m water column. At two points, the *Grüne Grotte* and the Maschinen Shaft which are 140 m apart from each other, the flooded part of

the mine is accessible. Dietrich (1965) calculated a flooding time of 12 weeks for the two underground levels, which seems to be rather low considering a mean flow of $12 \text{ m}^3 \text{ d}^{-1}$. Between 2002 and 2005 the electrical conductivity of the mine water was relatively constant at $1,190\text{--}1,260 \text{ }\mu\text{S cm}^{-1}$, with a pH of 6.5–8.9, an E_H of 350–510 mV, and a temperature of 7.5–8.7 °C. Slightly different values could be measured at the *Grüne Grotte*, from which the water flows through the flooded part to the Maschinen Shaft: electrical conductivity $800\text{--}1,000 \text{ }\mu\text{S cm}^{-1}$, pH 7.1–8.7, E_H 340–510 mV, and a temperature of 4.8–6.8 °C.

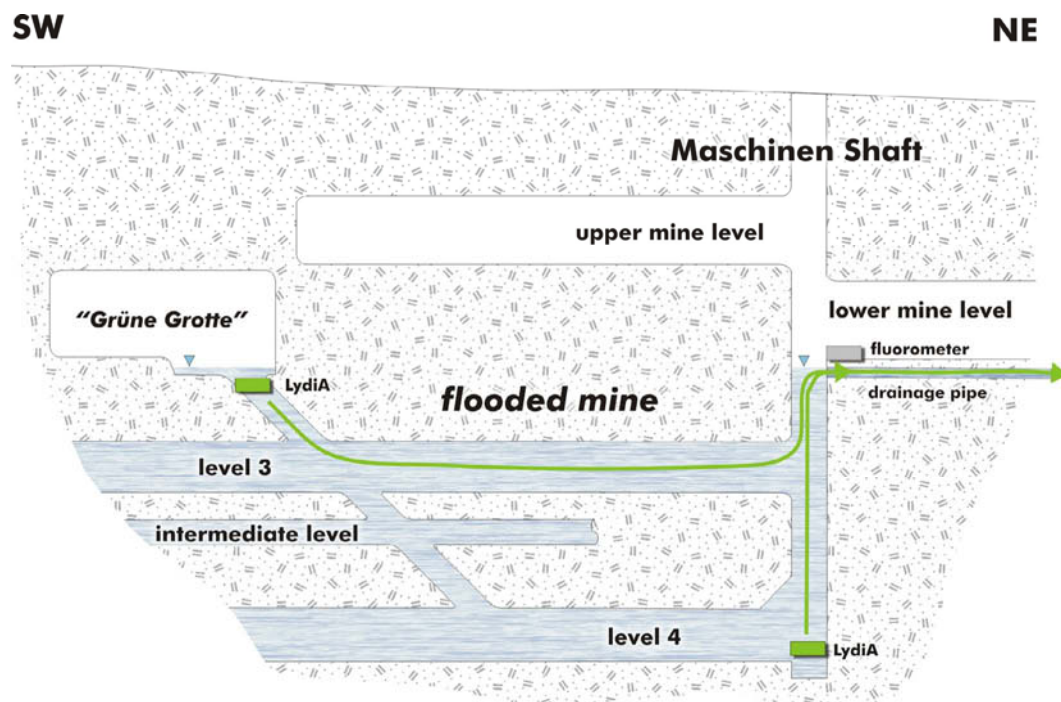


Fig. 114: Sketch of the Felsendome Rabenstein/Germany visitor mine with the two tracer injection sites, the location of the fluorometer, and the likely flow paths of the tracer.

LydiA: Tracer injection Apparatus (Fig. 78). Not to scale in horizontal direction and above the water table.

Geologically, the mined strata comprise mainly metamorphosed Palaeozoic limestone, a coloured calcite marble, in combination with phyllites, alum shale like black phyllites, and amphibolites. Both, in the limestone and especially in the phyllites, pyrite occurs frequently and in vugs copper pyrite, often oxidised to malachite, can be found. The whole formation is slightly folded with north-west striking folding axes, and faults in north-eastern direction with offsets in the range of 5 m are common (Dietrich 1965; Freyer et al. 1982; Geißler et al. 1986). Some of the faults are hydraulically active, as water penetrates from the surface to the mine during high precipitation periods (e.g. a fault between the *Blauer Salon* and the *Gnomengang*) and divers reported that they can regularly see air bubbling beading like and in straight lines from the 2nd deep level through the rock into the 1st deep level.

From the available data the volume of the Maschinen Shaft, the *Grüne Grotte* incline, and the flooded 1st and 2nd deep levels could be roughly calculated to be 124 m^3 , 120 m^3 , and $2,100 \text{ m}^3$. Only the parts which were thought to be directly affected by the tracer test were used in the calculation. The total volume of each level must be another 20–30,000 m^3 of volume. From that data a total flooding time of several years seems to be more likely than 12 weeks.

12.10.2 Appearance of Stratification

According to the known records and the reports of divers, the flooded and partly backfilled Maschinen Shaft is connected to the 1st and 2nd deep levels at depths of 6 and 12 m below the surface of the water. Several small blind shafts and the incline within the *Grüne Grotte* connect the 1st deep level with the 2nd deep level. No data are available about the shaft's outflow because no precise flow measurements are possible there, but flow estimations gave a result of 1–2 L min⁻¹. Only a poorly developed thermal stratification can be found in the Rabenstein Maschinen Shaft. The temperature in the shaft is relatively constant at 7.9 °C and small temperature fluctuations can be observed at the two onsetting stations (Fig. 115).

Such a quasi-stagnant situation is typical for single shaft systems (e.g. Georgi Unterbau) where the geothermal gradient is not high enough to start free convection, nor does a major regional forced convection exist. Water in the shaft flows very slowly and it can be assumed that there is a laminar Poiseuille flow situation in the upper part of the shaft and diffusive flow in the lower part of the shaft.

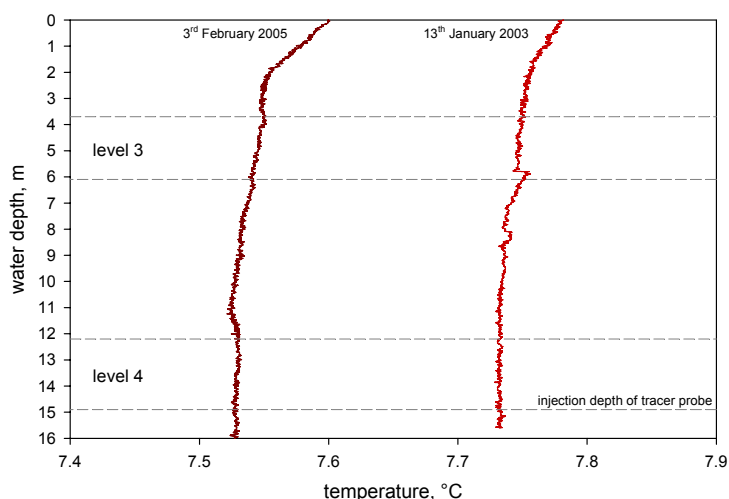


Fig. 115: Temperature—depth profile in the Felsendome Rabenstein machine shaft before and after the tracer test.

12.10.3 Investigations conducted

In 2002 the flooded part of the mine was investigated hydrogeologically and the potential for a mine water tracer test was evaluated. It could be shown that the water quality is good and that no negative impacts could be expected for a tracer test with sodium fluorescein, as the pH was slightly basic and therefore no decomposition of the tracer was expected. A discontinuous temperature log in the Maschinen Shaft showed a small temperature variation at the two flooded onsetting stations with no temperature shift. Later, continuous temperature logs revealed a small positive temperature shift of about 0.05 K / 10 m. Therefore, this shaft was selected for a tracer test in a mine with a low vertical flow characteristic. Thus that mine proved to be an example for the simple mine geometry type as in the case of the Georgi Unterbau mine.

Between 2002 and 2005 the mine was visited 16 times and the physico-chemical on-site parameters were measured regularly. One full analysis of the mine water was conducted showing that the water is a Ca-Mg-SO₄-HCO₃-type water. Five temperature and electrical conductivity logs were conducted in early 2003 and 2005 before and after a tracer test with sodium fluorescein.

On March 21st 2003 a tracer test with sodium fluorescein, scheduled for a duration of about 3 months, was started in the Felsendome Rabenstein mine. With a tracer probe (LydiA; Fig. 63, Fig. 78) 15 g of sodium fluorescein were lowered to a depth of 15 m in the Maschinen Shaft and 250 g into a depth of 2—3 m within the incline at the *Grüne Grotte* (Fig. 114). 96 weeks in the test the tracer concentration came down to the back ground values and was therefore brought to an end. Consequently, the Felsendome Rabenstein Tracer test has the longest duration of all tracer tests ever conducted in an underground mine (Fig. 116).

To clarify which of the two peaks belongs to which injection place, the vertical distribution of the tracer in the shaft was measured four times in the Maschinen Shaft and once in the *Grüne Grotte* incline.

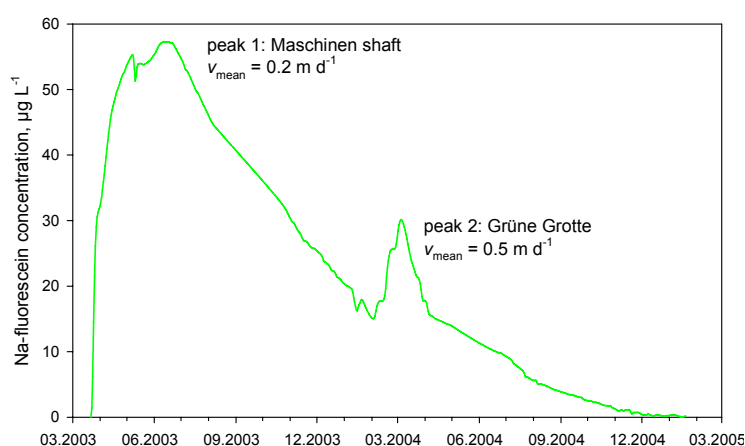


Fig. 116: Breakthrough curve of the 2003—2005 Felsendome/Rabenstein tracer test with Na-fluorescein.

12.10.4 Results and Interpretation

No relevant stratification could be observed in the Felsendome Rabenstein mine. Neither the temperature nor the electrical conductivity plots show signs of stratification, though small fluctuations in the area of the onsetting stations were observed. While the temperature slightly increases (by about 1.5 K) from bottom to top, the electrical conductivity decreases by about $100 \mu\text{S cm}^{-1}$, proving a mainly diffusion-based flow scenario in the Maschinen Shaft with only a small advective component.

Two tracer peaks could be observed, the first one 65 days, the second one 350 days after the tracer injection (Fig. 116, Tab. 27). Whilst the first peak comes from the tracer injected at a depth of 15 meters in the Maschinen Shaft, the second one is the result of the tracer injected 171 m away into the *Grüne Grotte* incline. From that data an effective velocity v_{eff} of 0.10 m d^{-1} was calculated for peak 1 and 0.48 m d^{-1} for peak 2. The smaller peak after peak 2 might be induced by divers training in the flooded underground mine.

In contrast to the first peak which has an extremely long tail of about 620 days, the second one has a tail of only 120 days, indicating a larger longitudinal dispersion coefficient of $7.1 \cdot 10^{-6} \text{ m}^2 \text{ s}^{-1}$ for peak 1 and a smaller one of $1.3 \cdot 10^{-6} \text{ m}^2 \text{ s}^{-1}$ for peak 2. From the vertical tracer distribution in the Maschinen Shaft it became clear that the first peak results from the tracer injected into the Maschinen Shaft and the second one from the tracer in the *Grüne Grotte*. The long tail of the first peak shows that the mass transport from the 2nd deep level to the surface is possibly diffusive based and the flow from the *Grüne Grotte* must be via the connecting adit between the incline and the Maschinen Shaft on the 1st level. If the water had

followed the connecting adit at the 2nd level a tailing similar to peak 1 would have been observed. It can be concluded that the transport from the 2nd level into the region of the 1st deep level is probably diffusion dominated whereas the shaft above the 1st deep level shows Poiseuille dominated flow.

Tab. 27: Results of the Rabenstein tracer test. Peak 2' is the reconstruction of the travel times for the *Grüne Grotte* tracer from the *Grüne Grotte* to the onsetting station of the 1st deep level at the Maschinen Shaft. It has been calculated from the travel times and distances of peak 2 less the distance to the onsetting station and the mean travel times calculated from peak 1 and the mean of the effective and mean velocities of peak 1.

Peak	Effective velocity v_{eff}			Mean velocity v_{mean}		
	Time, d	Distance, m	Velocity, m d ⁻¹	Time, d	Distance, m	Velocity, m d ⁻¹
1	149	15	0.10	65	15	0.23
2	356	171	0.48	350	171	0.49
2'	320	165	0.53	314	165	0.52

12.11 Straßberg, Harz Mountains, Saxony-Anhalt, Germany

12.11.1 Description of Location

The German Straßberg mine is situated in the eastern Harz Mountains approximately 20 km south of Quedlinburg and 6 km west of Harzgerode and was the most important producer of fluorite in the former “German Democratic Republic” (Mohr 1978). At the beginning of mining, silver, copper, and lead were the targets of the miners. From the 18th century until 1990 mining focused on fluorspar, which was mainly found in the deeper parts of the mine (Bartels and Lorenz 1993). Between 1950 and 1970, the *VEB Harzer Spatgrube* joined the three main deposits of the Straßberg mining district by driving two deep adits on the 5th and 9th level (from north to south: Brachmannsberg pit: № 539 Shaft, Straßberg pit: Fluor Shaft and Glasebach pit: Glasebach Shaft). Finally, in 1991 the Straßberg fluorspar mine was closed due to economic and environmental reasons (Kuyumcu and Hartwig 1998) and the flooding of the mine began.

Geologically, the mine is situated in the Mid Harz Fault Zone close to the boundaries of the Lower Devonian Harzgerode zone and the Upper Devonian Tanne zone. Whilst the Harzgerode Zone consists of folded schists, shales, grey wake, limestone, lydites, metabasites, and metabasite tuffs, the Tanne zone comprises of grey wakes and shales (Schwab 1999). Besides fluorite, the Permian to Cretaceous hydrothermal poly-metallic mineralization of the vein structures comprises approximately eight ore and less than a dozen secondary minerals, among them pyrite, marcasite, chalcopyrite, galena, sphalerite, and arsenopyrite (Kuschka and Franzke 1974; Beck 1981; Junker et al. 1991).

12.11.2 Appearance of Stratification

On May 31st 1991, by stopping the drainage water pumps, the Straßberg and Glasebach underground pits started to be flooded. Between July 1992 and August 1998 accompanying *in-situ* temperature and electrical conductivity measurements within the № 539 shaft and the Fluor shaft (310 and 147 m deep, respectively) clearly showed that stratification within the water body was taking place (Kahmann and Heinrich 1998; Kindermann 1998; Rüterkamp and Meßer 2000). In the Fluor shaft, three water bodies, being separated from each other at the 2nd (328 m a.s.l.) and 5th (243 m a.s.l.) levels, established. Only two water bodies, separated by the 4th (357 m a.s.l.) level, could be recognised in the № 539 shaft. Evidence for the

stratification were differences in temperature, electrical conductivity, and metal-concentration between each of the water bodies, the uppermost always low in those parameters, the lowermost with an increased contamination by iron, manganese, and sulphate.

Consequently, in 1993 the DMT – German Mining Technology, proposed to construct three new adits (Brachmannsberg adit, Biwender adit, new Glasebach adit), to drain and treat the lower contaminated mine water within the uppermost water bodies at a water level of 357.7 m a.s.l. (Rüterkamp and Meßer 2000). These three adits were built between 1995 and 1998 and are draining the mine since then. Two provisional active water treatment plants near the Fluor shaft and in the Uhlenbach Valley (close to the entrances of the Brachmannsberg and Biwender adits) are cleaning the circum-neutral Fe- and Mn-enriched mine water (pH 6.2–8.0, $n = 22$, 95% conf.) by using conventional liming technology.

After completion of the 3-adit-system in 1998 the annual water budget of the Straßberg mine increased by almost $2 \cdot 10^6 \text{ m}^3$. Furthermore, the stratification totally broke down in the № 539 shaft and partly in the Fluor shaft (Rüterkamp and Meßer (2000), appendixes 1 and 5), resulting in a generally higher contaminant load than expected. Similar circumstances were already found and investigated during the flooding of the Niederschlema/Alberoda mine (Erzgebirge/Germany, Wolkersdorfer et al. 1997a). There, as long as the water level was under a main working level, stratification could be seen in the shafts above the last flooded level. When the main working level was flooded a new loop established and the stratification broke down immediately (Wolkersdorfer 1996). A very similar behaviour has been observed at the Frazer's Grove/United Kingdom fluorspar mine where the stratification broke down after the water started to drain through the main shaft (Johnson and Younger 2000).

12.11.3 *Forced Flow Situation and Tracer Investigations*

Historically, the abandoned and flooded Straßberg/Germany fluorspar mine consisted of three mine departments, with one central shaft in each department: № 539, Fluor shaft, and Glasebach shaft. In the 2nd half of the 20th century, those departments were connected with each other through two deep galleries. Today most of the mine water discharges through three drainage adits which were constructed in the 1990s and in which the flow is controlled through Thomson weirs. In June 2002, the water densities in the three shafts № 539, Fluor shaft, and Glasebach shaft were $999.468 \text{ kg m}^{-3}$, $999.173 \text{ kg m}^{-3}$, and $999.524 \text{ kg m}^{-3}$, respectively. To prevent forced flow between the three mine parts – as preferred by the mine operator –, the weirs at the outflows therefore should have been fixed at 357.69 m a.s.l., 357.70 m a.s.l., and 357.59 m a.s.l. In fact, they were fixed at 357.79 m a.s.l., 357.70 m a.s.l., and 357.76 m a.s.l., forcing the water to flow from the № 539 shaft to the two other shafts.

To investigate the hydraulic conditions, a tracer test was conducted by Wolkersdorfer and Hasche (2001a). At the same time, an extensive water sampling campaign (Wolkersdorfer 2002a) and down hole measurements in the № 539 and Fluor shafts were used to support the tracer test (Fig. 117, Fig. 118). A second tracer test at the same location was conducted in 2003 with results similar to the 2000 tracer test.

Between the two shafts significant differences in the temperature and electrical conductivity profiles emerged. Whilst the № 539 shaft lacks a clear stratification, the Fluor shaft is stratified as indicated by the temperature profiles and to a lesser degree the conductivity. From the two tracer tests it is known that the water in the Fluor shaft flows upward with a mean effective velocity of $\text{ca. } 1.4 \cdot 10^{-2} \text{ m min}^{-1}$. This explains why the stratification is not as developed as in shafts with smaller water velocities. What can clearly be seen is the shafts sump at the 9th level and the colder, inflowing water from the Brachmannsberg mining area at the 5th level. Furthermore, there seems to be some inflow from the 3rd level.

The Straßberg/Harz case study clearly shows that the stratification in flooded mines is not a static condition but might be disturbed easily. As in the case of the Frazer's grove № 2 shaft, the stratification broke down at the time the water was allowed to flow freely out of the shaft (in Straßberg this was the case when the three-adit-system came into full functioning). The stratification in the Straßberg mine would have been only stable if the three distinct parts of the mine would have been hydraulically separated from each other. Without that hydraulic separation the water flows, driven by forced convective flow, from the northern Brachmannsberg mining field to the central Straßberg and the southern Glasebach mining field.

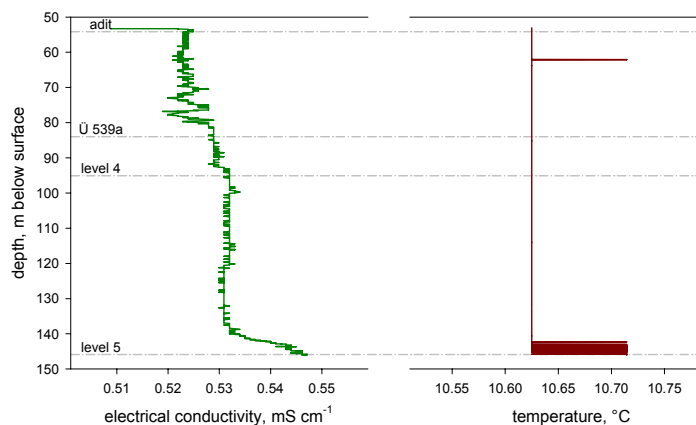


Fig. 117: Temperature and electrical conductivity measurement in the Straßberg/Harz № 539 shaft. Both, conductivity and temperature show that there is no stratification in the shaft and there seems to be some minor inflow from the 4th level.

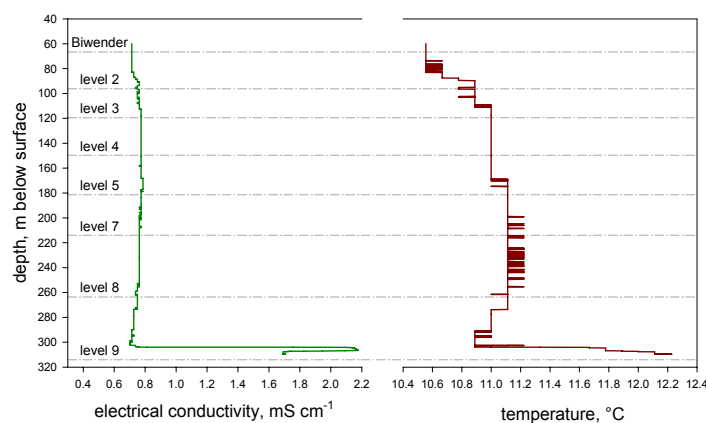


Fig. 118: Temperature and electrical conductivity measurement in the Straßberg/Harz Flour shaft. Both, conductivity and temperature show that there is no significant stratification in the shaft except at the 5th level as an indication of the water flowing into the shaft from the northern adit, connecting the Brachmannsberg mining field to the Fluor shaft.

12.12 Reiche Zeche, Erzgebirge, Saxony, Germany

12.12.1 Description of Location

The Freiberg/Saxony mining district in Germany once belonged to the richest silver deposits in Europe (Jobst et al. 1994). In 1168 AD mining started at a small place called Christiansdorf

which later became a part of Freiberg. Due to economic reasons the last mine producing in Freiberg was closed in 1969 and thereafter the uncontrolled flooding of the mine workings started. Because the deepest drainage adit is the 128 years old *Rothschönberger Stollen*, nearly all the mine water drains through that adit into a northerly direction and discharges 18 km north of Freiberg into the rivers Triebisch and Elbe. According to different sources, the total flooded mine volume is about $2\text{--}5 \cdot 10^6 \text{ m}^3$ (Kolitsch et al. 2001) and the discharge volume $0.4\text{--}1.1 \text{ m}^3 \text{ s}^{-1}$ (Weyer 2003). Since the adit was completed in 1877 the mine water has been discharging into the Triebisch without further treatment, and even after the collapse in 2002 nothing changed in this regard.

Below the flooding level of *Rothschönberger Stollen* are 494 m of flooded mine workings which were driven into the polymetallic ore veins of the Erzgebirge (Oelsner 1958). Therefore, the water penetrates through the flooded shafts, galleries, backfilled veins and open veins and gets enriched in nearly all metals of the periodic table, as can be seen from the chemical analyses (e.g. Dzombak and Morel 1990; Baacke 2000). Although the *Rothschönberger Stollen* is the main drainage gallery, several other drainage galleries for isolated parts of the mine exist, thereunder the *Königliche Verträgliche Gesellschaft Stollen* and the *Fürstentollen*, which dewater into the river *Freiberger Mulde* (Becke et al. 1986).

Though numerous studies were conducted about the Freiberg mine waters, no comprehensive hydrogeological investigation has been done so far. This is mainly because only a small part of the mine is accessible from the surface and underground but also to the complicated responsibilities concerning the mine and the flooded part of the mine.

Many parts of the Freiberg mining district, especially above the flooding water level, have extremely acid mine waters with pH as low as 2 (Baacke 2000), but it reaches 6–8 at the *Rothschönberger Stollen* portal (Yefimochkina 2004) with a conductivity of $0.7\text{--}0.9 \text{ mS cm}^{-1}$. This shows that enough buffer capacity exists, which mainly comes from the carbonates in the *Braunspatformation* (siderite formation) and the *BiCoNiAg-Formation* (bismuth, cobalt, nickel, silver formation). Geochemically, the discharge water can be classified as a Ca-SO₄-type, and in some parts of the drainage water system as a Ca-Mg-SO₄-type, thus showing that calcite and dolomite are relevant buffering minerals.

Tab. 28 lists some important parameters of the *Rothschönberger Stollen* drainage waters between 1996 and 2004. Dzombak and Morel (1990) calculated the average mass loads for some selected parameters for the three relevant discharge adits (Dzombak and Morel 1990) and if they are used as a conservative data for estimating the mass loads having left the mine since its closure in 1969 some extreme values can be calculated: 560 kg of As, 4,300 t of Zn, 1,400 t of Mn, and 108 t of Fe.

Tab. 28: Selection of mine water parameters from the drainage adit *Rothschönberger Stollen* (from Baacke et al. 1996; Beuge et al. 2001; Yefimochkina 2004). Electrical conductivity (Cond) in $\mu\text{S cm}^{-1}$, temperature in $^{\circ}\text{C}$, concentrations in mg L^{-1} . Values are rounded in accordance with the original data.

Parameter	pH	Cond	Temp	Ca	Mg	Cl	SO ₄	Zn	As	Cd	Pb	Mn
Min	6.8	620	12	100	24	35	155	2	0.001	0.01	0.002	0.8
Max	7.5	980	14	112	40	54	300	5	0.003	0.03	0.4	2

Some of the already existing publications gave a conceptual model of the hydrogeological situation in the flooded and the non-flooded part of the Freiberg mining district (e.g. Baacke and Degner 2000). Based on that conceptual model my working group conducted a tracer test in the shaft *Reiche Zeche Schacht* (Fig. 15) in 2002 but was not able to detect any tracer at the outflow of the shaft, though the existing models predicted that a tracer should be found at the

shafts outfall. Currently, it can only be concluded that the published conceptual hydrogeological and hydrodynamic models of the Freiberg mine are not complete and that an extended conceptual model is necessary. Yet, as described above, the mine is not accessible in all parts and an artificial tracer test is therefore not possible at the current situation.

From a hydrodynamic point of view the existing conceptual model is impossible, in that water cannot flow down a shaft against the hydrostatic pressure. Water always flows in the direction of the lowest hydrodynamic pressure. Furthermore, water can flow in fluid loops if the hydrodynamic situation allows fluid loops or it can flow through flooded levels in the direction of the lowest hydrodynamic pressure. Moreover, the water flow can split in one of two directions relative to the different hydrostatic pressures at each of the end-points of the possible flows. Tracer tests conducted by the Hydrogeology Department of Freiberg University in other mines (e.g. Wolkersdorfer and Hasche 2001a) clearly showed that the conceptual model of the Freiberg mine needs relevant amendments.

12.12.2 Investigations, Appearance of Stratification, and Interpretation

The only accessible place in the Reiche Zeche mine where the water below the flooding level can be studied is the *Reiche Zeche Schacht*. No other location, so far, is known, where the flooded part of the mine could be investigated to a certain degree. Therefore, the only depth dependent measurements known are from this location. First continuous physico-chemical measurements have been conducted by Peter (1983) and Zittnan et al. (1990). Their measurements showed that the water in the flooded shaft was stratified at the $\frac{1}{2}$ 10. Sohle (level $\frac{1}{2}$ 10) and the 13. Sohle (level 13) and consequently three mine water bodies could be separated from each other. Though the temperature showed a clear stratification in 1982, this was not true for the chemical composition. Zittnan et al. (1990) also conducted velocity measurements by using radioactive tracers and found up- and downward movements of the mine water. Between the water surface and a depth of 412 meters they measured velocities between 0.005 and 0.47 m min⁻¹ within 4 convection cells. Interestingly, the cells are not associated with the different levels in the shaft and they also differ from the experiences obtained by Czolbe et al. (1992) in the Peißen shaft, where water was flowing upwards on the shaft's walls and downward in the middle of the shaft. Because their findings are therefore not easily explainable they must have another reason. Flow measurements in 2001 proved velocities between 0.02 and 0.06 m min⁻¹ at 12 different depths and an overall trend for upward movement of the water (Sebastian Kolitsch, pers. comm., the range of that measurement is drawn with two vertical lines in Fig. 119; more than a decade lies between the two velocity measurements). Yet, also “no flow” was measured, thus indicating that turbulent convection cells are existing in the shaft. Therefore, it can be concluded that Zittnan et al. (1990) probably measured partly the up welling part of the convection cell and partly the down welling one. On the other hand, a tracer test conducted by the author seems to back the latter investigations.

In April 2002 the author started to conduct a tracer test in the *Reiche Zeche Schacht*, based on the results of the previous investigations (Fig. 120). Na-fluorescein was chosen as a tracer which proved to be stable in the mine water as has been shown by an earlier laboratory investigation with the Reiche Zeche mine water and Na-fluorescein. 50 g of Na-fluorescein were injected into the shaft at a depth of 75 m, between the $\frac{1}{2}$ 5. and 6. Sohle (level $\frac{1}{2}$ 5 and 6). From the velocity measurements (ca. 0.06 m min⁻¹), the shafts cross section (4–5 m²) and a piston flow model the tracer was expected to reappear after a minimum of 17 hours plus the opening time of the tracer probe (approx. 6–10 hours). Yet, even after 17 days no tracer could be observed at the outflow of the shaft. Interestingly this finding complies with the velocity measurements of Zittnan et al. (1990) who observed down welling between those two levels. If that result would represent the real flow situation in the flooded Reiche Zeche shaft – and from all the measurements conducted so far this hypothesis cannot be rejected – the

often published conceptual hydrogeological model of the mine (e.g. Baacke 2000; Baacke and Degner 2000; Bayer without year) must be reconsidered. The real situation in the mine seems to be that the out flowing mine water mainly comes from the $\frac{1}{2}$ 5. Sohle (level $\frac{1}{2}$ 5) and that the water below this level forms convective cells that are almost separated from the upper water body.

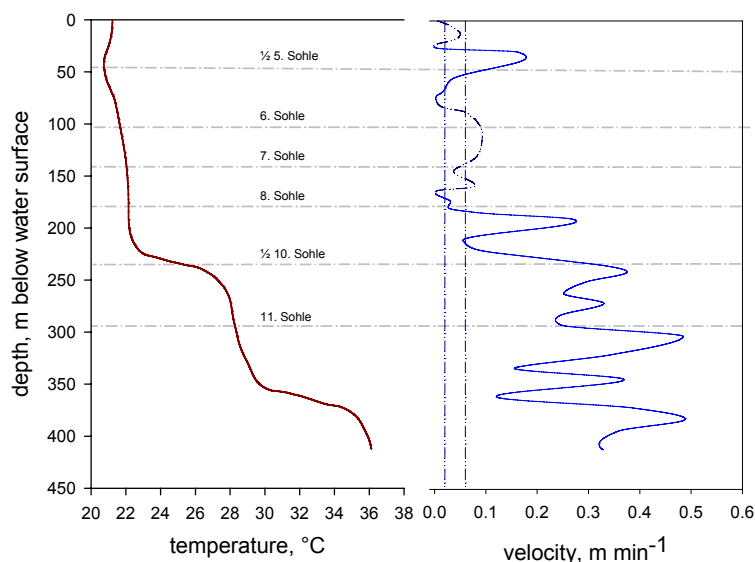


Fig. 119: Temperature and velocity measurement in the Reiche Zeche shaft (modified after Zittnan et al. 1990). Sohle: level. No error-bars were given by the authors.

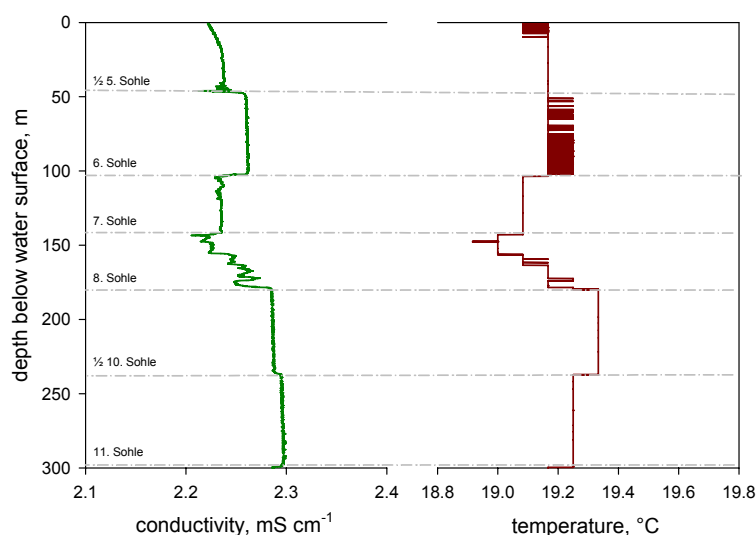


Fig. 120: Temperature and electrical conductivity measurements in the Reiche Zeche shaft (February 2nd 2001). Sohle: level.

12.13 Georgi Unterbau, Tyrol, Austria

12.13.1 Description of Location

40 km north-east of Innsbruck in Tyrol/Austria, close to Brixlegg, the historical silver-barite mines of the famous Kogel mining area can be found. Since medieval times silver-bearing

fahlore and in the 19th century as well as from 1947 to 1968 barite was mined there. One of these underground mines is the Georgi-Unterbau with the huge *Schwerspat* stope (pit) and its flooded underground works.

In 1887 the miners started to drive the Georgi-Unterbau adit, which opened up rich fahlore and barite resources. Starting in 1900, a 100 m deep two compartment blind shaft 320 m south-east of the mine entrance was sunk. This shaft opened up the 20, 40, 70 and 100 m main levels and the 10, 75 and 80 m sublevels (Pirkl 1961; Mutschlechner 1984; Krischker 1990; Hanneberg and Schuster 1994) and connected the 14-Nothelfer stope and the Barbara stope with each other. It was partially to fully flooded since the 1950s and the pre-flooding water make was about 100 L min⁻¹ (Hießleitner 1951; Schmidegg 1953) and pumped out again in 1984 and 1988 for ore prospecting (Krischker 1990). Finally, in 1990 the blind shaft has been flooded for the last time and therefore, stationary hydraulic conditions exist today. A typical characteristic for the Georgi-Unterbau mine is the existence of only one shaft that connects all the flooded levels.

All parts of the mine are within the Devonian Schwaz Dolostone (*Schwazer Dolomit*) of the Northern Tyrol Greywacke area (*Nordtiroler Grauwackenzone*). Typically, the Schwaz Dolostone is a very hard, light white to light grey dolostone, being highly brecciated and fissured in the area investigated. The dolostone hosts silver and mercury bearing fahlores (among them schwazite) as well as barite, the mineralisation being bound, but not restricted, to the breccia zones (Pirkl 1961; Wöbking 1982; Arlt and Diamond 1998). Grundmann and Martinek (1994), and Schnorrer (1994, 1996) described 20 ore minerals being characteristic for the Schwaz Dolostone and a total of over 132 minerals for all the rocks in the Schwaz-Brixlegg area.

A first physico-chemical investigation of the flooded blind shaft was conducted in December 2000. It became clear that the mine water is stratified, clearly showing two water bodies separated from each other at the 40 m level. Further investigations, including two tracer tests with 15 µm microspheres and Na-fluorescein, were conducted in August 2001 and February 2002, where 22 L min⁻¹ (32 m³ d⁻¹) and 31 L min⁻¹ (45 m³ d⁻¹) of mine water were flowing out, respectively (Unger 2002; Wackwitz 2002; Wolkersdorfer et al. 2002). About 300 m³ d⁻¹ are coming from other inflows, thereunder 150–200 m³ from a major fault zone, the Grünwies fault, summing up to 300–400 m³ d⁻¹ of total flow (Tab. 12).

Based on a piston-flow-model (non-dispersive laminar Poiseuille flow), the mean effective velocity of the mine water in the shaft was calculated to be $4 \cdot 10^{-3}$ m min⁻¹. However, the tracer test resulted in a mean effective velocity of $3 \cdot 10^{-2}$ m min⁻¹ above the 40-m-level. This difference is typical for all tracer tests conducted by the author so far, whereas the velocity itself does not fit into the 95%-confidence interval of 0.3–1.7 m min⁻¹ (Wolkersdorfer 2002b). Similar results for porous media were described by Ren et al. (1996). The low velocity in the Georgi-Unterbau blind shaft is due to the mine being a “single shaft mine”, whereas most of the other mines studied are “multiple shaft mines” or include both, shafts and galleries. None of the tracers injected below the 40-m-level could be retrieved.

Tab. 29: Exemplified composition of the Georgi-Unterbau mine water (samples BRX-0208-BS0.1 and BRX-0308-BS55 of August 2nd and 3rd 2001, filtered through 0.45 µm cellulose acetate filter). Concentrations in mg L⁻¹, depth in m, temp in °C, electrical conductivity in µS cm⁻¹; K, Cl, Ag, and Hg below the detection limits.

depth	temp	pH	cond	Ca	Mg	Sr	Na	Ba	Li
0.15	8.6	7.1	410	60.4	30.5	1.0	0.6	0.08	0.003
55	8.3	7.6	411	62.3	30.7	1.0	0.7	0.06	0.002

depth	HCO ₃ ⁻	SO ₄ ²⁻	NO ₃ ⁻	Sb	Cu	Zn	As
0.15	214	40.6	4.9	1.0	0.5	0.2	0.2
55	218	41.0	4.9	1.0	0.6	0.2	0.2

12.13.2 Investigations and Appearance of Stratification

Between December 2000 and February 2002 eight temperature and electrical conductivity profiles and two time series were measured in the Georgi-Unterbau blind shaft. Before the tracer test in August 2001, two mine water bodies indicated by different temperatures which were separated at the 40-m-level could be identified. Whilst the upper one had a mean temperature of 8.53 ± 0.01 °C and electrical conductivity of 324 ± 26 $\mu\text{S cm}^{-1}$, the lower one's mean temperature and electrical conductivity was 8.63 ± 0.03 °C and 332 ± 23 $\mu\text{S cm}^{-1}$, respectively. In contrast to the temperature, no stratification could be deduced from the electrical conductivity profile, which showed a relatively continuous increase from ca. 335 $\mu\text{S cm}^{-1}$ to ca. 345 $\mu\text{S cm}^{-1}$. A similar behaviour can be seen from the chemical composition of the mine water (Tab. 29). After the injection of 200 kg of sodium chloride tracer as a saturated brine in August 2001, the before described stratification significantly changed, albeit the brine was not injected into the blind shaft itself but into a small drop-hole 4 m west from it. Instead of two mine water bodies, four of them occurred, each one separated by the 20-, 40-, and 70-m-levels. Yet, six month later the previous stratification was restored with a lower temperature (ca. 8.1 °C) and a higher electrical conductivity (ca. 480 $\mu\text{S cm}^{-1}$).

Interestingly, the stratification pattern in the Georgi-Unterbau blind shaft was stable for at least 19 months, although the temperature difference of 0.1 K between the two mine water bodies is extremely small. On the other hand, the injection of the sodium chloride brine shows that the stratification can easily be destroyed but almost established again during the six months thereafter. Unger (2002) numerically modelled a part of the flooded mine and showed that the water seeping through the Schwerspat pit into the 14-Nothelfer pit and from there to the 40-m-level is responsible for the observed stratification; at least the temperature drop in the water column above the 40-m-level. Based on the physico-chemical measurements and the chemical analyses of the water, a good mixing within the upper and lower layers can be seen, whereas the tracer results of the test clearly show that there is restricted mixing between the two layers. Furthermore, the temperature and electrical conductivity patterns confirm that there is a turbulent flow regime in the shaft (Fig. 121).

12.13.3 Results and Interpretation

To investigate the situation, two multiple tracer tests were conducted in the mine. Tracers were injected into different depths of the blind shaft and an underground mining lake connected to the shaft. From the tracers injected, only the tracer from the underground mining lake (green microspheres) and sodium fluorescein from the blind shaft could be detected. A subsequent measurement of the sodium fluorescein concentration in the shaft showed a nearly-normal distribution of the tracer cloud around the tracer's injection depth at 55 m, with tracer concentrations between 200 and 600 $\mu\text{g L}^{-1}$ between 40 and 80 m depth. Thus, an indication for diffusive mass flow in that part of the shaft could be proven. An evaluation of the breakthrough curve of the green microspheres results in a mean flow duration of 52 hours for a distance of 48 m. Thus, a mean effective velocity of 0.02 m min^{-1} can be calculated for the flow between the "Big Lake" and the blind shaft's outflow.

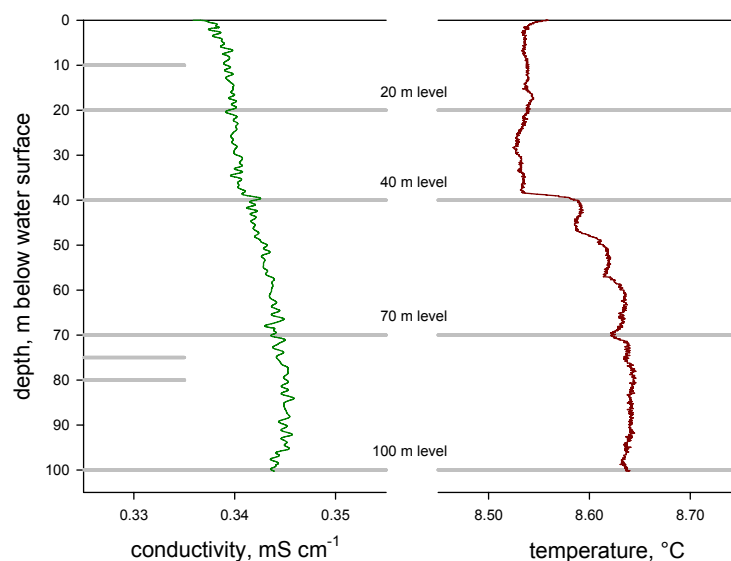


Fig. 121: Temperature and electrical conductivity measurement Georgi Unterbau Shaft. The temperature stratification at the 40 m level, even when significantly lower than 1 K was stable for more than 3 years. Measurement in August 2001.

In February 2002 the negative experiences of the first tracer test could be taken in consideration and the results therefore were better. The sodium fluorescein showed a well established breakthrough curve within the expected time frame being similar to that one of August 2001. During the period of investigation (February 18th to 24th 2002) the water temperature of the shaft stayed stable at about 9.2 °C and the conductivity around 430 $\mu\text{S cm}^{-1}$. Also stable was the outflow of the shaft with 31.5 L min⁻¹ (43—48 m³ d⁻¹) and the pumping capacity of the mini piston pump (sampling pump) at 0.726 L min⁻¹ (1 m³ d⁻¹). From the 50 g of Na-fluorescein used only 3.2 g could be recovered, which equals a recovery rate of 6.4%. As can be seen from the breakthrough curve (Fig. 81), the tracer concentration increases after the first peak at February 19th. This might be caused by a density flow of the tracer. The tracer was injected at a relatively high concentration and due to its higher density it first moved downward and was later, after dilution, transported out of the shaft again. It can therefore be concluded that after a longer time period more tracer would have been discharged and the recovery rate been higher.

The peak concentration of the tracer (excluding the background) was about 300 $\mu\text{g L}^{-1}$ at a background concentration of 70 $\mu\text{g L}^{-1}$ which resulted from the 2001 tracer test. At the end of the tracer test the tracer concentration is still as high as 13 $\mu\text{g L}^{-1}$. The calibration of the fluorometer gave a lower detection limit of 7 $\mu\text{g L}^{-1}$ and the maximum concentration design dependent was 7,000 $\mu\text{g L}^{-1}$ (the concentrations in Wolkersdorfer et al. (2002) are wrong as the fluorometer was back calibrated afterwards).

From those results, a mean effective velocity of 0.03 m min⁻¹ could be calculated, which fits well into the effective velocity calculated at the previous tracer test (0.02 m min⁻¹). This is 7—8 times the velocity expected from a piston flow model calculated ahead of the tracer test (0.004 m min⁻¹) and confirms a fact often observed in mine water tracer tests. So far, the reasons are unclear but the wall roughness of the shafts and adits might play an important role therein. When comparing the results of the two tracer tests it becomes clear that the first tracer tests has a larger dispersion than the second one. This facts results from the longer flow path of the first tracer test (48 m) compared to the second one (10 m).

Unfortunately, the reasons for the stratification in the Georgi Unterbau blind shaft could not be found by the two tracer tests, because none of the tracer in the deeper parts of the flooded

shaft could be recovered. Possibly the deeper parts of the shaft are to a lesser degree hydraulically connected and therefore diffusive flow might dominate. At least from the tracer concentration between the 40- and 70-m-levels a diffusive transport can be assumed, otherwise the tracer distribution around the injection location should have been distributed in another way. On the other hand, as the Rabenstein tracer test shows, extremely slow water movements cannot be excluded.

In conclusion this tracer experiment – as well as the Rabenstein experiment – showed that no fast convective mixing within the deeper parts of the flooded mine exist as it could be observed at the Niederschlema and Straßberg tracer tests. Therefore, it can be concluded that the missing hydraulic connection between the workings in the deeper parts prevents a convective mixing of the mine water. This fact so far aroused little attention, though Uerpmann (1980) already noticed that fact. It might therefore be concluded that the flooded mine between the 40- and 70-m-levels might act as a hydraulic isolation preventing the mass transport between the upper and lower parts of the shaft.

12.14 Salt mines

12.14.1 Description of Locations and Investigations

Germany host numerous abandoned salt mines that were flooded in a controlled or uncontrolled manner (Schwandt and Seifert 1999). They have intensively been studied, because Germany's radioactive waste disposal policy, for a long time, focused on salt mines and Germany has a long history in underground salt mining. Compared to metal or coal mines the situation in flooded salt mines differs significantly, because in addition to thermal effects causing the stratification, chemical effects have to be considered. Furthermore, the density differences in salt mines are usually larger than in metal mines because the mineralization in the first ones normally exceeds the latter ones. This fact would make salt mines an excellent example to study mine water stratification, but with the aim in mind, to reduce the environmental impacts of metal and coal mining, such case studies are transferable with restrictions only. Of course, there are non-salt mines with chloride contents as high as 130 g L^{-1} , and conductivities of up to 190 mS cm^{-1} (Coldewey et al. 1999), but such mines are seldom. Though this publication focuses on metal and coal mines some salt mines will be included here because the first monitored controlled flooding of a mine was conducted in the Hope salt mine north of Hannover/Germany. Furthermore, several of the experiences described in papers about that mine flooding where also used in mine water investigations, monitoring programmes, and tracer tests conducted by the author.

Rauche et al. (2002) describe that the evaluation of the stratification patterns in a flooded salt mine can be used to estimate the dissolution processes occurring in the flooded mine. In combination with chemical analyses of the mine water and their evaluation in Jänecke-plots (triangular $\text{Mg-SO}_4\text{-K}_2$ diagram) it is possible to give first information about potentially dangerous reactions occurring in the shaft. The example they give anonymously shows a density stratification indicated by a slight jump in the electrical conductivity but the temperature log clearly shows that no convective flow seems to dominate the flooded shaft. Because no details are given, their work will not be further discussed here.

All the other examples given here are based on the investigations of Uerpmann (1980), Sander and Herbert (1985), GSF – Gesellschaft für Strahlen- und Umweltforschung (1986), Herbert (1989), Herbert and Sander (1989), and Czolbe et al. (1992). From the examples in those publications, four salt mines, three in Lower Saxony (Hope and Ronnenberg near Hannover, Beienrode near Wolfsburg) and another one in Saxony-Anhalt (Peißen near Bernburg) are be described.

Hope is located 30 km north of Hannover/Germany and was flooded with saturated brine from salt cavern dissolution. Flooding started on March 12th 1984 after the mines abandonment in January 1984. Before flooding 16 *in-situ* measuring installations were installed to monitor the whole flooding process. Furthermore, an underground dam was constructed to study safety dams for radioactive waste disposal. The whole programme was initiated in a relatively short time period, only six month ahead of the planned flooding. Besides the geochemical, geomechanical, geophysical and the dam monitoring installations, the aim was to investigate “mass and heat transport in flooded, not backfilled galleries, mine workings, and shafts” (GSF – Gesellschaft für Strahlen- und Umweltforschung 1985; referred sentence translated from German). Mining in the Hope salt dome began in 1907 with the 521.2 m deep shaft *Adolphsglück* and in 1909 the 628.5 m deep shaft *Hope* north of it was sunk. The salt dome has a depth of about 4,000 m, a maximal diameter of 6 km and consists of different salt types of Permian (Zechstein) age: rock salt, sylvite, kieserite, and anhydrite. A total volume of about $1.6 \cdot 10^6 \text{ m}^3$ was flooded with saturated NaCl brine (density: $1,200 \text{ g m}^{-3}$) from the Empelede cavern field. During the flooding process and as long as the unflooded parts of the mine were accessible, the brine was regularly analysed by taking several samples underground. Also the flow path of the brine was controlled and monitored (Herbert 1989). In October 1988 the last measurements were conducted by the F&E research project participants.

The 477 m deep Peißen shaft near Bernburg has a diameter of about 5 m, was sunk in 1900 and controlled flooded between 1972 and 1974. It is one of the 11 shafts of the salt mines within the Bernburg main anticline and is located within a small syncline, the Peißen syncline. Mining in the Bernburg area started on January 31st 1852 with the Manteuffel shaft and it was there, where the world's first potassium salt mine was found (Riemann 1913; Hiltcher 1999). All the mines are within the Zechstein salt formation. Czolbe et al. (1992) investigated the flooded shaft through temperature and density logs as well as chemical analyses and a tracer test with a radioactive γ -tracer. Furthermore, they modelled the hydrodynamic situation of the shaft with the code KASOMO.

The salt mines investigated by Uerpmann (1980) are Asse I, Asse III (near Wolfenbüttel), shaft Beienrode near Braunschweig, shaft Desdemona near Göttingen and shaft Deutschland of the flooded Ronnenberg salt mine (Peter Uerpmann, pers. comm.). His work is entitled “Hydrogeological Questions at Radioactive Waste Disposal” and therefore he focuses on the safe disposal of such waste types. His work does not contain details about the single mines.

12.14.2 Appearance and Interpretation of Stratification

Herbert and Sander (1982), who were responsible for the Hope flooding experiment, had already noticed earlier, by measuring in several flooded mines, that the brine in those mines is commonly stratified into three layers: an upper layer with slightly saline or fresh water, a middle NaCl saturated layer with a density of about 1.20 g cm^{-3} , and a lower MgCl_2 -rich layer with a density of $1.29\text{—}1.30 \text{ g cm}^{-3}$. Between those three layers they found intermediate layers. They concluded that the formation of the main and intermediate layers is controlled by the chemical composition of the rock being in contact with the shaft and the geothermal gradient. Furthermore, they observed that in homogenous layers convective flow and in layers with density- or temperature gradients no flow could be observed (Herbert and Sander 1982; Sander and Herbert 1985). Over and above, more than one convective water body and also no flow and convective flow cells can adjoin each other. Which of those situations occurs is, according to their opinion, a result of the geological setting. The mine geometry is only seen as an additional reason for the stratification. In another paper (Sander and Herbert 1985) they concluded that the NaCl/ MgCl_2 boundary can be successfully used as a barrier against the transport as they noticed that crystalline halite (rock salt) was forming at the boundary. Yet,

and this fact is missing in their investigation, such a barrier can only be stable if no forced convection of flow occurs in the shaft.

During the active flooding of the Hope Salt mine numerous physico-chemical measurements were conducted (GSF – Gesellschaft für Strahlen- und Umweltforschung 1986; Herbert 1989). 14 of the continuous measurements in the Hope, Adolfsgrück, and the well Abbau 4 were published. Apart from their differences they show many similarities; for example the stratification in the shaft. Unfortunately, the results of the Hope and Adolfsgrück shaft cannot directly be compared as the Adolfsgrück shaft was backfilled in 1985 and the measurements had to be conducted in a pipe within the backfill. From the differences of the measurements before and after the backfill it becomes clear that the post-backfill situation is hydraulically significant different from the pre-backfill conditions. While the water in the Adolfsgrück shaft shows only a less developed stratification before the backfill the situation changes after its backfilling, where at least two separate mine water bodies can be identified (Fig. 122). Again, as in the case of the wells in the Ruhr area, wells cannot be used to interpret the stratification in a mine as the flow situation in a small diameter well differs significantly from the flow in a large volumed shaft. While the Adolfsgrück “shaft” stratified after the backfill, this was not the case for the Hope shaft which was not backfilled (Fig. 123). The interpretation of GSF – Gesellschaft für Strahlen- und Umweltforschung (1986) that the backfill prevents an overall turbulent mixing is definitely true, but cannot be verified from a measurement in a small diameter well within the backfill of the Adolfsgrück shaft. Though a minor stratification could be observed during the early stage of the flooding process this stratification broke down towards the end of the flooding. Just a lesser mineralised water lens with a thickness of 30–40 m was laying on top of the 180–190 m thick shaft mine water body. Out of the findings of the “Hope – geochemistry sub project” the following findings are relevant for the study of flooded shafts (GSF – Gesellschaft für Strahlen- und Umweltforschung 1986; Herbert 1989):

- a shaft can consist of two or more turbulent mixed water bodies that border on each other, but also no-flow layers can border on turbulent mixed ones
- water bodies with turbulent flow are characterised by a uniform temperature, electrical conductivity, density and chemical composition
- no-flow layers are reflected by concentration and temperature gradients
- the mass transport in no-flow layers is dominated by diffusion that means slowly, and in turbulent mixed layers mainly by fast flow
- the boundaries between the single layers act similar to mechanical barriers resulting in a turnover of the flow direction
- the mass transport across the boundaries is diffusion dominated even when both zones are mixed by turbulent flow
- backfilled mine workings and open ones show significantly different flow patterns, as in open mine workings the flow is dominated by convective flow and in backfilled ones by no-flow.

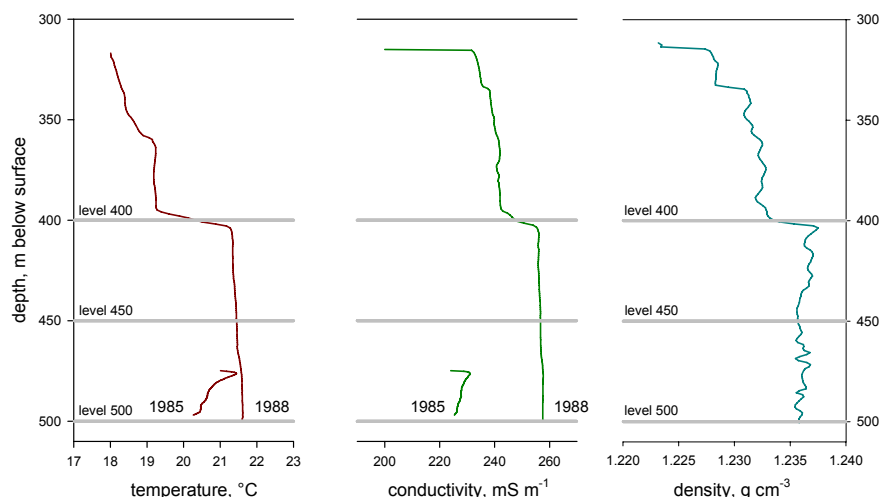


Fig. 122: Comparison of the post- and pre-backfilling situation in the Adolfsgrück shaft in 1985 and 1988. The 1988 situation is typical for measurements in small diameter boreholes or wells (modified after GSF – Gesellschaft für Strahlen- und Umweltforschung 1986 and Herbert 1989).

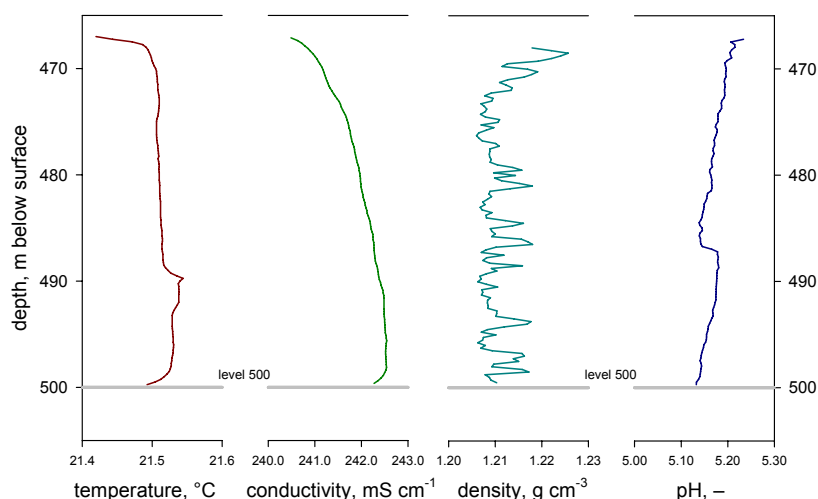


Fig. 123: Results of the multi-parameter measurements in the Hope shaft on September 20th 1985. Only a minor stratification can be observed which – except that the water level rose – did not change significantly until the end of the mine flooding (modified after Herbert 1989).

In the Peißen shaft a clear density and temperature stratification could be observed which stayed stable for a period of at least 4 months (Fig. 124). Densities of the water and brine ranged between 1.1 g cm^{-3} and 1.3 g cm^{-3} and the temperature was in a range of $13.2\text{--}30.5 \text{ }^{\circ}\text{C}$. A total of 4 mine water bodies with no or small intermediate layers could be observed. They are characterised by their chemical composition and are similar to the observations of Herbert and Sander (1982): a slightly mineralised “fresh” water layer, a NaCl-layer, a NaCl-KCl-layer and a lowermost MgCl_2 -layer. Besides the physico-chemical investigations the investigation included a tracer test. At 10 locations in the shaft a radioactive γ -tracer was injected and its distribution and velocity measured with velocities between 0.003 and 1.17 m min^{-1} which fits well into the overall range of mine water tracer tests (though their velocities are *in-situ* velocities of the water in the shaft and not the water flowing in the mine).

Furthermore, they could identify several, at least 5 and at a maximum 8 convective cells within the shaft, which can directly be linked to the different mine water bodies. From their measurements they also concluded that warm water is up welling near the shafts walls and down welling in the middle of the shaft as has been postulated for the Niederschlema/Alberoda mine as well (section 12.9). By using the hydrodynamic code KASOMO, originally written to model flow in caverns for radioactive waste disposal, they were able to model and understand the observations in the shaft. Their measurements fall within the single shaft mines as there is no flow between the Peißen and other shafts. It becomes clear that intermediate layers form relatively stable barriers which cannot be crossed by the different types of water in the shaft. This fact has been proven by Czolbe et al. (1992) because they were not able to detect the injected tracers outside of each of the single cells.

Uerpmann (1980) investigated temperature, conductivity, velocity, and chemical compositions of mine waters in five flooded salt mine shafts with depths between ca. 200 and ca. 800 m (shafts Asse III, Desdemona, Deutschland, Ronnenberg, Beienrode; Peter Uerpmann, pers. comm.). As in the case studies listed before he was able to identify different water bodies separated from each other by smaller, intermediate layers of different temperature (Fig. 125). This resulted in staircase profiles, which, according to section 7.1.1, might be seen because of double-diffusive convection. Due to the density differences in flooded salt mines this effect cannot be excluded as an explanation for the staircase temperature distribution. Uerpmann (1980) also refers to the saline Kivu lake between Rwanda and the Democratic Republic Congo (Zaire), which was investigated by Tietze (1978) and where also staircase like steps could be detected. An interesting observation was made during the flooding of the salt mines: already several weeks after the flooding the water temperatures reached nearly the final temperatures at the end of the flooding process. This observation is an indication for fast convective mixing processes during the first stages of the flooding (see section 7.1) and smaller exchanges at the later stages. In a laboratory experiment Uerpmann (1980) measured mean velocities of $0.06\text{--}0.3\text{ m min}^{-1}$ and he was able to observe that during his experiment the water stratified and layers with different physico-chemical properties were separated by one or more distinct intermediate layers. Tracer injected in the two different layers showed relatively good mixing after several days while in the intermediate layers the tracer distribution and the tracer velocities were significantly smaller.

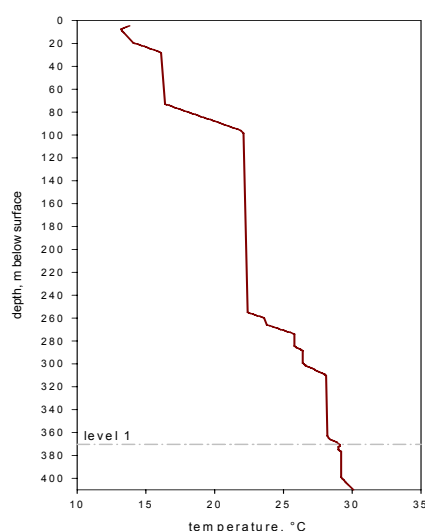


Fig. 124: Temperature profile in the Peißen shaft. The lowermost horizontal line is the mines' 1st level. Each of the “stable” temperature zones represents a convective cell as has been proven by tracer investigations (modified after Czolbe et al. 1992).

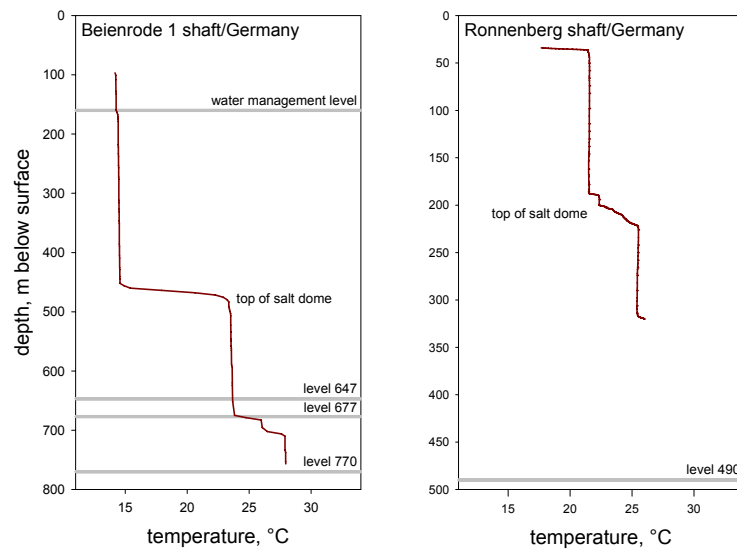


Fig. 125: Temperature—depth profiles in two flooded north-west German salt mines. Left: Beienrode № 1 shaft, right: Ronnenberg shaft (modified after Uerpmann 1980).

13 Literature

- ABELIN, H. & BIRGERSSON, L. (1985): Migration experiments in the Stripa mine, design and instrumentation. – In: COME, B., JOHNSTON, P. & MÜLLER, A.: Joint CEC/NEA workshop on design and instrumentation of in situ experiments in underground laboratories for radioactive waste disposal **CONF 8405165**. – p. 180-190, 9 fig.; Rotterdam u.a. (Balkema).
- ACKERS, P., WHITE, W. R., PERKINS, J. A. & HARRISON, A. J. M. (1978): Weirs and flumes for flow measurement. – 327 p., 39 plates; Chichester (Wiley & Sons).
- ACKMAN, T. E. (2003): An Introduction to the Use of Airborne Technologies for Watershed Characterization in Mined Areas. – *Mine Water Env.*, **22** (2): 62-68, 5 fig.; Berlin.
- ACKMAN, T. E. & JONES, J. R. (1988): Stream sealing to reduce surface water infiltration into underground mines. – Bureau of Mines Information Circular, **IC-9183**: 232-239, 6 fig., 3 tab.; Washington.
- ADAMS, N., CARROLL, D., MADALINSKI, K., ROCK, S., WILSON, T., PIVETZ, B., ANDERSON, T., CHAPPELL, J., HULING, S., PALMIOTTI, J. & SAYRE, P. (2000): Introduction to Phytoremediation **EPA-600/R-99/107**. – 72 p., 13 fig., 17 tab.; Cincinnati (U.S. Environmental Protection Agency).
- ADAMS, R. & YOUNGER, P. L. (2001): A Strategy for Modeling Ground Water Rebound in Abandoned Deep Mine Systems. – *Ground Water*, **39** (2): 249-261, 10 fig.; Worthington.
- ADELMAN, F. L., BACIGALUPI, C. M. & MOMYER, F. E. (1960): Final report on the Pinot experiment [Colorado]. – Lawrence Radiation Lab. Rept., **UCRL-6274**: 21; Livermore.
- ADLEM, C. J. L., MAREE, J. P. & MCCRINDLE, R. I. (1994): Water Softening in the Barium Sulphide Process for Sulphate Removal. – Proceedings, 5th International Mine Water Congress, Nottingham, U.K., **2**: 601-611, 9 fig., 2 tab.; Nottingham.
- AGRICOLA, G. (1556): De re metallica libri XII. – 273 Holzschnitte; Basel.
- AHONEN, L. & TUOVINEN, O. H. (1995): Bacterial Leaching of Complex Sulfide Ore Samples in Bench-Scale Column Reactors. – *Hydrometallurgy*, **37** (1): 1-21, 9 fig., 2 tab.; Amsterdam.
- AL, T. A. & BLOWES, D. W. (1996): Storm-water hydrograph separation of run off from a mine-tailings impoundment formed by thickened tailings discharge at Kidd Creek, Timmins, Ontario. – *J. Hydrol.*, **180** (1-4): 55-78, 15 fig., 2 tab.; Amsterdam.
- AL-FOUZAN, F., HARBERT, W., DILMORE, R., HAMMACK, R., SAMS, J., VELOSKI, G. & ACKMAN, T. E. (2004): Methods for Removing Signal Noise from Helicopter Electromagnetic Survey Data. – *Mine Water Env.*, **23** (1): 28-33, 7 fig.; Berlin.
- ALDOUS, P. J. & SMART, P. L. (1987): Tracing Ground-Water Movement in Abandoned Coal Mined Aquifers Using Fluorescent Dyes. – *Ground Water*, **26** (2): 172-178, 8 fig., 2 tab.; Worthington.
- ALEY, T. & FLETCHER, M. W. (1976): The water tracer's cookbook. – *Missouri Speleology*, **16**: 1-32, 2 fig., 1 tab.; Columbia.
- ALJOE, W. W. & HAWKINS, J. W. (1993): Neutralization of Acidic Discharges from Abandoned Underground Coal Mines by Alkaline Injection. – Report of Investigations – US Bureau of Mines, **9468**: 1-37, 40 fig., 3 tab.; Washington.
- ALJOE, W. W. & HAWKINS, J. W. (1994): Application of Aquifer Testing in Surface and Underground Coal Mines. – Proceedings, 5th International Mine Water Congress, Nottingham, U.K., **1**: 3-21, 8 fig., 1 tab.; Nottingham.

- ALPERS, C. N. & NORDSTROM, D. K. (1999): Geochemical Modelling of Water-Rock Interactions in Mining Environments. – In: PLUMLEE, G. S. & LOGSDON, M. J.: The Environmental Geochemistry of Mineral Deposits **6A** (14). – p. 289-323, 8 fig., 5 tab.; Littleton (Society of Economic Geologists).
- ALPERS, C. N., NORDSTROM, D. K. & BURCHARD, J. M. (1992): Compilation and interpretation of water-quality and discharge data for acidic mine waters at Iron Mountain, Shasta County, California, 1940-91. – Water resources investigations report, **WRI 91-4160**: 173, 9 fig., 5 tab., 1 app.; Reston.
- ALPERS, C. N., BLOWES, D. W., NORDSTROM, D. K. & JAMBOR, J. L. (1994): Secondary minerals and acid mine-water chemistry. – In: JAMBOR, J. L. & BLOWES, D. W.: Short Course Handbook on Environmental geochemistry of Sulfide Mine-Wastes **22**. – p. 247-270, 4 fig., 12 tab.; Waterloo, Ontario (Mineralogical Association of Canada).
- AMERICAN PUBLIC HEALTH ASSOCIATION, AMERICAN WATER WORKS ASSOCIATION & WATER ENVIRONMENT FEDERATION (1998): Standard methods for the examination of water and wastewater. – In: CLESCERI, L. S., EATON, A. D., FRANSON, M. A. H. & GREENBERG, A. E. – 20th edn., Washington (American Public Health Association).
- AMEZAGA, J. M. & YOUNGER, P. L. (2002): Environmental Regulation of Mine Waters in the European Union – D 3 Institutional research – 2 Great Britain. – p. 2.1-2.20, 2 fig.; Newcastle upon Tyne (University of Newcastle).
- AMEZAGA, J. M., KROLL, A., JARVIS, A. P., REES, B., PUURA, E., MISERÉ, W. M. H., WINGS, R. W. M. G., BAJTOŠ, P., WEBER, L., SÁRVÁRYNÉ-SZENTKATOLNAY, B., GARZONIO, C. A., WOLKERSDORFER, CH., YOUNGER, P. L. & BOWELL, R. (2004): Contemporary Reviews of Mine Water Studies in Europe, Part 1. – Mine Water Env., **23** (4): 163-183, 16 fig., 3 tab.; Berlin.
- AMMANN, G. (1990): Silber, Erz und weißes Gold – Bergbau in Tirol – Tiroler Landesausstellung 1990. – 478 p.; Innsbruck (Tiroler Landesmuseum Ferdinandeum).
- AMORIM, L. Q., GRANDCHAMP, C. A. P. & BERTACHINI, A. C. (2001): Dewatering of Iron Ore Mines and Capability to Predict Environmental Impacts – The Experience of MBR. – p. 1-14 [CD-ROM], 9 fig.; Belo Horizonte (Proceedings, IMWA Symposium Belo Horizonte).
- ANCKORN, J. F. & DEL OLMO, A. G. (1999): The implications of current European directives and the future of mining in Europe. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 791-803, 3 fig., 2 tab.; Sevilla (International Mine Water Association).
- ANDERSON, B. C. (1987): Use of temperature logs to determine intra-well-bore flows, Yucca Mountain, Nevada. – 59 p.; Golden (Master's Thesis Colorado School of Mines).
- ANGERER, J. & LECHLEITNER, H. (1893): Bergbau und Hüttenwesen, Gewerbe, Industrie und Handel in Tirol und Vorarlberg. – In: ERZHERZOG RUDOLF: Die österreichisch-ungarische Monarchie in Wort und Bild – Tirol und Vorarlberg. – p. 585-600, 4 fig.; Wien (K.K. Hof- u. Staatsdr.).
- ANNANDALE, J. G. (1998): Irrigating with gypsiferous mine water. – p. Afr. J. Sci., **94** (7): 359-360; Johannesburg.
- ANNANDALE, J. G., JOVANOVIĆ, N. Z., BENADE, N. & TANNER, P. D. (1999): Modelling the long-term effect of irrigation with gypsiferous water on soil and water resources. – Agric. Ecosyst. Environ., **76** (2-3): 109-119, 8 fig., 1 tab.; Amsterdam.
- ANNANDALE, J. G., JOVANOVIĆ, N. Z., PRETORIUS, J. J. B., LORENTZ, S. A., RETHMAN, N. F. G. & TANNER, P. D. (2001): Gypsiferous mine water use in irrigation on rehabilitated open-cast mine land: Crop Production, soil water and salt balance. – Ecol. Eng., **17** (2-3): 153-164, 3 fig., 4 tab.; Amsterdam.
- ANNANDALE, J. G., JOVANOVIĆ, N. Z., TANNER, P. D., BENADE, N. & DU PLESSIS, H. M. (2002): The Sustainability of Irrigation with Gypsiferous Mine Water and Implications for the Mining Industry in South Africa. – Mine Water Env., **21** (2): 81-90, 5 fig., 2 tab.; Berlin.

- ANONYMOUS (1901): La fabrique d'absinthe Pernod en feu. – L'illustration – journal universel, **17.8.1901**; Paris.
- ANONYMOUS (1934): The Henry Process for the Clarification of Polluted Water. – Engineering, **138** (8, 9): 213-215, 293-295, 17 fig.; London.
- ANONYMOUS (1935): Wallsend – At their Rising Sun pit the Wallsend & Hebburn Coal Co. Ltd. have erected what is probably the most complete coal preparation plant in the world. – Colliery Engineering; London.
- ANONYMOUS (1936): Mine-Sealing Program to Reduce Acid Pollution in Streams. – Engineering news record, **2**: 42-43, 3 fig.; New York.
- ANONYMOUS (1994): Urteil des Oberverwaltungsgerichts Lüneburg vom 15. Juni 1994 (7 L 5295/92) – Schadensvorsorge bei der Stilllegung eines Bergwerksbetriebes. – Z. f. Bergrecht, **135**: 277-286; Bonn.
- ANONYMOUS (1995): Urteil des Bundesverwaltungsgerichts vom 9. November 1995 (4 C 25.94) – Unternehmerverantwortung bei der teilweisen Betriebseinstellung eines Bergwerks für die Abwehr gemeinschädlicher Einwirkungen – Beseitigung von Sauerwasser im Erzbergwerk Rammelsberg/Harz. – Z. f. Bergrecht, **136**: 290-301; Bonn.
- ANONYMOUS (1997): Environmental pumping tasks at Wheal Jane. – Mining Magazine, **177** (5): 278-279, 2 fig.; London.
- ANONYMOUS (1999a): Scottish Environment Protection Agency to take Action on Mine Water Pollution. – Coal International, **247** (4): 139, 1 fig.; Redhill, Surrey.
- ANONYMOUS (1999b): Mining history matters. – Mining Journal, **332** (8523): 198-199; London.
- ANONYMOUS (2000): On January 11, Jiangsu Province's Dahuangshan coal mine at Xuzhou was hit by an inrush. – Engineering & Mining Journal, **2001** (3): w10; Prairieville.
- ANONYMOUS (2002): NW China Coal Mine Flooded, 15 Trapped. – People's Daily Online, **4.7.2002**; Beijing.
- APPELO, C. A. J. & POSTMA, D. (1994): Geochemistry, groundwater and pollution. – 2nd edn., 536 p.; Rotterdam (Balkema).
- AQUILINA, L., ROSE, P., VAUTE, L., BRACH, M., GENTIER, S., JEANNOT, R., JACQUOT, E., AUDIGANE, P., TRAN-VIET, T., JUNG, R., BAUMGÄRTNER, J., BARIA, R. & GÉRARD, A. (1998): A Tracer Test at the Soultz-Sous-Forêts Hot Dry Rock Geothermal Site. – Proceedings, 23rd Workshop on Geothermal Reservoir Engineering, **SGP-TR-158**: 1-8, 3 fig.; Stanford.
- ARANCIBIA, E. & FLORES, G. (2004): Transición de Chuqui a mina subterránea. – Minería Chilena, **280** (10): 25, 1 fig.
- ARANYAKANON, P. & YONGSAKUL, K. (1981): Offshore tin deposits of Thailand and related mining problems. – In: HALBOUTY, M. T.: Energy Resources of the Pacific region. – p. 167-178, 9 fig., 1 tab.; Tulsa (American Association of Petroleum Geologists).
- ARBEITSKREIS "HUMAN- UND ÖKOTOXIKOLOGISCHE BEWERTUNG VON MARKIERUNGSMITTELN IN GEWÄSSERN" (1997): Human- und ökotoxikologische Bewertung von Markierungsmitteln in Gewässern. – Grundwasser, **2** (2): 61-64, 2 tab.; Heidelberg.
- ARDEJANI, F. D., SINGH, R. N., BAAFI, E. & PORTER, I. (2003): A Finite Element Model to: 1. Predict Groundwater Inflow to Surface Mining Excavations. – Mine Water Env., **22** (1): 31-38, 7 fig., 2 tab.; Berlin.
- ARIZA, L. M. (1998): River of vitriol. – Sci. Amer., **279** (3): 26-27, 2 fig.; New York.
- ARLT, T. & DIAMOND, L. W. (1998): Composition of tetrahedrite-tennantite and "schwazite" in the Schwaz silver mines, North Tyrol, Austria. – Mineral. Mag., **62** (6): 801-820, 8 fig., 5 tab.; London.

- ARNOLD, I. & BAUMGARTEN, U. (1996): Wege des Wassers – Kohleförderung und Wasserhebung im Lausitzer Braunkohlerevier. – 11 p., 31 fig.; Senftenberg (Lausitzer Braunkohle Aktiengesellschaft).
- ARNOLD, I. & KUHLMANN, K. (1994): Wasserwirtschaft und Bergbau in der Lausitz. – Braunkohle (7): 10-21, 10 fig.; Düsseldorf.
- AURADA, K. D. (1970): Berechnung des Wiederanstiegsprozesses im Grubengebäude des Kupferschieferbergbaus in der Mansfelder Mulde mit Hilfe der Monte-Carlo-Methode. – Wasserwirtschaft Wassertechnik, **20** (8): 264-270, 6 fig., 3 tab.; Berlin.
- AXLER, R., YOKOM, S., TIKKANEN, C., McDONALD, M., RUNKE, H., WILCOX, D. & CADY, B. (1998): Restoration of a Mine Pit Lake from Aquacultural Nutrient Enrichment. – Restoration Ecology, **6** (1): 1-19, 11 fig., 5 tab.; Cambridge.
- BAACKE, D. (2000): Geochemisches Verhalten umweltrelevanter Elemente in stillgelegten Polysulfiderzgruben am Beispiel der Grube "Himmelfahrt" in Freiberg/Sachsen. – Freiburger Dissertationen On-line, **35**: 139, 72 fig., 62 tab., 7 app.; Freiberg.
- BAACKE, D. & DEGNER, T. (2000): Hydrogeochemie, Wärmehaushalt und Strömung des Flutungswassers einer untertägigen Erzgrube. – In: WIPPERMANN, T.: Bergbau und Umwelt – Langfristige geochemische Einflüsse. – p. 163-174, 6 fig., 2 tab.; Berlin (Springer).
- BAACKE, D., KLUGE, A., MARTIN, M. & BEUGE, P. (1996): Geochemische Prozesse beim Stofftransport in Stollnsystemen gefluteter Grubenbaue. – GeoCongress, **2**: 1-6, 5 fig., 1 tab.; Köln.
- BAAS BECKING, L. G. M., KAPLAN, I. R. & MOORE, D. (1960): Limits of the Natural Environment in Terms of pH and Oxidation-Reduction Potentials. – J. of Geol., **68** (3): 243-284, 34 fig., 3 tab.; Chicago.
- BAGDY, I. & SULYOKM, P. (1990): Case Example on Acid-Resistent Bulkhead in Hungary. – Int. J. Mine Water, **9** (1-4): 97-109, 7 fig.; Castro Verde.
- BAKER, A. J. M., REEVES, R. D. & MCGRATH, S. P. (1991): *In Situ* Decontamination of Heavy Metal Polluted Soils Using Crops of Metal-Accumulating Plants – A Feasibility Study. – In: HINCHEE, R. E. & OLFENBUTTEL, R. F.: *In Situ* bioreclamation – Applications and investigations for hydrocarbon and contaminated site remediation. – p. 600-605, 1 fig.; Stoneham (Butterworth-Heinemann).
- BALKAU, F. (1999): Abandoned mine sites. – In: NATURAL RESOURCES MANAGEMENT UNIT DIVISION „PROTECTION OF THE ENVIRONMENT AND NATURAL RESOURCES“: Report on the International Round Table on Mining and the Environment ("Berlin II Declaration"). – p. 40-43; Köln (Carl Duisberg Gesellschaft).
- BALL, B. R. & BRIX, K. V. (2000): Passive Treatment of Metalloids Associated with Acid Rock Drainage. – In: SOCIETY FOR MINING METALLURGY AND EXPLORATION: New frontiers in mining: meeting 21st century challenges. – 20 p., 4 fig., 1 tab.; Littleton (Society for Mining, Metallurgy, and Exploration).
- BANKS, D. (1994): The Abandonment of the Killingdal Sulphide Mine, Norway: a Saga of Acid Mine Drainage and Radioactive Waste Disposal. – Mine Water Env., **13** (3-4): 35-48, 4 fig., 1 tab.; Wollongong.
- BANKS, D. (2001): A Variable-Volume, Head-Dependent Mine Water Filling Model. – Ground Water, **39** (3): 362-365, 2 fig., 1 tab.; Worthington.
- BANKS, D. (2004): Geochemical Processes controlling Minewater Pollution: Conference Papers **35**. – p. 17-44, 14 fig., 4 tab.; Wien (Umweltbundesamt).
- BANKS, D., YOUNGER, P. L., ARNESEN, R. T., IVERSEN, E. R. & BANKS, S. B. (1997): Mine-water chemistry: the good, the bad and the ugly. – Environ. Geol., **32** (3): 157-174, 11 fig., 5 tab.; New York, Berlin.

- BANKS, D., SKARPHAGEN, H., WILTSHIRE, R. & JESSOP, C. (2003): Mine water as a resource – space heating and cooling via use of heat pumps. – Land Contam. Reclam., **11**: 191-198, 5 fig.; Richmond.
- BARNES, I. & CLARKE, F. E. (1964): Geochemistry of ground water in mine drainage problems. – US Geological Survey Professional Paper, **473, A**: 1-6, 1 fig., 1 tab.; Washington.
- BARNES, I., STUART, W. T. & FISHER, D. W. (1964): Field investigation of mine waters in the northern anthracite field, Pennsylvania. – US Geological Survey Professional Paper, **0473-B**: B1-B8, 3 fig., 3 tab.; Reston.
- BARRY, M. E., IVEY, G. N., WINTERS, K. B. & IMBERGER, J. (2001): Measurements of diapycnal diffusivities in stratified fluids. – J. Fluid Mech., **442**: 267-291, 11 fig., 2 tab.; Cambridge.
- BARTELS, C. & LORENZ, E. (1993): Die Grube Glasebach – ein Denkmal des Erz- und Fluoritbergbaus im Ostharz. – Der Anschnitt, **45** (4): 144-158, 10 fig.; Bochum.
- BARTHEL, F. H. (1993): Die Urangewinnung auf dem Gebiet der ehemaligen DDR von 1945 bis 1990. – Geol. Jb., **A 142**: 335-346, 2 fig., 5 tab.; Hannover.
- BARTON, C. D. & KARATHANASIS, A. D. (1999): Renovation of a failed constructed wetland treating acid mine drainage. – Environ. Geol., **39** (1): 39-50, 9 fig., 4 tab.; New York, Berlin.
- BATCHELOR, G. K. (2000): An introduction to fluid dynamics. – 615 p., 172 fig.; Cambridge (Cambridge University Press).
- BATES, R. L. & JACKSON, J. A. (1987): Glossary of geology. – 3rd edn., 788 p.; Alexandria (American Geological Institute).
- BATTY, L. C., BAKER, A. J. M. & WHEELER, B. D. (2002): Aluminium and phosphate uptake by *Phragmites australis*: the role of Fe, Mn and Al root plaques. – Ann. Bot., **89** (4): 443-449, 2 fig., 2 tab.; Oxford.
- BATTY, L. C. (2005): The Potential Importance of Mine Sites for Biodiversity. – Mine Water Env., **24** (2): 101-103; Berlin.
- BATTY, L. C. & YOUNGER, P. L. (2003): Effects of External iron Concentration upon Seedling Growth and Uptake of Fe and Phosphate by the Common Reed, *Phragmites Australis* (Cav.) Trin Ex. Steudel. – Ann. Bot., **92** (6): 801-806, 2 fig., 2 tab.; Oxford.
- BATTY, L. C. & YOUNGER, P. L. (2002): Critical role of macrophytes in achieving low iron concentrations in mine water treatment wetlands. – Environ. Sci. Technol., **36** (18): 3997-4002, 6 fig.; Washington.
- BAU, H. H. & TORRANCE, K. E. (1981a): Transient and Steady behavior of an open, symmetrically-heated, free convection loop. – Int. J. Heat Mass Transfer, **24**: 597-609, 15 fig.; Oxford.
- BAU, H. H. & TORRANCE, K. E. (1981b): On the stability and flow reversal of an asymmetrically heated open convection loop. – J. Fluid Mech., **109**: 417-433, 10 fig., 2 tab.; Cambridge.
- BAU, H. H. & TORRANCE, K. E. (1983): On the effects of viscous dissipation and pressure work in free convection loops. – Int. J. Heat Mass Transfer, **26** (5): 727-734, 5 fig., 2 tab.; Oxford.
- BAUMANN, L. (1992): Zur Metallogenie des Erzgebirges sowie Darstellung einiger seiner bedeutenden Lagerstätten. – Schriftenreihe Gesellschaft Deutscher Metallhütten- und Bergleute, **64**: 1-30, 17 fig., 2 tab.; Clausthal-Zellerfeld.
- BÄUMLE, R. (2003): Geohydraulic Characterisation of Fractured Rock Flow Regimes Regional Studies in Granite (Lindau, Black Forest, Germany) and Dolomite (Tsumeb Aquifers, Northern Namibia). – 129 p., 61 fig., 17 tab., 4 app.; Karlsruhe (unpubl. Diss. Uni Karlsruhe).

- BÄUMLE, R., BEHRENS, H., EINSIDL, F., GOLDSCHNEIDER, N., GRUST, K., HÖTZL, H., KÄSS, W., KENNEDY, K., KINZELBACH, W., KOZEL, R., MÜLLER, I., MÜLLER, J., NIEHREN, S., ROSSI, P., SCHNEGG, P.-A., SEILER, K.-P., WITTHÜSER, K., WOHNLICH, S. & ZOJER, H. (2001): Comparative Tracer Studies in Groundwater. – *Beitr. Hydrogeologie*, **52**: 103-230, 55 fig., 21 tab.; Graz.
- BAYER, D. (without year): Die Himmelfahrt Fundgrube – Ein Führer durch das Lehr- und Besucherbergwerk der TU Bergakademie Freiberg. – 144 p., 83 fig., 3 tab.; Freiberg (Eigenverlag TU Bergakademie Freiberg).
- BAYLESS, E. R. & OLYPHANT, G. A. (1993): Acid-generating salts and their relationship to the chemistry of groundwater and storm runoff at an abandoned mine site in southwestern Indiana, U.S.A. – *J. Contam. Hydrol.*, **12** (4): 313-328, 6 fig., 2 tab.; Amsterdam.
- BEAR, J. & BACHMAT, Y. (1991): Introduction to Modeling of Transport Phenomena in Porous Media. – In: BEAR, J.: *Theory and Application of Transport in Porous Media* **4**. – 553 p.; Dordrecht (Kluwer).
- BECK (1981): Zwischenbericht "Sulfid-Schwefel". – 18 p., 10 fig.; Rottleberode (VEB Schwerspat).
- BECKE, A., DOUFFET, H., JOBST, W., PFORR, H., SENNEWALD, R., WÄCHTLER, E. & WAGENBRETH, O. (1986): Der Freiburger Bergbau – Technische Denkmale und Geschichte. – 1st edn., 382 p., 315 fig., 28 tab.; Leipzig (VEB Deutscher Verlag für Grundstoffindustrie).
- BEHRENS, H., BEIMS, U., DIETER, H., DIETZE, G., EIKMANN, T., GRUMMT, T., HANISCH, H., HENSELING, H., KÄSS, W., KERNDORFF, H., LEIBUNDGUT, C., MÜLLER-WEGENER, U., RÖNNEFAHRT, I., SCHARENBERG, B., SCHLEYER, R., SCHLOZ, W. & TILKES, F. (2001): Toxicological and ecotoxicological assessment of water tracers. – *Hydrogeol. J.*, **9** (3): 321-325, 2 tab.; Berlin.
- BEI, X. & CHINA DAILY (2004): Chinese Mine Accidents Provide Tragic Lessons. – *Mine Water Env.*, **23** (1): 54; Berlin.
- BEKENDAM, R. F. & PÖTTGENS, J. J. (1995): Ground movements over the coal mines of southern Limburg, the Netherlands, and their relation to rising mine waters. – *IAHS-AISH Publication*, **234**: 3-12, 6 fig.; The Hague.
- BENCH, D. W. (2000): PCBs, Mining, and Water Pollution. – *Engineering & Mining Journal*, **201** (8): 36-38, 1 fig.; Overland.
- BENEDIX, E. H., CASPER, S. J., DANERT, S., HÜBSCH, P., LINDNER, K. E., SCHMIEDEKNECHT, M. & SCHUBERT, R. (1991): *Urania Pflanzenreich in vier Bänden – Viren Bakterien Algen Pilze*. – 1st edn., 664 p., fig.; Leipzig Jena Berlin (Urania Verlag).
- BENKOVICS, I., CSICSÁK, J., CSÖVÁRI, M., LENDVAI, Z. & MOLNÁR, J. (1997): Mine Water Treatment – Anion-exchange and Membrane Process. – *Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia*, **1**: 149-157, 6 fig., 2 tab.; Ljubljana.
- BENNER, S. G., BLOWES, D. W. & PTACEK, C. J. (1997): A full-scale porous reactive wall for prevention of acid mine drainage. – *Ground Water Monit. Remediat.*, **17**: 99-107, 5 fig., 1 tab.; Dublin.
- BENNETT, J. C. & TRIBUTSCH, H. (1978): Bacterial Leaching Patterns on Pyrite Crystal Surfaces. – *J. Bacteriol.*, **134** (1): 310-317, 3 fig.; Washington.
- BENNING, H.-J., MITTRACH, J. & HEIMING, M. (2002): Aufwältigung und Sanierung eines abgeworfenen Schachtes im Bereich einer Deponie. – In: BUSCH, W., MAAS, K., KLAPPERICH, H., TONDERA, D., SROKA, A. & MEIER, G.: *2. Altbergbau-Kolloquium – Tagungsband*. – p. 258-267, 2 fig.; Clausthal-Zellerfeld (Papierflieger).
- BENTHAUS, F. C. & NITSCHKE, C. (2005): Wiederherstellung des Wasserhaushaltes in den vom Braunkohlenbergbau beeinflussten Gebieten Mitteldeutschlands und der Lausitz. – *Wissenschaftliche Mitteilungen*, **28**: 15-19, 4 fig.; Freiberg.

- BERG, M., TRAN, H. C., NGUYEN, T. C., PHAM, H. V., SCHERTENLEIB, R. & GIGER, W. (2001): Arsenic Contamination of Groundwater and Drinking Water in Vietnam: A Human Health Threat. – Environ. Sci. Technol., **35** (13): 2621-2626, 4 fig., 2 tab.; Washington.
- BERK, W. v. (1987): Hydrochemische Stoffumsetzung in einem Grundwasserleiter – beeinflusst durch eine Steinkohlebergehalde. – Besondere Mitteilungen zum Deutschen Gewässerkundlichen Jahrbuch, **49**: 1-175, 49 fig., 6 tab.; Düsseldorf.
- BERK, W. v. (2000): Hydrogeochemische Modellierung als Instrument zur Wirkungsanalyse hydrogeochemischer Prozesse. – 270 p., 63 fig., 89 tab.; Aachen (Shaker).
- BERNASCONI, C. & BETLEM, G. (2003): Licences, liability and cross-border pollution. – Environ. Liability, **3**: 83-88; London.
- BESSLER, F. T., SAVIC, D. A. & WALTERS, G. A. (2003): Water Reservoir Control with Data Mining. – J. Water Resour. Plan. Manage.-ASCE, **129** (1): 26-34, 8 fig., 4 tab.; Reston.
- BETLEM, G. & BRANS, E. H. P. (2002): The Future Role of Civil Liability for Environmental Damage in the EU. – Yearbook of European Environmental Law, **2**: 183-221, 2 tab.; Oxford.
- BEUGE, P., KLUGE, A. & DEGNER, T. (2001): Prognose geochemischer Auswirkungen der Nachnutzung stillgelegter Bergbau Stollensysteme – Abschlußbericht über ein Forschungsprojekt gefördert unter dem Az: 15339 von der Deutschen Bundesstiftung Umwelt. – In: DEUTSCHE BUNDESSTIFTUNG UMWELT. – 88 p., 25 fig., 24 tab., 6 app.; Freiberg (TU Bergakademie Freiberg).
- BEYER, W. (1986): Zur mikrobiellen In-situ-Laugung komplexer Sulfiderze des Rammelsbergs bei Goslar. – 118 p., 83 fig., 16 tab.; Berlin (Weidler).
- BIAGLOW, J. A. (1988): Study to determine the effectiveness of the hydraulic mine seal at Big Four Hollow Creek near Lake Hope State Park in southeastern Ohio. – 169 p.; Dayton OH (Master's Thesis).
- BIEHLER, D. & FALCK, W. E. (1999): Simulation of the effects of geochemical reactions on groundwater quality during planned flooding of the Königstein uranium mine, Saxony, Germany. – Hydrogeol. J., **7** (3): 284-293, 11 fig., 2 tab.; Berlin.
- BIGHAM, J. M. (1994): Mineralogy of Ochre Deposits Formed by Sulfide Oxidation. – In: JAMBOR, J. L. & BLOWES, D. W.: Short Course Handbook on Environmental geochemistry of Sulfide Mine-Wastes **22**. – p. 103-132, 12 fig., 3 tab.; Waterloo, Ontario (Mineralogical Association of Canada).
- BIRGERSSON, L., WIDEN, H., AAGREN, T. & NERETNIEKS, I. (1992): Tracer migration experiments in the Stripa mine 1980-1991 **STRIPA-TR-92-25**. – 151 p.; Stockholm (Swedish Nuclear Fuel and Waste Management Co.).
- BISWAS, A. K. (1970): History of hydrology. – 336 p.; Amsterdam u.a. (North-Holland Publishing).
- BLANC, G. & ANSCHUTZ, P. (1995): New stratification in the hydrothermal brine system of the Atlantis II Deep, Red Sea. – Geology, **23** (6): 543-546, 2 fig., 2 tab.; Boulder.
- BLEEKER, P. M., TEIGA, P. M., SANTOS, M. H., DE KOE, T. & VERKLEIJ, J. A. C. (2003): Ameliorating effects of industrial sugar residue on the Jales gold mine spoil (NE Portugal) using *Holcus lanatus* and *Phaseolus vulgaris* as indicators. – Environ. Pollut., **125** (2): 237-244, 2 fig., 3 tab.; Oxford.
- BLOCKEEL, H., DZEROSKI, S. & GRBOVIC, J. (1999): Simultaneous prediction of multiple chemical parameters of river water quality with TILDE. – Lecture notes in computer science, **1704**: 32-40, 1 fig., 3 tab.; Berlin.
- BLOWES, D. W., PTACEK, C. L., FRIND, E. O., JOHNSON, R. H., ROBERTSON, W. D. & MOLSON, J. W. (1994): Acid-Neutralization Reactions in inactive Mine Tailings Impoundments and their Effect on the Transport of dissolved Metals. – Proceedings, International Land Reclamation and Mine Drainage Conference, **1**: 429-438, 4 fig., 1 tab.; Pittsburgh.

- BOEHRER, B. (2000): Modal Response of a Deep Stratified Lake: Western Lake Constance. – J. Geophys. Res. C, **105** (12): 28,837-28,845, 9 fig., 1 tab.; Washington.
- BOEHRER, B., ILMBERGER, J. & MÜNNICH, K. O. (2000): Vertical Structure of Currents in Western Lake Constance. – J. Geophys. Res. C, **105** (12): 28,823-28,835, 11 fig., 1 tab.; Washington.
- BOGNER, J. E. & DOEHLER, R. W. (1984): Ettringite Formation following Chemical Neutralization Treatment of Acidic Coal Mine Drainage. – Int. J. Mine Water, **3** (1): 13-24, 3 fig.; Granada.
- BOHL, W. (1998): Technische Strömungslehre. – 317 p.; Würzburg (Vogel).
- BOICE, J. D., MUMMA, M., SCHWEITZER, S. & BLOT, W. J. (2003): Cancer Mortality in a Texas County with prior Uranium Mining and Milling Activities, 1950-2001. – J. Radiol. Prot., **23** (3): 247-262, 4 fig.; Bristol.
- BOLIS, J. L., WILDEMAN, T. R. & DAWSON, H. E. (1992): Hydraulic Conductivity of Substrates used for passive Acid Mine Drainage Treatment. – p. 79-89, 2 fig., 1 tab.; Duluth (Proceedings, 9th Annual National Meeting – American Society for Surface Mining and Reclamation).
- BOLZICCO, J., AYORA, C., RÖTTING, T. & CARRERA, J. (2003): Performance of the Aznalcóllar permeable reactive barrier. – In: NEL, P. J. L.: Mine Water and the Environment. – p. 287-299, 10 fig., 2 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- BONFIG, K. W. (2002): Technische Durchflussmessung unter Berücksichtigung neuartiger Durchflussmessverfahren. – 3rd edn., 280 p., 221 fig., 17 tab.; Essen (Vulkan).
- BONGAERTS, J. C. (2002): European Water Law: water policy and water resources management in France: the project de loi sur l'eau. – Europ. Env. Law Rev. (8-9, 10): 239-244; 260-270.
- BORDEN, R. K. (2003): Environmental geochemistry of the Bingham Canyon porphyry copper deposit, Utah. – Environ. Geol., **43** (7): 752-758, 6 fig., 1 tab.; Berlin.
- BOSECKER, K. (1980): Bakterieller Leaching – Metallgewinnung mit Hilfe von Bakterien. – Metall, **34** (1): 36-40; Berlin.
- BOSMAN, D. J., CLAYTON, J. A., MAREE, J. P. & ADLEM, C. J. L. (1990): Removal of Sulphate from Mine Water with Barium Sulphide. – Int. J. Mine Water, **9** (1-4): 149-163, 5 fig., 6 tab.; Castro Verde.
- BÖTTCHER, J., BÜDER, W., HAMANN, M., HAMANN, S., LINKERT, K.-H., LIPP, U., MÜLLER, F., ROCHHAUSEN, D., ROSENHAHN, L., SCHRÖDER, B., SCHUPPAN, W. & SCHWARZE, K.-H. (1991): Seilfahrt – Auf den Spuren des sächsischen Erzbergbaus. – In: WISMUT AG. – 2nd edn., 153 p.; Haltern (Bode).
- BOUDJEMAA, R., FORBES, A. B., HARRIS, P. M. & LANGDELL, S. (2003): Multivariate empirical models and their use in metrology. – Reports NPL-CMSC, **CMSC 32/03**: 55, 30 fig.; Teddington.
- BOULT, S. (1996): Fluvial Metal Transport near Sources of Acid Mine-Drainage: Relationships of Soluble, Suspended and Deposited Metal. – Mineral. Mag., **60** (399): 325-335, 5 fig., 2 tab.; London.
- BOURGEOIS, W., GARDEY, G., SERVIERES, M. & STUETZ, R. M. (2003): A Chemical Sensor Array based System for Protecting Wastewater Treatment Plants. – Sens. Actuator B-Chem., **91** (1-3): 109-116, 9 fig.; Lausanne.
- BOWELL, R. J., CONNELLY, R. J., ELLIS, J., COWAN, J., WOOD, A., BARTA, J. & EDWARDS, P. (1997): A review of sulfate removal options from mine waters. – Open-File Report – US Geological Survey, **OF 97-0496**: 14; Reston.
- BOWELL, R. J., BARTA, J., GRINGRICH, M., MANSANARES, W. & PARSHLEY, J. (1998): Geological Controls on Pit Lake Chemistry: Implications for the assessment of Water Quality in Inactive Open Pits. – In: NEL, P. J. L.: Mine Water and Environmental Impacts **2**. – p. 375-386, 5 fig., 1 tab.; Johannesburg (Proceedings International Mine Water Association Symposium).

- BOWELL, R. J. (2001): Hydrogeochemistry of the Getchell Underground Mine – Part 1: Mine Water Chemistry. – *Mine Water Env.*, **20** (2): 81-97, 10 fig., 7 tab.; Berlin.
- BOWELL, R. J. (2002): The hydrogeochemical dynamics of mine pit lakes. – *Spec. Publ. – Geol. Soc. London*, **198**: 159-185, 18 fig., 7 tab.; London.
- BOWELL, R. J. (2003): Pit Lake Systematics: A Special Issue. – *Mine Water Env.*, **22** (4): 167-169; Berlin.
- BOWELL, R. J. (2004): A review of sulphate removal options for mine waters. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: *mine water 2004 – Proceedings International Mine Water Association Symposium 2*. – p. 75-91, 6 fig., 7 tab.; Newcastle upon Tyne (University of Newcastle).
- BOX, G. E. P. (1976): Science and Statistics. – *J. American Statistical Association*, **71** (356): 791-799, 3 fig., 1 tab.; Alexandria.
- BOX, G. E. P. (1979): Robustness in the strategy of scientific model building. – In: LAUNER, R. L. & WILKINSON, G. N.: *Robustness in Statistics*. – p. 201-236, 14 fig., 1 tab.; New York (Academic Press).
- BOX, G. E. P., HUNTER, W. G. & HUNTER, J. S. (1978): *Statistics for Experimenters – An Introduction to Design, Data Analysis, and Model Building*. – 653 p., 129 fig., 124 tab.; New York (Wiley).
- BRANDBERG, F., GRUNDFELT, B., HOGLUND, L. O. & LINDGREN, M. (1993): Prediction of metal discharges from sulphidic mine tailings covered with soil. – *IAHS-AISH Publication*, **220**: 193-202, 3 fig., 2 tab.; Louvain.
- BRANDT, A. & FERNANDO, H. J. S. (1995): *Double-Diffusive Convection: Geophysical Monograph 94*. – 334 p.; Washington (American Geophysical Union).
- BRASSINGTON, R. (1999): *Field Hydrogeology*. – 2nd edn., 248 p., 104 fig., 43 tab., 2 app.; Chichester u.a. (Wiley & Sons).
- BRENDEL, K., BRÜCKNER, G., KNITZSCHKE, G., SCHWANDT, A. & SPILKER, M. (1982): Montanhydrologische Aspekte zur Gewährleistung der Bergbausicherheit beim Abbau zechsteinzeitlicher Lagerstätten. – *Z. geol. Wiss.*, **10** (1): 7-31, 13 fig.; Berlin.
- BRENNER, F. J. (2000): Wildlife and Fishery Considerations in Surface Mine Reclamation. – In: BARNHISEL, R. I., DARMODY, R. G. & DANIELS, W. L.: *Reclamation of Drastically Disturbed Lands 15*. – p. 399-413, 6 fig., 1 tab.; Madison, Wis. (American Society of Agronomy).
- BRETHERTON, B. (1989): Near-surface acid mine water pools and their implications for mine abandonment, Coeur d'Alene mining district, Idaho. – 152 p., 32 fig., 17 tab.; Moscow (Master's Thesis University of Idaho).
- BREWITZ, W., KULL, H. & TEBBE, W. (1985): In situ experiments for the determination of macro-permeability and RN-migration in faulted rock formations such as the oolitic iron ore in the Konrad Mine: Design and instrumentation in underground laboratories for radioactive waste disposal. – p. 289-302, 4 fig., 1 tab.; Rotterdam (Balkema).
- BRIECHLE, D., HOLL-HAGEMEIR, C., LÖGTERS, C., MAYERS-BEECKS, E., MILBERT, G., PAAS, W. & PRÜFERT, J. (1998): Raw material extraction in the Rhenish lignite district – the effect of the deep open-cast mining on the population and environment. – *Chron. Rech. Minière*, **66** (533): 13-29, 19 fig., 1 tab.; Orleans.
- BRIX, H. (1987a): The applicability of the wastewater treatment plant in Othfresen as scientific documentation of the root-zone method. – *Water Sci. Technol.*, **19** (10): 19-24, 1 fig., 1 tab.; Oxford.
- BRIX, H. (1987b): Treatment of wastewater in the rhizosphere of wetland plants – the root-zone method. – *Water Sci. Technol.*, **19** (Rio): 107-118, 5 fig., 2 tab.; Oxford.

- BRIX, H. (2003): Plants used in Constructed Wetlands and their Function. – Proceedings, 1st International Seminar on the Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands: 81-109, 9 fig., 2 tab.; Lisboa.
- BRONTĚ, C. (1993 [1860]): Emma: Unfinished Novels. – p. 98-113; Stroud (Sutton).
- BROOK, C. & CRESSWELL, P. J. (1989): Environmental Aspects of Opencast Mining with reference to the Rother Valley Country Park. – Min. Eng.: 191-196, 5 fig.; London.
- BROWN, A. (1982): A validated theory of underground mine flooding. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **B**: 411-428, 9 fig.; Budapest.
- BROWN, E. B. & HOLLEY, M. C. (1982): Metal Levels Associated with Tin Dredging and Smelting and Their Effect Upon Intertidal Reef Flats at Ko Phuket, Thailand. – Coral Reefs, **1**: 131-137, 9 fig., 2 tab.; Berlin.
- BROWN, M. C. & FORD, D. C. (1971): Quantitative tracer methods for investigation of karst hydrological systems. – Trans. Cave Research Group of Great Britain, **13** (1): 37-51, 13 fig., 2 tab.; Bridgewater.
- BROWN, M., BARLEY, B. & WOOD, H. (2002): Minewater Treatment – Technology, Application and Policy. – 500 p.; London (IWA Publishing).
- BRUNNER, H., STRASSER, H., PÜMPEL, T. & SCHINNER, F. (1993): Mikrobielle Laugung von Kupfer und Zink aus Tetraedrit-haltigem Baryt (Region Kleinkogel/Tirol) durch *Thiobacillus ferrooxidans*. – Arch. f. Lagerst.forsch. Geol. B.-A., **16**: 5-12, 6 fig., 3 tab.; Wien.
- BRYAN, C. G., HALLBERG, K. B. & JOHNSON, D. B. (2004): Microbial populations in surface spoil at the abandoned Mynydd Parys copper mines. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: mine water 2004 – Proceedings International Mine Water Association Symposium **1**. – p. 107-112, 3 tab.; Newcastle upon Tyne (University of Newcastle).
- BRYAN, G. W. & LANGSTON, W. J. (1992): Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. – Environ. Pollut., **76** (2): 89-131, 9 fig., 5 tab.; Oxford.
- BUCKSTEEG, K. (1986): Sumpfpflanzenkläranlage – Verfahrensvarianten, Betriebserfahrungen, Problem Bodenhydraulik. – GWF, Gas- Wasserfach: Wasser/Abwasser, **127** (9): 429-434, 4 fig., 1 tab.
- BÜDER, W. & SCHUPPAN, W. (1992): Zum Uranerzbergbau im Lagerstättenfeld Schneeberg-Schlema-Alberoda im Westerzgebirge. – Schriftenreihe Gesellschaft Deutscher Metallhütten- und Bergleute, **64**: 203-221, 9 fig., 2 tab.; Clausthal-Zellerfeld.
- BUHRMANN, F., VAN DER WALDT, M., HANEKOM, D. & FINLAYSON, F. (1999): Treatment of industrial wastewater for reuse. – Desalination, **124** (1-3): 263-269, 3 fig., 4 tab.; Amsterdam.
- BUNDESMINISTER FÜR WIRTSCHAFT (1993): Wismut – Stand der Stilllegung und Sanierung. – BMWI Dokumentation, **335**: 1-35; Bonn.
- BURBEY, T. J., YOUNOS, T. & ANDERSON, E. T. (2000): Hydrologic analysis of discharge sustainability from an abandoned underground coal mine. – J. Am. Water Resour. Assoc., **36** (5): 1161-1172, 10 fig., 4 tab.; Herndon.
- BURCHI, S. & MECHLEM, K. (2005): Groundwater in international law – Compilation of treaties and other legal instruments [sic!]: FAO Legislative Studies **86**. – 584 p., 3 tab.; Rome (United Nations Educational Scientific, and Cultural Organization; Food and Agriculture Organization of the United Nations).
- BURKE, S. P. & YOUNGER, P. L. (2000): Groundwater Rebound in the South Yorkshire Coalfield: A First Approximation using the GRAM Model. – Q. J. Eng. Geol. Hydrogeol., **33** (2): 149-160, 9 fig., 5 tab.; London.

- BURNETT, J. M., BURNETT, M., ZIEMKIEWICZ, P. & BLACK, D. C. (1995): Pneumatic backfilling of coal combustion residues in underground mines. – In: HYNES, T. P. & BLANCHETTE, M. C.: Sudbury '95 – Mining and the Environment **3** (REHAB 18). – p. 1211-1220, 6 fig.; Ottawa (CANMET).
- BUSCH, W., MAAS, K., KLAPPERICH, H., TONDERA, D., SROKA, A. & MEIER, G. (2002): 2. Altbergbaukolloquium – Tagungsband. – 388 p.; Clausthal-Zellerfeld (Papierflieger).
- CABRAL, A., LEFEBVRE, G., PROULX, M., AUDET, C., LABBE, M. & MICHAUD, C. (1997): Use of deinking residues as cover material in the prevention of AMD generation at an abandoned mine site. – Proceedings, International Conference on Tailings and Mine Waste, **4**: 257-266, 6 fig., 1 tab.; Rotterdam.
- CACAS, M. C., LEDOUX, E., DE MARSILY, G., TILLIE, B., BARBREAU, A., DURAND, E., FEUGA, B. & PEAUDE CERF, P. (1990): Modeling Fracture Flow With a Stochastic Discrete Fracture Network: Calibration and Validation. 1. The Flow Model. – Water Resour. Res., **26** (3): 479-489, 12 fig., 1 tab.; Washington.
- CAI, S., LIN, Y., ZHINENG, H., XIANZU, Z., ZHALOU, Y., HUIDONG, X., YUANRONG, L., RONGDI, J., WENHAU, Z. & FANGYUAN, Z. (1990): Cadmium exposure and health effects among residents in an irrigation area with ore dressing wastewater. – Sci. Total Environ., **90**: 67-73, 3 tab.; Amsterdam.
- CALLAGHAN, T., BRADY, K., CHISHOLM, W. & SAMES, G. (2000): Hydrology of the Appalachian Bituminous Coalbasin. – In: KLEINMANN, R. L. P.: Prediction of Water Quality at Surface Coal Mines (3). – p. 36-72, 5 fig., 4 tab.; Morgantown (National Mine Land Reclamation Center).
- CALOW, R. (1991): Quality Control/Quality Assurance in Geochemical Laboratories. – Explore – Newsletter for the Association of Exploration Geochemists, **72**: 23-24; Reno.
- CANTY, G. A. & EVERETT, J. (1998): Using tracers to understand the hydrology of an abandoned underground coal mine **15**. – p. 62-72, 5 fig., 2 tab.; Princeton (Proceedings, 15th Annual National Meeting – American Society for Surface Mining and Reclamation).
- CANTY, G. A. & EVERETT, J. (1999): Remediation of Underground Mine Areas through Treatment with Fly Ash. – 167 p., 30 fig., 43 tab.; Oklahoma City (Oklahoma Conservation Commission).
- CARLIN, J. F. J. (1999): Tin. – In: PLUNKERT, P. A. & JONES, T. S.: Metal Prices in the United States through 1998. – p. 159-160, 1 fig., 1 tab.; Washington (US Department of the Interior, US Geological Survey).
- CARLOWITZ, H. C. v. (1713): Sylvicultura Oeconomica, oder haußwirthliche Nachricht und Naturmäßige Anweisung zur Wilden Baum-Zucht, nebst Gründlicher Darstellung, wie zufförderst durch Göttliches Benedeyen dem allenthalben und insgemein einreissenden Grossen Holtz-Mangel, vermittelst Säe-Pflantz- und Versetzung vielerhand Bäume zu prospiciere, auch also durch Anflug und Wiederwachs des so wohl guten und schleunig anwachsend, als anderen gewüchsig und nützlichen Holtzes, ganz öde und abgetriebene Holtz-Ländereyen, Plätze und Orte widerum Holzreich, nütz und brauchbar zu machen; Bevorab von Saam-Bäumen und wie der wilde Baum-Saamen zu sammeln, der Grund und Boden zum Säen zuzurichten, solche Saat zu bewerkstelligen, auch der junge Anflug und Wiederwachs zu beachten. Daneben das sogenannte lebendige, oder Schlag-an Ober- und Unter-Holz aufzubringen und zu vermehren, welchen beygefügt die Arten des Tangel- und Laub Holzes, theils deren Eigenschafften und was besagtes Holtz für Saamen trage, auch wie man mit frembden Baum-Gewächsen sich zu verhalten, ferner wie das Holz zu fällen, zu verkohlen, zu äschern und sonst zu nutzen. Alles zu nothdürfftiger Versorgung des Hauß-Bau-Brau-Berg- und Schmeltz-Wesens, und wie eine immerwährende Holtz-Nutzung, Land und Leuten, auch jedem Hauß-Wirthe zu unschätzbaren großen Auffnehmen, pfleglich und füglich zu erziehen und einzuführen, wobey zugleich eine gründliche Nachricht von den in Churfl. Sächß. Landen Gefundenen Turff Dessen Natürliche Beschaffenheit, grossen Nutzen, Gebrauch und

- nützlichen Verkohlung Aus Liebe zu Beförderung des allgemeinen Bestens beschrieben. – 432 p.; Leipzig (Braun).
- CARLSON, A. B., CRILLY, P. B. & RUTLEDGE, J. C. (2002): Communication systems – An introduction to signals and noise in electrical communication: McGraw-Hill series in electrical and computer engineering. – 4th edn., 864 p., 479 fig.; Boston (McGraw-Hill).
- CARLSSON, E. (2004): Sulphide-Rich Tailings Remediated by Soil Cover – Evaluation of cover efficiency and tailings geochemistry, Kristineberg, northern Sweden. – Doctoral Thesis, 44: 56, 20 fig., 7 app.; Lulea.
- CARVAJAL, D. J. & GONZÁLEZ, A. (2000): Mining Heritage & Closure Mines. – p. 158-165; Rio de Janeiro (CYTED – IMACC – UNIDO).
- CASSITY, W. D. & PESIC, B. (1999): Interaction of *Thiobacillus ferrooxidans* with arsenite, arsenate and arsenopyrite. – In: AMILS, R. & BALLESTER, A.: Biohydrometallurgy and the environment toward the mining of the 21st century A. – p. 521-532, 10 fig., 1 tab.; Amsterdam (Elsevier).
- CASTRILLI, J. F. (2002): Barriers to Collaboration – Orphaned/Abandoned Mines in Canada (Final Report). – 67 p.; Ontario (National Orphaned/Abandoned Mines Advisory Committee [<http://www.abandoned-mines.org/>]).
- CASTRO, J. M. & MOORE, J. N. (2000): Pit lakes: their characteristics and the potential for their remediation. – Environ. Geol., **39** (11): 1254-1260, 2 fig.; Berlin.
- CHALUPNIK, S. & WYSOCKA, M. (2000): Removal of radium from mine waters – the experience from the coal mine. – Proceedings, 7th International Mine Water Association Congress: 352-362, 3 fig.; Ustron.
- CHALUPNIK, S., MICHALIK, B., WYSOCKA, M., SKUBACZ, K. & MIELNIKOW, A. (2001): Contamination of settling ponds and rivers as a result of discharge of radium-bearing waters from Polish coal mines. – J. Environ. Radioact., **54** (1): 85-98, 1 fig., 7 tab.; Oxford.
- CHAN, G. Y. S., YE, Z. H. & WONG, M. H. (2003): Comparison of four *Sesbania* Species to remediate Pb/Zn and Cu Mine Tailings. – Environ. Manage., **32** (2): 246-251, 6 tab.; New York.
- CHAN, T., REID, J. A. K. & GUVANASEN, V. (1987): Numerical modelling of coupled fluid, heat, and solute transport in deformable fractured rock. – In: TSANG, C.-F.: Coupled processes associated with nuclear waste repositories (44). – p. 605-625, 11 fig.; Orlando (Academic Press).
- CHANEY, R. L., BROWN, S. L., LI, Y. M., ANGLE, J. S., STUCZYNSKI, T. I., DANIELS, W. L., HENRY, C. L., SIEBELEC, G., MALIK, M., RYAN, J. A. & COMPTON, H. (2000): Progress in Risk Assessment for Soil Metals, and In-situ Remediation and Phytoextraction of Metals from Hazardous Contaminated Soils. – In: U.S. ENVIRONMENTAL PROTECTION AGENCY: Phytoremediation: State of Science **625/R-01/011b**. – p. 1-26, 4 fig., 5 tab.; Boston (U.S. Environmental Protection Agency).
- CHANSANG, H. (1988): Coastal tin mining and marine pollution in Thailand. – Ambio, **17** (3): 223-228, 6 fig., 2 tab.; Stockholm.
- CHARBONNIER, P. (2001): Management of Mining, Quarrying and Ore-Processing Waste in the European Union **BRGM/RP-50319-FR**. – 79 p., 7 fig., 17 tab., 7 app.; Orleans (BRGM).
- CHEKAN, G. J. (1985): Design of Bulkheads for Controlling Water in Underground Mines. – Bureau of Mines Information Circular, **IC-9020**: 36, 10 fig., 3 tab., 2 app.; Washington.
- CLARK, L. C. j., WOLF, R., GRANGER, D. & TAYLOR, Z. (1953): Continuous Recording of Blood Oxygen Tensions by Polarography. – J. Appl. Physiol., **6** (3): 189-193, 4 fig., 1 tab.; Bethesda.
- CLAUSER, C., HEITFELD, M., ROSNER, P., SAHL, H. & SCHETELIG, K. (2005): Beispiel Aachener Steinkohlenrevier – Nutzung von Erdwärme in aufgelassenen Bergwerken. – Beratende Ingenieure (6): 14-17, 4 fig.; Wiesbaden.

- CLEMENS, T., HUCKINGHAUS, D., LIEDL, R. & SAUTER, M. (1999): Simulation of the development of karst aquifers: role of the epikarst. – *Int. J. Eart. Sciences (Geol. Rundsch.)*, **88** (1): 157-163, 6 fig.; Berlin.
- COETZEE, H. (1995): Radioactivity and the leakage of radioactive waste associated with Witwatersrand gold and uranium mining. – *Proceedings, Uranium-Mining and Hydrogeology, Freiberg, Germany; GeoCongress*, **1**: 90-99, 6 fig.; Freiberg.
- COHEN, R. R. H. & STAUB, M. W. (1992): Technical manual for the design and operation of a passive mine drainage system. – 69 p.; Golden (unpublished Report to the US Bureau of Land Reclamation).
- COL, M. & COL, C. (2003): Environmental Boron Contamination in Waters of Hisarcik Area in the Kutahya Province of Turkey. – *Food Chem. Toxicol.*, **41** (10): 1417-1420; Oxford.
- COLDEWEY, W. G. & SEMRAU, L. (1994): Mine Water in the Ruhr Area (Federal Republic of Germany). – *Proceedings, 5th International Mine Water Congress, Nottingham, U.K.*, **2**: 613-629, 7 fig.; Nottingham.
- COLDEWEY, W. G., HEWIG, R., RICHTER, R., RÜTERKAMP, P. & WEDEWART, M. (1999): Mittelfristige Entwicklung des Chemismus und der Dichte-Schichtungen von Grubenwässern in Bergwerken und ihre Auswirkungen auf nutzbares Grund- und Oberflächenwasser. – Für: LANDESÖBERBERGAMT NORDRHEIN-WESTFALEN. – 83 p., 3 tab., 73 app.; Essen (Deutsche Montan Technologie GmbH).
- COLLON-DROUAILLET, P., FABRIOL, R. & BUÈS, M. (2005): A semi-distributed modelling of the evolution of groundwater quality in flooded mines in the lorraine iron basin. – p. 1-15 [CD-ROM], 10 fig., 3 tab.; Nancy (Proceedings, Post-Mining 2005).
- COMMONWEALTH OF PENNSYLVANIA (1973): Scarlift Report – Clearfield Creek / Moshannon Watershed **SL 1-73:1-101.7**. – 333 p., 21 fig.; Harrisburg (Commonwealth of Pennsylvania).
- COPA, L. V., FERDERER, D. A., GOKLANY, I. M. & KAAS, L. M. (1995): Developing Priorities for Verifying and Inventorying Suspected Abandoned Mine Sites on Public Lands. – In: HOTCHKISS, W. R., DOWNEY, J. S., GUTENTAG, E. D. & MOORE, J. E.: *Water resources at risk*. – p. 125-140, 10 fig.; Minneapolis (American Institute of Hydrology).
- COPPER MINES OF TASMANIA (1995): Mt Lyell Mine Redevelopment – Environmental Management Plan. – 181 p.; Queenstown Tas. (Copper Mines of Tasmania).
- CORWIN, R. F. (1997): The Self-Potential Method for Environmental and Engineering Applications. – In: WARD, S. H.: *Geotechnical and environmental geophysics* **1**. – 4th edn., S. 127-145, 7 fig.; Tulsa (Society of Exploration Geophysicists).
- COULTON, R., BULLEN, C., WILLIAMS, K., DEY, M. & JARVIS, A. (2004): Active treatment of high salinity mine water. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: *mine water 2004 – Proceedings International Mine Water Association Symposium* **2**. – p. 119-124, 3 fig., 6 tab.; Newcastle upon Tyne (University of Newcastle).
- COUPLAND, K., ROWE, O., HALLBERG, K. B. & JOHNSON, D. B. (2004): Biogeochemistry of a subterranean acidic mine water body at an abandoned copper mine. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: *mine water 2004 – Proceedings International Mine Water Association Symposium* **1**. – p. 113-119, 4 fig.; Newcastle upon Tyne (University of Newcastle).
- COURY, L. (1999): Conductance Measurements – Part 1: Theory. – *Current Separations*, **18** (3): 91-96, 4 fig., 1 tab.; West Lafayette.
- CRAVOTTA, C. A. & HILGAR, G. M. (2000): Considerations for Chemical Monitoring at Coal Mines. – In: KLEINMANN, R. L. P.: *Prediction of Water Quality at Surface Coal Mines* **Appendix A**. – p. 195-218, 5 fig.; Morgantown (National Mine Land Reclamation Center).

- CREVELING, H. F., DE PAZ, J. F., BALADI, J. Y. & SCHOENHALS, R. J. (1975): Stability characteristics of a single-phase free convection loop. – *J. Fluid Mech.*, **67** (1): 65-84, 10 fig., 1 tab.; Cambridge.
- CROXFORD, S. J., ENGLAND, A. & JARVIS, A. P. (2004): Application of the PHREEQC geochemical Computer Model during the Design and Operation of UK Mine Water Treatment Schemes. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: mine water 2004 – Proceedings International Mine Water Association Symposium **2**. – p. 125-134, 8 tab.; Newcastle upon Tyne (University of Newcastle).
- CSÖVÁRI, M., BERTA, Z., CSICSÁK, J., HIDEG, J. & VARHEGYI, A. (2004): Case Study: Remediation of a former Uranium Mining/Processing Site in Hungary: Conference Papers **35**. – p. 81-100, 17 fig., 5 tab.; Wien (Umweltbundesamt).
- CUNNINGHAM, S. D., SHANN, J. R., CROWLEY, D. E. & ANDERSON, T. A. (1997): Phytoremediation of Contaminated Water and Soil. – ACS Symposium Series, **664**: 2-17; Washington.
- CVJIĆ, J. (1893): Das Karstphänomen – Versuch einer morphologischen Monographie. – *Geogr. Abh.*, **5** (3): 217-329, 5 fig., 8 tab.; Wien.
- CZOLBE, P., KRETZSCHMAR, H.-J., KLAFFKI, M. & HEIDENREICH, H. (1992): Strömungszellen im gefluteten Salzschant. – *Neue Bergbautechnik*, **22** (6): 213-218, 10 fig.; Leipzig.
- DAHLKAMP, F. J. (1993): Uranium ore deposits. – 460 p.; Berlin (Springer).
- DAMERELL, P. S. & SCHOENHALS, R. J. (1979): Flow in a Toroidal Thermosyphon with Angular Displacement of Heated and Cooled Sections. – *J. Heat Transfer*, **101**: 672-676, 7 fig.; New York.
- DARCY, H. P. G. (1856): Les fontaines publiques de la ville de Dijon – Exposition et application des principes a suivre et des formules a employer dans les questions de distribution d'eau – Ouvrage terminé par un appendice relatif aux fournitures d'eau de plusieurs villes au filtrage des eaux à la fabrication des tuyaux de fonte, de plomb, de toile et de bitume. – 647 p.; Paris (Dalmont).
- DAVIS, J. C. (1986): Statistics and Data Analysis in Geology. – 2nd edn., 646 p.; New York u.a. (Wiley & Sons).
- DAVIS, M. W. (1994a): Identification and remediation of a mine flooding problem in Rico, Dolores County, Colorado with a discussion on the use of tracer dyes. – Colorado Geological Survey Open-File Report, **91-1**: 15, 1 tab.; Denver.
- DAVIS, M. W. (1994b): The use of tracer dyes for the identification of a mine flooding problem, Rico, Dolores County, Colorado. – Colorado Geological Survey Open-File Report, **91-2**: 1-20, 7 fig., 1 tab.; Denver.
- DAVIS, S. N. (1983): Measurement, Prediction, and Hazard Evaluation of Earth Fissuring and Subsidence due to Ground-Water Overdraft. – Office of Water Policy Project, **B-092-ARIZ**: 44, 12 fig., 1 tab.; Arizona.
- DE LA VERGNE, J. (2003): Hard Rock Miner's Handbook. – 3rd edn., 314 p., 13 fig., 98 tab.; Tempe (McIntosh Engineering).
- DEHAY, K. L. (2003): Assessment and Comparison of 1976-77 and 2002 Water Quality in Mineshafts in the Picher Mining District, Northeastern Oklahoma and Southeastern Kansas. – Water-Resources Investigations Report, **03-4248**: 65, 6 fig., 2 tab., 2 app.; Washington.
- DEHAY, K. L., ANDREWS, W. J. & SUGHRU, M. P. (2004): Hydrology and Ground-Water Quality in the Mine Workings within the Picher Mining District, Northeastern Oklahoma, 2002-03. – Scientific Investigations Report, **2004-5043**: 62, 24 fig., 4 tab., 4 app.; Washington.
- DEMCHAK, J., SKOUSEN, J. G. & McDONALD, L. M. (2004): Longevity of Acid Discharges from Underground Mines Located above the Regional Water Table. – *J. Environ. Qual.*, **33** (2): 656-668, 6 fig., 6 tab.; Madison.

- DEMING, D. (2002): Introduction to Hydrogeology. – 1st edn., 468 p.; Boston u.a. (McGraw-Hill).
- DEMPSEY, B. A. & JEON, B.-H. (2001): Characteristics of sludge produced from passive treatment of mine drainage. – *Geochem.-Explor. Environ. Anal.*, **1**: 89-94, 5 fig., 5 tab.; London.
- DIAZ, W. D. (1990): Hydrogeochemistry and physical hydrogeology of the Newfoundland zinc mine, Daniel's Harbour, Newfoundland. – 282 p., Refs: 93; illus. incl. 18 tables, 2 plates; St. John's, NF (Master's Thesis).
- DIETRICH, P. (1965): Zur Geologie des Kalksteinvorkommens von Niederrabenstein. – 17 p., 30 fig., 47 app.; Freiberg (unpubl. Thesis Bergakademie Freiberg).
- DINELLI, E. & TATEO, F. (2002): Different types of fine-grained sediments associated with acid mine drainage in the Libiola Fe-Cu mine area (Ligurian Apennines, Italy). – *Appl. Geochem.*, **17** (8): 1081-1092, 4 fig., 4 tab.; Oxford.
- DIZ, H. R. (1998): The Selective Oxide System™: Acid Drainage Treatment that avoids the Formation of Sludge. – *Mine Water Env.*, **17** (1): 1-7, 3 fig.; Lakewood, CO.
- DIZDAREVIĆ, T. & REŽUN, B. (1998): Environmental Problems related to the closing down of the Idrija Mercury Mine **2**. – p. 367-374, 4 fig.; Johannesburg (Proceedings International Mine Water Association Symposium).
- DOBLER, L. (1999): Der Einfluß der Bergbauschichte im Ostharz auf die Schwermetall-tiefengradienten in historischen Sedimenten und die fluviale Schwermetalldispersion in den Einzugsgebieten von Bode und Selke im Harz. – 120 p., 22 fig., 41 tab., 9 maps, 24 app.; Halle (unpubl. Diss. Univ. Halle-Wittenberg).
- DOHERTY, J. (1990): The interpretation of pump-test data from a disused underground mine. – *J. Hydrol.*, **114** (1-2): 109-123, 7 fig., 2 tab.; Amsterdam.
- DOMENICO, P. A. & SCHWARTZ, F. W. (1998): Physical and Chemical Hydrogeology. – 2nd edn., 506 p.; New York u.a. (Wiley & Sons).
- DOORNBOS, M. H. (1989): Hydrogeologic controls upon the water quality of the acid mine drainage in the Sand Coulee-Stochett area, Montana. – 103 p., Refs: 57; Butte (Master's Thesis Montana College of Mineral Science & Technology).
- DÖRING, M. (1993): Unterirdische Wasserkraftwerke im Bergbau. – *Wasserwirtschaft*, **83** (5): 272-278, 6 fig.; Stuttgart.
- DÖRING, M. (1999): Eisen und Silber – Wasser und Wald. – 226 p.; Kreuztal (wielandschmiede).
- DOUGLAS, G. E., JOHN, D. M., WILLIAMSON, D. B. & REID, G. (1998): The aquatic algae associated with mining areas in Peninsula Malaysia and Sarawak: their composition, diversity and distribution. – *Nova Hedwigia*, **67** (1-2): 189-211, 9 fig., 4 tab.; Berlin.
- DOWDALL, M., VICAT, K., FREARSON, I., GERLAND, S., LIND, B. & SHAW, G. (2004): Assessment of the radiological impacts of historical coal mining operations on the environment of Ny-Alesund, Svalbard. – *J. Environ. Radioact.*, **71** (2): 101-114, 5 fig., 5 tab.; Oxford.
- DOWNING, B. W. & MILLS, C. (1998): Quality Assurance/Quality Control for Acid Mine Drainage. – 1 p., 7 fig. (<http://www.enviromine.com/ard/Acid-Base%20Accounting/Quality.htm> 5.1.2005).
- DOYLE, G. A. & RUNNELLS, D. D. (1997): Physical limnology of existing mine pit lakes. – *Mining Engineering*, **49** (12): 76-80, 5 fig., 1 tab.; Littleton.
- DRAZIN, P. G. (2002): Introduction to Hydrodynamic Stability: Cambridge texts in applied mathematics **32**. – 1st edn., 258 p., 78 fig.; Cambridge (Cambridge University Press).
- DRISCOLL, F. G. (1987): Groundwater and Wells. – 2nd edn., 1089 p.; Minnesota (Johnson Division).
- DROSTE, R. L. (1997): Theory and Practice of Water and Wastewater Treatment. – 800 p.; New York (Wiley & Sons).

- DU PLESSIS, H. M. (1983): Using Lime Treated Acid-Mine Water for Irrigation. – *Water Sci. Technol.*, **15** (2): 145-154, 5 fig., 3 tab.; Oxford.
- DUANE, M. J., PIGOZZI, G. & HARRIS, C. (1997): Geochemistry of some deep gold mine waters from the western portion of the Witswatersrand Basin, South Africa. – *J. Afr. Earth Sci.*, **24** (1-2): 105-123, 6 fig., 2 tab.; Oxford.
- DUARTE, J. C., ESTRADA, P., BEAUMONT, H., SITIMA, M. & PEREIRA, P. (1990): Biotreatment of Tailings for Metal recovery. – *Int. J. Mine Water*, **9** (1-4): 193-206, 4 fig., 3 tab.; Castro Verde.
- DUFFUS, J. H. (2002): “Heavy Metals” – A meaningless term? – *Pure Appl. Chem.*, **74** (5): 793-807, 3 fig., 4 tab.; Oxford.
- DUMPLETON, S., ROBINS, N. S., WALKER, J. A. & MERRIN, P. D. (2001): Mine water rebound in South Nottinghamshire: risk evaluation using 3-D visualization and predictive modelling. – *Q. J. Eng. Geol. Hydrogeol.*, **34** (3): 307-319, 6 fig., 3 tab.; London.
- DUNHAM, K. C. (1990 [1948]): Geology of the Northern Pennine Orefield – Tyne to Stainmore. – In: BRITISH GEOLOGICAL SURVEY: Economic Memoir of the British Geological Survey – England and Wales **1**. – 2nd edn., 299 p., 46 fig., 75 tab., 10 plates; London (Her Majesty’s Stationery Office).
- DURGIN, W. W. (1981): Flow: Its Measurement and Control in Science and Industry **2**. – 867 p.; Triangle Park NC (ISA, Research).
- DUSHENKOV, S., VASUDEV, D., KAPULNIK, Y., GLEBA, D., FLEISHER, D., TING, K. C. & ENSLEY, B. (1997): Removal of Uranium from Water Using Terrestrial Plants. – *Environ. Sci. Technol.*, **31** (12): 3468-3474, 9 fig., 2 tab.; Washington.
- DZOMBAK, D. A. & MOREL, F. M. M. (1990): Surface Complexation Modeling – Hydrous Ferric Oxides. – 393 p.; New York (Wiley).
- EBELING, W. (1986): Betriebserfahrungen mit dem Wurzelraumverfahren. – *Schriftenr. WAR*, **26**: 31-49, 6 fig., 1 tab.; Darmstadt.
- EBENA, G. (2003): Sulfidic mine waste microorganisms in an ecological context. – *Linköping studies in science and technology*, **841**: 51, 8 fig., 1 tab., 6 app.; Linköping.
- EBNER, H. G. & SCHWARZ, W. (1973): Untersuchungen über die Laugung von Uranerzen mit Hilfe von Bakterien. – *Erzmetall*, **26** (10): 484-490, 3 fig., 7 tab.; Clausthal-Zellerfeld.
- ECKHART, E. & RIEPLER, F. (2004): Sicherungsmaßnahmen in den Altbergbauen Nassereith (Tirol) und Schlaining (Burgenland) unter besonderer Berücksichtigung der hydrogeologischen Verhältnisse. – In: MEIER, G., SROKA, A., LÖBEL, K.-H., KLAPPERICH, H., TONDERA, D., BUSCH, W., WAGNER, H. & RANDJBAR, B.: 4. Altbergbau-Kolloquium – Tagungsband. – p. 154-170, 15 fig.; Essen (Glückauf).
- EDWARDS, K. J., BOND, P. L., DRUSCHEL, G. K., MCGUIRE, M. M., HAMERS, R. J. & BANFIELD, J. F. (2000): Geochemical and biological aspects of sulfide mineral dissolution: lessons from Iron Mountain, California. – *Chem. Geol.*, **169**: 383-397, 11 fig.; Amsterdam.
- EDWARDS, P. J., BOLTON, C. P., RANSON, C. M. & SMITH, A. C. (1997): The River Pelenna minewater treatment project. – In: YOUNGER, P. L.: *Minewater Treatment Using Wetlands*. – p. 17-32; London (Chartered Institution of Water and Environmental Management).
- ELANDER, P., LINDVALL, M. & HAKANSSON, K. (1998): MiMi – Prevention and control of pollution from mining waste products: State-of-the-art-report **MiMi 1998:2**. – 29 p.; Linköping (MiMi Print).
- ELLIS, D. V., POLING, G. W. & BAER, R. L. (1995): Submarine Tailings Disposal (STD) for Mines – An Introduction. – *Mar. Geores. Geotechnol.*, **13** (1-2): 3-18, 4 fig.; London.

- ELOFF, E., GREBEN, H. A., MAREE, J. P., RADEBE, B. V. & GOMES, R. E. (2003): Biological sulphate removal using hydrogen as the energy source. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 99-108, 3 fig., 5 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- ENDERLE, G. J. & FRIEDRICH, K. (1995): East-German Uranium Miners (Wismut) – Exposure Conditions and Health Consequences. – *Stem Cells*, **13** (Suppl. 1): 78-89, 9 fig., 2 tab.; Dayton.
- ENVIRONMENT AGENCY (1997): Use and Design of Oil Separators in Surface Water Drainage Systems. – Pollution prevention guidelines, **PPG 3**: 1-4; Almondsbury.
- ENVIRONMENTAL PROTECTION AGENCY (2001): Environmental Protection Agency Annual Report and Account 2000. – 84 p.; Wexford (Environmental Protection Agency).
- ERICKSON, P. M., KLEINMANN, R. L. P., POSLUSZNY, E. T. & LEONARD-MAYER, P. J. (1982): Hydrogeochemistry of a large mine pool. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **A**: 27-42, 7 fig., 2 tab.; Budapest.
- ERMITE CONSORTIUM, [EDS. YOUNGER, P. & WOLKERSDORFER, CH.]. (2004): Mining Impacts on the Fresh Water Environment: Technical and Managerial Guidelines for Catchment Scale Management. – *Mine Water Env.*, **23** (Supplement 1): S2-S80, 28 fig., 6 tab.; Berlin.
- ERNST, W. H. O. (1996): Bioavailability of heavy metals and decontamination of soils by plants. – *Appl. Geochem.*, **11** (1 & 2): 163-167, 4 fig.; Oxford.
- ERNST, W. H. O. (2004): Phytoextraction – options and impossibilities. – Abstracts, Geomicrobiology in remediation of mine waste (3. Jenaer Sanierungskolloquium): 11:30-12:10; Jena.
- EUROPEAN COMMISSION (2000): Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. – *Official Journal of the European Communities*, **L 327**: 1-72, 2 fig.; Luxembourg.
- EVANGELOU, V. P. (1995): Pyrite oxidation and its control – solution chemistry, surface chemistry, acid mine drainage (AMD), molecular oxidation mechanisms, microbial role, kinetics, control, ameliorites and limitations, microencapsulation. – 293 p.; Boca Raton (CRC Press).
- EXNER, C. (1952): Zur Geologie und Petrographie des Radhausberg-Unterbaustollens bei Badgastein. – *Tschermaks Min. u. Petr. Mitt.*, **3** (1): 79-81; Wien.
- FAHLE, W. & ARNOLD, I. (1996): Der Einsatz von Dichtwänden im Lausitzer Braunkohlerevier. – *Braunkohle*, **48** (4): 407-412, 9 fig.; Düsseldorf.
- FAUST, K. M., DENTON, R. A. & PLATE, E. J. (1983): Schichtungseffekte in Flußstauhaltungen: Laborversuche. – *Wasserwirtschaft*, **73** (11): 386-392, 17 fig.; Stuttgart.
- FAUVILLE, A., MAYER, B., FROMMICHEN, R., FRIESE, K. & VEIZER, J. (2004): Chemical and isotopic evidence for accelerated bacterial sulphate reduction in acid mining lakes after addition of organic carbon: laboratory batch experiments. – *Chem. Geol.*, **204** (3-4): 325-344; Amsterdam.
- FAWCETT, R. J., HIBBERD, S. & SINGH, R. N. (1984): An Appraisal of Mathematical Models to predict Water Inflows into underground Coal Workings. – *Int. J. Mine Water*, **3** (2): 33-54, 5 fig.; Granada.
- FEATHERSTONE, A. M. & O'GRADY, B. V. (1997): Removal of Dissolved Copper and Iron at the Freshwater-Saltwater Interface of an Acid Mine Stream. – *Mar. Pollut. Bull.*, **34** (5): 332-337, 4 fig., 1 tab.; Oxford-New York.
- FEDOTOV, S., BASHKIRTSEVA, I. & RYASHKO, L. (2002): Simple stochastic model for laminar-to-turbulent subcritical transition. – *Annual Research Briefs*, **2002**: 397-404, 4 fig.; Stanford.
- FELEGY, E. W., JOHNSON, L. H. & WESTFIELD, J. (1948): Acid mine-water in the anthracite region of Pennsylvania. – *Tech. Paper*, **710**: 49, 13 fig., 11 tab.; Washington, DC.

- FELLOWS, P. A. (1937): Sealing Projects – Sharply Reduce Stream Pollution From Abandoned Mines. – Coal Age, **42**: 158-161, 13 fig., 2 tab.; New York.
- FERNÁNDEZ RUBIO, R. (2004): Sustainable mining – Environmental assets. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: mine water 2004 – Proceedings International Mine Water Association Symposium **2**. – p. 251-271, 37 fig.; Newcastle upon Tyne (University of Newcastle).
- FERNÁNDEZ-RUBIO, R. & FERNANDEZ LORCA, D. (1993): Mine Water drainage. – Mine Water Env., **12** (1-4): 107-130, 22 fig., 1 tab.; Wollongong.
- FERNÁNDEZ-RUBIO, R., FERNÁNDEZ-LORCA, S. & ESTEBAN ARLEGUI, J. (1987): Preventive Techniques for Controlling Acid Water in Underground Mines by Flooding. – Int. J. Mine Water, **6** (3): 39-52, 11 fig.; Budapest.
- FETTER, C. W. (1988): Applied hydrogeology. – 2nd edn., 592 p.; Columbus u.a. (Merrill).
- FICKLIN, W. H., PLUMLEE, G. S., SMITH, K. S. & MCHUGH, J. B. (1992): Geochemical classification of mine drainages and natural drainages in mineralized areas. – Proceedings, International Symposium on Water-Rock Interaction, **7**: 381-384, 2 fig., 2 tab.; Park City.
- FICKLIN, W. H., MOSIER & E. L. (1999): Field Methods for Sampling and Analysis of Environmental Samples for Unstable and Selected Stable Constituents. – In: PLUMLEE, G. S. & LOGSDON, M. J.: The Environmental Geochemistry of Mineral Deposits **6A** (12). – p. 249-264, 2 fig., 3 tab.; Littleton (Society of Economic Geologists).
- FIELD, M. S. (2002): The QTRACER2 Program for Tracer-Breakthrough Curve Analysis for Tracer Tests in Karstic Aquifers and Other Hydrologic Systems **EPA/600/R-02/001**. – 179 p., 51 fig., 13 tab.; Washington (U.S. Environmental Protection Agency – Office of Research and Development).
- FIELD, M. S. (2003): Tracer-Test Planning Using the Efficient Hydrologic Tracer-Test Design (EHTD) Program **EPA/600/R-03/034**. – 175 p., 32 fig., 34 tab.; Washington (U.S. Environmental Protection Agency – Office of Research and Development; National Center for Environmental Assessment).
- FISHER, J. & GREENBANK, L. (2000): The Rogerley Mine, Weardale County Durham, England. – Rocks and Minerals, **75** (1): 54-62, 16 fig.; Peekskill.
- FISHMAN, M. J. & FRIEDMAN, L. C. (1989): Methods for determination of inorganic substances in water and fluvial sediments. – Techniques of Water-Resources Investigations, **TWRI 05-A1** (3): 545, 16 tab.; Reston.
- FITZPATRICK, R. W. & SELF, P. G. (1997): Iron Oxyhydroxides, Sulfides and Oxyhydroxysulfates as Indicators of Acid Sulfate Weathering Environments. – Advances in Geocology, **30**: 227-240, 2 fig., 2 tab.; Reiskirchen.
- FLACH, D. & HEICK, C. (1986): Untersuchungen zum mikroseismischen Verhalten eines mit Lauge gefluteten Salzbergwerkes am Beispiel Hope. – gsf-Bericht (20): 130-195, 39 fig.; München.
- FLYNN, R. M., SCHNEGG, P.-A., COSTA, R., MALLIN, G. & ZWAHLIN, F. (2005): Identification of zones of preferential groundwater tracer transport using a mobile downhole fluorometer. – Hydrogeol. J., **13** (2): 366-377, 9 fig., 1 tab.; Berlin.
- FOFONOFF, N. P. & MILLARD JR., R. C. (1983): Algorithms for computation of fundamental properties of seawater. – UNESCO Tech. Pap. Mar. Sci., **44**: 53, 13 fig.; Paris.
- FONTAINE, R. C., DAVIS, A. & FENNEMORE, G. G. (2003): The Comprehensive Realistic Yearly Pit Transient Infilling Code (CRYPTIC): A Novel Pit Lake Analytical Solution. – Mine Water Env., **22** (4): 187-193, 8 fig., 2 tab.; Berlin.
- FOREMAN, J. W. (1971): Deep Mine Sealing. – Proceedings, 1st Acid Mine Drainage Workshop, **1**: 19-45, 21 fig.; Athens, Ohio.

- FOREMAN, J. W. (1972): Evaluation of mine sealing in Butler County, Pennsylvania. – Papers Presented Before the Symposium on Coal Mine Drainage Research, **4**: 83-95, 4 fig., 3 tab.; Washington.
- FREEZE, R. A. & CHERRY, J. A. (1979): Groundwater. – 604 p.; Englewood Cliffs (Prentice Hall).
- FREYER, G., GEISSLER, E., HOTH, K. & TRAN, T. N. (1982): Die Altersstellung des Karbonatgesteinshorizonts Rabenstein-Auerswalde und ihre Bedeutung für die Geologie des südöstlichen Schiefermantels des sächsischen Granulitgebirges. – Z. geol. Wiss., **10**: 1403-1424, 6 fig., 1 tab., 4 plates; Berlin.
- FRIED, J. J. (1972): Miscible pollutions of groundwater – a study in methodology. – In: BISWAS, A. K.: Modelling of water resources systems **2**. – p. 520-529, 3 fig.; Montreal (Harvest House).
- FRIESE, K., WENDT-POTTHOFF, K., ZACHMANN, D. W., FAUVILLE, A., MAYER, B. & VEIZER, J. (1998): Biochemistry of iron and sulfur in sediments of an acidic mining lake in Lusatia, Germany. – Water Air Soil Pollut., **108**: 231-247, 8 fig., 2 tab.; Dordrecht.
- FRIGAARD, I. A. & NOUAR, C. (2003): Predicting Transition to Turbulence in Well Construction Flows. – 8th SPE Latin American and Caribbean Petroleum Engineering Conference, **SPE 81150**: 1-9, 6 fig.; Port-of-Spain.
- FROSTAD, S., KLEIN, B. & LAWRENCE, R. W. (2002): Evaluation of Laboratory Kinetic Test Methods for Measuring Rates of Weathering. – Mine Water Env., **21** (4): 183-192, 4 fig., 6 tab.; Berlin.
- FRYDRYCH, D., MARYŠKA, J., MUŽÁK, J. & SEVERÝN, O. (2000): Mathematical Modelling of a Uranium Mine Flooding. – In: HANDLOVICOVÁ, A.: Algoritmy 2000 – Proceedings of Contributed Papers and Posters **15**. – p. 110-119, 6 fig.; Bratislava (Slovenská Technická Univerzita).
- FUGE, R., PEARCE, F. M., PEARCE, N. J. G. & PERKINS, W. T. (1994): Acid mine drainage in Wales and influence of ochre precipitation on water chemistry. – ACS Symposium Series, **550**: 261-274, 3 fig., 4 tab.; Washington.
- FURRER, G., PHILLIPS, B. L., ULRICH, K.-U., PÖTHIG, R. & CASEY, W. H. (2002): The Origin of Aluminum Floccs in Polluted Streams. – Science, **297** (5590): 2245-2247, 3 fig., 1 tab. + 3 fig., 1 tab.; Washington.
- FYSON, A. & NIXDORF, B. (2005): The Lignite Pit Lakes of Lusatia, Germany – Chemistry, Ecology and Acidity Removal Strategies. – p. 1-20, 6 fig., 5 tab.; Princeton (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- FYSON, A., NIXDORF, B., KALIN, M. & STEINBERG, C. E. W. (1997): Mesocosm Studies to Assess Acidity Removal from Acidic Mining Lakes through Controlled Eutrophication. – Ecol. Eng., **10**: 229-245, 5 fig., 1 tab.; Amsterdam.
- FYTAS, K. & BOUSQUET, P. (2002): Silicate Micro-encapsulation of Pyrite to Prevent Acid Mine Drainage. – Bull. Can. Inst. Min. & Met., **95** (1063): 96-99, 6 fig., 3 tab.; Calgary.
- GALINSKY, G., LEISTNER, J., SCHEUERMANN, G., EBERT, S. & FÖRDERVEREIN DREI-BRÜDER-SCHACHT E.V. (2001): Kavernenkraftwerk Drei-Brüder-Schacht – Geschichte und Überlegungen zur Rekonstruktion. – 2nd edn., 72 p., 70 fig., 1 tab.; Freiberg (Saxonia Standortentwicklungs- und -verwaltungsgesellschaft).
- GALLOWAY, D. L. & ERICKSON, J. R. (1985): Tracer test for evaluating nonpumping intraborehole flow in fractured media. – Trans. Amer. Nucl. Soc., **50**: 192-193; La Grange Park.
- GAMMONS, C. H., POULSON, S. R., METESH, J. J., DUAIME, T. E. & HENNE, A. R. (2003): Geochemistry and Isotopic Composition of H₂S-rich Water in Flooded Underground Mine Workings, Butte, Montana, USA. – Mine Water Env., **22** (3): 141-148, 3 fig., 3 tab.; Berlin.
- GATZWEILER, R. & MAGER, D. (1993): Altlasten des Uranbergbaus – Der Sanierungsfall Wismut. – Die Geowissenschaften, **11** (5-6): 164-172, 10 fig., 1 tab.; Berlin.

- GATZWEILER, R., HÄHNE, R. E. M., MEYER, J. & SNAGOVSKY, S. (1997): Prognosis of the Flooding of Uranium mining Sites in East Germany with the Help of numerical Box-Modeling. – Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia, **1**: 57-63, 2 tab.; Ljubljana.
- GATZWEILER, R., JAKUBICK, A. T., MEYER, J., PAUL, M. & SCHREYER, J. (2002): Flooding of the WISMUT mines – learning by doing or applying a comprehensive systematic approach? – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 745-754; Heidelberg (Springer).
- GEBHART, B., JALURIA, Y., MAHAJAN, R. L. & SAMMAKIA, B. (1988): Buoyancy-Induced Flows and Transport. – 1001 p.; Berlin (Springer).
- GEISENHEIMER, P. (1913): Die Wasserversorgung des Oberschlesischen Industriebezirks. – 97 p., 29 fig., 30 tab., 1 Kart.; Kattowitz (Selbstverl. des Oberschlesischen Berg- und Hüttenmännischen Vereins).
- GEISSLER, E., NHUAN, T. T. & STRAUSS, R. (1986): Die Rabensteiner unterirdischen Felsendome – zur Geologie und Geschichte des Schaubergwerkes. – 4th edn., 38 p., 20 fig.; Karl-Marx-Stadt [Chemnitz] (Städtische Museen).
- GELETNEKY, J. W. (2002): Hydrogeologische/Hydrologische Untersuchung einer Prä-Flutungs-situation am Beispiel des Gessentals im ehemaligen ostthüringischen Uranbergbauegebiet. – 262 p., 75 fig., 28 tab., 9 app.; Jena (unpubl. Diss. Universität Jena).
- GELLER, W., KLAPPER, H. & SALOMONS, W. (1998): Acidic Mining Lakes – Acid Mine Drainage, Limnology and Reclamation. – 435 p., 155 fig., 55 tab.; Berlin (Springer).
- GELLER, W., FRIESE, K., HERZSPRUNG, P., KRINGEL, R., SCHIMMELE, M., SCHULTZE, M., WENDT-POTTHOFF, K. & WÖLFL, S. (2000a): Geogene Versauerung von Tagebaurestgewässern. – In: GUDERIAN, R. & GUNKEL, G.: Handbuch der Umweltveränderlichkeiten und Ökotoxikologie **3A** (3.5). – p. 352-380, 8 fig., 2 tab.; Heidelberg (Springer).
- GELLER, W., FRIESE, K., HERZSPRUNG, P., KRINGEL, R. & SCHULTZE, M. (2000b): Limnology of sulphur-acidic mining lakes. II Chemical properties: The main constituents and buffering systems. – Verh. Internat. Verein. Limnol., **27**: 2475-2479, 5 fig., 3 tab.; Stuttgart.
- GENSER, C. (1933): Radioaktive Heilquellen in Deutschland. – Z. dt. geol. Ges., **85**: 482-495, 3 fig., 1 plates; Berlin.
- GEORGAKI, I., DUDENEY, A. W. L. & MONHEMIUS, A. J. (2004): Characterisation of Iron-rich Sludge: Correlations Between Reactivity, Density and Structure. – Miner. Eng., **17** (2): 305-316, 13 fig., 1 tab.; Oxford.
- GEORGESCU, D., STOICA, L., RADULESCU, R. & FILIP, D. (1995): Radium Content Reduction from Radioactive Effluents resulted in Uranium ores Mining and Processing. – Proceedings, Uranium-Mining and Hydrogeology, Freiberg, Germany; GeoCongress, **1**: 197-206, 8 fig., 3 tab.; Freiberg.
- GERTH, A., HEBNER, A., KIESSIG, G. & ZELLMER, A. (2005): Passive Treatment of Minewater at the Schlema-Alberoda Site. – In: MERKEL, B. J. & HASCHE-BERGER, A.: Uranium in the Environment. – p. 409-414, 2 fig.; Heidelberg (Springer).
- GERTSCH, R. E. & BULLOCK, R. L. (1998): Techniques in Underground Mining – Selections From Underground Mining Handbook. – 823 p.; Littleton (Society for Mining, Metallurgy, and Exploration).
- GEYH, M. A. & SCHLEICHER, H. (1990): Absolute Age Determination – Physical and Chemical Dating Methods and Their Application. – 503 p., 146 fig.; Berlin (Springer).

- GIBERT, O., RÖTTING, T., DE PABLO, J., CORTINA, J. L., BOLZICCO, J., CARRERA, J. & AYORA, C. (2004): Metal Retention Mechanisms for the Aznalcóllar permeable reactive Barrier (SW Spain). – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: mine water 2004 – Proceedings International Mine Water Association Symposium **2**. – p. 61-68, 7 fig.; Newcastle upon Tyne (University of Newcastle).
- GIESE, S., GRUBERT, A. & HÜBNER, C. (1994): Ergebnisse geophysikalischer Untersuchungen des Carolinen-Gangzuges (Bergbaurevier Freiamt-Sexau, Mittlerer Schwarzwald) und des darauf umgegangenen historischen Bergbaus. – Abh. geol. Landesamt Baden-Württemberg, **14**: 343-360, 6 fig., 1 tab.; Freiburg.
- GILLMORE, G. K., GRATTAN, J., PYATT, F. B., PHILLIPS, P. S. & PEARCE, G. (2002): Radon, water and abandoned metalliferous mines in the UK: Environmental and Human Health implications. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 65-76, 2 tab.; Heidelberg (Springer).
- GISMAN, S. & SZCZYPA, H. (1982): Trends in Utilization of Mine Waters in the Upper-Silesian Coal Basin. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **D**: 17-23, 1 tab.; Budapest.
- GOETTE, H. (1934): Das Henry-Verfahren zum Klären von Ab- und Schlammwässern. – Technische Blätter – Wochenschr. z. Dtsch. Bergwerksztg., **42**: 683-684; Essen.
- GOETZ, D., ROUSSEL, V. & TINCELIN, E. (1994): Cave Ins Linked to Mine Flooding. – Proceedings, 5th International Mine Water Congress, Nottingham, U.K., **1**: 29-42, 5 fig., 4 tab.; Nottingham.
- GOLDBRUNNER, J. E., RAMSPACHER, P., ZOJER, H., ZÖTL, J. G., MOSER, H., RAUERT, W. & STICHLER, W. (1982): Die Anwendung natürlicher und künstlicher Tracer in einem hochalpinen Magnesitbergbau. – Beitr. Geol. Schweiz – Hydrol., **28** (II): 407-422, 4 fig.; Bern.
- GOMBKÖTÖ, I., KOVÁCS, F., BÖHM, J. & DEBRECZENI, Á. (2002): Removal of Uranium from contaminated Groundwater at a former Uranium Mining Site. – p. 97-102, 7 fig.; Karlsruhe (Proceedings, First Image-Train Cluster-Meeting).
- GOODMAN, A. E., KHALID, A. M. & RALPH, B. J. (1981): Microbial ecology of Rum Jungle II: environmental study of two flooded opencuts and smaller, associated water bodies. – 61 p., 10 fig., 36 tab.; Lucas Heights (Australian Atomic Energy Commission Research Establishment).
- GOSPODARIČ, R. & HABIČ, P. (1976): Underground Water tracing – Investigations in Slovenia 1972-1975. – 309 p., 85 fig., 46 tab., 11 plates; Ljubljana (Institute Karst Research).
- GOTTSCHALK, V. H. & BUEHLER, H. A. (1912): Oxidation of Sulphides. – Econ. Geol., **7**: 15-34, 1 fig., 3 tab.; Lancaster.
- GOURDON, R. & FUNTOWICZ, N. (1995): Bioleaching of metals from industrial contaminated soil using sulphuric acid produced by bacterial activity – a feasibility study. – Soil & Environment, **5**: 1049-1056, 5 fig., 4 tab.; Dordrecht - Boston - London.
- GOVERNMENT OF IRELAND (1997): Sustainable Development – A Strategy for Ireland. – 258 p., 48 fig., 48 tab.; Dublin (Government Publications).
- GRANDCHAMP, C. A. P., SPERLING, E. v. & ALMEIDA, D. C. d. (2001): Águas Claras Mine – A successful dewatering story. – p. 1-8 [CD-ROM], 6 fig., 2 tab.; Belo Horizonte (Proceedings, IMWA Symposium Belo Horizonte).
- GRAPES, T. R. & CONNELLY, R. J. (1998): Evaluation of Mine Water Inflows and associated Impacts at the Olympias mine, Halkidiki, Greece **1**. – p. 13-23, 2 fig.; Johannesburg (Proceedings International Mine Water Association Symposium).
- GRAPS, A. (1995): An Introduction to Wavelets. – IEEE Computational Science & Engineering, **2** (2): 1-18, 8 fig.; Los Alamitos.

- GRAY, D. M. (1970): Handbook on the Principles of Hydrology: A water information center publication. – 691 p.; Ottawa (Canadian Committee for the International Hydrological Decade).
- GRAY, D. M., MCKAY, G. A. & WIGHAM, J. M. (1970): Energy, Evaporation and Evapotranspiration. – In: GRAY, D. M.: Handbook on the Principles of Hydrology (III). – p. 3.1-3.66, 26 fig., 13 tab.; Ottawa (Canadian Committee for the International Hydrological Decade).
- GRIFFIN, B. C., AHLGREN, K., COLEMAN, M., JURGENSMEYER, V., LITTLEFIELD, R., MARTIN, T., MCCAULEY, N., ROBERTS, L., SEYBOLT, D. & SMITH, M. (2000): Final Report – Governor Frank Keating's Tar Creek Superfund Task Force. – 30 p., 16 fig., 11 tab.; Oklahoma City (Office of the Secretary of Environment).
- GRIMES, D. J., FICKLIN, W. H., MEIER, A. L. & MCHUGH, J. B. (1995): Anomalous gold, antimony, arsenic, and tungsten in ground water and alluvium around disseminated gold deposits along the Getchell Trend, Humboldt County, Nevada. – J. Geochem. Explor., **52** (3): 351-371, 13 fig., 1 tab.; Amsterdam-New York.
- GRISAK, G. E. & PICKENS, J. F. (1981): An analytical Solution for solute Transport through fractured Media with Matrix Diffusion. – J. Hydrol., **52** (1-2): 47-57, 7 fig., 1 tab.; Amsterdam.
- GRUNDMANN, G. & MARTINEK, K.-P. (1994): Erzminerale und Gangarten des Bergbaubereiches Schwaz-Brixlegg. – Mineralien Magazin, Lapis, **19** (7-8): 28-40, 44 fig.; München.
- GRÜNEWALD, U. (2001): Water resources management in river catchments influenced by lignite mining. – Ecol. Eng., **17** (2-3): 143-152, 7 fig., 2 tab.; Amsterdam.
- GSF – GESELLSCHAFT FÜR STRAHLEN- UND UMWELTFORSCHUNG (1985): Untersuchungen endlager-relevanter Vorgänge vor, während und nach der Flutung des Kalisalzbergwerkes Hope. 1. Bericht. – gsf-Bericht, **12/85**: 286; Neuherberg.
- GSF – GESELLSCHAFT FÜR STRAHLEN- UND UMWELTFORSCHUNG (1986): Untersuchungen endlager-relevanter Vorgänge vor, während und nach der Flutung des Kalisalzbergwerkes Hope. Statusseminar zum FE-Projekt Hope am 31. Oktober 1985 in Braunschweig. – gsf-Bericht, **20/86**: 286; Neuherberg.
- GULATI, M. S., LIPMAN, S. C. & STROBEL, C. J. (1978): Tritium Tracer Survey at the Geysers. – In: GEOTHERMAL RESOURCE COUNCIL: Geothermal Resource Council Annual Meeting: Geothermal Energy – A Novelty Becomes Resource, Hilo, Hawaii, Jul 25-27 1978 **2**. – p. 237-239, 1 tab.; Davis, Calif. (Geothermal Resource Council).
- GUNDELWEIN, A. (2001): Die Roeder'sche Zeit – Das Ende des "Mittelalters" am Rammelsberg. – In: ROSENECK, R.: Der Rammelsberg – Tausend Jahre Mensch-Natur-Technik **2**. – p. 172-181, 8 fig.; Goslar (Goslarsche Zeitung).
- GÜNTHER, W., EIBNER, C., LIPPERT, A. & PAAR, W. ([ca. 1993]): 5000 Jahre Kupferbergbau Mühlbach am Hochkönig-Bischofshofen. – 396 p.; Bischofshofen (Eigenverlag Marktgemeinde Bischofshofen).
- GURRIERI, J. T. (1998): Distribution of metals in water and sediment and effects on aquatic biota in the upper Stillwater River Basin, Montana. – J. Geochem. Explor., **64**: 83-100, 6 fig., 3 tab.; Amsterdam.
- GUSEK, J. J. (2002): Sulfate-Reducing Bioreactor Design and Operating Issues – Is this the Passive Treatment Technology for your Mine Drainage? – p. 1-14 [CD-ROM], 4 fig.; Park City (Proceedings, Annual Conference – National Association of Abandoned Mine Land Programs).
- GUSEK, J. J. & WILDEMAN, T. R. (2002): Passive Treatment of Aluminum-Bearing Acid Rock Drainage. – Proceedings, West Virginia Surface Mine Drainage Task Force Symposium, **23**: 1-9 [CD-ROM], 3 fig., 1 tab.; Morgantown.
- HÄGE, K. (1995): Methoden und Erfahrungen zur Vermeidung von Umweltauswirkungen im Lausitzer Braunkohlenrevier. – Braunkohle (7): 4-8, 6 fig.; Düsseldorf.

- HAGEN, M. & LANGE, G. (1995): Der Flutungsprozess ehemaliger Uranerzgruben in Ostdeutschland als Sanierungsschwerpunkt der Wismut GmbH. – *Erzmetall*, **48** (11): 790-804, 8 fig.; Weinheim.
- HALLBERG, K. B. & JOHNSON, D. B. (2005): *Mine Water Microbiology*. – *Mine Water Env.*, **24** (1): 28-32, 2 fig., 1 tab.; Berlin.
- HÁMAR, T., D'ALESSANDRO, M. & BIDAGLIO, G. (2002): Legislation on Mining Waste Management in Central and Eastern European Candidate Countries **EUR 20545 EN**. – 196 p.; Ispra (Joint Research Centre of the European Commission).
- HAMILTON, Q. U. I., LAMB, H. M., HALLETT, C. & PROCTOR, J. A. (1997): Passive Treatment Systems for the Remediation of Acid Mine Drainage Wheal Jane, Cornwall, UK. – In: YOUNGER, P. L.: *Minewater Treatment Using Wetlands*. – p. 33-56, 9 fig., 2 tab.; London (Chartered Institution of Water and Environmental Management).
- HAMILTON, R. M., TABERHAM, J., WAITE, R. R. J., CAMBRIDGE, M., COULTON, R. H. & HALLEWELL, M. P. (1994a): The development of a temporary treatment solution for the acid mine water discharge at Wheal Jane. – *Proceedings, 5th International Mine Water Congress*, Nottingham, U.K., **2**: 643-656, 7 fig., 5 tab.; Nottingham.
- HAMILTON, R. M., BOWEN, G. G., POSTLETHWAITE, N. A. & DUSSEK, C. J. (1994b): The abandonment of Wheal Jane, a tin mine in South West England. – *Proceedings, 5th International Mine Water Congress*, Nottingham, U.K., **2**: 543-551, 4 fig., 2 tab.; Nottingham.
- HAMM, V., FABRIOL, R., VAUTE, L., FOURNIQUET, G. & BAUBRON, J.-C. (2005): Modelling the impact of back-filling material on the water quality of flooded iron mines (Lorraine, France). – p. 1-12 [CD-ROM], 7 fig., 2 tab.; Nancy (Proceedings, Post-Mining 2005).
- HAMMACK, R. W., SAMS, J. I., VELOSKI, G. A. & MABIE, J. S. (2003a): Geophysical Investigation of the Sulphur Bank Mercury Mine Superfund Site, Lake County, California. – *Mine Water Env.*, **22** (2): 69-79, 15 fig.; Berlin.
- HAMMACK, R. W., LOVE, E. I., VELOSKI, G. A., ACKMAN, T. E. & HARBERT, W. (2003b): Using Helicopter Electromagnetic Surveys to Identify Environmental Problems at Coal Mines. – *Mine Water Env.*, **22** (2): 80-84, 7 fig.; Berlin.
- HANNEBERG, A. & SCHUSTER, H. (1994): Geschichte des Bergbaus in Schwaz und Brixlegg. – *Mineralien Magazin, Lapis*, **19** (7-8): 13-21, 16 fig., 3 tab.; München.
- HANSEN, C. (1998): Modellierung der hydrogeochemischen Entwicklung saurer Grubenwässer – Flutung der Schwefelkieslagerstätte Einheit bei Elbingerode (Harz). – 137 p., 76 fig., 6 tab.; Clausthal-Zellerfeld (unpubl. Diplomarbeit TU Clausthal).
- HANZLÍK, J. & VYDRA, J. (1985): Liquidation of abandoned Mine Excavations by Flooding. – *Proceedings, 2nd International Mine Water Association Congress*, **2**: 953-965, 6 fig., 2 tab.; Granada.
- HARRIES, J. R. (1990): Acid Drainage from Waste rock Dumps at Mine Sites (Australia and Scandinavia) **ANSTO/E692**. – 30 p., 21 fig., 6 tab.; Lucas Heights (Australian Nuclear Science and Technology Organisation).
- HARRIS, D. (2004): Post-mining regeneration at the Garden of Eden. – *Mining Environmental Management*, **12** (1): 8-11, 3 fig.; London.
- HARVEY, R. W., GEORGE, L. H., SMITH, R. L. & LEBLANC, D. R. (1989): Transport of microspheres and indigenous bacteria through a sandy aquifer; results of natural- and forced-gradient tracer experiments. – *Environ. Sci. Technol.*, **23** (1): 51-56, 5 fig., 3 tab.; Washington.
- HARVEY, R. W., KINNER, N. E., McDONALD, D., METGE, D. W. & BUNN, A. L. (1993): Role of physical heterogeneity in the interpretation of small-scale laboratory and field observations of bacteria, microbial-sized microsphere, and bromide transport through aquifer sediments. – *Water Resour. Res.*, **29** (8): 2713-2721, 2 fig., 2 tab.; Washington.

- HASCHE, A. (2001): Multitracerversuch in einem gefluteten Bergwerk – Vorbereitung, Durchführung, Auswertung. – 99 p., 16 fig., 29 tab., 36 app., 2 app.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- HASCHE, A. & WOLKERSDORFER, CH. (2004): Mine Water Treatment with a Pilot Scale RAPS-System. – Wissenschaftliche Mitteilungen, **25**: 93-99, 8 fig., 1 tab.; Freiberg.
- HASSE, J. (2000): Flutung des Metallerzbergwerkes Meggen. – In: WISMUT GMBH: Proceedings Wismut 2000 – Bergbausanierung Internationale Konferenz. – 9 p., 7 fig.; Chemnitz (Eigenverlag Wismut GmbH).
- HAUSE, D. R. & WILLISON, L. R. (1986): Deep Mine Abandonment Sealing and Underground Treatment to Prelude Acid Mine Drainage. – Proceedings, West Virginia Surface Mine Drainage Task Force Symposium, **7** (Paper 14): 14, 13 fig.; Morgantown.
- HAWKINS, J. W. (1994): Statistical characteristics of coal-mine discharges on western Pennsylvania remining sites. – Water Resour. Bull., **30** (5): 861-869, 5 fig., 3 tab.; Bethesda.
- HAWKINS, J. W. (1995): Characterization and Effectiveness of Remining Abandoned Coal Mines in Pennsylvania. – Report of Investigations – US Bureau of Mines, **RI 9562**: 1-38, 13 fig., 14 tab.; Washington.
- HAWKINS, J. W. (1998): Remining. – In: THE PENNSYLVANIAN DEPARTMENT OF ENVIRONMENTAL PROTECTION: Coal mine drainage prediction and pollution prevention in Pennsylvania (17). – p. 17.1-17.6, 3 fig.; Harrisburg (Pennsylvanian Bureau of Mining and Reclamation).
- HAWKINS, J. W. & ALJOE, W. W. (1992): Pseudokarst Groundwater Hydrologic Characteristics of a Mine Spoil Aquifer. – Mine Water Env., **11** (2): 37-52, 11 fig., 1 tab.; Wollongong.
- HAYEK, E. & WIERER, H. (1951): Über den Arsengehalt von Stollenwässern in der Umgebung Badgasteins. – Tscherma's Min. u. Petr. Mitt., **2** (3): 290-291; Wien.
- HEDIN, R. S. (2003): Recovery of marketable iron oxide from mine drainage in the USA. – Land Contam. Reclam., **11** (2): 93-97, 6 fig., 2 tab.; Richmond.
- HEDIN, R. S., NAIRN, R. W. & KLEINMANN, R. L. P. (1994a): Passive Treatment of Coal Mine Drainage. – Bureau of Mines Information Circular, **IC-9389**: 1-35, 13 fig., 19 tab.; Washington.
- HEDIN, R. S., WATZLAF, G. R. & NAIRN, R. W. (1994b): Passive treatment of acid-mine drainage with limestone. – J. Environ. Qual., **23** (6): 1338-1345, 3 fig., 6 tab.; Madison.
- HEDIN, R. S., STAFFORD, S. L. & WEAVER, T. J. (2005): Acid Mine Drainage Flowing from Abandoned Gas Wells. – Mine Water Env., **24** (2): 104-106, 1 fig., 2 tab.; Berlin.
- HEIDE, B. & HASSE, J. (1997): Stilllegung des Metallerzbergwerks Meggen nach 140 Jahren aktivem Bergbau. – Glückauf, **133**: 233-239, 6 fig., 4 tab.; Essen.
- HEIDE, B. & HASSE, J. (1998): Der Anfang und das Ende des Metallerzbergwerkes Meggen. – Bergbau, **49** (3): 115-122, 6 fig., 4 tab.; Gelsenkirchen.
- HEIDENREICH, H., KLAFFKI, M., KRETZSCHMAR, H. J. & ZIEGERT, H. R. (1991): Strömungsmechanisch-mathematische Simulation eines gefluteten Endlagers. – Kernenergie, **34** (6): 270-275, 7 fig.; Berlin.
- HEITFELD, K.-H., HEITFELD, M., ROSNER, P., SAHL, H. & SCHETELIG, K. (2002): Mine Water Recovery in the Coal Mining District of Aachen – Impacts and Measures to Control Potential Risks. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 1027-1036, 6 fig.; Heidelberg (Springer).
- HENTON, M. P. (1976): Predicting the quality of mine water discharges. – Effluent & Water Treatment Journal, **16** (11): 568, 572, 1 tab.; London.
- HENTSCHEL, T., HRUSCHKA, F. & PRIESTER, M. (2002): Global Report on Artisanal & Small-Scale Mining. – Mining, Minerals and Sustainable Development, **70**: 1-67, 16 fig., 7 tab.; London.

- HERBERT, H.-J. (1989): Geochemische Vorgänge bei der Flutung des Kalisalzbergwerkes Hope – Abschlußbericht des Teilvorhabens Geochemie des FE-Vorhabens Hope. – gsf-Bericht, **5/89**: 62, 4 tab., 72 fig. im Anhang; Neuherberg.
- HERBERT, H.-J. & SANDER, W. (1982): Mineralogisch-chemische Prozesse in laugenerfüllten Schächten des Zechsteinsalinars. – Fortsch. d. Min., **60**: 96-98; Stuttgart.
- HERBERT, H.-J. & SANDER, W. (1989): Verfahren zur verfälschungsfreien Messung und Probennahme von hochkonzentrierten Salzlösungen im Untertagebereich. – Kali und Steinsalz, **10** (4/5): 137-141, 3 fig.; Kassel.
- HERBERT JR., R. B. (1999): MiMi – Sulfide oxidation in mine waste deposits – A review with emphasis on dysoxic weathering. – 45 p., 10 fig., 3 tab.; Luleå (MiMi Print).
- HERSCHY, R. W. (1995): Streamflow measurement. – 2nd edn., 524 p.; London (Spon).
- HERZSPRUNG, P., FRIESE, K., PACKROFF, G., SCHIMMELE, M., WENDT-POTTHOFF, K. & WINKLER, M. (1998): Vertical and Annual Distribution of Ferric and Ferrous Iron in Acidic Mining Lakes. – Acta Hydrochim. Hydrobiol., **26** (5): 253-262, 9 fig., 7 tab.; Weinheim.
- HETTLER, J. (1995): Bergbau und Umwelt in Papua-Neuguinea – Die Ok Tedi-Mine und das Fly River-Flußökosystem. – Berliner geowissenschaftliche Abhandlungen: Reihe A, **176**: 110, 29 fig., 17 tab., 8 app.; Berlin.
- HEWETT, R. (2003): Data Mining for generating Predictive Models of local Hydrology. – Appl. Intell., **19** (3): 157-170, 11 fig.; Dordrecht.
- HIESSLEITNER, G. (1951): Geologischer Gutachtensbericht über den Schwerspatbergbau Großkogel bei Brixlegg **Archiv Montanwerke Brixlegg**. – 26 p., 5 fig., 5 tab., 8 Beil.; Brixlegg (Montanwerke G.m.b.H.).
- HILTSCHER, A. (1999): 140 Jahre Salzbergbau in der Region Anhalt – Ein Überblick. – Der Aufschluss, **Sonderband**: 103-116, 13 fig., 3 tab.; Heidelberg.
- HIMMELSBACH, T. & WENDLAND, E. (1999): Schwermetalltransport in Sandsteinen unter Bedingungen einer hochsalinaren Porenwasserlösung – Laborversuch und Modellierung. – Grundwasser, **4** (3): 103-112; Berlin.
- HIMMELSBACH, T., HÖTZL, H., KÄSS, W., LEIBUNDGUT, C., MAŁOSZEWSKI, P., MEYER, T., MOSER, H., RAJNER, V., RANK, D., STICHLER, W., TRIMBORN, P. & VEULLIET, E. (1992): Fractured Rock – Test site Lindau/Southern Black Forest (Germany). – Steir. Beitr. Hydrogeol., **43**: 159-228, 28 fig., 9 tab.; Graz.
- HISCOCK, K. M. (2005): Hydrogeology – Principles and Practice. – 389 p., 266 fig., 60 tab.; Oxford (Blackwell).
- HJULSTRÖM, F. H. (1935): Studies of the morphological activity of rivers as illustrated by the river Fyris. – Bull. Geol. Ass. Uppsala, **25**: 221-525, 51 fig., 38 tab., 8 plates; Uppsala.
- HODGKINSON, R. A. & ARMSTRONG, A. C. (1996): Field studies of runoff processes on restored land in South Wales and the design of channels for erosion control. – In: ANDERSON, M. G. & BROOKS, S. M.: British Geomorphological Research Group Symposia Series – Advances in hillslope processes **12** (27). – p. 613-633, 11 fig., 6 tab.; Chichester - New York (Wiley).
- HOEHN, E., EIKENBERG, J., FIERZ, T., DROST, W. & REICHLMAYR, E. (1998): The Grimsel Migration Experiment - field injection-withdrawal experiments in fractured rock with sorbing tracers. – J. Contam. Hydrol., **34** (1-2): 85-106, 9 fig., 4 tab.; Amsterdam.
- HÖGLUND, L. O., HERBERT, R., LÖVGREN, L., ÖHLANDER, B., NERETNIEKS, I., MORENO, L., MALMSTRÖM, M. E., ELANDER, P., LINDVALL, M. & LINDSTRÖM, B. (2004): MiMi – Performance Assessment Main Report. – 345 p.; Luleå (MiMi Print).
- HOLGATE, R. (1991): Prehistoric Flint Mines: Shire archaeology **67**. – 56 p.; Princes Risborough (Shire).

- HÖLTING, B. & COLDEWEY, W. G. (2005): Hydrogeologie – Einführung in die Allgemeine und Angewandte Hydrogeologie. – 6th edn., 326 p., 118 fig., 69 tab.; München (Elsevier).
- HOPKINS, W. B. & SIMPSON, T. A. (1960): Montana Earthquakes Noted in Pennsylvania Mine-Water Pools. – Trans. Amer. Geophys. Union, **41** (2): 435-436; Washington.
- HORNE, R. N., JOHNS, R. A., ADAMS, M. C., MOORE, J. N. & STIGER, S. G. (1987): The use of tracers to analyze the effects of reinjection into fractured geothermal reservoirs. – p. 37-52; Washington (Proceedings 5th geothermal program review).
- HOSKIN, W. (2003): Abandoned Mine Sites: Problems, Issues and Options. – In: PUURA, E., MARMO, L. & D'ALESSANDRO, M.: Workshop on Mine and Quarry Waste – The Burden from the Past. – p. 53-55; Luxembourg (Office for Official Publications of the European Communities).
- HOSKIN, W. M. A. (2000): Mine Closure – The 21st Century Approach. – In: VILLAS BÔAS, R. C. & BARRETO, M. L.: Mine Closure in Iberoamerica. – p. 109-116; Rio de Janeiro (CYTED – IMACC – UNIDO).
- HÖTZL, H. & WERNER, A. (1992): Tracer Hydrology – 6th International Symposium on Water Tracing. – 464 p.; Rotterdam (Balkema).
- HOUNSLOW, A. W. (1995): Water Quality Data – Analysis and Interpretation. – 397 p., 88 fig., 73 tab.; London (Lewis Publishers).
- HOUSTON, K. S., MILIONIS, P. N., EPPLEY, R. L., HARRINGTON, J. M. & HARRINGTON, J. G. (2005): Field Demonstration of In-Situ Treatment and Prevention of Acid Mine Drainage in the Abandoned Tide Mine, Indiana County, Pennsylvania. – Proceedings, West Virginia Surface Mine Drainage Task Force Symposium, **26**: 1-9, 6 fig.; Morgantown.
- HUDSON, N. W. (1993): Field measurement of soil erosion and runoff: Food and Agriculture Organization of the United Nations soils bulletin **68**. – 139 p.; Rome (Food and Agriculture Organization of the United Nations).
- HUNT, D. J. & REDDISH, D. J. (1997): The stability implications of groundwater recharge upon shallow, abandoned mineworkings in the UK. – In: MARINOS, P. G., KOUKIS, G. C., TSIAMBAOS, G. C. & STOURNARAS, G. C.: Engineering geology and the environment **3**. – p. 2425-2430, 3 fig.; Rotterdam (Balkema).
- HUNT, L. E. & HOWARD, A. G. (1994): Arsenic Speciation and Distribution in the Carnon Estuary Following the Acute Discharge of Contaminated Water from a Disused Mine. – Mar. Pollut. Bull., **28** (1): 33-38, 5 fig., 1 tab.; Oxford-New York.
- INTEGRATED GEOTECHNICAL ENGINEERING SERVICES SPECIALISTS IN UNSATURATED ZONE HYDROLOGY (2003): Evaluation of the Long-Term Performance of Dry Cover Systems Final Report. – Für: INTERNATIONAL NETWORK FOR ACID PREVENTION. – 45 p., 15 fig., 4 tab., 9 app.; Saskatoon (Integrated Geotechnical Engineering Services Specialists in Unsaturated Zone Hydrology).
- INTERNATIONAL INSTITUTE FOR ENVIRONMENT AND DEVELOPMENT (2002): Mining for the Future – Appendix C: Abandoned Mines Working Paper. – Mining, Minerals and Sustainable Development, **28**: C-1-C-20, 3 tab.; London.
- JACKSON, C. B. & HACH COMPANY (2004): Report on the Validation of Proposed EPA Method 360.3 (Luminescence) for the Measurement of Dissolved Oxygen in Water and Wastewater. – 42 p., 14 fig., 3 tab.; Loveland (Hach Company).
- JACOBI, O. & EVERLING, G. (1981): Praxis der Gebirgsbeherrschung. – 2nd edn., 576 p.; Essen (Glückauf).
- JACOBS, M. L. & DENHAM, R. (2001): Stormwater Management and the Limited Discharge System of the Batu Hijau Mine. – p. 1-10 [CD-ROM], 1 fig., 3 tab., 3 Fot.; Belo Horizonte (Proceedings, IMWA Symposium Belo Horizonte).

- JACOBS, M. L. & WHITE, T. (2001): Sediment Control in a Tropical, High Rainfall Environment. – p. 1-11 [CD-ROM], 4 fig., 4 tab., 4 photographs; Belo Horizonte (Proceedings, IMWA Symposium Belo Horizonte).
- JÄGER, B., OBERMANN, P. & WILKE, F. L. (1990): Studie zur Eignung von Steinkohlebergwerken im rechtsrheinischen Ruhrkohlebezirk zur Untertageverbringung von Abfall- und Reststoffen. – In: LANDESAMT FÜR WASSER UND ABFALL NORDRHEIN-WESTFALEN. – 628 p.; Düsseldorf (LWA Studie).
- JAKUBICK, A. T., JENK, U. & KAHNT, R. (2002): Modelling of mine flooding and consequences in the mine hydrogeological environment: flooding of the Königstein mine, Germany. – *Environ. Geol.*, **42** (2-3): 222-234, 11 fig., 6 tab.; Heidelberg.
- JAMBOR, J. L., BLOWES, D. W. & RITCHIE, A. I. M. (2003): Environmental Aspects of Mine Wastes. – In: RAESIDE, R.: Short Course Series Volume **31**. – 430 p.; Waterloo, Ontario (Mineralogical Association of Canada).
- JANISCHEWSKIJ, E. M. & KONSTANTINOW, W. M. (1962): Der Einfluß tektonischer und lithologischer Faktoren auf die Uranvererzung im Erzgebirge. – *Z. Angew. Geol.*, **8** (3): 124-128, 5 fig.; Berlin.
- JANNECK, E. & KRÜGER, H.-G. (1999): Sanierung von Bergbauwässern aus Altablagerungen des Schieferbergbaus – Pilotanlage für die Behandlung saurer Bergbauwässer. – In: DEUTSCHE BUNDESSTIFTUNG UMWELT FKZ 04600. – 75 p., 27 fig., 12 tab., 5 app.; Freiberg (unpubl. Ergebnisbericht G.E.O.S. Freiberg).
- JARVIS, A. P. & YOUNGER, P. L. (1999): Design, Construction and Performance of a Full-Scale Compost Wetland for Mine-Spoil Drainage Treatment at Quaking Houses. – *Water Environ. Manage.*, **13** (5): 313-318, 4 fig., 2 tab.; London.
- JARVIS, A. & ENGLAND, A. (2002): Operational and treatment performance of an unique Reducing and Alkalinity Producing System (RAPS) for acidic leachate remediation in Lancashire, UK. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 1019-1026, 4 fig., 1 tab.; Heidelberg (Springer).
- JAVERI, V. (1995): Orientierende Analysen zur Naturkonvektion und Dispersion in porösen Medien mit dem Rechenprogramm TOUGH2. – Schriftenreihe Reaktorsicherheit und Strahlenschutz, **BMU-1995-416**: 121, 50 fig.; Bonn.
- JENK, U., SCHREYER, J. & KLINGER, C. (2003): Fe(0)/lignitic coal: An efficient and mechanically stable reactive material for purification of water containing heavy metals, radionuclides, and nitroaromatics. – *Environ. Sci. Technol.*, **37** (3): 644-651, 9 fig., 9 tab.; Washington.
- JENK, U., ZIMMERMANN, U. & ZIEGENBALG, G. (2005): The use of BaSO₄ supersaturated solutions for in-situ immobilization of heavy metals in the abandoned Wismut GmbH uranium mine at Königstein. – In: MERKEL, B. J. & HASCHE-BERGER, A.: Uranium in the Environment. – p. 721-727, 5 fig.; Heidelberg (Springer).
- JENSEN, E. B. (1983): Mine Water Used to Heat Ventilation Air at Henderson Molybdenum Mine. – *Mining Engineering*, **35** (1): 17-20, 4 fig., 1 tab.; Littleton.
- JESSOP, A. M. (1998): Geothermal Energy in Canada. – *Geosci. Can.*, **25** (1): 33-41, 8 fig.; Waterloo.
- JESSOP, A. M., MACDONALD, J. K. & SPENCE, H. (1995): Clean Energy from Abandoned Mines at Springhill, Nova Scotia. – *Energy Sources*, **17** (1): 93-106, 5 fig., 2 tab.; Bristol.
- JESTER, W. A. & RAUPACH, D. C. (1987): A Survey of applications of non-radioactive but neutron activatable groundwater tracers. – In: IAEA: Isotope techniques in water resources development. – p. 623-633, 4 fig.; Wien (International symposium on the use of isotope techniques in water resources development).
- JIRIES, A., EL-HASAN, T., AL-HWEITI, M. & SEILER, K.-P. (2004): Evaluation of the Effluent Water Quality Produced at Phosphate Mines in Central Jordan. – *Mine Water Env.*, **23** (3): 133-137, 4 fig., 4 tab.; Berlin.

- JOB, B. (1987): Inrushes at British collieries: 1851 to 1970. – *Colliery Guardian*, **235** (5, 6): 192-199; 232-235, 8 fig., 2 tab.; London.
- JOBST, W., RENTZSCH, W., SCHUBERT, W. & TRACHBROD, K. (1994): *Bergwerke im Freiburger Land*. – 2nd edn., 227 p., 167 fig., 38 tab.; Freiberg (Medienzentrum der TU Bergakademie Freiberg).
- JOHNSON, D. B. (2003): Chemical and microbiological characteristics of mineral spoils and drainage waters at abandoned coal and metal mines. – *Water Air Soil Pollut.*, **Focus 3**: 47-66, 2 fig., 3 tab.; Dordrecht.
- JOHNSON, D. B. & HALLBERG, K. B. (2002): Pitfalls of passive mine water treatment. – *Re/Views in Environmental Science & Bio/Technology*, **1**: 335-343, 4 fig., 1 tab.; Amsterdam.
- JOHNSON, K. L. & YOUNGER, P. L. (2000): Abandonment of Frazer's Grove Fluorspar Mine, North Pennines, UK: Prediction and observation of Water level and chemistry changes after closure. – p. 271-279, 4 fig.; Ustron (Proceedings, 7th International Mine Water Association Congress).
- JOHNSON, K. L. & YOUNGER, P. L. (2002): Hydrogeological and geochemical consequences of the abandonment of Frazer's Grove carbonate hosted Pb/Zn fluorspar mine, North Pennines, UK. – *Spec. Publ. – Geol. Soc. London*, **198**: 347-363, 13 fig., 2 tab.; London.
- JOHNSTON, D. (2004): A metal mines strategy for Wales. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: *mine water 2004 – Proceedings International Mine Water Association Symposium 1*. – p. 17-23, 1 fig., 2 tab.; Newcastle upon Tyne (University of Newcastle).
- JONES, S. G. & ELLIS, D. V. (1995): Deep-water STD at the Misima Gold and Silver Mine, Papua, New Guinea. – *Mar. Geores. Geotechnol.*, **13** (1-2): 183-200, 11 fig., 2 tab.; London.
- JORDAN, G., D'ALESSANDRO, M., HAMOR, T., LIEDEKERKE, M. V., PANAGOS, P., PUURA, E., SOMMER, S. & VIJDEA, A.-M. (2004): Mining, mining waste and related environmental issues: problems and solutions in Central and Eastern European Candidate Countries **EUR 20868 EN**. – 207 p.; Ispra (Joint Research Centre of the European Commission).
- JORDAN, H. & WEDER, H.-J. (1995): *Hydrogeologie – Grundlagen und Methoden*. – 2nd edn., 409 p., 427 fig., 178 tab., 3 Kart; Stuttgart (Enke Verlag).
- JOVANOVIĆ, N. Z., ANNANDALE, J. G., CLAASSENS, A. S., LORENTZ, S. A. & TANNER, P. D. (2001): Modeling irrigation with gypsiferous Mine Water – a case study in Botswana. – *Mine Water Env.*, **20** (2): 65-72, 2 fig., 4 tab.; Berlin.
- JUNG, H. G. (2002): *Sekundäre Prozesse in Bergbauhalden und Aufbereitungsabgängen*. – 202 p., 148 fig., 25 tab., 18 app.; Hannover (Bundesanstalt für Geowissenschaften und Rohstoffe).
- JUNKER, R., KRAUSE, H., SCHUMANN, K. & SIEMROTH, J. (1991): Sekundärminerale des Neudorf-Straßberger-Gangzuges im Unterharz. – *Der Aufschluss*, **42** (2): 95-100, 6 fig.; Heidelberg.
- JURJOVEC, J., BLOWES, D. W. & PTACEK, C. J. (1995): Acid Neutralization in Mill Tailings and the Effect of Natrojarosite Addition. – *Proceedings, Sudbury '95 – Mining and the Environment*, **I**: 29-38, 5 fig., 1 tab.; Sudbury, Ontario, Canada.
- KADEN, S. & SCHRAMM, M. (2000): Control model Spree/Schwarze Elster – a tool to optimise rehabilitation of water resources in the Lusatian mining district. – *Landsc. Urban Plan.*, **51** (2-4): 101-108, 3 fig., 2 tab.; Amsterdam.
- KADEN, S. (1997): Water Resources Modeling for Decision Support in Open-pit Lignite Mining Areas. – *Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia*, **2**: 385-395, 6 fig., 2 tab.; Ljubljana.
- KAHMANN, H.-J. & HEINRICH, K. (1998): Zur Sauerwasserproblematik bei der Verwahrung der Südharzer Flußspatgrube Straßberg [Vortragsmanuskript]. – *Proceedings, Deutsche Rohstoff- und Metalltage 1998*: 56, 16 fig.; Clausthal-Zellerfeld.
- KALIN, M. (2004): Passive mine water treatment – The correct approach. – *Wissenschaftliche Mitteilungen*, **25**: 107-112; Freiberg.

- KALIN, M., SMITH, M. P. & LANDRY, G. (1994): The Application of Ecological Engineering to Acid Coal Seepages in Eastern Canada. – Proceedings, International Land Reclamation and Mine Drainage Conference, **2**: 296-302, 8 fig., 1 tab.; Pittsburgh.
- KALKA, H., MÄRTEN, H. & MÜNZE, R. (2002): Flooding and Post-flooding Scenarios – Dynamics and Geochemistry. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 1037-1044, 7 fig.; Heidelberg (Springer).
- KALKA, H., MÄRTEN, H. & KAHNT, R. (2005): Dynamical Models for Uranium Leaching – Production and Remediation Cases. – In: MERKEL, B. J. & HASCHE-BERGER, A.: Uranium in the Environment. – p. 235-245, 7 fig.; Heidelberg (Springer).
- KARAKAS, G., BROOKLAND, I. & BOEHRER, B. (2003): Physical characteristics of Acidic Mining Lake 111. – *Aquat. Sci.*, **65** (3): 297–307, 11 fig., 2 tab.; Basel.
- KARATHANASIS, A. D. & JOHNSON, C. M. (2003): Metal Removal Potential by Three Aquatic Plants in an Acid Mine Drainage Wetland. – *Mine Water Env.*, **22** (1): 22-30, 7 fig., 1 tab.; Berlin.
- KARRENBURG, H. (1981): Hydrogeologie der nichtverkarstungsfähigen Festgesteine. – 284 p., 83 fig.; Wien u.a. (Springer).
- KÄSS, W. (1964): Die unmittelbare Bestimmung von Uranin-Spuren bei Färbeversuchen. – *Steir. Beitr. Hydrogeol.*, **15/16**: 37-65, 7 fig., 6 tab.; Graz.
- KÄSS, W. (1998): Tracing Technique in Geohydrology. – 581 p., 270 fig., 40 tab., 8 Plat.; Rotterdam (Balkema).
- KÄSS, W. (2004): Geohydrologische Markierungstechnik. – In: MATTHESS, G.: Lehrbuch der Hydrogeologie **9**. – 2nd edn., 557 p., 239 fig., 43 tab., 8 plates; Berlin (Borntraeger).
- KECOJEVIC, V. J., MRUGALA, M. J., SIMIC, R. D. & STANKOVIC, R. M. (2003): A Computer-Aided System for Design of Drainage Facilities in Surface Mining. – *Mine Water Env.*, **22** (3): 149-154, 10 fig.; Berlin.
- KEGEL, K. (1950): Bergmännische Wasserwirtschaft einschließlich Grundwasserkunde, Wasserversorgung und Abwasserbeseitigung. – In: KEGEL, K., SPACKELER, G. & RAMMLER, E.: Berg- und Aufbereitungstechnik, Band III Geologische und technologische Grundlagen des Bergbaues **III** (Teil 1). – 3rd edn., 374 p., 240 fig., 49 tab.; Halle (Saale) (Knapp).
- KELLEY, D. E., FERNANDO, H. J. S., GARGETT, A. E., TANNY, J. & ÖZSOY, E. (2003): The Diffusive Regime of Double-Diffusive Convection. – *Prog. Oceanogr.*, **56**: 461-481, 7 fig.; Oxford.
- KELLNER, R., MERMET, J.-M., OTTO, M. & WIDMER, H. M. (1998): Analytical Chemistry. – 916 p.; Weinheim (Wiley-VCH).
- KELLY, D. P. & WOOD, A. P. (2000): Reclassification of some species of *Thiobacillus* to the newly designated genera *Acidithiobacillus* gen. nov., *Halothiobacillus* gen. nov. and *Thermithiobacillus* gen. nov. – *Int. J. Syst. Evol. Microbiol.*, **50**: 511-516, 1 fig., 1 tab.; Reading.
- KEMP, P. & GRIFFITHS, J. (1999): Quaking Houses – Art, science and the community: a collaborative approach to water pollution. – 142 p., 41 fig.; Charlbury (Jon Carpenter).
- KENNEDY, C. (2002): Alternatives for the Reclamation of Surface Mined Lands. – In: MUDROCH, A., STOTTMEISTER, U., KENNEDY, C. & KLAPPER, H.: Remediation of Abandoned Surface Coal Mining Sites. – p. 37-56, 8 tab.; Heidelberg (Springer).
- KEPLER, D. A. & MC CLEARY, E. C. (1994): Successive Alkalinity-Producing Systems (SAPS) for the Treatment of Acid Mine Drainage. – Proceedings, International Land Reclamation and Mine Drainage Conference, **1**: 195-204; Pittsburgh.
- KESTER, D. R., BYRNE, R. H. & LIANG, Y.-J. (1975): Redox Reactions and Solution Complexes of Iron in Marine Systems. – ACS Symposium Series, **18**: 56-79, 6 fig., 9 tab.; Washington.

- KICKUTH, R. (1977): Degradation and incorporation of nutrients from rural wastewaters by plant hydrosphere under limnic conditions. – Comm. of the Europ. Communitite – Rapports, **EUR 5672e**: 335-343, 2 fig., 2 tab.; London.
- KIESSIG, G. & HERMANN, E. (2000): Nachfolgelösungen für die konventionelle Behandlung kontaminierter Wässer. – In: WISMUT GMBH: Proceedings Wismut 2000 – Bergbausanierung Internationale Konferenz. – 9 p., 10 fig., 2 tab.; Chemnitz (Eigenverlag Wismut GmbH).
- KIESSIG, G., KUNZE, C., KÜCHLER, A., ZELLMER, A., MEYER, J. & KALIN, M. (2004): Kostengünstige passive Nachsorgelösung mit einem Constructed Wetland auf der Grundlage von Prognosen der Entwicklung des Grubenwassers der Grube Pöhla. – Wissenschaftliche Mitteilungen, **25**: 113-118, 5 fig., 1 tab.; Freiberg.
- KIMBALL, B. A., RUNKEL, R. L., BENCALA, K. E. & WALTON-DAY, K. (1999): Use of tracer injections and synoptic sampling to measure metal loading from acid mine drainage. – Water resources investigations report, **WRI 99-4018A** (1): 31-36, 6 fig.; Reston.
- KIMBALL, B. A., RUNKEL, R. L. & GERNER, L. J. (2001): Quantification of mine-drainage inflows to Little Cottonwood Creek, Utah, using a tracer-injection and synoptic-sampling study. – Environ. Geol., **40** (11-12): 1390-1404, 11 fig., 1 tab.; Heidelberg.
- KIMBALL, B. A., RUNKEL, R. L., WALTON-DAY, K. & BENCALA, K. E. (2002): Assessment of metal loads in watersheds affected by acid mine drainage by using tracer injection and synoptic sampling: Cement Creek, Colorado, USA. – Appl. Geochem., **17** (9): 1183-1207, 10 fig., 5 tab.; Oxford.
- KINDERMANN, L. (1998): Kontrolle geochemischer Parameter beim Wiedereinbau von Reststoffen in ein stillgelegtes Bergwerk. – Wissenschaftliche Mitteilungen, **7**: 196-201, 5 fig.; Freiberg.
- KINDSVATER, C. E. & CARTER, R. W. (1959): Discharge Characteristics of Rectangular Thin-Plate Weirs. – Trans. American Society of Civil Engineers, **124**: 772-801, 11 fig., 1 tab.; Reston.
- KING, C. Y., KING, B. S. & EVANS, W. C. (1996): Spatial radon anomalies on active faults in California. – Appl. Geochem., **11** (4): 497-510, 15 fig., 1 tab.; Oxford.
- KINZELBACH, W. & RAUSCH, R. (1995): Grundwassermodellierung – Eine Einführung mit Übungen. – 283 p., 223 fig., 15 tab., 2 Disk; Berlin Stuttgart (Borntraeger).
- KIPKO, E. Y., SPICHAK, Y. N., POLOZOV, Y. A., KIPKO, A. E. & HEPNAR, P. (1993): Grouting of Old Flooded Workings at M. Mayerova Mine in Czechoslovakia. – Mine Water Env., **12** (1-4): 21-26, 1 fig., 2 tab.; Wollongong.
- KIRBY, C. S. & CRAVOTTA, C. A. I. (2005a): Net alkalinity and net acidity 2: Practical considerations. – Appl. Geochem., **20** (10): 1920-1940, 8 fig., 9 tab.; Amsterdam.
- KIRBY, C. S. & CRAVOTTA, C. A. I. (2005b): Net alkalinity and net acidity 1: Theoretical considerations. – Appl. Geochem., **20** (10): 1920-1940, 11 fig., 2 tab.; Amsterdam.
- KIRSCHNER, F. E. & WILLIAMS, R. E. (1993): Hydrogeological evaluation of an in-situ leach cell in abandoned mine and mill waste at the Kellogg, Idaho Superfund site. – p. 77-87, 6 fig.; Jackson Hole (International Biohydrometallurgy Symposium).
- KLAPPER, H. & GELLER, W. (2001): Water Quality Management of Mining Lakes – a New Field of Applied Hydrobiology. – Acta Hydrochim. Hydrobiol., **29** (6-7): 363-374, 7 fig., 5 tab.; Weinheim.
- KLAPPER, H. & SCHULTZE, M. (1996): Geogene Versauerung von Bergbaurestseen – Ursachen, Folgen, Möglichkeiten zur Schadensbegrenzung. – In: ARBEITSGRUPPE DES GBL-GEMEINSCHAFTSVORHABENS: GBL Gemeinschaftsvorhaben – Vortragsband des 2. GBL-Kolloquiums vom 6.-8. März 1996 in Freiberg – Methoden und Ergebnisse **3**. – p. 121-131, 12 fig.; Stuttgart (Schweizerbart).

- KLAPPER, H., GELLER, W. & SCHULTZE, M. (1996): Abatement of acidification in mining lakes in Germany. – *Lakes and Reservoirs Research and Management*, **2** (1-2): 7-16, 6 fig., 3 tab.; Carlton.
- KLAPPERICH, H., MEIER, G., SROKA, A., TONDERA, D. & BUSCH, W. (2001): 1. Altbergbau-Kolloquium – Tagungsband. – 212 p.; Essen (Glückauf).
- KLEINMANN, R. L. P. (1990): Acid Mine Water Treatment using Engineered Wetlands. – *Int. J. Mine Water*, **9** (1-4): 269-276; Castro Verde.
- KLIMANT, I., MEYER, V. & KOHLS, M. (1995): Fibre-optic oxygen microsensors, a new tool in aquatic biology. – *Limnol. Oceanogr.*, **40** (6): 1159-1165, 5 fig., 1 tab.; Waco.
- KLINGER, C., HANSEN, C., RÜTERKAMP, P. & HEINRICH, H. (2000a): In situ tests for interactions between acid mine water and ferrihydrite sludge in the pyrite mine “Elbingerode” (Harz Mts.; Germany). – p. 137-145, 6 fig., 1 tab.; Ustron (Proceedings, 7th International Mine Water Association Congress).
- KLINGER, C., JENK, U. & SCHREYER, J. (2000b): Investigation of efficacy of reactive materials for reduction of pollutants in acid mine water in the former uranium mine of Königstein (Germany). – *Proceedings, 7th International Mine Water Association Congress*: 292-298, 1 fig., 2 tab.; Ustron.
- KLINGER, C., JENK, U. & SCHREYER, J. (2002): Processes in passive mine water remediation with zero-valent iron and lignite as reactive materials. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: *Uranium in the Aquatic Environment*. – p. 569-576, 7 fig., 2 tab.; Heidelberg (Springer).
- KOETTER, G. (2001): *Bergbau im Muttental – Geologie und Geschichte des Wittener Bergbauwanderwegs*. – 198 p., 108 fig.; Witten (Wöhrle).
- KOLITSCH, S., JUNGHANS, M., KLEMM, W. & TICHOMIROVA, M. (2001): Der Flutungsraum des Grubenfeldes Freiberg – Hydrochemie, Isotopenchemie und Hydraulik. – *Wissenschaftliche Mitteilungen*, **18**: 14-26, 10 fig., 4 tab.; Freiberg.
- KOLUPAILA, S. (1961): *Bibliography of hydrometry*. – 975 p.; Notre Dame (University of Notre Dame Press).
- KÖNIG, C. & BLÖMER, C. (1999): Berechnung von Temperatur- und dichteabhängiger Strömung in gefluteten Schächten. – Für: DMT-GESELLSCHAFT FÜR FORSCHUNG UND PRÜFUNG MBH GESCHÄFTSBEREICH GUC ESSEN. – 37 p., 17 fig., 4 tab.; Bochum (GKW-Ingenieurgesellschaft mbH).
- KÖRTZINGER, A., SCHIMANSKI, J. & SEND, U. (2005): High-quality oxygen measurements from profiling floats: A promising new techniques. – *J. Atmos. Ocean. Technol.*, **22** (3): 302-308, 7 fig.; Boston.
- KOSCHORRECK, M., HERZSPRUNG, P., WENDT-POTTHOFF, K., LORKE, A., GELLER, W., LUTHER, G., ELSNER, W. & MÜLLER, M. (2002): An In-lake Reactor to Treat an Acidic Lake: the Effect of Substrate Overdosage. – *Mine Water Env.*, **21** (3): 137-149, 10 fig., 3 tab.; Berlin.
- KOSTENBADER, P. D. & HAINES, G. F. (1970): High-density sludge treats acid mine water. – *Coal Age* (September): 90-97, 5 fig., 3 tab.; New York.
- KRANAWETTREISER, J. (1989): Dichteströmungen. – In: BOLLRICH, G. & PREISLER, G.: *Technische Hydromechanik* **2** (6). – 1st edn., S. 328-350, 9 fig., 1 tab.; Berlin (VEB Verlag für Bauwesen).
- KRANJC, A. (1997): Karst Hydrogeological Investigations in South-Western Slovenia. – *Acta Carsologica*, **26** (1): 1-388, 188 fig., 83 tab.; Ljubljana.
- KRAUT, K. H. & BONDAREVA, O. (2000): Anwendung von Flockungsverfahren bei der Regenwasserbehandlung. – 126 p., 43 fig., 27 tab., 3 app.; Stuttgart (unpubl. Report Universität Stuttgart – FZKA-BWPLUS).

- KREIBICH, M. & EISBEIN, M. (ca. 2003): Georgius Agricola und die Ehrenfriedersdorfer Radpumpe. – 34 p., 24 fig.; Ehrenfriedersdorf (Zweckverband Sächsisches Industriemuseum Zinngrube Ehrenfriedersdorf).
- KRENTZ, O., WITTHAUER, B., MITTAG, R., NOVAK, E., WENDT, S., BUCHHOLZ, P., KORN, M., FUNKE, S., FORKMANN, B., DOCEKAL, A., DONNER, F., HEINICKE, J. & TEMMLER, T. (2004): Materialien zur Geologie – Erdbebenbeobachtung im Freistaat Sachsen – Zweijahresbericht 2002–2003. – 54 p., 35 fig.; Dresden (Sächsisches Landesamt für Umwelt und Geologie).
- KRISCHKER, A. G. (1990): Die Baryt-Fahlerz-Lagerstätte St. Gertraudi/Brixlegg. – 206 p.; Innsbruck (unpubl. Diploma Thesis Univ. Innsbruck).
- KROKER, E. & FARRENKOPF, M. (1999): Grubenunglücke im deutschsprachigen Raum: Veröffentlichungen aus dem Deutschen Bergbau-Museum Bochum **79**. – 2nd edn., 653 p.; Bochum (Deutsches Bergbau-Museum).
- KROLL, A., BETLEM, G., BRANS, E., GETLIFFE, K., GROEN, F., SANTAMARÍA, L., LUCAS, M. & LÓPEZ, B. (2002a): Environmental Regulation of Mine Waters in the European Union – D 4 European Policies and Mine Waters: ERMITE Report – EU R&D Project EVK1-CT-2000-00078. – 189 p., 5 fig., 8 tab., 7 app.; Sevilla (Institute for Prospective Technological Studies).
- KROLL, A., AMEZAGA, J. M., YOUNGER, P. L. & WOLKERSDORFER, CH. (2002b): Regulation of Mine Waters in the European Union: Contribution of Scientific Research to Policy Development. – *Mine Water Env.*, **21** (4): 193-200, 1 tab.; Berlin.
- KRUSE, N. A. S. & YOUNGER, P. L. (2005): Computational methods for acid mine drainage management: decision-making for post-closure decision making. – In: *MINE WATER 2005 – MINE CLOSURE*: Loredó, Jorge; Pendás, Fernando. – p. 427-433, 2 fig., 3 tab.; Oviedo (University of Oviedo).
- KUGLER, J. & WEYER, J. (2002): Der Einfluß des Hochwassers am 12.-14. 08. 2002 auf Anlagen des Montanwesens im Freiburger Revier – Größenordnung potentieller Gefahren und bergmännische Aufgaben nach dem Verbruch des Rothschnöberger Stollens. – 37 p., 24 fig.; Freiberg (Eigenverlag).
- KUNNERT, H. (1974): Bergbauwissenschaft und technische Neuerungen im 18. Jahrhundert – Die “Anleitung zu der Bergbaukunst” von Chr. Tr. Delius (1773). – In: MITTERAUER, M.: Österreichisches Montanwesen – Produktion, Verteilung, Sozialformen. – p. 181-198, 1 tab.; München (Oldenburg).
- KUNZE, C. & KÜCHLER, A. (2003): Passive biological treatment systems of mine waters at WISMUT sites. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 187-199, 10 fig., 1 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- KUSCHKA, E. & FRANZKE, H. J. (1974): Zur Kenntnis der Hydrothermalite des Harzes. – *Z. geol. Wiss.*, **2** (12): 1417-1436, 3 fig.; Berlin.
- KUYUMCU, M. & HARTWIG, H.-J. (1998): Aufgaben der GVV – Gesellschaft zur Verwahrung und Verwertung von stillgelegten Bergwerksbetrieben mbH. – *Bergbau*, **49** (1): 18-22, 5 fig., 1 tab.; Gelsenkirchen.
- KWAKWA, K. A. (1989): Tracer results. – In: PARKER, R. H.: *Hot dry rock geothermal energy; Phase 2B final report of the Camborne School of Mines Project* (7). – p. 1037-1099, 18 fig., 3 tab.; Oxford (Pergamon Press).
- KWONG, Y. T. J. & VAN STEMPOORT, D. R. (1994): Attenuation of Acid Rock Drainage in a Natural Wetland System. – In: ALPERS, C. N. & BLOWES, D. W.: *Environmental geochemistry of Sulfide Oxidation* (25). – p. 382-392, 3 fig., 2 tab.; New York (American Chemical Society).
- LACHMAR, T. E. (1994): Application of fracture-flow hydrogeology to acid-mine drainage at the Bunker Hill Mine, Kellogg, Idaho. – *J. Hydrol.*, **155** (2): 125-149, 11 fig., 3 tab.; Amsterdam.

- LADWIG, K. J., ERICKSON, P. M., KLEINMANN, R. L. P. & POSLUSZNY, E. T. (1984): Stratification in Water Quality in Inundated Anthracite Mines, Eastern Pennsylvania. – Report of Investigations – US Bureau of Mines, **8837**: 35, 19 fig., 8 tab.; Washington.
- LAI, R. W. M. (1980): Liquid membrane metal extraction process for recovering copper from mine water (Abstract). – Journal of Metals, **32** (12): 49; New York.
- LAINE, D. M. & DUDENEY, A. W. L. (2000): Bullhouse mine-water project. – Trans. Inst. Min. Metall. Sect. A, **109**: A224-A227, 3 fig.; London.
- LAINE, D. M. & JARVIS, A. P. (2003): Engineering design aspects of passive in situ remediation of mining effluents. – Land Contam. Reclam., **11** (2): 113-125, 2 fig., 1 tab.; Richmond.
- LAKE, P. S., COLEMAN, D., MILLS, B. & NORRIS, R. (1977): A Reconnaissance of Pollution of the King River in the Comstock-Crotty Area, Western Tasmania. – In: BANKS, M. R. & KIRKPATRICK, J. B. – p. 157-173, 4 fig., 4 tab.; Hobart (Royal Society of Tasmania).
- LANDAU, L. D. & LIFSCHITZ, E. M. (1991): Hydrodynamik. – In: ZIESCHE, P.: Lehrbuch der theoretischen Physik **VI**. – 5th edn., 683 p., 136 fig.; Berlin (Akademie Verlag).
- LÄNDERARBEITSGEMEINSCHAFT WASSER [LAWA] (2001): Tagebaurestseen – Anforderungen an die Wasserqualität. – 11 p., 1 fig., 2 tab.; Berlin (Kulturbuch-Verlag).
- LANDESUMWELTAMT NORDRHEIN-WESTFALEN (2002): Gewässergütebericht 2001 Nordrhein-Westfalen: Berichtszeitraum 1995-2000. – In: LANDESUMWELTAMT NORDRHEIN-WESTFALEN. – 274 p., 59 fig., 68 tab.; Essen (Landesumweltamt Nordrhein-Westfalen).
- LANDRIALT, D. (2001): Backfill in Underground Mining. – In: HUSTRULID, W. A. & BULLOCK, R. L.: Underground Mining Methods – Engineering Fundamentals and International Case Studies (69). – p. 601-614, 16 fig., 3 tab.; Littleton (Society for Mining, Metallurgy, and Exploration).
- LANG, B. (1999): Permanent Sealing of Tunnels to retain Tailings or Acid Rock Drainage. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 647-655, 6 fig., 3 tab.; Sevilla (International Mine Water Association).
- LANGE, G. & FREYHOFF, G. (1991): Geologie und Bergbau in der Uranlagerstätte Ronneburg/Thüringen. – Erzmetall, **44**: 262-269, 8 fig.; Weinheim.
- LANGE, G., MÜHLSTEDT, P., FREYHOFF, G. & SCHRÖDER, B. (1991): Der Uranerzbergbau in Thüringen und Sachsen – ein geologisch-bergmännischer Überblick. – Erzmetall, **44** (3): 162-171; Weinheim.
- LANGGUTH, H. R. & VOIGT, R. (2004): Hydrogeologische Methoden. – 2nd edn., 1005 p., 304 fig.; Berlin (Springer).
- LAROSA, P. J., KARNAVAS, J. A. & PELCZARSKI, E. A. (1971): Evaluation of a New Acid Mine Drainage Treatment Process. – Water pollution control research series, **14010 DYI 02/71**: 155, 60 fig., 23 tab.; Washington.
- LÄSSL, L. (1556): Schwazer Bergbuch. – 378 (396) p., 98 (99) fig.; Schwaz (Handschrift).
- LAUBRICH, D., NEUBERT, G., TOMSCHI, H. & UNLAND, W. (1996): Sanierung von Altlasten im Bereich der Wismut GmbH – Das Beispiel des ehemaligen Urantagebaus Lichtenberg. – VDF Führungskraft, **1996** (9/10): 33-40, 5 fig., 7 tab.; Essen.
- LAUSITZER BRAUNKOHLE AKTIENGESellschaft (2000): Braunkohlenbergbau im Osten Deutschlands. – 64 p.; Senftenberg (Lausitzer Braunkohle Aktiengesellschaft).
- LAWSON, D. C. & SONDEREGGER, J. L. (1978): Geothermal data-base study – Mine-water temperatures. – Special Publication – State of Montana Bureau of Mines and Geology, **79**: 38, 4 fig., 1 tab., 1 app.; Butte.
- LE CARLIER DE VESLUD, C., ROYER, J.-J., GÉRARD, B. & PAGEL, M. (1998): Modélisation des transferts hydrologiques et thermiques lors de l'évolution de la paléo-marge ardéchoise (France). – Bull. Soc. géol. France, **t. 169** (1): 81-89, 2 fig., 6 tab.; Paris.

- LEBECKA, J., CHALUPNIK, S., MICHALIK, B., WYSOCKA, M., SKUBACZ, K. & MIELNIKOW, A. (1994): Radioactivity of Mine Waters in Upper Silesian Coal Basin and its Influence on Natural Environment. – Proceedings, 5th International Mine Water Congress, Nottingham, U.K., **2**: 657-662, 1 fig., 1 tab.; Nottingham.
- LEBECKA, J., MIELNIKOW, A., CHALUPNIK, S., WYSOCKA, M., SKUBACZ, K. & MICHALIK, B. (1997): Radium in Mine Waters in Poland – Occurrence and Impact on River Waters. – Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia, **2**: 415-421, 1 fig.; Ljubljana.
- LEBLANC, M., MORALES, J. A., BORREGO, J. & ELBAZ-POULICHET, F. (2000): 4,500-Year-Old Mining Pollution in Southwest Spain: Long-Term Implications from Modern Mining Pollution. – *Econ. Geol.*, **95**: 655-662, 3 fig., 1 tab.; Lancaster.
- LECHNER, E. (2003): Heilende Wasser in Tirol – Geschichte und Entwicklung von Heilbädern, Bauernbädern und Kraftquellen. – 153 p.; Innsbruck, Wien (Tyrolia).
- LEDIN, M. & PEDERSEN, K. (1996): The environmental impact of mine wastes; roles of microorganisms and their significance in treatment of mine wastes. – *Earth-Science Reviews*, **41** (1-2): 67-108, 3 fig., 6 tab.; Amsterdam.
- LEE, W. L. (1984): Performance assessment of nuclear waste isolation systems: Shaping our energy future. – 59 p.; Knoxville (Proceedings, 11th annual WATtec conference and exposition).
- LEITCH, R. D. (1935): Sealing of Coal Mines – Will Reduce Acidity of their Effluent Waters. – *Coal Age*, **40**: 323-326, 7 fig.; New York.
- LERNER, D. N., ISSAR, A. S. & SIMMERS, I. (1990): Groundwater recharge – A guide to understanding and estimating natural recharge: *International Contributions to Hydrogeology* **8**. – 345 p.; Hannover (Heise).
- LEUPOLT, G. & HOCKER, M. (ca. 1999): *Befahrerhandbuch – Streitschrift zu Arbeitsweisen der praktischen bergbauhistorischen Forschung*. – 264 p.; Dresden, Freiberg (Eigenverlag).
- LEVY, D. B., CUSTIS, K. H., CASEY, W. H. & ROCK, P. A. (1997): A comparison of metal attenuation in mine residue and overburden material from an abandoned copper mine. – *Appl. Geochem.*, **12** (2): 203-211, 5 fig., 3 tab.; Oxford.
- LEWIS, R. L., MCGOWAN, I. R., HERSHMAN, J. I. & TILK, J. (1997): Numerical simulation of bulkheads to reduce uncontrolled discharge from an underground copper mine. – *Mining Engineering*, **49** (4): 68-72, 3 fig.; Littleton.
- LEWIS, R. A. (1990): Borate adsorption by some volcanic tuffs; evaluation of borate as a potential tracer for unsaturated flow at Yucca Mountain: Borate adsorption by some volcanic tuffs; evaluation of borate as a potential tracer for unsaturated flow at Yucca Mountain. – Golden (PhD Thesis Colorado School of Mines).
- LI, G. & ZHOU, W. (1999): Sinkholes in karst mining areas in China and some methods of prevention. – *Eng. Geol.*, **52** (1-2): 45-50, 5 fig.; Amsterdam.
- LIEBER, D. (2003): Säure- und Basekapazität – Vergleich unterschiedlicher vor-Ort-Bestimmungsmethoden bei unterschiedlichen Bergwerkswässern und Bearbeitern. – , 22 fig., 23 tab.; Freiberg (unpubl. Thesis).
- LIEBISCH, K. & REMUS, W. (1966): Hydrogeologische Ergebnisse mit Färbversuchen im Kupferschieferbergbau der Mansfelder Mulde. – *Z. Angew. Geol.*, **12** (9): 486-489, 2 fig.; Berlin.
- LIEDL, R. & SAUTER, M. (2000): Charakterisierung von Karstgrundwasserleitern durch Simulation der Aquifergenese und des Wärmetransports. – *Grundwasser*, **5** (1): 9-16, 8 fig.; Heidelberg.
- LIESSMANN, W. (2002): Die Harzer Erbstollen des 17. und 18. Jahrhunderts – Planung, Bau und wirtschaftliche Bedeutung am Beispiel des St. Andreasberger Reviers. – p. 84-91, 3 fig., 2 tab.; Ridnau/Schneeberg (Proceedings, 5. Internationaler Bergbau Workshop).

- LINDGREN, G. A. & DESTOUNI, G. (2002): Environmental Regulation of Mine Waters in the European Union – D 3 Institutional research – 3 Sweden: ERMITE Report – EU R&D Project EVK1-CT-2000-00078. – p. 3.1-3.11, 3 fig.; Stockholm (Royal Institute of Technology).
- LINSLEY, R. K., KOHLER, M. A. & PAULHUS, J. L. H. (1988): Hydrology for engineers – SI Metric edition: McGraw-Hill series in water resources and environmental engineering. – 492 p.; London (McGraw-Hill).
- LIPPMANN, J., STUTE, M., TORGERSEN, T., MOSER, D. P., HALL, J. A., LIN, L., BORCSIK, M., BELLAMY, R. E. S. & ONSTOTT, T. C. (2003): Dating ultra-deep mine waters with noble gases and ^{36}Cl , Witwatersrand Basin, South Africa. – *Geochim. Cosmochim. Acta*, **67** (23): 4597-4619, 8 fig., 8 tab.; Oxford.
- LLOYD, J. W. & HEATHCOTE, J. A. (1985): Natural inorganic chemistry in relation to groundwater – An introduction. – 296 p., 116 fig., 74 tab.; Oxford (Clarendon).
- LOCHER, H. & KELLER, R. (1995): Managing impacts of large artificial sediment loads in rivers – a case study. – Proceedings, International Association for Hydraulic Research Congress [Congres de l'Association Internationale de Recherches Hydrauliques], **26** (4): 356-361, 4 fig., 1 tab.; London.
- LOOP, C. M. (2004): Lessons learned from full-scale, non-traditional Placement of Fly Ash. – Proceedings, State Regulation of Coal Combustion By-Product Placement at Mine Sites: A Technical Interactive Forum: 43-46; Harrisburg.
- LOOP, C. M., SCHEETZ, B. E. & WHITE, W. B. (2003): Geochemical Evolution of a Surface Mine Lake with Alkaline Ash Addition: Field Observations vs. Laboratory Predictions. – *Mine Water Env.*, **22** (4): 206-213, 5 fig., 3 tab.; Berlin.
- LORAX ENVIRONMENTAL (2003): Treatment of Sulphate in Mine Effluents. – 129 p., 39 fig., 33 tab.; London (INAP – International Network of Acid Prevention).
- LORD, G. J., PARDO-IGÚQUIZA, E. & SMITH, I. M. (2000): A Practical Guide to Wavelets for Metrology. – Reports NPL-CMSC, **CMSC 02/00**: 39, 26 fig.; Teddington.
- LOTTERMOSER, B. (2003): Mine Wastes – Characterization, Treatment and Environmental Impacts. – 277 p., 65 fig., 45 tab.; Heidelberg (Springer).
- LUCKNER, L. & HUMMEL, J. (1982): Modelling and Prediction of the Quality of Mine Drainage Water used for the Drinking Water Supply in the GDR. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **D**: 25-36; Budapest.
- LUDWIG, K.-H. & GRUBER, F. (1987): Gold- und Silberbergbau im Übergang vom Mittelalter zur Neuzeit – das Salzburger Revier von Gastein und Rauris. – 400 p.; Köln (Böhlen).
- LUNDGREN, D. G. & SILVER, M. (1980): Ore leaching by bacteria. – *Ann. Rev. Microbiol.*, **34**: 263-283, 1 fig.; Palo Alto.
- LUSZNAT, M. (1985): Grube Stahlberg bei Müsen. – *Geol. Jb.*, **D 77**: 149-154, 2 fig.; Hannover.
- LYNCH, M. (2002): Mining in world history. – 350 p., 11 fig.; London (Reaktion).
- MACADAM, J. & SHAIL, R. (1999): Abandoned Pits and Quarries: a Resource for Research, Education, Leisure and Tourism. – In: SPALDING, A.: The conservation value of abandoned pits and quarries in Cornwall (6). – p. 71-80, 8 fig.; Truro (The Historic Environment Service).
- MACLEOD, R., BURSEY, G. G., ASTON, T. R. C. & GALE, J. E. (1993): Preliminary modelling of groundwater inflows at Gays River mine, Nova Scotia, Canada. – *Trans. Inst. Min. Metall. Sect. A*, **102**: A153-A164, 15 fig., 3 tab.; London.
- MADIGAN, M. T., MARTINKO, J. M., PARKER, J. & BROCK, T. D. (2003): An Overview of Microbial Life: Brock biology of microorganisms (2). – 10th edn., S. 22-37, 24 fig., 1 tab.; Upper Saddle River (Prentice Hall).

- MAGDZIORZ, A. & SEWERYNSKI, J. (2000): The use of membrane technic in mineralised water treatment for drinking and domestic purposes at "Pokój" coal mine district under liquidation. – Proceedings, 7th International Mine Water Association Congress: 430-442, 4 fig., 4 tab.; Ustron.
- MAIR, C. (2002): Hydrogeologie, Hydrogeochemie und Isotopie der Grund- und Grubenwässer im Einzugsgebiet des Burgfeyer Stollens bei Mechernich, Eifel. – 270 p., 70 fig., 62 tab.; Aachen (unpubl. Diss. RWTH Aachen).
- MÄKINEN, T. (1996): Information on Mines, Beneficiation Plants and Disposed Mining Waste in Finland. – (Report of the fifth meeting of HELCOM TC Point, Agenda Items 5.38-5.41 on impact from mining industry).
- MAŁOSZEWSKI, P. & ZUBER, A. (1984): Interpretation of artificial and environmental tracers in fissured rocks with a porous matrix: *Isotope Hydrology* 1983 **1**. – p. 635-654, 6 fig.; Wien (Proceedings Series – International Atomic Energy Agency).
- MAŁOSZEWSKI, P. & ZUBER, A. (1985): On the theory of tracer experiments in fissured rocks with a porous matrix. – *J. Hydrol.*, **79** (3-4): 333-358, 12 fig.; Amsterdam.
- MAŁOSZEWSKI, P. & ZUBER, A. (1990): Mathematical Modeling of Tracer Behavior in Short-Term Experiments in Fissured Rocks. – *Water Resour. Res.*, **26** (7): 1517-1528, 11 fig., 2 tab.; Washington.
- MAŁOSZEWSKI, P., HARUM, T. & BENISCHKE, R. (1992): Mathematical Modelling of Tracer Experiments in the Karst of the Lurbach System. – *Steir. Beitr. Hydrogeol.*, **43**: 116-143, 27 fig., 9 tab.; Graz.
- MANDELBROT, B. B. (1982): *The Fractal Geometry of Nature*. – 480 p.; New York (Freeman & Company).
- MANYIN, T., WILLIAMS, F. M. & STARK, L. R. (1997): Effects of iron concentration and flow rate on treatment of coal mine drainage in wetland mesocosms: An experimental approach to sizing of constructed wetlands. – *Ecol. Eng.*, **9**: 171-185, 4 fig., 3 tab.; Amsterdam.
- MAREE, J. P., HLABELA, P., NENGOVHELA, R., GELDENHUYS, A. J., MBHELE, N., NEVHULAUDZI, T. & WAANDERS, F. B. (2004a): Treatment of Mine Water for Sulphate and Metal Removal Using Barium Sulphide. – *Mine Water Env.*, **23** (4): 196-204, 7 fig., 10 tab.; Berlin.
- MAREE, J. P., BEER, M. d., STRYDOM, W. F., CHRISTIE, A. D. M. & WAANDERS, F. B. (2004b): Neutralizing Coal Mine Effluent with Limestone to Decrease Metals and Sulphate Concentrations. – *Mine Water Env.*, **23** (2): 81-86, 3 fig., 6 tab.; Berlin.
- MAREE, J. P., GÜNTHER, P., STROBOS, G. & WAANDERS, F. B. (2004c): Optimizing the Effluent Treatment at a Coal Mine by Process Modelling. – *Mine Water Env.*, **23** (2): 87-90, 6 tab.; Berlin.
- MAREE, J. P., STROBOS, G., GREBEN, H., NETSHIDAU, E., STEYN, E., CHRISTIE, A., GÜNTHER, P. & WAANDERS, F. B. (2004d): Treatment of Acid Leachate from Coal Discard using Calcium Carbonate and Biological Sulphate Removal. – *Mine Water Env.*, **23** (3): 144-151, 6 fig., 8 tab.; Berlin.
- MASCHER, R., NEAGOE, A., VOIGT, K. D., FISCHER, S., SCHEIDING, W. & BERGMANN, H. (2004): Amino alcohols – an alternative tool in phytoremediation of heavy metal polluted soils? – Abstracts, Geomicrobiology in remediation of mine waste (3. Jenaer Sanierungskolloquium): 12:50-13:10; Jena.
- MATHER, J. D., GRAY, D. A. & JENKINS, D. G. (1969): The use of tracers to investigate the relationship between mining subsidence and groundwater occurrence at Aberfan, South Wales. – *J. Hydrol.*, **9** (2): 136-154, 6 fig., 5 tab.; Amsterdam.
- MATTSSON, K. T. & STENBACKA, S. (2004): The Role of Social Capital in Local Development – The Case of Leksand and Rättvik, Sweden. – 55 p., 3 fig., 1 tab.; Uppsala (EU Project Restructuring in Marginal Rural Areas [RESTRIM]).

- MC CLEARY, E. C. & KEPLER, D. A. (1994): Ecological Benefits of passive Wetland Treatment Systems designed for Acid Mine Drainage. – Proceedings, International Land Reclamation and Mine Drainage Conference, **3**: 111-119; Pittsburgh.
- MCCOURT, J. L. (1999): Environmental Legislation and Water Management Issues during Mine Closure in South Africa. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 743-749, 1 tab.; Sevilla (International Mine Water Association).
- MCKENZIE, R. (2005): Software Update to Better Predict Costs of Treating Mine Drainage. – Mine Water Env., **24** (4): 213-215; Berlin.
- MCKNIGHT, E. T. & FISCHER, R. P. (1970): Geology and Ore Deposits of the Picher Field Oklahoma and Kansas. – US Geological Survey Professional Paper, **588**: 165, 45 fig., 10 tab., 11 Kart.; Washington, DC.
- MEIER, G., SROKA, A., LÖBEL, K.-H., KLAPPERICH, H., TONDERA, D., BUSCH, W., WAGNER, H. & RANDJBAR, B. (2004): 4. Altbergbau-Kolloquium – Tagungsband. – 492 p.; Essen (Glückauf).
- MEIER, J., BABENZIEN, H.-D. & WENDT-POTTHOFF, K. (2004): Microbial cycling of iron and sulfur in sediments of acidic and pH-neutral mining lakes in Lusatia (Brandenburg, Germany). – Biogeochemistry, **67** (2): 135-156, 6 fig., 2 tab.; Dordrecht-Boston-Lancaster.
- MEINEL, G. (1993): Mögliche Beziehungen zwischen Uranlagerstätten und der Kratonisierung am Beispiel Thüringens. – Geowiss. Mitt. Thüringen, **1**: 59-63, 1 fig., 1 tab.; Weimar.
- MEINRATH, G., MERKEL, B. J., ØDEGAARD-JENSEN, A. & EKBERG, C. (2004): Sorption of Iron on Surfaces: Modelling, Data Evaluation, and Measurement Uncertainty. – Acta Hydrochim. Hydrobiol., **32** (2): 154-160, 3 fig., 1 tab.; Weinheim.
- MEND [MINE ENVIRONMENT NEUTRAL DRAINAGE PROGRAM] (1991): Acid Rock Drainage Prediction Manual – A manual of chemical evaluation procedures for the prediction of acid generation from mine wastes: MEND Project **1.16.Ib**. – Ottawa (The Mine Environment Neutral Drainage [MEND] Program).
- MEND [MINE ENVIRONMENT NEUTRAL DRAINAGE PROGRAM] (1994): Evaluation of alternate dry covers for the inhibition of acid mine drainage from tailings: MEND Project **2.20.1**. – Richmond Hill (The Mine Environment Neutral Drainage [MEND] Program).
- MENTZ, J. W., WARG, J. B. & SKELLY AND LOY INC. (1975): Up-dip versus down-dip mining – an evaluation. – Environmental protection technology series, **EPA-670/2-75-047**: 74; Cincinnati.
- MERKEL, B. (2002): Flooding of the Königstein uranium mine – Aquifer reactivity versus dilution. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 267-275, 6 fig., 1 tab.; Heidelberg (Springer).
- MERKEL, B. (2005): Alkalinitätserhöhung in sauren Grubenwässern durch CO₂-Zugabe. – Wissenschaftliche Mitteilungen, **28**: 51-55, 3 tab.; Freiberg.
- MERKEL, B. J. (2004): Modelling of Arsenic Removal from Water – Means and Limitations. – Wissenschaftliche Mitteilungen, **25**: 119-124, 3 fig., 1 tab.; Freiberg.
- MERKEL, B. J. & HELLING, C. (1995): Proceedings Uranium-Mining and Hydrogeology 1995, Freiberg, Germany: GeoCongress **1**. – 583 p.; Köln (von Loga).
- MERKEL, B. J. & PLANER-FRIEDRICH, B. (2002): Grundwasserchemie – Praxisorientierter Leitfaden zur numerischen Modellierung von Beschaffenheit, Kontamination und Sanierung aquatischer Systeme. – 219 p.; Heidelberg (Springer).
- MERKEL, B. J. & SPERLING, B. (1998): DVWK Schriften – Hydrogeochemische Stoffsysteme Teil 2. – In: DEUTSCHER VERBAND FÜR WASSERWIRTSCHAFT UND KULTURBAU E.V. (DVWK) **117**. – p. 1-415; Bonn (Kommissionsvertrieb Wirtschafts- und Verlagsgesellschaft Gas und Wasser mbH).

- MERKEL, B. J., HARTSCH, J., PINKA, F. & PETERS, H. [. (1997): Chemische Veränderungen nach Auflassen von Uranbergwerken unter besonderer Berücksichtigung von Radionukliden. – 48 p., 32 fig., 9 tab.; Freiberg (unpubl. Manuskript).
- MERKEL, B. J., WERNER, F. & WOLKERSDORFER, CH. (2005a): Carbon dioxide elimination by using acid mine lakes and calcium oxide suspensions (CDEAL). – Geotechnologien – Science Report, **6**: 4-12, 1 fig., 1 tab.; Potsdam.
- MERKEL, B. J., PLANER-FRIEDRICH, B. & NORDSTROM, D. K. (2005b): Groundwater Geochemistry – A Practical Guide to Modeling of Natural and Contaminated Aquatic Systems. – 200 p., 76 fig.; Heidelberg (Springer).
- MERTEN, D., KOTHE, E. & BÜCHEL, G. (2004): Studies on Microbial Heavy Metal Retention from Uranium Mine Drainage Water with Special Emphasis on Rare Earth Elements. – Mine Water Env., **23** (1): 34-43, 5 fig., 5 tab.; Berlin.
- METROPOLIS, N. & ULAM, S. (1949): The Monte Carlo method. – J. Amer. Stat. Assoc., **44** (247): 335-341; Alexandria.
- MEYER, J. & SCHRAMM, A. (2004): Grubenflutung Schlema-Alberoda – Stand und Zielsetzung der Flutung der Uranerzgrube Schlema-Alberoda. – 22 overheads, 2 fig.; Bad Schlema (Vortrag Workshop Geothermie).
- MEYER, J., JENK, U., GÖHRS, A. S. & WERNER (2002): Characterisation of final mine flooding at the Schlema-Alberoda site of WISMUT GmbH with particular emphasis on flooding water quality evolution. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 777-782, 2 fig.; Heidelberg (Springer).
- MIELCAREK, W. (1988): Application of Deep Diaphragm Walls for Groundwater Control in Open cast Lignite Mining in the German Democratic Republic. – Int. J. Mine Water, **7** (3): 35-47, 9 fig.; Nottingham.
- MILITZER, H., SCHÖN, J. & STÖTZNER, U. (1986): Angewandte Geophysik im Ingenieur- und Bergbau. – 2nd edn., 419 p., 341 fig., 53 tab.; Stuttgart (Enke).
- MILLER, D. J., SEMMENS, K. J., VIADERO JR., R. C. & TIERNEY, A. E. (2005): The resource potential of mining discharge water for aquaculture. – World Aquaculture, **35** (4): 57-59 + 70, 2 fig.; Baton Rouge.
- MILLER, R. W. (1996): Flow measurement engineering handbook. – 3rd edn., 1168 p.; New York (McGraw-Hill).
- MILLER, S. D., JEFFERY, J. J. & MURRAY, G. S. C. (1990): Identification and Management of Acid Generating Mine Wastes – Procedures and Practices in South-East Asia and the Pacific Regions. – Int. J. Mine Water, **9** (1-4): 57-67, 5 fig.; Castro Verde.
- MILLS, A. L. (1999): The Role of Bacteria in Environmental Geochemistry. – In: PLUMLEE, G. S. & LOGSDON, M. J.: The Environmental Geochemistry of Mineral Deposits **6A** (5). – p. 125-132, 4 fig., 4 tab.; Littleton (Society of Economic Geologists).
- MINING MINERALS AND SUSTAINABLE DEVELOPMENT [MMSD] PROJECT (2002): Breaking New Ground – Mining, Minerals, and Sustainable Development – The Report of the MMSD Project. – 441 p.; London (Earthscan Publications).
- MINISTER OF PUBLIC WORKS AND GOVERNMENT SERVICES (2001): Guidance document for flow measurement of metal mining effluents. – Environmental Protection Series, **EPS 2/MM/4**: 28, 5 fig., 1 tab.; Ottawa.
- MODAK, J. M., NATARAJAN, K. A. & MUKHOPADHYAY, S. (1996): Development of temperature-tolerant strains of *Thiobacillus ferrooxidans* to improve bioleaching kinetics. – Hydrometallurgy, **42** (1): 51-61, 6 fig., 2 tab.; Amsterdam.
- MOHR, K. (1978): Geologie und Minerallagerstätten des Harzes. – 388 p., 138 fig.; Stuttgart (Schweizerbart).

- MORGENSTERN, P., FRIESE, K., WENDT-POTTHOFF, K. & WENNRICH, R. (2001): Bulk Chemistry Analysis of Sediments from Acid Mine Lakes by Means of Wavelength Dispersive X-ray Fluorescence. – *Mine Water Env.*, **20** (3): 105-113, 6 fig., 2 tab.; Berlin.
- MORTON, K. L. & MEKERK, F. A. v. (1993): A Phased Approach to Mine Dewatering. – *Mine Water Env.*, **12** (1-4): 27-33; Wollongong.
- MORTON, K. L. & NIEKERK, F. A. v. (1994): Mine Drainage Control and Environment Protection by using Grouting Technology and the Hydrogeological Approach. – *Mine Water Env.*, **13** (2): 41-43; Wollongong.
- MOSER, H. & RAUERT, W. (1980): Isotopenmethoden in der Hydrologie. – In: MATTHESS, G.: *Lehrbuch der Hydrogeologie* **8**. – 400 p., 227 fig., 32 tab.; Berlin (Borntraeger).
- MOTYKA, I. & SKIBINSKI, L. (1982): Utilization of Salt Mine Waters in the Means of Environmental Protection. – *Proceedings, 1st International Mine Water Congress, Budapest, Hungary*, **D**: 37-47, 2 fig.; Budapest.
- MUDROCH, A., STOTTMEISTER, U., KENNEDY, C. & KLAPPER, H. (2002): Remediation of Abandoned Surface Coal Mining Sites. – 269 p., 73 fig., 22 tab.; Heidelberg (Springer).
- MUELLER, D. K., MARTIN, J. D. & LOPES, T. J. (1997): Quality-Control Design for Surface-Water Sampling in the National Water-Quality Assessment Program. – Open-File Report – US Geological Survey, **OF 97-223**: 1-17, 1 fig., 1 app.; Reston.
- MÜHLSTEDT, P. (1992): Der Uranbergbau in Sachsen und seine Besonderheiten in exogenen Lagerstätten. – *Schriftenreihe Gesellschaft Deutscher Metallhütten- und Bergleute*, **64**: 183-201, 10 fig., 5 tab.; Clausthal-Zellerfeld.
- MULENGA, S. C., FERNÁNDEZ-RUBIO, R., LEON, A. & BAQUERO, J. C. (1992): Estimation of Quantitative Water Inflow from Different Sources in Konkola Mine, Zambia. – *Mine Water Env.*, **11** (4): 1-22, 16 fig.; Wollongong.
- MÜLLER-SALZBURG, L. (1978): *Der Felsbau* **3**. – 945 p., 612 fig., 50 plates; Stuttgart (Enke).
- MUNRO, L. D., CLARK, M. W. & MCCONCHIE, D. (2004): A Bauxsol™-based Permeable Reactive Barrier for the Treatment of Acid Rock Drainage. – *Mine Water Env.*, **23** (4): 183-194, 6 fig., 4 tab.; Berlin.
- MURPHY, W. M. (1997): Are pit lakes susceptible to limnic eruptions? – *Proceedings, International Conference on Tailings and Mine Waste*, **4**: 543-548; Rotterdam.
- MUTSCHLECHNER, G. (1984): *Erzbergbau und Bergwesen im Berggericht Rattenberg*. – 163 p., 68 fig., 1 tab.; Innsbruck (Eigenverlag Gemeinden Alpbach, Brixlegg, Rattenberg und Reith im Alpbachtal).
- NAIRN, R. W. & MERCER, M. N. (2000): Alkalinity generation and metals retention in a successive alkalinity producing system. – *Mine Water Env.*, **19** (2): 124-135, 3 fig., 2 tab.; Berlin.
- NATURAL RESOURCES MANAGEMENT UNIT DIVISION „PROTECTION OF THE ENVIRONMENT AND NATURAL RESOURCES“ (1999): Report on the International Round Table on Mining and the Environment („Berlin II Declaration“). – 105 p., 4 tab.; Köln (Carl Duisberg Gesellschaft).
- NEAL, C., FOX, K. K., HARROW, M. & NEAL, M. (1998): Boron in the major UK rivers entering the North Sea. – *Sci. Total Environ.*, **210** (1-6): 41-51; Amsterdam.
- NEEF, T. (2004): Hydrogeochemischen Verhältnisse in einem natürlichen Feuchtgebiet zur passiven Grubenwasserreinigung. – 141 p.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- NEL, P. J. L. (2000): Cover drilling and cementation of water fissure during shaft sinking and the use of micro fine cement as a grouting medium. – p. 585-594, 4 fig.; Ustron (*Proceedings, 7th International Mine Water Association Congress*).

- NEUBERT, J. T. (2000): Naturally Degraded Surface Waters Associated with Hydrothermally Altered Terrane in Colorado. – Colorado Geological Survey Open-File Report, **00-16**: 1-158, 61 fig., 12 tab.; Denver.
- NEWTON, J. G. (1984): Review of induced sinkhole development. – p. 3-9, 3 fig.; Rotterdam Boston (Balkema).
- NITZSCHE, O., MEINRATH, G. & MERKEL, B. (2000): Database uncertainty as a limiting factor in reactive transport prognosis. – J. Contam. Hydrol., **44** (3-4): 223-237, 7 fig., 3 tab.; Amsterdam.
- NIXDORF, B. (1999): Mining Lakes in East Germany – The Problem of Acidification and Chances for Ecosystem Development. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 557-561, 6 fig., 1 tab.; Sevilla (International Mine Water Association).
- NIXDORF, B. & DENEKE, R. (2004): Grundlagen und Maßnahmen zur biogenen Alkalinisierung von sauren Tagebauseen. – 281 p.; Berlin (Weißensee).
- NIYOGI, D. K., MCKNIGHT, D. M., LEWIS, W. M. J. & KIMBALL, B. A. (1999): Experimental diversion of acid mine drainage and the effects on a headwater stream. – Water resources investigations report, **WRI 99-4018-A**: 123-130, 4 fig.; Reston.
- NORDSTROM, D. K. (1977): Hydrogeochemical and microbiological factors affecting the heavy metal chemistry of an acid mine drainage system. – 230 p.; Stanford (Degree: Doctoral).
- NORDSTROM, D. K. (2000): Advances in the Hydrogeochemistry and Microbiology of Acid Mine Waters. – Int. Geol. Rev., **42**: 499-515, 4 fig., 2 tab.; Silver Spring.
- NORDSTROM, D. K. (2004a): Modeling Low-temperature Geochemical Processes: Treatise on Geochemistry **5** (5.02). – 37–72 p., 10 fig., 3 tab.; Amsterdam (Elsevier).
- NORDSTROM, D. K. (2004b): From research to remediation: application of hydrogeochemical research for effective mine site remediation. – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: mine water 2004 – Proceedings International Mine Water Association Symposium **1**. – p. 141-148, 3 fig., 2 tab.; Newcastle upon Tyne (University of Newcastle).
- NORDSTROM, D. K. (2004c): Negative pH, Efflorescent Mineralogy, and the Challenge of Environmental Restoration at the Iron Mountain Superfund site or why not plug a mine. – Wissenschaftliche Mitteilungen, **25**: 125-131, 3 fig., 1 tab.; Freiberg.
- NORDSTROM, D. K. & ALPERS, C. N. (1995): Remedial Investigation, Decisions and Geochemical Consequences at Iron Mountain Mine, California. – Proceedings, Sudbury '95 – Mining and the Environment, **II**: 633-642, 2 fig., 1 tab.; Sudbury, Ontario, Canada.
- NORDSTROM, D. K. & ALPERS, C. N. (1999): Geochemistry of Acid Mine Waters. – In: PLUMLEE, G. S. & LOGSDON, M. J.: The Environmental Geochemistry of Mineral Deposits **6A** (6). – p. 133-160, 7 fig., 14 tab.; Littleton (Society of Economic Geologists).
- NORDSTROM, D. K., PLUMMER, L. N., WIGLEY, T. M. L., WOLERY, T. J., BALL, J. W., JENNE, E. A., BASSETT, R. L., CRERAR, D. A., FLORENCE, T. M., FRITZ, B., HOFFMAN, M., HOLDREN JR, G. R., LAFON, G. M., MATTIGOD, S. V., MCDUFF, R. E., MOREL, F., REDDY, M. M., SPOSITO, G. & THRAILKILL, J. (1979): A Comparison of computerized chemical models for equilibrium calculations in aqueous systems. – ACS Symposium Series, **93**: 857-892, 12 tab.; Washington.
- NORDSTROM, D. K., OLSSON, T., CARLSSON, L. & FRITZ, P. (1989): Introduction to the hydrogeochemical investigations within the International Stripa Project. – Geochim. Cosmochim. Acta, **53** (8): 1717-1726, fig., 8 tab.; Oxford.
- NORDSTROM, D. K., ALPERS, C. N., PTACEK, C. J. & BLOWES, D. W. (2000): Negative pH and Extremely Acidic Mine Waters from Iron Mountain, California. – Environ. Sci. Technol., **34**: 254-258, 3 fig., 2 tab.; Washington.

- NORTON, D. & KNAPP, R. (1977): Transport phenomena in hydrothermal systems: the nature of porosity. – *American Journal of Science*, **277**: 913-936, 10 fig., 3 tab.; New Haven.
- NOUBACTEP, C., MEINRATH, G., VOLKE, P., PETER, H.-J., DIETRICH, P. & MERKEL, B. J. (2002): Mechanism of Uranium Fixation by Zero Valent Iron: The Importance of Co-precipitation. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: *Uranium in the Aquatic Environment*. – p. 581-590, 3 fig., 3 tab.; Heidelberg (Springer).
- NOVAK, P. (1994): Improvement of water quality in rivers by aeration at hydraulic structures. – In: HINO, M.: *Water Quality and its Control*. – p. 147-168, 7 fig., 3 tab.; Rotterdam (Balkema).
- NOVÁK, J. (2001): Groundwater Remediation in the Stráz Leaching Operation. – *Mine Water Env.*, **20** (4): 158-167, 10 fig., 1 tab.; Berlin.
- NOVÁK, J., MARECEK, P. & SMETANA, R. (1995): System of the Model Solutions of the Physical-Chemical Processes in the Straz Deposit Used for Rock Environment Restoration. – *Proceedings, Uranium-Mining and Hydrogeology, Freiberg, Germany; GeoCongress*, **1**: 423-432, 8 fig.; Freiberg.
- NUNES, J. F., RICHARDS, D. G. & GAMA, H. A. (1990): Mine Water and Environmental Protection – The Somincor Case. – *Int. J. Mine Water*, **9** (1-4): 237-253, 5 fig., 2 tab.; Castro Verde.
- NUTTALL, C. A., ADAMS, R. & YOUNGER, P. L. (2002): Integrated hydraulic-hydrogeochemical assessment of flooded deep mine voids by test pumping at the Deerplay (Lancashire) and Frances (Fife) Colliery. – *Spec. Publ. – Geol. Soc. London*, **198**: 315-326, 11 fig., 5 tab.; London.
- NUTTALL, C. A. & YOUNGER, P. L. (2004): Hydrochemical stratification in flooded underground mines: an overlooked pitfall. – *J. Contam. Hydrol.*, **69** (1-2): 101-114, 6 fig., 3 tab.; Amsterdam.
- OBERSTE-BRINK, K. (1940): Die Frage der Hebungen bei Bodenbewegungen infolge Bergbaues [Surface uplift due to ground movements as a result of mining]. – *Glückauf*, **76** (18): 249-256, 14 fig., 2 tab.; Essen.
- OELSNER, O. W. (1958): Die erzgebirgischen Granite, ihre Vererzung und die Stellung der Bi-Co-Ni-Formation innerhalb der Vererzung. – *Geologie*, **7** (3-6): 682-697, 5 fig., 6 photographs; Berlin.
- OERTEL, H. (2001): *Introduction to fluid mechanics*. – 1st edn., 167 p.; Braunschweig (Viehweg).
- OFFICE OF SURFACE MINING (2002): *Office of Surface Mining – 2002 Annual Report (CD Version)*. – 93 p.; Washington (Office of Surface Mining).
- OGASAWARA, H., FUJIMORI, K., KOIZUMI, N., HIRANO, N., FUJIWARA, S., OTSUKA, S., NAKAO, S., NISHIGAMI, K., TANIGUCHI, K., IIO, Y., NISHIDA, R., OIKE, K. & TANAKA, Y. (2002): Microseismicity induced by heavy rainfall around flooded vertical ore veins. – *Pure Appl. Geophys.*, **159** (1-3): 91-109, 11 fig., 2 tab.; Basel.
- OHIMAIN, E. I. (2003): Available options for the bioremediation and restoration of abandoned pyritic dredge spoils causing the death of fringing mangroves in the Niger Delta. – 9 p., 2 fig., 1 tab.; Athen (Proceedings International Biohydrometallurgy Symposium).
- OHLY, M. (2005): *Schutzwürdigkeit und Schutzfähigkeit des Wassereinzugsgebietes des Teichtalstollens in Königshütte/Ldkr. Wernigerode*. – 69 p., 15 fig., 2 tab., 2 app.; Freiberg (unpubl. Dipl.-Arbeit TU Bergakademie Freiberg).
- OSSANDÓN, G. C., FRÉRAUT, R. C., GUSTAFSON, L. B., LINDSAY, D. D. & ZENTILLI, M. (2001): *Geology of the Chuqicamata Mine – A Progress Report*. – *Econ. Geol. Bull. Soc. Econ. Geol.*, **96** (2): 249-270, 8 fig., 1 tab.; Lancaster.
- O’SULLIVAN, A. D., MURRAY, D. A. & OTTE, M. L. (2004): Removal of Sulfate, Zinc, and Lead from Alkaline Mine Wastewater Using Pilot-scale Surface-Flow Wetlands at Tara Mines, Ireland. – *Mine Water Env.*, **23** (2): 58-65, 3 fig., 1 tab.; Berlin.

- OTT, A. N. (1986): Estimating iron and aluminum content of acid mine discharge from a north-central Pennsylvania coal field by use of acidity titration curves. – Water resources investigations report, **WRI 84-4335**: 25, 9 fig., 9 tab.; Washington.
- OTT, A. N. (1988): Dual-acidity titration curves – fingerprint, indicator of redox state, and estimator of iron and aluminum content of acid mine drainage and related waters. – US Geological Survey Water-Supply Paper, **W 2330**: 19-33, 11 fig., 5 tab.; Reston.
- OTTO, M. (2000): Analytische Chemie. – 2nd edn., 671 p., 326 fig., 104 tab.; Weinheim (Wiley-VCH).
- PALME, R. & INGENHAEFF-BERENKAMP, W. (1993): Stollen, Schächte, fahle Erze – Zur Geschichte des Schwazer Bergbaus. – 3rd edn., 111 p., 96 fig.; Schwaz (Berenkamp).
- PALME, R., GSTREIN, P. & INGENHAEFF, W. (2002): Glück Auf! – Faszination Schwazer Silberbergwerk. – 111 p., 56 fig.; Innsbruck (Berenkamp).
- PARISH, R. V. (1977): The metallic elements. – 254 p., 124 fig., 89 tab.; New York (Longman).
- PARKHURST, D. L. (1987): Chemical analyses of water samples from the Picher mining area, Northeast Oklahoma and Southeast Kansas. – Open-File Report – US Geological Survey, **OF 87-0453**: 43, 3 fig., 6 tab.; Denver.
- PARKHURST, D. L. (1988): Mine-water discharge, metal loading, and chemical reactions. – Open-File Report – US Geological Survey, **OF 86-0481**: D5-D9, 4 fig., 1 tab.; Reston.
- PARLIAMENT OF THE COMMONWEALTH OF AUSTRALIA (1970): Water Pollution in Australia – Report from the Senate Select Committee on Water Pollution. – 214 p., 13 fig., 6 tab., 14 plates; Canberra (Commonwealth Printing Office).
- PARRY, W. T., FORSTER, C. B., SOLOMON, D. K. & JAMES, L. P. (2000): Ownership of Mine-Tunnel Discharge. – Ground Water, **38** (4): 487-496; Westerville.
- PARSONS, A. S. & HUNTER, M. D. (1972): Investigation into the movement of groundwater from Bryn Pit above Ebbw Vale, Monmouthshire. – Water Pollut. Control, **71**: 568-572, 2 fig., 1 tab.; London.
- PASCALY, P. (1997): Forum – Leserbrief. – Grundwasser, **2** (4): 185; Heidelberg.
- PASCHOA, A. S. (1998): Potential Environmental and Regulatory Implications of Naturally Occurring Radioactive Materials (NORM). – Appl. Radiat. Isot., **49** (3): 189-196, 3 fig., 3 tab.; Oxford.
- PAUL, M., GENGNAGEL, M., VOGEL, D. & KUHN, W. (2002): Four years of flooding WISMUT's Ronneburg uranium mine – a status report. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 783-792, 4 fig., 4 tab.; Heidelberg (Springer).
- PAULEY, E., SHUMAKER, T. & COLE, B. (2003): Non-Injury Mine Inundation Accident (Entrapment) July 24, 2002 – Black Wolf Coal Company, Inc. Quecreek No. 1 Mine – CMAP # 56981301 – Quecreek, Somerset County, Pennsylvania. – Report of Investigations – Pennsylvania Department of Environmental Protection Bureau of Deep Mine Safety: 52, 26 fig., 23 app.; Pittsburgh.
- PEARCE, P. F. & RIES, E. R. (1982): Protection of environmental resources by effective mine water management. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **C**: 80-95, 6 fig., 3 tab.; Budapest.
- PELZEL, J., HEMMANN, M. & SEIFERT, G. (1972): Ursachen und Entwicklung der Senkungserscheinungen und Erdfälle bei der Flutung der Grube Friedenshall bei Bernburg. – Ber. Dtsch. Ges. Geol. Wiss., Reihe A, Geol. Paläontol., **17** (2): 191-219, 17 fig., 4 tab.; Berlin.
- PENROSE, R. G. J. (1974): Limestone selection for permeable plug mine seals. – Papers Presented Before the Symposium on Coal Mine Drainage Research, **5**: 133-145, 6 fig., 1 tab.; Washington.

- PERRIN, C. J., WILKES, B. & RICHARDSON, J. S. (1992): Stream periphyton and benthic insect responses to additions of treated and mine drainage in a continuous-flow on-site mesocosm. – *Environ. Toxicol. Chem.*, **11** (11): 1513-1525, 3 fig., 5 tab.; Pensacola.
- PERRY, E. F. (2000): Mine Drainage Prediction, a Summary of Concepts and Processes. – In: KLEINMANN, R. L. P.: *Prediction of Water Quality at Surface Coal Mines* (7). – p. 168-194, 5 fig., 6 tab.; Morgantown (National Mine Land Reclamation Center).
- PERRY, E. F. & EVANS, R. S. (1999): Application of geochemical modeling and hydrologic techniques to interpret sources, mixing and evolution of mine drainage: Mining and Reclamation for the Next Millennium **2**. – p. 444-452, 1 fig., 4 tab.; Boulder (Proceedings, 16th Annual National Meeting – American Society for Surface Mining and Reclamation).
- PERRY, E. F., GARDNER, M. D. & EVANS, R. S. (1997): Effects of acid material handling and disposal on coal mine drainage quality **3**. – p. 1007-1025, 8 fig., 8 tab.; Vancouver (Proceedings, 4th International Conference on Acid Rock Drainage).
- PERRY, R. S., DUDENEY, A. W. L. & LEAK, D. J. (2001): Microbiology and 16S rDNA molecular ecology of a polluting neutral mine drainage sediment. – In: CIMINELLI, V. S. T. & GARCIA JR, O.: *Biohydrometallurgy – Fundamentals, Technology and Sustainable Development* **11A**. – p. 335-344; Amsterdam (Elsevier).
- PETER, G. (1983): Untersuchungen zu Dynamik, Chemismus und Temperaturregime von Wässern ersoffener Grubenbaue. – Freiberg (unpubl. Thesis, Bergakademie Freiberg).
- PEUKERT, D., REISNER, H. & KUNZE, W. (1982): Application of a continuously operating Model for Forecasting a Regional Groundwater Flow Situation Caused by Open-Cast Mine Dewatering Measures. – Proceedings, 1st International Mine Water Congress, Budapest, Hungary, **A**: 209-225, 6 fig.; Budapest.
- PFINGSTEN, W. & MULL, R. (1990): Transportprozesse in Kluftgrundwasserleitern. – *Deutsche Gewässerkundliche Mitteilungen*, **34** (4): 116-123, 9 fig., 1 tab.; Koblenz.
- PIETSCH, W. & SCHÖTZ, A. (2004): Constructed Wetlands – ein Beitrag zur Behandlung extrem saurer Tagebaugewässer. – In: NIXDORF, B. & DENEKE, R.: *Grundlagen und Maßnahmen zur biogenen Alkalinisierung von sauren Tagebauseen* (11). – p. 175-198, 7 fig., 5 tab.; Berlin (Weißensee).
- PIPER, A. M. (1944): A graphic procedure in the geochemical interpretation of water analyses. – *Trans. Amer. Geophys. Union*, **25**: 914-923, 6 fig., 2 Tab; Washington.
- PIPER, A. M. (1953): A graphic procedure in the geochemical interpretation of water analyses. – *US Geological Survey Ground Water Notes – Geochemistry*, **12**: 1-14, 6 fig., 2 Tab; Washington.
- PIRAMID CONSORTIUM (2003): Engineering Guidelines for the Passive Remediation of Acidic and/or Metalliferous Mine Drainage and similar Wastewaters. – 166 p., 10 fig., 27 tab.; Newcastle Upon Tyne (University of Newcastle Upon Tyne).
- PIRKL, H. (1961): Geologie des Trias-Streifens und des Schwazer Dolomits südlich des Inn zwischen Schwaz und Wörgl (Tirol). – *Jb. Geol. B.-A.*, **104** (1): 1-150, 9 fig., 7 plates; Wien.
- PLAYTON, S. J. & DAVIS, R. E. (1978): Quality of water in abandoned zinc mines in northeastern Oklahoma and southeastern Kansas. – *US Geological Survey Professional Paper*, **1100**: 111, 4 fig., 4 tab.; Washington.
- PLAYTON, S. J., DAVIS, R. E. & MCCLAFLIN, R. G. (1978): Water quality in abandoned zinc mines in the Picher Field, Tri-State mining district. – *Abstracts with Programs – Geological Society of America*, **10** (1): 24; Boulder.
- PLAYTON, S. J., DAVIS, R. E. & MCCLAFLIN, R. G. (1980): Chemical Quality of Water in abandoned Zinc Mines in Northeastern Oklahoma and Southeastern Kansas. – *Oklahoma Geological Survey Circular*, **82**: 1-49, 13 fig., 19 tab.; Norman.

- PLINKE, M. & WILDHAGEN, E. (2003): Mining and water in Germany – Legal framework and technical aspects. – In: NEL, P. J. L.: Mine Water and the Environment. – p. 279-286; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- PLOTNIKOV, I. I. (1988): Method of protecting mineral deposits in karst rocks from inundation. – Soviet Eng. Geol., **1**: 51-56, 1 fig.; New York.
- PLOTNIKOV, N. I., ROGINETS, I. I. & VISWANATHAN, S. (1989): Hydrogeology of Ore Deposits: Russian translations series **72**. – 290 p., 76 fig., 14 tab.; Rotterdam (Balkema).
- PLUTA, I. (2000): Barium discharged in waters from the closed Zory Coal Mine in the Upper Silesian Coal Basin, Poland. – Proceedings, 7th International Mine Water Association Congress: 595-603, 6 fig., 2 tab.; Ustron.
- POHL, W. L. (2005): W. u. W. E. Petrascheck's Lagerstättenlehre – Mineralische und Energie-Rohstoffe – Eine Einführung zur Entstehung und nachhaltigen Nutzung von Lagerstätten. – 5th edn., 527 p., 189 fig.; Stuttgart (Schweizerbart).
- POKRYSKA, Z., TAUZIČDE, C., LAGNY, C., GUISE, Y., GOBILLOT, R., PLANCHENAULT, J.-M. & LAGARDE, R. (2005): Gas migration from closed coal mines to the surface – Risk assessment methodology and prevention means. – p. 1-15 [CD-ROM], 8 fig.; Nancy (Proceedings, Post-Mining 2005).
- POLAND, J. F. (1984): Guidebook to studies of land subsidence due to ground-water withdrawal: Studies and reports in hydrology **40**. – 305 p.; Paris (UNESCO).
- PÖTTGENS, J. J. E. (1985): Bodenhebung durch ansteigendes Grubenwasser [Uplift as a result of rising mine waters]: The Development Science and Art of Minerals Surveying **2**. – p. 928-938, 7 fig.; Harrogate (Proceedings, 6th International Congress International Society for Mine Surveying).
- POTVIN, Y., THOMAS, E. G. & FOURIE, A. B. (2005): Handbook on mine fill. – 179 p.; Nedlands (Australian Centre for Geomechanics).
- PREDICTION WORKGROUP OF THE ACID DRAINAGE TECHNOLOGY INITIATIVE (2000): Prediction of Water Quality at Surface Coal Mines. – In: KLEINMANN, R. L. P. – 241 p.; Morgantown (National Mine Land Reclamation Center).
- PRESTOR, J., ŠTRUCL, S. & PUNGARTNIK, M. (2003): Mežica lead and zinc mine closure impact on hydrogeological conditions in upper Meža valley. – RMZ – Materials and Geoenvironment, **50** (1): 313-316, 1 fig., 1 tab.; Ljubljana.
- PRING, G. W. (1999): International law and mineral resources: Mining, Environment and Development. – 55 p.; Genf (UNCTAD – United Nations Conference on Trade and Development).
- PUURA, E., MARMO, L. & D'ALESSANDRO, M. (2003): Workshop on Mine and Quarry Waste – The Burden from the Past. – In: EUROPEAN COMMISSION JOINT RESEARCH CENTRE **EUR 20661 EN**. – 79 p.; Luxembourg (Office for Official Publications of the European Communities).
- QIANG, W., YUJIE, J., BAOLIN, T., DEAN, L. & YNOGHUA, X. (1992): An application of artificial tracers in determining vertical groundwater recharge locations in a coal mine. – In: HÖTZL, H. & WERNER, A.: Tracer Hydrology – 6th International Symposium on Water Tracing. – p. 365-367, 1 fig., 1 tab.; Rotterdam (Balkema).
- QUINN, J. J., NEGRI, M. C., HINCHMAN, R. R., MOOS, L. P. & WOZNIAK, J. B. ., G. E. G. (2001): Predicting the Effect of Deep-Rooted Hybrid Poplars on the Groundwater Flow System at a Large-Scale Phytoremediation Site. – Int. J. Phytoremediation, **3** (1): 41-60, 8 fig., 2 tab.; Boca Raton.
- RACKLEY, R. I. (1976): Origin of Western-States Type Uranium Mineralization. – In: WOLF, K. H.: Handbook of strata-bound and stratiform ore deposits **9** (3). – p. 89-156, 22 fig.; Amsterdam (Elsevier).

- RAMMLMAIR, D. & GRISSEMANN, C. (2000): Natural attenuation in slag heaps versus remediation. – In: RAMMLMAIR, D., MEDERER, J., OBERTHÜR, T., HEIMANN, R. B. & PENTINGHAUS, H.: *Applied Mineralogy in Research, Economy, Technology, Ecology and Culture* **2**. – p. 645-648, 2 fig.; Göttingen (Proceedings, 6th International Congress on Applied Mineralogy).
- RANDALL, M. M., LANYON, G. W., NICHOLLS, J. & WILLIS-RICHARDS, J. (1990): Evaluation of jointing in the carmenellis granite. – In: BARIA, R.: *Hot dry rock: Geothermal Energy*. – p. 108-123, 7 fig., 2 tab.; Camborne (Robertson Scientific Publications).
- RANVILLE, J. F. & SCHMIERMUND, R. L. (1999): General Aspects of Aquatic Colloids in Environmental Geochemistry. – In: PLUMLEE, G. S. & LOGSDON, M. J.: *The Environmental Geochemistry of Mineral Deposits* **6A** (8). – p. 183-199, 8 fig., 4 tab.; Littleton (Society of Economic Geologists).
- RÄTSEP, A. & LIBLIK, V. (1998): Changes in physico-chemical composition of mine water in closed Kiviõli oil shale mine. – *Oil Shale*, **15** (4): 341-352, 1 fig., 3 tab.; Tallin.
- RAUCHE, H., JOCKEL, A. & RAUCHE, H. (2002): Monitoring von Altkalischächten – Instrument zur Beurteilung der Gefährdungspotentiale. – In: BUSCH, W., MAAS, K., KLAPPERICH, H., TONDERA, D., SROKA, A. & MEIER, G.: *2. Altbergbau-Kolloquium – Tagungsband*. – p. 232-246, 4 fig.; Clausthal-Zellerfeld (Papierflieger).
- RAWLINGS, D. E. (2002): Heavy Metal Mining using Microbes. – *Ann. Rev. Microbiol.*, **56**: 56-91, 5 fig.; Palo Alto.
- RAWLINGS, D. E. (2005): Characteristics and adaptability of iron- and sulfur-oxidizing microorganisms used for the recovery of metals from minerals and their concentrates. – *Microbial Cell Factories*, **4** (13): [1-15], 3 fig.; London.
- REES, B. & CONNELLY, R. (2003): Review of design and performance of the Peleenna wetland systems. – *Land Contam. Reclam.*, **11** (2): 293-300, 5 fig., 3 tab.; Richmond.
- REES, S. B., BOWELL, R. J. & WISEMAN I. (2002): Influence of mine hydrogeology on minewater discharge chemistry. – *Spec. Publ. – Geol. Soc. London*, **198**: 379-390, 7 fig., 3 tab.; London.
- REGENSPURG, S., MEIMA, J. A., KASSAHUN, A. & RAMMLMAIR, D. (2005): Krustenbildung in Bergbauhalden. – *Wissenschaftliche Mitteilungen*, **28**: 63-67, 3 fig., 2 tab.; Freiberg.
- REGIONALER PLANUNGSVERBAND WESTSACHSEN (1998): *Braunkohlenplan Tagebau Vereinigtes Schleenhain*. – 94 p., 2 fig., 2 app., 7 Kart.; Bautzen (Lausitzer Druck- und Verlagshaus).
- REIMUS, P. W. & HAGA, M. J. (1999): Analysis of Tracer Responses in the BULLION Forced-Gradient Experiment at Pahute Mesa, Nevada. – *Los Alamos Report*, **LA-13615-MS**: 32, 16 fig., 3 tab., 2 app.; Los Alamos.
- REN, G.-L., IZADI, B., KING, B. & DOWDING, E. (1996): Preferential Transport of Bromide in undisturbed Cores under different Irrigation Methods. – *Soil Sci.*, **161** (4): 214-225, 5 fig., 6 tab.; Baltimore.
- REZNIK, Y. M. (1990): Determination of groundwater paths using method of streaming potentials. – In: GRAVES, D. H.: *National symposium on mining: a new beginning*. – p. 217-221, 2 fig.; Knoxville, TN (Proceedings National Symposium on Mining).
- RIEDEL, L. (1993): *Zur Geschichte des Bergbaues um Chemnitz*. – Veröff. Mus. Naturk. Chemnitz, **16**: 1-88, 28 fig., 1 plates; Chemnitz.
- RIEMANN, C. (1913): *Die deutschen Salzlagerstätten – ihr Vorkommen, ihre Entstehung und die Verwertung ihrer Produkte in Industrie und Landwirtschaft*. – 97 p., 29 fig.; Leipzig (Teubner).
- ROBBINS, E. I., CRAVOTTA, C. A., SVELA, C. E. & NORD, G. L. (1999): Hydrobiogeochemical interactions in 'anoxic' limestone drains for neutralization of acidic mine drainage. – *Fuel*, **78** (2): 259-270, 4 fig., 2 tab.; Oxford.

- ROBBINS, E. I., RODGERS, T. M., ALPERS, C. N. & NORDSTROM, D. K. (2000): Ecogeochemistry of the subsurface food web at pH 0-2.5 in Iron Mountain, California, USA. – *Hydrobiologia*, **433** (1-3): 15-23, 3 fig., 2 tab.; Dordrecht.
- ROBERTSON, A. M., DEVENNY, D. & SHAW, S. C. (1998): Post Mining Sustainable use Plans vs Closure Plans: Mine Reclamation and Remediation. – p. 95-110, 4 fig., 1 tab.; Penticton (Proceedings, 22nd Annual British Columbia Mine Reclamation Symposium).
- ROBERTSON, A. M. & SHAW, S. C. (1999): The Concept of Custodial Transfer of Mined Land. – In: FERNÁNDEZ RUBIO, R.: *Mine, Water & Environment II*. – p. 765-771, 3 fig., 1 tab.; Sevilla (International Mine Water Association).
- ROBILLARD, L., NGUYEN, T. H. & VASSEUR, P. (1986): Free Convection in a Two-Dimensional Porous Loop. – *J. Heat Transfer*, **108** (2): 277-283, 4 fig.; New York.
- ROEHL, K. E. (2004): Passive in situ Remediation of Contaminated Groundwater: Permeable Reactive Barriers – PRBS: Conference Papers **35**. – p. 57-70, 4 fig.; Wien (Umweltbundesamt).
- ROGOZ, M. (1994): Computer Simulation of the Process of Flooding up a Group of Mines. – Proceedings, 5th International Mine Water Congress, Nottingham, U.K., **1**: 369-377, 3 fig., 1 tab.; Nottingham.
- ROHWERDER, T., GEHRKE, T., KINZLER, K. & SAND, W. (2003): Bioleaching review part A – Progress in bioleaching – fundamentals and mechanisms of bacterial metal sulfide oxidation. – *Appl. Microbiol. Biotechnol.*, **63**: 239-248, 4 fig.; Berlin.
- ROLLAND, W., WAGNER, H., CHMIELEWSKI, R. & GRÜNEWALD, U. (2001): Evaluation of the long term Groundwater Pollution by the open cast Lignite mine Jänschwalde (Germany). – *J. Geochem. Explor.*, **73** (2): 97-111, 12 fig., 2 tab.; Amsterdam.
- ROMMEL, K. (2004): Oxygen Measurement: Optically or Electrochemically? – *Asian Environ. Technol.*, **8** (4): 32-33, 2 fig.; Hertfordshire.
- ROSCHLAU, H. (1994): Bohr- und Sprengarbeiten im Uranerzbergbau der AG/SDAG Wismut. – *Erzmetall*, **47** (6/7, 8, 10): 411-416, 480-488, 627-631, 28 fig., 4 tab.; Weinheim.
- ROSE, A. W. (2004): Vertical Flow Systems – Effects of Time and Acidity Relations. – p. 1595-1616, 6 fig., 5 tab.; Lexington (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- ROSE, A. W. & DIETZ, J. M. (2002): Case Studies of Passive Treatment Systems – Vertical Flow Systems. – p. 776-797, 7 fig., 4 tab.; Lexington (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- ROSENECK, R. (2001): Der Rammelsberg als UNESCO-Weltkulturerbe. – In: ROSENECK, R.: *Der Rammelsberg – Tausend Jahre Mensch-Natur-Technik I*. – p. 16-39, 39 fig.; Goslar (Goslarsche Zeitung).
- ROSSUM, J. R. (1975): Checking the accuracy of water analyses through the use of conductivity. – *J. Am. Water Works Assoc.*, **67** (4): 204-205, 2 tab.; New York.
- ROTHENHÖFER, P., SAHIN, H. & PEIFFER, S. (2000): Verringerung der Schwermetall- und Sulfatbelastung in sauren bergbaubelasteten Gewässern durch Aluminiumpräzipitate? – *Acta Hydrochim. Hydrobiol.*, **28** (3): 136-144, 8 fig., 2 tab.; Weinheim.
- ROY, S. & WORRAL, F. (1999): Pyrite Oxidation in Coal-Bearing Strata – The Use of P-Blocking Techniques. – In: FERNÁNDEZ RUBIO, R.: *Mine, Water & Environment II*. – p. 547-550, 1 fig., 1 tab.; Sevilla (International Mine Water Association).
- RÜDE, T. (2004): Natural acid drainage laboratory: lessons from the Rötzbach (Zillertal Alps/Austria). – In: JARVIS, A. P., DUDGEON, B. A. & YOUNGER, P. L.: *mine water 2004 – Proceedings International Mine Water Association Symposium I*. – p. 219-224, 3 fig., 1 tab.; Newcastle upon Tyne (University of Newcastle).

- RUDISELL, M. T., STUART, B. J., NOVAK, G., PAYNE, H. & TOGNI, C. S. (2001): Use of flue gas desulfurization by-product for mine sealing and abatement of acid mine drainage. – *Fuel*, **80** (6): 837-843, 6 fig., 1 tab.; Oxford.
- RUDOLPH, C. (1997): Die untertägige Sanierung der abgebauten Uranerzlagerstätte Niederschlema-Alberoda als komplexer Sanierungsprozeß. – *Glückauf*, **133** (5): 250-254, 2 fig., 3 tab.; Essen.
- RUSHTON, K. R. (2003): Groundwater hydrology – Conceptual and computational models. – 416 p.; Chichester (Wiley & Sons).
- RÜTERKAMP, P. & MESSER, J. (2000): Untersuchungen zur hydraulischen und hydrochemischen Situation in den drei Teilrevieren der gefluteten Flussspatgrube Straßberg. – Für: BST MANSFELD GMBH & CO KG NIEDERRÖBLINGEN. – 46 p., 14 app., 3 app.; Essen (Deutsche Montan Technologie GmbH).
- SABIR, I. H., TORGENSEN, J., HALDORSEN, S. & ALESTROM, P. (1999): DNA tracers with information capacity and high detection sensitivity tested in groundwater studies. – *Hydrogeol. J.*, **7** (3): 264-272, 6 fig., 1 tab.; Berlin.
- SÄCHSISCHES LANDESAMT FÜR UMWELT UND GEOLOGIE (1997): Merkblätter zur Grundwasserbeobachtung, Grundwasserprobenahme. – Materialien zur Wasserwirtschaft, **L II-1/6** (4): 1-20, 7 app.; Radebeul.
- SALKIELD, L. U. (1987): A technical history of the Rio Tinto mines – some notes on exploitation from pre-Phoenician times to the 1950s. – 114 p., 22 fig., 12 app.; London (Institution of Mining and Metallurgy).
- SALMON, S. U. & DESTOUNI, G. (2001): Environmental Regulation of Mine Waters in the European Union – D 1 National Case Studies – Sweden: ERMITE Final Report – EU R&D Project EVK1-CT-2000-00078. – 21 p., 6 fig., 5 tab.; Stockholm (Royal Institute of Technology).
- SALMON, S. U. & MALMSTRÖM, M. E. (2002): MiMi – Steady state, geochemical box model of a tailings impoundment – Application to Impoundment 1. Kristineberg, Sweden, and prediction of effect of remediation **MiMi 2002:2**. – 88 p., 32 fig., 23 tab.; Luleå (MiMi Print).
- SAMMARCO, O. (1986): Spontaneous Inrushes of Water in Underground Mines. – *Int. J. Mine Water*, **5** (2): 29-41, 6 fig.; Granada.
- SAMMARCO, O. (1995): Storage of Water in the Voids of Abandoned Mines and Fractured Rocks. – In: HOTCHKISS, W. R., DOWNEY, J. S., GUTENTAG, E. D. & MOORE, J. E.: Water resources at risk. – p. IMWA.102-114, 4 fig., 1 tab.; Minneapolis (American Institute of Hydrology).
- SAMS, J. I. I. & VELOSKI, G. A. (2003): Evaluation of Airborne Thermal Infrared Imagery for Locating Mine Drainage Sites in the Lower Kettle Creek and Cooks Run Basins, Pennsylvania, USA. – *Mine Water Env.*, **22** (2): 85-93, 7 fig., 2 tab.; Berlin.
- SAMS, J. I. I., VELOSKI, G. A. & ACKMAN, T. E. (2003): Evaluation of Airborne Thermal Infrared Imagery for Locating Mine Drainage Sites in the Lower Youghiogheny River Basin, Pennsylvania, USA. – *Mine Water Env.*, **22** (2): 94-103, 10 fig., 2 tab.; Berlin.
- SAND, W. & GEHRKE, T. (2006): Extracellular polymeric substances mediate bioleaching/biocorrosion via interfacial processes involving iron(III) ions and acidophilic bacteria. – *Research in Microbiology*, **157**: 49-56, 4 fig., 1 tab.; Amsterdam.
- SAND, W., GEHRKE, T., HALLMANN, R. & SCHIPPERS, A. (1995): Sulfur chemistry, biofilm, and the (in)direct attack mechanism – critical evaluation of bacterial leaching. – *Appl. Microbiol. Biotechnol.*, **43**: 961-966; Berlin.
- SANDÉN, P., KARLSSON, S., DUKER, A., LEDIN, A. & LUNDMAN, L. (1997): Variations in hydrochemistry, trace metal concentration and transport during a rain storm event in a small catchment. – *J. Geochem. Explor.*, **58** (2-3): 145-155, 4 fig., 2 tab.; Amsterdam-New York.

- SANDER, W. & HERBERT, H.-J. (1985): NaCl crystallization at the MgCl_2 / NaCl solution boundary; a possible natural barrier to the transport of radionuclides. – Mineral. Mag., **49** (2): 265-270, 11 fig.; London.
- SARES, M. A., HAUFF, P. L., PETERS, D. C., COULTER, D. W., BIRD, D. A., III, F. B. H. & PROSH, E. C. (2004): Characterizing Sources of Acid Rock Drainage and Resulting Water Quality Impacts Using Hyperspectral Remote Sensing – Examples from the Upper Arkansas River Basin, Colorado. – Proceedings, 2004 Advanced Integration of Geospatial Technologies in mining and Reclamation: 1-20, 10 fig., 1 tab.; Atlanta.
- SARPKAYA, T., KOLUPAILA, S., POWELL, R. W., PAULL, J. W., OKI, I., CARSTENS, M. R., KINDSVATER, C. E. & CARTER, R. W. (1959): Discharge Characteristics of Rectangular Thin-Plate Weirs – Discussion. – Trans. American Society of Civil Engineers, **124**: 802-822, 5 fig., 1 tab.; Reston.
- SAUL, H. (1948): Mine Water. – Transactions of the Institution of Mining Engineers, **107**: 294-310, 7 fig., 6 tab.; London.
- SAVIC, D. A. & WALTERS, G. A. (1999): Hydroinformatics, data mining and maintenance of UK water networks. – Anti-Corros. Methods Mater., **46** (6): 415-425, 3 fig., 1 tab.; Bradford.
- SAWADA, A., UCHIDA, M., SHIMO, M., YAMAMOTO, H., TAKAHARA, H. & DOE, T. W. (2000): Non-sorbing tracer migration experiments in fractured rock at the Kamaishi Mine, Northeast Japan. – Eng. Geol., **56** (1-2): 75-96, 16 fig., 2 tab.; Amsterdam.
- SCHALICH, J., SCHNEIDER, F. K. & STADLER, G. (1986): Die Bleierzlagerstätte Mechernich – Grundlage des Wohlstandes, Belastung für den Boden. – Fortschr. Geol. Rheinld. u. Westf., **34**: 11-91, 25 fig., 5 tab., 2 plates; Krefeld.
- SCHIFFNER, C. (1908): Radioaktive Wässer in Sachsen **I**. – p. 1-57, 16 fig., 15 tab.; Freiberg (Craz & Gerlach).
- SCHIFFNER, C. & WEIDIG, M. (1909): Radioaktive Wässer in Sachsen **II**. – p. 59-144, 35 fig., 36 tab.; Freiberg (Craz & Gerlach).
- SCHIFFNER, C., WEIDIG, M. & FRIEDRICH, R. (1911): Radioaktive Wässer in Sachsen **III**. – p. 145-216, 51 fig., 52 tab.; Freiberg (Craz & Gerlach).
- SCHINDLER, I., KUTSCHERA, U. & WIESENBERGER, H. (2004): Medienübergreifende Umweltkontrolle in ausgewählten Gebieten – Brixlegg: Monographien des Umweltbundesamtes Wien **168** (2). – p. 81-118, 14 fig., 15 tab.; Wien (Umweltbundesamt).
- SCHINDLER, S. (2003): Hydrogeochemische Untersuchung des Grubenwassers im Hagenbachtal bei Gernrode/Harz. – 146 p., 77 fig., 11 tab., 29 app.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- SCHINDLER, W., GERLACH, F. & KADEN, H. (1995): pH-Sensor für Präzisionsmessungen in Tiefenwässern. – Sensor Magazin, **1995** (3): 6-9, 6 fig.; Bad Nenndorf.
- SCHLAEGER, F. & KÖNGETER, J. (2001): A long-term water quality management model for a lignite mining affected river catchment area. – Water Sci. Technol., **44** (6): 147-154, 6 fig.; Oxford.
- SCHMID, S. & WIEGAND, J. (2003): Radionuclide Contamination of Surface Waters, Sediments, and Soil Caused by Coal Mining Activities in the Ruhr District (Germany). – Mine Water Env., **22** (3): 130-140, 10 fig., 5 tab.; Berlin.
- SCHMIDEGG, O. (1953): Die Erzlagerstätten am Reiter Kopf und am Reiter Kogel. – Schlern-Schriften, **101**: 17-25, 3 fig.; Innsbruck.
- SCHNEIDER, P., OSENBRÜCK, K., NEITZEL, P. L. & NINDEL, K. (2002): In-Situ Mitigation of Effluents from Acid Waste Rock Dumps Using Reactive Surface Barriers – a Feasibility Study. – Mine Water Env., **21** (1): 36-44, 4 fig., 3 tab.; Berlin.

- SCHNEIDER, P., NEITZEL, P. L., OSENBRÜCK, K., NOUBACTEB, C., MERKEL, B. J. & HURST, S. (2001): In-situ treatment of radioactive mine water using reactive materials – Results of laboratory and field experiments in uranium ore mines in Germany. – *Acta Hydrochim. Hydrobiol.*, **29** (2-3): 129-138, 9 fig., 2 tab.; Weinheim.
- SCHNORRER, G. (1994): Die Sekundärminerale des Bergbaugesbietes Schwaz-Brixlegg in Tirol. – *Mineralien-Magazin, Lapis*, **19** (7-8): 41-68, 87 fig.; München.
- SCHNORRER, G. (1996): Neue Sekundärminerale des alten Bergreviers Schwaz-Brixlegg in Tirol. – *Mineralien-Magazin, Lapis*, **21** (12): 16-19, 14 fig.; München.
- SCHOLZ, G. (2005): Inlake-Technik – ein effizientes Verfahren zur Alkalinisierung und Sulfatreduzierung schwefelsaurer Tagebaurestseen. – *Wissenschaftliche Mitteilungen*, **28**: 75-80, 9 fig., 2 tab.; Freiberg.
- SCHÖNENBERG, R. & NEUGEBAUER, J. (1997): Einführung in die Geologie Europas. – 7th edn., 385 p., 48 fig.; Freiburg im Breisgau (Rombach).
- SCHRECK, M. (1999): Research on the Application of Distributed Intrinsic Fibre Optic Temperature Sensing for Geotechnical Engineering. – 114 p., 42 fig., 13 tab.; Okayama (unpubl. Diss. Okayama University).
- SCHRECK, M., NISHIGAKI, M., GLÄSER, H.-R., CHRISTOPH, G., GROSSWIG, S., HURTIG, E., KASCH, M. & KÜHN, K. (1998): Determination of Heat Production Zones at Open-Cast Mine Dump. – *J. Environm. Engineering*, **124** (7): 646-651, 6 fig., 1 tab.; New York.
- SCHREYER, J., ZIMMERMANN, U. & JENK, U. (2002): Flooding of the Königstein mine up to 80 m above sea level – Prediction and Reality. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: *Uranium in the Aquatic Environment*. – p. 763-770, 5 fig.; Heidelberg (Springer).
- SCHRÖDER, B. & LIPP, U. (1990a): Der Uranerzbergbau der SDAG-Wismut im Raum Schneeberg-Aue-Schlema und seine Mineralien (II). – *Mineralien Welt*, **3** (1): 21-44; Haltern.
- SCHRÖDER, B. & LIPP, U. (1990b): Der Uranerzbergbau der SDAG-Wismut im Raum Schneeberg-Aue-Schlema und seine Mineralien (I) – Zur Geologie und Mineralogie der Lagerstätte Schlema-Alberoda. – *Mineralien Welt*, **2** (1): 42-47; Haltern.
- SCHULTZE, M., FRIESE, K., SÁNCHEZ, J. & LÓPEZ, E. (2006): The Aznalcóllar Pit Lake – Water Quality and Options of Control: ICARD 2006. – p. 1927-1935 [CD-ROM], 1 fig., 1 tab.; St. Louis (Proceedings, International Conference of Acid Rock Drainage ICARD).
- SCHWAB, M. (1999): Beiträge zur Mineralogie und Geologie von Sachsen-Anhalt. – *Der Aufschluss, Sonderband*: 1-38, 21 fig.; Heidelberg.
- SCHWANDT, A. (1993): Salzlösungs- und Wasservorkommen im Salz- und Kupferschieferbergbau am Bernburger Hauptsattel und im Sangerhäuser Revier. – *GSF-Bericht*, **21/93**: 1-232, 68 fig., tab.; Neuherberg.
- SCHWANDT, A. & SEIFERT, G. (1999): Natürliche und gelenkte Flutung von Salzbergwerken in Mitteldeutschland. – *Exkurs.f. u. Veröfftl. GGW*, **205**: 61-72, 13 fig.; Berlin.
- SCHWARZ, M., FRIEDMANN, L., SCHMEDERER, J. & ZAHN, M. T. (2002): Hinweise für die Durchführung und die Begutachtung von Markierungsversuchen in Gewässern – Merkblatt Nr. 3.1/1 vom 6.6.2002. – 16 p.; München (Bayerisches Landesamt für Wasserwirtschaft).
- SCHWEDT, G. (1995): Analytische Chemie – Grundlagen, Methoden und Praxis. – 442 p., 243 fig., 158 tab.; Stuttgart (Thieme).
- SCHWERTMANN, U., BIGHAM, J. M. & MURAD, E. (1995): The first occurrence of schwertmannite in a natural stream environment. – *Eur. J. Mineral.*, **7** (3): 547-552, 6 fig., 1 tab.; Stuttgart.
- SCIENCE APPLICATIONS INTERNATIONAL CORPORATION (2001): Performing Quality Flow Measurements at Mine Sites. – 78 p., 9 fig., 8 tab.; Cincinnati (National Risk Management Research Laboratory).

- SCOFFIN, T. P. & BRADSHAW, C. (2000): The Taphonomic Significance of Endoliths in Dead – versus Live – Coral Skeletons. – *Palaios*, **15** (3): 248-254, 3 fig.; Tulsa.
- SCOTT, R. L. & HAYS, R. M. (1975): Inactive and abandoned Underground Mines – Water Pollution Prevention and Control. – In: MICHAEL BAKER JR. INC. **EPA-440/9-75-007**. – 338 p., 54 fig., 14 tab.; Washington (U.S. Environmental Protection Agency).
- SEASOLVE SOFTWARE (2003): Auto Signal – Pioneering Automated Signal Analyses and Advanced Filtering (Software User's Manual). – p. 1.1-26.8; Framingham (SeaSolve Software).
- SEIDEL, K. (1966): Reinigung von Gewässern durch höhere Pflanzen. – *Naturwissenschaften*, **53** (12): 289-297, 8 fig., 9 tab.; Berlin.
- SEIDL, K. & STÄHLER, G. (1913): Die Technik im oberschlesischen Bergbau und Hüttenbetriebe – Erzbergbau. – In: OBERSCHLESISCHER BERG- UND HÜTTENMÄNNISCHER VEREIN: Handbuch des Oberschlesischen Industriebezirks **II** (2). – p. 651-680, 13 fig.; Kattowitz (Selbstverl. des Oberschlesischen Berg- und Hüttenmännischen Vereins).
- SEIFERT, G. & SCHWANDT, A. (1996): Beitrag zur Bergschadensentwicklung am Beispiel der SW-Flanke des Staßfurter Sattels. – *Exkurs.f. u. Veröfftl. GGW*, **198**: 68-81, 11 fig.; Berlin.
- SELLEY, R. C. (2000): Applied Sedimentology. – 2nd edn., 523 p.; San Diego u.a. (Academic Press).
- SENES CONSULTANTS LIMITED (1994): Acid Mine Drainage – Status of Chemical Treatment and Sludge Management Practices: MEND Report **3.32.1**. – 179 p.; Richmond Hill (The Mine Environment Neutral Drainage [MEND] Program).
- SHARP, K. V. & ADRIAN, R. J. (2004): Transition from laminar to turbulent flow in liquid filled microtubes. – *Experiments in Fluids*, **36** (5): 741-747, 6 fig.; Berlin.
- SHAW, E. M. (1994): Hydrology in Practice. – 3rd edn., 569 p.; London (Chapman & Hall).
- SHEIBACH, R. B., WILLIAMS, R. E. & GENES, B. R. (1982): Controlling Acid Mine Drainage from the Picher Mining District, Oklahoma, United States. – *Int. J. Mine Water*, **1** (1): 45-52, 3 fig.; Granada.
- SHELDON, C. G., STRONGMAN, J. E., WEBER-FAHR, M., MEKHARAT, N. & TALL, M. (2002): Mining and Development – It's Not Over When It's Over: Mine Closure Around the World. – In: WORLD BANK & INTERNATIONAL FINANCE CORPORATION: Global Mining. – 28 p., 1 tab., 3 app.; Washington (The World Bank Group's Mining Department).
- SHEN, J. (1981): Discharge Characteristics of Triangular Thin-Plate Weirs – Studies of Flow to Water over Weirs and Dams. – *Water Supply Paper*, **1617-B**: 45, 20 fig., 3 tab.; Washington.
- SHEPHERD, R. (1993): Ancient Mining. – 494 p., 69 fig.; London (Elsevier).
- SHERWOOD, J. M. & YOUNGER, P. L. (1994): Modelling Groundwater Rebound after Coalfield Closure: An Example from County Durham, UK. – *Proceedings, 5th International Mine Water Congress, Nottingham, U.K.*, **2**: 769-777, 4 fig.; Nottingham.
- SHILLER, A. M. (2003): Syringe Filtration Methods for Examining Dissolved and Colloidal Trace Element Distributions in Remote Field Locations. – *Environ. Sci. Technol.*, **37** (17): 3953-3957, 5 fig., 1 tab.; Washington.
- SIEFER, W. (2000): Heißer Bakterien-Zoo im Granit – Mikrobiologen entdecken ein neues Ökosystem: Die Erde ist kilometertief von Keimen besiedelt. – *Focus* (34): 108-112, 16 fig.; München.
- SIGG, L. & STUMM, W. (1994): Aquatische Chemie – eine Einführung in die Chemie wässriger Lösungen und natürlicher Gewässer. – 3rd edn., 498 p.; Zürich (Verlag der Fachvereine).
- SIMMERS, I. (1998): Groundwater recharge – an overview of estimation “problems” and recent developments. – *Spec. Publ. – Geol. Soc. London*, **130**: 107-115, 3 fig.; London.

- SIMON, F.-G., BIERMANN, V. & CSICSÁK, J. (2003): Uranium removal using elemental iron and hydroxyapatite in permeable reactive barriers. – In: NEL, P. J. L.: Mine Water and the Environment. – p. 43-55, 8 fig., 5 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- SIMON, J. (2003): Konzipierung einer passiven Grubenwasserreinigungsanlage im Hagenbachtal bei Gernrode/Harz. – 169 p., 97 fig., 24 tab.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- SINCLAIR, J. (1958): Water in Mines and Mine Pumps. – 130 p.; London (Pitman).
- SINGER, P. C. & STUMM, W. (1970): Acidic Mine Drainage – Rate-Determining Step. – Science, **167** (3921): 1121-1123, 2 fig.; Washington.
- SINGH, G. & BHATNAGAR, M. (1988): Inhibition of Bacterial Activity in Acid Mine drainage. – Int. J. Mine Water, **7** (4): 13-25, 10 fig., 1 tab.; Nottingham.
- SINGH, R. N. (1986): Mine Inundations. – Int. J. Mine Water, **5** (2): 1-27, 13 fig., 3 tab.; Granada.
- SINGH, R. N. & ATKINS, A. S. (1985): Analytical techniques for the estimation of mine water inflow. – International Journal of Mining Engineering, **3** (1): 65-77, 7 fig., 1 tab., 4 app.
- SINGH, R. N., HIBBERT, S. & FAWCETT, R. J. (1986): Studies in the Prediction of Water Inflows to Longwall Mine Workings. – Int. J. Mine Water, **5** (3): 29-45, 7 fig., 3 tab.; Madrid.
- SINGH, R. N. & ATKINS, A. S. (1984): Application of Analytical Solution to Simulate some Mine Inflow Problems in Underground Coal Mining. – Int. J. Mine Water, **3** (4): 1-27, 8 fig., 15 tab., 1 app.; Madrid.
- SINGH, R. N. & REED, S. M. (1988): Mathematical Modelling for Estimation of Minewater Inflow to a Surface Mining Operation. – Int. J. Mine Water, **7** (3): 1-33, 10 fig., 3 tab.; Nottingham.
- SKEAFF, J. M. & CAMPBELL, H. W. (1980): Physical, chemical and dewatering characteristics of Ba/RaSO₄ sludges from uranium milling. – Report Department of Energy Mines and Resources, **MRP/MSL 80-7; DE83701156/XAB**: 22, 4 fig., 6 tab.; Ottawa.
- SKINNER, E. H. & BECKETT, L. A. (1993): Field Trial for Sealing Abandoned Mine Shafts and Adits with Lightweight Concrete. – Report of Investigations – US Bureau of Mines, **9495**: 34; Washington.
- SKOUSEN, J. G. (1991): Anoxic limestone drains for acid mine drainage treatment. – Green Lands, **21** (4): 30-35; Charleston.
- SKOUSEN, J. G., ROSE, A., GEIDEL, G., FOREMAN, J., EVANS, R. & HELLIER, W. (1998): Handbook of Technologies for Avoidance and Remediation of Acid Mine Drainage. – 130 p., 3 fig.; Morgantown (The National Mine Land Reclamation Center).
- SKOUSEN, J. G., SEXSTONE, A. & ZIEMKIEWICZ, P. F. (2000): Acid Mine Drainage Control and Treatment. – In: BARNHISEL, R. I., DARMODY, R. G. & DANIELS, W. L.: Reclamation of Drastically Disturbed Lands (6). – p. 131-168, 1 fig., 3 tab.; Madison, Wis. (American Society of Agronomy).
- SKOUSEN, J. G., SIMMONS, J., McDONALD, L. M. & ZIEMKIEWICZ, P. (2002): Acid-Base Accounting to Predict Post-Mining Drainage Quality on Surface Mines. – J. Environ. Qual., **31** (6): 2034-2044, 6 fig., 7 tab.; Madison.
- SKOWRONEK, E. & ZMIJ, M. (1977): Okreslenie pochodzenia wody wypływającej zza obmurza szybu znakowaniem rodamina B [Determination of the Origin of Water Flowing Out from Behind a Shaft Lining Traced with Rhodamine B]. – Przegl. Gorn., **33** (1): 13-19, 4 fig., 1 tab.; Gliwice, Poland.
- SKOWRONEK, J., LIPOWCZAN, A. & DUBIŃSKI, J. (2004): NORM IV Conference – Naturally Occurring Radioactive Materials (Abstracts). – Quarterly Research Reports, **1**: 15-153; KATOWICE.

- SMETANA, R., MUZÁK, J. & NOVÁK, J. (2002): Environmental Impact of Uranium ISL in Northern Bohemia. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 703-712, 7 fig.; Heidelberg (Springer).
- SMITH, K. S. & HUYCK, H. L. O. (1999): An Overview of the Abundance, Relative Mobility, Bioavailability, and Human Toxicity of Metals. – In: PLUMLEE, G. S. & LOGSDON, M. J.: The Environmental Geochemistry of Mineral Deposits **6A** (2). – p. 29-70, 17 fig., 16 tab.; Littleton (Society of Economic Geologists).
- SMITH, K. S., FILIPEK, L. H., UPDEGRAFF, D. M. & PAPP, C. S. E. (1988): Distribution of microorganisms and selected metals in mine drainage, stream water, and sediment. – Open-File Report – US Geological Survey, **OF 86-0481**: D11-D15, 3 fig.; Reston.
- SMITH, M. W. & BRADY, K. B. C. (1998): Alkaline Addition. – In: THE PENNSYLVANIAN DEPARTMENT OF ENVIRONMENTAL PROTECTION: Coal mine drainage prediction and pollution prevention in Pennsylvania (13). – p. 13.1-13.13, 4 fig., 1 tab.; Harrisburg (Pennsylvanian Bureau of Mining and Reclamation).
- SMITH, M. W. (1988): Establishing baseline pollution load from preexisting pollutional discharges for remining in Pennsylvania. – Bureau of Mines Information Circular, **IC-9183**: 311-318, 8 fig., 1 tab.; Washington.
- SOBEK, A. A., SCHULLER, W. A., FREEMAN, J. R. & SMITH, R. M. (1978): Field and laboratory methods applicable to overburden and minesoils **600/2-78-054**. – 203 p.; Washington (U.S. Environmental Protection Agency).
- SOHRABI, M. (1998): The State-of-the-art on Worldwide Studies in some Environments with Elevated Naturally Occurring Radioactive Materials (NORM). – Appl. Radiat. Isot., **49** (3): 169-188, 5 fig., 4 tab.; Oxford.
- SOMERFIELD, P. J., GEE, J. M. & WARWICK, R. M. (1994): Benthic Community Structure in Relation to an Instantaneous Discharge of Waste Water from a Tin Mine. – Mar. Pollut. Bull., **28** (6): 363-369, 6 fig., 2 tab.; Oxford-New York.
- SPENCE, C. & MCPHIE, M. (1997): Streamflow measurement using salt dilution in tundra streams, Northwest Territories, Canada. – J. Am. Water Resour. Assoc., **33** (2): 285-292, 6 fig., 1 tab.; Herndon.
- SPIESSL, S. M., PROMMER, H., SAUTER, M. & ZHENG, C. (2002): Numerical simulation of uranium transport in flooded underground mines. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 277-286, 4 fig., 1 tab.; Heidelberg (Springer).
- SPRUILL, T. B. (1983): Effects of abandoned lead and zinc mines on water quality in Kansas. – US Geological Survey Professional Paper, **1375**: 1-226; Washington, DC.
- SPRUILL, T. B. (1987): Assessment of Water Resources in Lead-Zinc Mined Areas in Cherokee County, Kansas, and Adjacent Areas. – US Geological Survey Water-Supply Paper, **W 2268**: 1-68, 23 fig., 26 tab.; Reston, VA.
- SROKA, A., LÖBEL, K.-H., KLAPPERICH, H., TONDERA, D., MEIER, G. & BUSCH, W. (2003): 3. Altbergbau Kolloquium – Tagungsband. – 443 p.; Essen (Glückauf).
- STARK, L. R., WENERICK, W. R., WILLIAMS, F. M., STEVENS JR, S. E. & WUEST, P. J. (1994): Restoring the capacity of spent mushroom compost to treat coal mine drainage by reducing the inflow rate: a microcosm experiment. – Water Air Soil Pollut., **75** (3-4): 405-420, 9 fig., 1 tab.; Dordrecht.
- STARK, L. R., WILLIAMS, F. M., WENERICK, W. R., WUEST, P. J. & URBAN, C. (1996): The effects of substrate type, surface water depth, and flow rate on manganese retention in mesocosm wetlands. – J. Environ. Qual., **25** (1): 97-106, 10 fig., 3 tab.; Madison.
- STEFAN, H. G. & GU, R. (1992): Efficiency of Jet Mixing of Temperature-Stratified Water. – J. Environm. Engineering, **118** (3): 363-379, 17 fig., 1 tab.; New York.

- STEMPROK, M. (1992): Geochemical development of the Krusné hory/Erzgebirge granite pluton exemplified on its Czechoslovak part. – *Geophys. u. Geol.*, **IV** (4): 51-63, 10 fig.; Berlin.
- STENGEL-RUTKOWSKI, W. (1993): Trinkwasserversorgung aus Grubengebäuden des ehemaligen Roteisensteinbergbaus im Lahn-Dill-Kreis (Rheinisches Schiefergebirge). – *Geol. Jb. Hessen*, **121**: 125-140, 1 fig., 2 tab.; Wiesbaden.
- STENGEL-RUTKOWSKI, W. (2002): Trinkwasserversorgung aus Grubengebäuden des ehemaligen Bergbaus im Rheingau-Taunus-Kreis (Rheinisches Schiefergebirge). – *Jb. nass. Ver. Naturk.*, **123**: 125-138, 3 fig., 3 tab.; Wiesbaden.
- STEVENS, C. L. & LAWRENCE, G. A. (1998): Stability and meromixis in a water-filled mine pit. – *Limnol. Oceanogr.*, **43** (5): 946-954, 10 fig.; Waco.
- STOBER, I. (1995): Die Wasserführung des kristallinen Grundgebirges. – 191 p., 81 fig., 16 tab.; Stuttgart (Enke).
- STODDART, P. R., CADUSCH, P. J., PEARCE, J. B., VUKOVIC, D., NAGARAJAH, C. R. & BOOTH, D. J. (2005): Fibre optic distributed temperature sensor with an integrated background correction function. – *Meas. Sci. Technol.*, **16**: 1299-1304, 6 fig., 1 tab.; Bristol.
- STOERTZ, M. W., HUGHES, M. L., WANNER, N. S. & FARLEY, M. E. (2001): Long-term water quality trends at a sealed, partially flooded underground mine. – *Environ. Eng. Geosci.*, **7** (1): 51-65, 12 fig.; Texas.
- STOICA, L., IUHAS, T., BARDAN, N., RAZVAN, A., FILIP, D. & OPROIU, G. (1998): Removal of ²²⁶Ra(II) from the Uranium Mine and Pond Waters. – *Proceedings, Uranium-Mining and Hydrogeology II*, Freiberg, Germany; *GeoCongress*, **5**: 331-338, 4 fig., 4 tab.; Köln.
- STOLL, R. & BAUER, D. (1991): Anwendung geophysikalischer Verfahren zur Kontrolle und Überwachung technologischer Teilprozesse im untertägigen Erzbergbau. – *Neue Bergbautechnik*, **21** (12): 427-431, 11 fig., 1 tab.; Leipzig.
- STOLTZ, E. & GREGER, M. (2002): Cottongrass Effects on Trace Elements in Submersed Mine Tailings. – *J. Environ. Qual.*, **31** (5): 1477-1483, 5 fig., 2 tab.; Madison.
- STRASKRABA, V. & MORAN, R. E. (1990): Environmental Occurrence and Impacts of Arsenic at Gold Mining Sites in the Western United States. – *Int. J. Mine Water*, **9** (1-4): 181-191, 2 fig., 1 tab.; Castro Verde.
- STRONGMAN, J. (2000): Mine Closure – An Overview of the Issues. – p. 1-29; Jakarta (Presentation Indonesia Mine Closure Workshop).
- STRUNZ, H. & NICKEL, E. H. (2001): Strunz Mineralogical Tables – Chemical-Structural Mineral Classification System. – 9th edn., 870 p.; Stuttgart (Schweizerbart).
- STUART, W. T. & SIMPSON, T. A. (1961): Variations of pH with depth in anthracite mine-water pools in Pennsylvania; Article 37. – *US Geological Survey Professional Paper*, **0424-B**: B82-B84, 1 tab.; Reston.
- STUMM, W. & MORGAN, J. J. (1996): Aquatic chemistry – Chemical Equilibria and Rates in Natural Waters. – 3rd edn., 1022 p.; New York (Wiley & Sons).
- SUMAN, A. L., FREEMAN, S., ERDNER, B., SHARP, M., MEIER, L., DATIN, D., DALGARN, J., GARVIN, M., WOOD, F., KEHELEY, E., RAINBOLT, J. & CRAWFORD, J. (2000): Mine Shaft Subcommittee Final Report to Governor Frank Keating's Tar Creek Superfund Task Force. – 64 p., 9 fig., 2 tab., 3 app.; Oklahoma City (Tar Creek Superfund Task Force Mine Shaft Subcommittee).
- SUMIOKA, S. S. (1991): Quality of water in an inactive uranium mine and its effects on the quality of water in Blue Creek, Stevens County, Washington, 1984-85. – *Water resources investigations report*, **WRI 89-4110**: 1-70, 8 fig., 26 tab.; Reston.
- SUNDBORG, Å. (1956): The River Klarälven: a study of fluvial processes. – *Geografiska Annaler*, **38**: 127-316; Stockholm.

- SUSTAINABLE DEVELOPMENT ADVISORY COUNCIL & COPPER MINES OF TASMANIA (1995): Copper Mines of Tasmania Project – Final assessment report. – 93 p.; Hobart (The Council).
- ŠUŠTERŠIČ, F. (1996): Poljes and Caves of Notranjska. – *Acta Carsologica*, **15**: 252-289, 6 fig., 3 tab.; Ljubljana.
- SYSTAT SOFTWARE INC. (2002): TableCurve 2D 5.01 for Windows User's Manual – Automated curve fitting and equation discovery. – p. 1.1-13.14; Richmond (SYSTAT Software Inc. Marketing Department).
- SZUROWSKI, H. (1968): Petrographische, mineralogische und geochemische Untersuchungen in den Graptolithenschiefern im Raum südlich von Ronneburg. – 168 p., 34 fig., 19 tab.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- TAKAI, K., MOSER, D. P., ONSTOTT, T. C., SPOELSTRA, N., PFIFFNER, S. M., DOHNALKOVA, A. & FREDRICKSON, J. K. (2001): *Alkaliphilus transvaalensis* gen. nov., sp. nov., an extremely alkaliphilic bacterium isolated from a deep South African gold mine. – *Int. J. Syst. Evol. Microbiol.*, **51**: 1245-1256, 4 fig., 3 tab.; Reading.
- TAMME, S. (2002): Naturnahe Reinigung kontaminierter Bergwerkswässer – Bemessung und Entwurf eines konstruierten Feuchtgebietes bei Gernrode/Harz. – 132 p., 44 fig., 13 tab., 18 app.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- TANG, D. H., FRIND, E. O. & SUDICKY, E. A. (1981): Contaminant Transport in Fractured Porous Media: Analytical Solution for a Single Fracture. – *Water Resour. Res.*, **17** (3): 555-564, 10 fig.; Washington.
- TAR CREEK SUPERFUND TASK FORCE WATER QUALITY SUBCOMMITTEE (2000a): Final Report Governor Frank Keating's Tar Creek Superfund Task Force Water Quality Subcommittee Surface & Ground Water Monitoring. – 21 p., 17 fig., 42 tab., 4 app.; Oklahoma City (Tar Creek Superfund Task Force Water Quality Subcommittee).
- TAR CREEK SUPERFUND TASK FORCE WATER QUALITY SUBCOMMITTEE (2000b): Final Report Governor Frank Keating's Tar Creek Superfund Task Force Water Quality Subcommittee Water Quality Improvement Alternatives. – 35 p., 2 fig., 4 tab.; Oklahoma City (Tar Creek Superfund Task Force Water Quality Subcommittee).
- TAYLOR, B. J. (1978): British Regional Geology – Northern England. – 4th edn., 125 p., 29 fig., 4 tab., 13 plates; London (Her Majesty's Stationery Office).
- TAYLOR, J. & WARING, C. (2001): The Passive Prevention of ARD in Underground Mines by Displacement of Air with a Reducing Gas Mixture: GaRDS. – *Mine Water Env.*, **20** (1): 2-7, 1 fig.; Berlin.
- TAYLOR, J. R., WEAVER, T. R., MCPHAIL, D. C. & MURPHY, N. C. (1996): Characterisation and impact assessment of mine tailings in the King River system and delta, Western Tasmania. – In: SUPERVISING SCIENTIST **105**. – 135 p., 32 fig., 14 tab., 8 app.; Canberra (Supervising Scientist Report).
- TEAF, C., MERKEL, B., MULISCH, H.-M., KUPERBERG, M. & WCISLO, E. (2006): Industry, mining and military sites. – In: WORLD HEALTH ORGANISATION: Protecting groundwater for health: managing the quality of drinking-water sources (11, 23). – 640 p.; London (IWA Publishing).
- TERZAGHI, K. (1925): Erdbaumechanik auf bodenphysikalischer Grundlage. – 399 p., 65 fig., 63 tab.; Leipzig (Deuticke).
- THE PENNSYLVANIAN DEPARTMENT OF ENVIRONMENTAL PROTECTION (1998): Coal mine drainage prediction and pollution prevention in Pennsylvania. – 398 p., 68 fig., 193 tab.; Harrisburg (Pennsylvanian Bureau of Mining and Reclamation).
- THEIS, C. V. (1935): The relation between the lowering of the piezometric surface and the rate and duration of discharge of a well using groundwater storage. – *Trans. Amer. Geophys. Union*, **16**: 519-524, 3 fig.; Washington.

- THEIS, C. V. (1940): The source of water derived from wells – Essential factors controlling the response of an aquifer to development. – *Civil Engineering*, **10** (5): 277-280, 5 fig.; New York.
- THIEME, G. (2005): Wismut muss neuen Stollen auffahren – Flutung der Grube Gittersee seit über zwei Jahren gestoppt – Grund: unkontrollierter Wasseraustritt. – *Freie Presse*, **20.12.2005**: 2; Chemnitz.
- THIES, M. (2003): Vereinbarkeit von Fledermausschutz und berghistorischer Forschung. – In: REGER, K.: 10 Jahre – Besucherbergwerk Grube Wohlfahrt. – p. 178-180, 1 fig., 1 tab.; Rescheid (Tagungsband – 6. Internationaler Bergbau-Workshop).
- THOMAS, C. W. (1959): Errors in Measurement of Irrigation Water. – *Trans. American Society of Civil Engineers*, **124**: 319-332, 6 fig., 4 tab.; Reston.
- THOMAS, R. C. & ROMANEK, C. S. (2002): Passive Treatment of low-pH, Ferric Iron-Dominated Acid Rock Drainage in a Vertical Flow Wetland I: Acidity Neutralization and Alkalinity Generation. – p. 723-748, 3 fig., 3 tab.; Lexington (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- THOMAS, R. C., ROMANEK, C. S. & PADDOCK, L. S. (2004): Trace Metal Retention in LBOS-Amended, Vertical-Flow Constructed Wetlands Treating Low pH, Ferric Iron Dominated Acid Rock Drainage. – p. 1862-1893, 12 fig., 2 tab.; Lexington (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- TIETZE, K. (1978): Geophysikalische Untersuchung des Kivusees und seiner ungewöhnlichen Methangaslagerstätte – Schichtung, Dynamik und Gasgehalt des Seewassers. – 149 p.; Kiel (Naturwiss. Diss. Univ. Kiel).
- TIMMERMANN, J. G. & LANGAAS, S. (2004): Environmental Information in European Transboundary Water Management. – 258 p.; London (IWA Publishing).
- TIWARY, R. K. (2001): Environmental Impact of coal Mining on Water Regime and its Management. – *Water Air Soil Pollut.*, **132** (1-2): 185-199, 1 fig., 3 tab.; Dordrecht.
- TORRENCE, C. & COMPO, G. P. (1998): A Practical Guide to Wavelet Analysis. – *Bulletin of the American Meteorological Society*, **79** (1): 61-78, 11 fig., 2 tab.; Boston.
- TSANG, Y. W. & TSANG, C. F. (1987): Channel Model of Flow Through Fractured Media. – *Water Resour. Res.*, **23** (3): 467-479, 15 fig.; Washington.
- TUOVINEN, O. H. & BHATTI, T. M. (1999): Microbiological leaching of uranium ores. – *Minerals Metallurg. Process.*, **16** (4): 51-60, 4 fig., 1 tab.; Littleton.
- TUREK, M. (2004): Electrodialytic desalination and concentration of coal-mine brine. – *Desalination*, **162** (1-3): 355-359, 1 fig., 4 tab.; Amsterdam.
- TUREK, M. & GONET, M. (1996): Nanofiltration in the utilization of coal-mine brines. – *Desalination*, **108**: 171-177, 1 tab.; Amsterdam.
- TUREK, M., MROWIEC-BIALON, J. & GNOT, W. (1995): Utilization of coal mine brines in the chlorine production process. – *Desalination*, **101**: 57-67, 2 fig., 8 tab.; Amsterdam.
- TUTU, H., CUKROWSKA, E., MCCARTHY, T. S., MPHEPHU, N. F. & HART, R. (2003): Determination and modelling of geochemical speciation of uranium in gold mine polluted land in South Africa. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 137-155, 10 fig.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- TWINING, J. R. & CAMERON, R. F. (1997): Decision-making processes in ecological risk assessment using copper pollution of Macquarie Harbour from Mt. Lyell, Tasmania, as a case study. – *Hydrobiologia*, **5**: 207-218, 5 fig., 2 tab.; Dordrecht.
- UERPMANN, E.-P. (1980): Hydrogeologische Fragen bei der Endlagerung radioaktiver Abfälle. – 128 p., 21 fig., 5 tab.; Clausthal (unpubl. Diss. TU Clausthal).

- UHLMANN, W., BÜTTCHER, H., TOTSCHKE, O. & STEINBERG, C. E. W. (2004): Buffering of Acidic Mine Lakes: The Relevance of Surface Exchange and Solid-Bound Sulphate. – *Mine Water Env.*, **23** (1): 20-27, 7 fig., 4 tab.; Berlin.
- UNGER, K. (2002): Hydrodynamische Verhältnisse im gefluteten Unterbau des Bergwerks Großkogel/Tirol – Numerische Modellierung mit ANSYS-FLOTTRAN. – 117 p., 52 fig., 37 tab., 2 app.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- UNITED NATIONS CONFERENCE ON ENVIRONMENT AND DEVELOPMENT [UNCED] (1992): Agenda 21. – In: UNITED NATIONS **A/CONF. 151/26**. – 600 p.; New York (United Nations).
- UNITED NATIONS DEPARTMENT OF ECONOMIC AND SOCIAL AFFAIRS [UNDESA] & UNITED NATIONS ENVIRONMENT PROGRAMME INDUSTRY AND ENVIRONMENT [UNEP] (2002): Berlin II Guidelines for Mining and Sustainable Development. – 52 p., 1 tab., 11 app. (United Nations).
- UNITED NATIONS DEPARTMENT OF TECHNICAL CO-OPERATION FOR DEVELOPMENT & DEVELOPMENT POLICY FORUM OF THE GERMAN FOUNDATION FOR INTERNATIONAL DEVELOPMENT (1992): Mining and the Environment – The Berlin Guidelines – A study based on an international Round Table in June 1991. – 180 p.; London (Mining Journal Books).
- UNITED NATIONS ENVIRONMENT PROGRAMME [UNEP] (1994): Environmental management of mine sites – Training Manual. – Technical report, **30**; Kenia.
- UNITED STATES DEPARTMENT OF AGRICULTURE (1979): Field manual for research in agricultural hydrology. – In: BRAKENSIEK, D. L., OSBORN, H. B. & RAWLS, W. J.: US Department of Agriculture Handbook **224**. – 547 p., 355 fig., 40 tab.; Washington (US Department of Agriculture).
- UNLAND, W., ECKART, M., PAUL, M., KUHN, W. & OSTERMANN, R. (2002): Groundwater rebound compatible with the aquatic environment – technical solutions at WISMUT's Ronneburg mine. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: Uranium in the Aquatic Environment. – p. 667-676, 5 fig., 1 tab.; Heidelberg (Springer).
- U.S. ARMY CORPS OF ENGINEERS (2004): Tar Creek and Lower Spring River Watershed Management Plan – Reconnaissance Phase Draft. – 21 p., 10 fig., 9 app. (Unpublished Report).
- U.S. DEPARTMENT OF THE INTERIOR – BUREAU OF RECLAMATION (2001): Water Measurement Manual. – 3 revisedth edn., 272 p., 155 fig., 25 tab.; Washington (U.S. Government Printing Office).
- U.S. ENVIRONMENTAL PROTECTION AGENCY (1994): Guidance for the Data Quality Objectives Process **EPA-600/R-96/055**. – 69 p., 14 fig., 4 tab.; Washington (U.S. Environmental Protection Agency).
- U.S. ENVIRONMENTAL PROTECTION AGENCY (2000): Abandoned Mine Site Characterization and Cleanup Handbook **EPA 910-B-00-001**. – 408 p., 5 fig., 18 tab.; Denver, San Francisco, Seattle (U.S. Environmental Protection Agency).
- U.S. ENVIRONMENTAL PROTECTION AGENCY (2003): EPA and Hardrock Mining – A Source Book for Industry in the Northwest and Alaska. – 57 p., 3 fig., 11 tab.; Seattle (U.S. Environmental Protection Agency).
- U.S. ENVIRONMENTAL PROTECTION AGENCY & OFFICE OF WATER (1988): REMINE – Coal Remining – Best Professional Judgement Analyses. – 80 p.; Washington (U.S. Environmental Protection Agency).
- U.S. ENVIRONMENTAL PROTECTION AGENCY, OFFICE OF WATER & OFFICE OF SCIENCE AND TECHNOLOGY ENGINEERING AND ANALYSIS DIVISION (2001): Coal Remining best Management Practices Guidance Manual **EPA-821-B-01-010**. – 618 p., 71 fig., 76 tab.; Washington (U.S. Environmental Protection Agency).

- US GEOLOGICAL SURVEY (2000): Interim report on the scientific investigations in the Animas River watershed, Colorado to facilitate remediation decisions by the U.S. Bureau of Land Management and the U.S. Forest Service. – Open-File Report – US Geological Survey, **OF 00-245**: 1-34, 14 fig., 4 tab.; Reston.
- VALENZUELA, F., BASUALTO, C., TAPIA, C. & SAPAG, J. (1999): Application of hollow-fiber supported liquid membranes technique to the selective recovery of a low content of copper from a Chilean mine water. – *J. Membr. Sci.*, **155** (1): 163-168, 3 fig.; Amsterdam.
- VALENZUELA, F., VEGA, M. A., YANEZ, M. F. & BASUALTO, C. (2002): Application of a mathematical model for copper permeation from a Chilean mine water through a hollow fiber-type supported liquid membrane. – *J. Membr. Sci.*, **204** (1-2): 385-400, 9 fig., 7 tab.; Amsterdam.
- VAN ZYL, H. C., MAREE, J. P., VAN NIEKERK, A. M., VAN TONDER, G. J. & NAIDOO, C. (2001): Collection, Treatment and re-use of Mine Water in the Olifants River Catchment. – *J. S. Afr. Inst. Min. Metall.*, **101** (1): 41-46, 2 fig., 7 tab.; Johannesburg.
- VASAN, S. S., MODAK, J. M. & NATARAJAN, K. A. (2001): Some recent advances in the bioprocessing of bauxite. – *Int. J. Miner. Process.*, **62** (1-4): 173-186, 3 fig., 4 tab.; Amsterdam.
- VEGA, E., LESIKAR, B. & PILLAI, S. D. (2003): Transport and Survival of Bacterial and Viral Tracers Through Submerged-flow Constructed Wetland and Sand-filter System. – *Bioresour. Technol.*, **89** (1): 49-56.
- VEIL, J. A., KUPAR, J. M. & PUDER, M. G. (2003): Use of Mine Pool Water for Power Plant Cooling. – Für: U.S. DEPARTMENT OF ENERGY. – 52 p., 7 fig., 10 tab.; Argonne (Argonne National Laboratory).
- VEIT, U., PETZOLD, B. & PIEHL, H.-D. (1987): Klimatologische Normalwerte 1951/80: Klimadaten der Deutschen Demokratischen Republik – Ein Handbuch für die Praxis – Reihe B **14**. – 111 p., 47 tab.; Potsdam (Meteorologischer Dienst der Deutschen Demokratischen Republik).
- VESELIČ, M. & NORTON, P. J. (1997): Successes and Failures in Prediction of Groundwater Rebound in Abandoned Mines – Some Theory and Case Studies. – Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia, **1**: 127-147, 4 fig.; Ljubljana.
- VESELIČ, M., PREGL, M., VIŽINTIN, G. & SUPOVEC, I. (2001): Environmental Regulation of Mine Waters in the European Union – D 1 National Case Studies – Slovenia: ERMITE Final Report – EU R&D Project EVK1-CT-2000-00078. – 25 p., 7 fig., 4 tab.; Ljubljana (IRGO Slovenia).
- VIEIRA, F. M. C. C., DIERING, D. H. & DURRHEIM, R. J. (2001): Methods to Mine the Ultra-Deep Tabular Gold-Bearing Reefs of the Witwatersrand Basin, South Africa. – In: HUSTRULID, W. A. & BULLOCK, R. L.: *Underground Mining Methods – Engineering Fundamentals and International Case Studies* (76). – p. 691-704, 21 fig., 3 tab.; Littleton (Society for Mining, Metallurgy, and Exploration).
- VILLAS BÔAS, R. C. & BARRETO, M. L. (2000): Mine Closure in Iberoamerica. – 455 p.; Rio de Janeiro (CYTED – IMACC – UNIDO).
- VINCI, B. J. & SCHMIDT, T. W. (2001): Passive, Periodic Flushing Technology for Mine Drainage Treatment Systems. – In: BARNHISEL, R. I., BUCHANAN, B. A., PETERSON, D. & PFEIL, J. J.: *Land Reclamation – A Different Approach* **2**. – p. 611-625, 6 fig.; Lexington (Proceedings, Annual National Meeting – American Society for Surface Mining and Reclamation).
- VOGT, W. (2001): *Alte Vorarlberger Heilbäder – Eine Reise durch die Vorarlberger Bäderlandschaft*. – 1st edn., 187 p., 193 fig.; Feldkirch (Benvenuti).
- VON ROHDEN, C. & ILMBERGER, J. (2001): Tracer experiment with sulfur hexafluoride to quantify the vertical transport in a meromictic pit lake. – *Aquatic Sciences*, **63** (4): 417-431, 9 fig.; Basel.
- VOZNJUK, G. G. & GORSHKOV, V. A. (1983): Mine Water Utilization in the USSR National Economy. – *Int. J. Mine Water*, **2** (1): 23-30, 1 fig.; Granada.

- VUTUKURI, V. S. & SINGH, R. N. (1993): Recent Developments in Pumping Systems in Underground Metalliferous Mining. – *Mine Water Env.*, **12** (1-4): 71-94, 18 fig.; Wollongong.
- VUTUKURI, V. S. & SINGH, R. N. (1995): Mine Inundation – Case Histories. – *Mine Water Env.*, **14** (1-4): 107-130, 12 fig., 2 tab.; Wollongong.
- WACKWITZ, T. (2002): Multitracerversuch im gefluteten Blindschacht des Bergwerks Großkogel/Tirol: Vorbereitung, Durchführung und GIS unterstützte Auswertung. – 104 p., 37 fig., 17 tab., 48 app.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- WALDER, I. F., NILSSEN, S., RÄISÄNEN, M. L., HEIKKINEN, P., PULKKINEN, K., KORKKA-NIEMI, K., SALONEN, V.-P., DESTOUNI, G., HASCHE, A., WOLKERSDORFER, CH., WITKOWSKI, A. J., BLACHÈRE, A., MOREL, S., LEFORT, D., MIDŽIĆ, S., SILAJDŽIĆ, I., COULTON, R. H., WILLIAMS, K. P., REES, B., HALLBERG, K. B. & JOHNSON, D. B. (2005): Contemporary Reviews of Mine Water Studies in Europe, Part 2. – *Mine Water Env.*, **24** (1): 2-37, 10 fig., 13 tab.; Berlin.
- WALTER, R., GIESE, P., WALTHER, H. W. & DILL, H. G. (1992): *Geologie von Mitteleuropa*. – 5th edn., 561 p.; Stuttgart (Schweizerbart).
- WALTON-DAY, K. (1999): Geochemistry of the Processes that Attenuate Acid Mine Drainage in Wetlands. – In: PLUMLEE, G. S. & LOGSDON, M. J.: *The Environmental Geochemistry of Mineral Deposits* **6A** (10). – p. 215-228, 4 fig.; Littleton (Society of Economic Geologists).
- WARING, C. L. & TAYLOR, J. R. (1999): A new technique for building in-situ sub-surface hydrologic barriers: NBT. – In: FERNÁNDEZ RUBIO, R.: *Mine, Water & Environment II*. – p. 663-665, 3 fig., 1 tab.; Sevilla (International Mine Water Association).
- WATERHOUSE, J., MURIE, A. & THOMSON, D. (2003): Groundwater inflows and management at Emily Ann and Maggie Hays underground nickel mines, Western Australia. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 73-81, 5 fig.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- WATZLAF, G. R., SCHROEDER, K. T. & KAIRIES, C. L. (2000): Long-term performance of anoxic limestone drains. – *Mine Water Env.*, **19** (2): 98-110, 3 fig., 5 tab.; Berlin.
- WATZLAF, G. R., KAIRIES, C. L., SCHROEDER, K. T., DANEHY, T. & BEAM, R. (2002): Quantitative Results from the Flushing of four Reducing and Alkalinity-Producing Systems. – *Proceedings, West Virginia Surface Mine Drainage Task Force Symposium*, **23**: 1-11, 5 fig., 2 tab.; Morgantown.
- WAUGH, W. J., SMITH, G. M., BERGMAN-TABBERT, D. & METZLER, D. R. (2001): Evolution of Cover Systems for the Uranium Mill Tailings Remedial Action Project, USA. – *Mine Water Env.*, **20** (4): 190-197, 3 fig.; Berlin.
- WEBER, L. (2004): Die Nassereither Bergwasserexplosion als Folge einer unzureichenden Schließung eines Altbergbaus. – In: MEIER, G., SROKA, A., LÖBEL, K.-H., KLAPPERICH, H., TONDERA, D., BUSCH, W., WAGNER, H. & RANDJBAR, B.: *4. Altbergbau-Kolloquium – Tagungsband*. – p. 270-280, 9 fig.; Essen (Glückauf).
- WEBSTER, J. G., SWEDLUND, P. J. & WEBSTER, K. S. (1998): Trace metal adsorption onto acid mine drainage iron oxide. – In: AREHART, G. B. & HULSTON, J. R.: *Water-Rock Interaction* **9**. – p. 951-954, 2 fig.; Rotterdam u.a. (Balkema).
- WEIDIG, M. (1912): *Radioaktive Wässer in Sachsen IV*. – p. 217-359, 73 fig., 81 tab.; Freiberg (Craz & Gerlach).
- WEIERSBYE, I. M. & WITKOWSKI, E. T. F. (2003): Acid rock drainage (ARD) from gold tailings dams on the Witwatersrand basin impacts on tree seed fate, viability, inorganic content and seedling morphology. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 311-328, 4 fig., 3 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).

- WEISSENBORN, B. & BENTHIN, A. (2003): Idee, Planung und bergmännische Arbeiten zur Nutzung untertägiger Wetter für die Klimatisierung des Kreiskrankenhauses Freiberg. – In: SROKA, A., LÖBEL, K.-H., KLAPPERICH, H., TONDERA, D., MEIER, G. & BUSCH, W.: 3. Altbergbau Kolloquium. – p. 345-358; Essen (Glückauf).
- WENDEL, G. (1998): Radioactivity in mines and mine water – sources and mechanisms. – J. S. Afr. Inst. Min. Metall., **98** (2): 87-92, 4 fig., 2 tab.; Johannesburg.
- WENDLAND, E. & HIMMELSBACH, T. (2002): Underground Waste Repositories in Exhausted Coal Mines – Recent Developments. – Mine Water Env., **21** (4): 160-167, 11 fig.; Berlin.
- WERNER, F., BREMSTAHLER, F. & LUCKNER, L. (1999): Predictive Modeling of Water Quality in Post Mining Environments. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 445-448, 6 fig., 1 tab.; Sevilla (International Mine Water Association).
- WERNER, F., BILEK, F. & LUCKNER, L. (2001): Implications of Predicted Hydrologic Changes on Lake Senftenberg as Calculated Using Water and Reactive Mass Budgets. – Mine Water Env., **20** (3): 129-139, 8 fig., 4 tab.; Berlin.
- WEST VIRGINIA DIVISION OF ENVIRONMENTAL PROTECTION – OFFICE OF MINING AND RECLAMATION (2001): West Virginia Acid Mine Drainage Study 2000 – A Quantitative Inventory and Interpretive Review of Water Quality on Active Mining Operations in West Virginia. – 15 p., 10 fig., 11 tab.; Morgantown (West Virginia Division of Environmental Protection).
- WEYER, J. (2003): Collapse of the adit “Rothschoenberger Stollen” as a result of the flood on 12/13 August 2002. – In: NEL, P. J. L.: Mine Water and the Environment. – p. 477-492, 24 fig., 1 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- WHITWORTH, K. R. (2002): The monitoring and modelling of mine water recovery in UK coalfields. – Spec. Publ. – Geol. Soc. London, **198**: 61-73, 15 fig., 2 tab.; London.
- WILDEMAN, T., BRODIE, G. & GUSEK, J. (1993): Wetland Design for Mining Operations. – 408 p., 67 fig., 40 tab.; Richmond, BC (BiTech).
- WILLIAMS, D. J., BIGHAM, J. M., CRAVOTTA, C. A., TRAINA, S. J., ANDERSON, J. E. & LYON, J. G. (2002): Assessing Mine Drainage pH from the Color and Spectral Reflectance of Chemical Precipitates. – Appl. Geochem., **17** (10): 1273-1286; Oxford.
- WILLIAMS, M. (2001): Arsenic in mine waters: an international study. – Environ. Geol., **40** (3): 267-278, 8 fig., 2 tab.; New York, Berlin.
- WILLIAMS, R. E. & KIRSCHNER, F. E. J. (1992): Hydrogeological Evaluation of a Prototype In-situ Leach Cell in Unsaturated Lead-Zinc Mine and Mill Wastes. – Mine Water Env., **11** (4): 23-33, 7 fig.; Wollongong.
- WILSON, D. & BROWN, D. (1997): Fast-track Provision of St. Aidan’s Reverse-Osmosis Water-Treatment Plant. – Water Environ. Manage., **11**: 271-276, 2 fig., 4 tab.; London.
- WILSON, G. V. & FLETT, J. S. (1921): The Lead, Zinc, Copper and Nickel Ores of Scotland – Islay. – Memoirs of the Geological Survey, Scotland, **XVIII**: 65-73, 1 fig., 1 tab.; Edinburgh.
- WINKELMANN, H. (1956): Schwazer Bergbuch – Codex Vindobonensis 10.852. – 284 p.; Lünen (Gewerkschaft Eisenhütte Westfalia).
- WINKLER, C. (2001): Einsatz passiver Reinigungsmethoden zur Behandlung von Bergbauwässern am Beispiel der Grube Strassberg/Harz. – Wissenschaftliche Mitteilungen, **16**: 179-192, 4 fig., 3 tab.; Freiberg.
- WIREMAN, M., DAVIES, G. J. & STOVER, B. K. (2001): Characterizing Ground-Water Flow Paths in high-altitude fractured Rock Settings impacted by Mining Activities. – In: KUEPER, B. H., NOVAKOWSKI, B. H. & REYNOLDS, D. A. – p. 1-11 [CD-ROM], 9 fig.; Toronto (Proceedings, International Conference on Fractured rock).

- WIRSING, G. (1995): Hydrogeologische Untersuchungen im Grubengebäude des Schauinslandbergwerks – Beurteilung des Grundwasservorkommens und Abgrenzung eines Wasserschutzgebietes für die Quelle 14 der Freiburger Energie- und Wasserversorgungs-AG. – Für: UMWELTSCHUTZAMT STADT FREIBURG I. BR. – 45 p., 7 tab., 16 app.; Freiburg (Geologisches Landesamt Baden-Württemberg).
- WISMUT GMBH (1994): Ergebnisse der Umweltüberwachung und Sanierungstätigkeit 1993 – Sanierungsbetrieb Aue. – 68 p., 16 app.; Chemnitz (Eigenverlag Wismut GmbH).
- WISOTZKY, F. (2001): Prevention of Acidic Groundwater in Lignite Overburden Dumps by the Addition of Alkaline Substances: Pilot-scale Field Experiments. – *Mine Water Env.*, **20** (3): 122-128, 6 fig., 2 tab.; Berlin.
- WITKOWSKI, A. J., MOTYKA, J. & WRÓBEL, J. (2003): Assessment of potential risk of groundwater contamination in areas subjected to the intensive mining drainage – Case study from Poland – Olkusz Zn-Pb ore mining region. – In: NEL, P. J. L.: *Mine Water and the Environment*. – p. 221-234, 9 fig., 4 tab.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- WITTRUP, M. B., KYSER, T. K. & DANYLUK, T. (1986): The use of stable isotopes to determine the source of brine in Saskatchewan potash mines. – Special Publication – Saskatchewan Geological Society, **8**: 159-165, 5 fig.; Regina.
- WÖBKING, H. (1982): Untertägige Eigenpotentialmessungen in den Bergbauen Großkogel bei Brixlegg und Falkenstein bei Schwaz. – *Berg- u. Hüttenm. Mh.*, **127** (12): 476-483, 8 fig.; Wien.
- WOLKERSDORFER, CH. (1991): Geschichte des Bergbaues im westlichen Mieminger Gebirge/Tirol. – *Der Aufschluss*, **42** (6): 359-379, 18 fig.; Heidelberg.
- WOLKERSDORFER, CH. (1995): 2. Bericht zum Kooperationsvertrag zwischen der WISMUT GmbH und der Abteilung Ingenieurgeologie. – 107 p., 45 fig., 16 tab.; Clausthal (unpubl. Report).
- WOLKERSDORFER, CH. (1996): Hydrogeochemische Verhältnisse im Flutungswasser eines Uranbergwerks – Die Lagerstätte Niederschlema/Alberoda. – *Clausthaler Geowissenschaftliche Dissertationen*, **50**: 1-216, 131 fig., 61 tab.; Clausthal.
- WOLKERSDORFER, CH. (2001): Tracer Tests in Flooded Underground Mines. – In: SEILER, K.-P. & WOHNICH, S.: *New Approches Characterizing Groundwater Flow* **1**. – p. 229-233, 4 fig., 3 tab.; Rotterdam (Balkema).
- WOLKERSDORFER, CH. (2002a): Rare Earth Elements (REEs) as Natural Tracers in Mine Waters. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: *Uranium in the Aquatic Environment*. – p. 967-974, 2 fig., 1 tab.; Heidelberg (Springer).
- WOLKERSDORFER, CH. (2002b): Mine water tracing. – Spec. Publ. – *Geol. Soc. London*, **198**: 47-61, 5 fig., 6 tab.; London.
- WOLKERSDORFER, CH. (2004): Mine Water Literature in ISI's Science Citation Index Expanded™. – *Mine Water Env.*, **23** (2): 96-99, 3 fig., 3 tab.; Berlin.
- WOLKERSDORFER, CH. (2006): Acid Mine Drainage Tracer Tests: ICARD 2006 7. – S. 2490-2501 [CD-ROM]; St. Louis (Proceedings, International Conference of Acid Rock Drainage (ICARD)).
- WOLKERSDORFER, CH. & GÖBEL, J. (2001a): Environmental Regulation of Mine Waters in the European Union – D 2 Overview of the EU and Eastern Europe – 22 Malta: ERMITE Final Report – EU R&D Project EVK1-CT-2000-00078. – 4 p.; Freiberg (TU Bergakademie Freiberg).

- WOLKERSDORFER, CH. & GÖBEL, J. (2001b): Environmental Regulation of Mine Waters in the European Union – D 2 Overview of the EU and Eastern Europe – 31 Turkey: ERMITE Final Report – EU R&D Project EVK1-CT-2000-00078. – 4 p.; Freiberg (TU Bergakademie Freiberg).
- WOLKERSDORFER, CH. & GÖBEL, J. (2004): Hydrogeologie der Troianischen Landschaft – Eine Bestandsaufnahme. – *Studia Troica*, **14**: 157-167, 9 fig., 1 tab.; Mainz.
- WOLKERSDORFER, CH. & HASCHE, A. (2001a): Tracer Test in the abandoned Fluorspar Mine Sträßberg/Harz Mountains, Germany. – *Wissenschaftliche Mitteilungen*, **16**: 57-67, 5 fig., 5 tab.; Freiberg.
- WOLKERSDORFER, CH. & HASCHE, A. (2001b): Tracer Test in the abandoned Fluorspar Mine Sträßberg/Harz Mountains, Germany. – p. 1-11 [CD-ROM], 5 fig., 5 tab.; Belo Horizonte (Proceedings, IMWA Symposium Belo Horizonte).
- WOLKERSDORFER, CH. & HASCHE, A. (2001c): Environmental Regulation of Mine Waters in the European Union – D 1 National Case Studies – Germany: ERMITE Final Report – EU R&D Project EVK1-CT-2000-00078. – 33 p., 5 fig., 5 tab.; Freiberg (TU Bergakademie Freiberg).
- WOLKERSDORFER, CH. & HASCHE, A. (2003): Multitracerversuch Königshütte/Harz. – Für: GEOMONTAN GESELLSCHAFT FÜR GEOLOGIE UND BERGBAU MBH & CO. KG SACHSEN. – 13 p., 5 tab., 7 app.; Freiberg (TU Bergakademie Freiberg, unpublished).
- WOLKERSDORFER, CH. & HASCHE, A. (2004): Tracer Investigations in flooded mines – The Sträßberg/Harz Multitracer Test: Conference Papers **35**. – p. 45-56, 5 fig., 5 tab.; Wien (Umweltbundesamt).
- WOLKERSDORFER, CH. & THIEM, G. (1999): Ground Water Withdrawal and Land Subsidence in Northeastern Saxony (Germany). – *Mine Water Env.*, **18** (1): 81-92, 3 fig., 4 tab.; Lakewood, CO.
- WOLKERSDORFER, CH. & YOUNGER, P. L. (2002): Passive Grubenwassereinigung als Alternative zu aktiven Systemen. – *Grundwasser*, **7** (2): 67-77, 7 fig., 2 tab.; Heidelberg.
- WOLKERSDORFER, CH., TREBUŠAK, I. & FELDTNER, N. (1997a): Development of a Tracer Test in a flooded Uranium Mine using *Lycopodium clavatum*. – In: KRANJC, A.: Tracer Hydrology 97 **7**. – p. 377-385, 4 fig., 2 tab.; Rotterdam (Balkema).
- WOLKERSDORFER, CH., FELDTNER, N. & TREBUŠAK, I. (1997b): LydiA – A new Method for tracing Mine Water. – Proceedings, 6th International Mine Water Association Congress, Bled, Slovenia, **1**: 43-55, 3 fig., 2 tab.; Ljubljana.
- WOLKERSDORFER, CH., HASCHE, A., UNGER, K. & WACKWITZ, T. (2002): Tracer Techniken im Bergbau – Georgi-Unterbau bei Brixlegg/Tirol. – *Wissenschaftliche Mitteilungen*, **19**: 37-43, 4 fig., 1 tab.; Freiberg/Sachsen.
- WOLKERSDORFER, CH., TAMME, S. & HASCHE, A. (2003): Natural attenuation of iron rich mine water by a surface brook. – In: NEL, P. J. L.: Mine Water and the Environment. – p. 433-439, 7 fig.; Johannesburg (Proceedings, 8th International Mine Water Association Congress).
- WOLKERSDORFER, CH., YOUNGER, P. L. & BOWELL, R. (2004): PADRE – Partnership for Acid Drainage Remediation in Europe. – *Mine Water Env.*, **23** (4): 181-182; Berlin.
- WONG, M. H. (2003): Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils. – *Chemosphere*, **50** (6): 775-780, 4 tab.; Oxford.
- WOOD, S. C., YOUNGER, P. L. & ROBINS, N. S. (1999): Long-term changes in the quality of polluted minewater discharges from abandoned underground coal workings in Scotland. – *Q. J. Eng. Geol.*, **32**: 69-79, 9 fig., 2 tab.; London.
- WORRALL, F. (1999): Chemical Fingerprinting of acid Minewaters: the Use of Rare Earth Elements (REE) Patterns. – In: FERNÁNDEZ RUBIO, R.: Mine, Water & Environment **II**. – p. 509-511, 3 fig.; Sevilla (International Mine Water Association).

- WORTHINGTON, S. R. H. (1999): A comprehensive strategy for understanding flow in carbonate aquifer. – In: PALMER, A. N., PALMER, M. V. & SASOWSKY, I. D.: Karst Modeling. – p. 30-37, 4 fig., 5 tab.; Charles Town (Karst Waters Institute).
- WRIGHT, J. B., NILSEN, D. N., HUNDLEY, G. & GALVAN, G. J. (1995): Field-Test of liquid Emulsion Membrane Technique for Copper Recovery from Mine Solutions. – *Miner. Eng.*, **8** (4-5): 549-556, 2 fig., 2 tab.; Oxford.
- WROBEL, M., RÜDE, T. R. & WOHNLIICH, S. (2000): Small scale mining in the area of the Choapa river, Central Chile: an environmental risk? – Proceedings, 7th International Mine Water Association Congress: 195-203, 5 fig.; Ustron.
- YAGER, D. B., LUND, K., LUEDKE, R. G., BOVE, D. J., O'NEILL, M. J. & SOLE, T. C. (1998): Digital geologic compilations of the upper Animas River and Boulder River watersheds; geographic information systems technology used as a scientific interpretation tool. – Open-File Report – US Geological Survey, **OF 98-0297**: 27; Reston.
- YANG, X. J., GU, Z. M. & FANE, A. G. (1999): Extraction and group separation of rare earths by a combined extraction/electrostatic pseudo liquid membrane from simulated rare earths mine water. – *Materials Science Forum*, **315-3**: 275-281, 2 fig., 4 tab.; Aedermannsdorf.
- YEFIMOCHKINA, E. (2004): Hydrochemische Untersuchungen an Sicker- und Grubenwässern des Freiburger Bergbaureviers und ihre ökologische Bewertung. – 126 p., 95 fig., 7 tab.; Freiberg (unpubl. Diploma Thesis TU Bergakademie Freiberg).
- YELPATYEVSKY, P. V. (1995): Factors controlling metal content of mining waters. – In: KHARAKA, Y. K. & CHUDADEV, O. V.: *Water-Rock Interaction* **8**. – p. 901-904, 2 fig.; Rotterdam u.a. (Balkema).
- YOUNGER, P. L., JENKINS, D. A., REES, S. B., ROBINSON, J., JARVIS, A. P., RALPH, J., JOHNSTON, D. N. & COULTON, R. H. (2004): Mine waters in Wales – pollution, risk management and remediation. – *Geological Series – National Museum & Art Galleries of Wales*, **23**: 138-154, 9 fig., 4 tab.; Cardiff.
- YOUNGER, P. L., COULTON, R. H. & FROGGATT, E. C. (2005): The Contribution of Science to Risk-Based Decision-Making: Lessons from the Development of Full-Scale Treatment Measures for Acidic Mine Waters at Wheal Jane, UK. – *Sci. Total Environ.*, **338** (1-2): 137-154, 3 fig., 3 tab.; Amsterdam.
- YOUNGER, P. L. (1998): Adit Hydrology in the Long-Term: Observations from the Pb-Zn Mines of Northern England. – In: NEL, P. J. L.: *Mine Water and Environmental Impacts* **2**. – p. 347-356, 4 fig., 2 tab.; Johannesburg (Proceedings International Mine Water Association Symposium).
- YOUNGER, P. L. (1993): Possible Environmental Impact of the Closure of two Collieries in County Durham. – *Water Environ. Manage.*, **7** (5): 521-531, 2 fig., 3 tab.; London.
- YOUNGER, P. L. (1995): Hydrogeochemistry of minewaters flowing from abandoned coal workings in the Durham coal field. – *Q. J. Eng. Geol.*, **28** (4): S101-S113, 4 fig., 4 tab.; London.
- YOUNGER, P. L. (1997): The longevity of minewater pollution – a basis for decision-making. – *Sci. Total Environ.*, **194-195**: 457-466, 5 fig.; Amsterdam.
- YOUNGER, P. L. (1998): Design, construction and initial operation of full-scale compost-based passive systems for treatment of coal mine drainage and spoil leachate in the UK. – In: NEL, P. J. L.: *Mine Water and Environmental Impacts* **2**. – p. 413-424; Johannesburg (Proceedings International Mine Water Association Symposium).
- YOUNGER, P. L. (2000a): Holistic remedial strategies for short- and long-term water pollution from abandoned mines. – *Trans. Inst. Min. Metall. Sect. A*, **109**: A210-A218, 5 fig., 1 tab.; London.
- YOUNGER, P. L. (2000b): Nature and practical implications of heterogeneities in the geochemistry of zinc-rich, alkaline mine waters in an underground F-Pb mine in the UK. – *Appl. Geochem.*, **15** (9): 1383-1397, 5 fig., 4 tab.; Amsterdam.

- YOUNGER, P. L. (2000c): The adoption and adaptation of passive treatment technologies for mine waters in the United Kingdom. – *Mine Water Env.*, **19** (2): 84-97, 5 fig., 1 tab.; Berlin.
- YOUNGER, P. L. (2000d): Predicting temporal changes in total iron concentrations in groundwaters flowing from abandoned deep mines: a first approximation. – *J. Contam. Hydrol.*, **44**: 47-69, 5 fig., 3 tab.; Amsterdam.
- YOUNGER, P. L. (2002a): Shilbottle Colliery remediation scheme: the UK's first large-scale permeable reactive barrier for mine spoil leachate remediation. – *CL:AIRE View*, **Autumn 2002**: 1-3, 3 fig.; London.
- YOUNGER, P. L. (2002b): Coalfield closure and the water environment in Europe. – *Trans. Inst. Min. Metall. Sect. A*, **111**: A201-A209, 1 fig., 5 tab.; London.
- YOUNGER, P. L. (2002c): A reducing and alkalinity-producing system (RAPS) for the passive treatment of acidic, aluminium-rich leachates emanating from revegetated colliery spoil materials at Bowden Close, County Durham. – 21 p.; London (Proceedings of the CL:AIRE Annual Project Conference, Imperial College).
- YOUNGER, P. L. (2002d): Deep mine hydrogeology after closure – insights from the UK. – In: MERKEL, B. J., PLANER-FRIEDRICH, B. & WOLKERSDORFER, CH.: *Uranium in the Aquatic Environment*. – p. 25-40, 1 fig., 2 tab.; Heidelberg (Springer).
- YOUNGER, P. L. (2002e): Mine Water Pollution from Kernow to Kwazulu-Natal: Geochemical Remedial Options and their Selection in Practice. – *Proceedings, Ussher Society*, **10**: 255-266, 8 fig., 3 tab.; Callington.
- YOUNGER, P. L. (2003): Frazer's Grove: the life and death of the last great Weardale fluorspar mine. – In: FAIRBAIRN, R. A.: *Fluorspar in the North Pennines*. – p. 47-72, 6 fig., 3 tab.; Durham (Friends of Killhope).
- YOUNGER, P. L. (2004a): Environmental impacts of coal mining and associated wastes – a geochemical perspective. – *Spec. Publ. – Geol. Soc. London*, **236**: 169-209, 6 fig., 7 tab.; London.
- YOUNGER, P. L. (2004b): "Making water": the hydrogeological adventures of Britain's early mining engineers. – *Spec. Publ. – Geol. Soc. London*, **225**: 121-157, 4 fig., 2 tab.; London.
- YOUNGER, P. L. & ADAMS, R. (1999): *Predicting Mine Water Rebound*. – 109 p., 32 fig., 88 tab.; Bristol (Environment Agency).
- YOUNGER, P. L. & ELLIOTT, A. (2003): The Shilbottle Spoil Heap Permeable Reactive Barrier. – London (Proceedings of the CL:AIRE Annual Project Conference 2003, University College).
- YOUNGER, P. L. & HARBOURNE, K. J. (1995): "To Pump or Not to Pump" – Cost-Benefit Analysis of Future Environmental Management Options for the Abandoned Durham Coalfield. – *Water Environ. Manage.*, **9** (4): 405-415, 5 fig., 7 tab.; London.
- YOUNGER, P. L. & LAPIERRE, A. B. (2000): "Uisge Mèinne" – Mine Water hydrogeology in the Celtic lands, from *Kernow* (Cornwall, UK) to *Ceap Breattain* (Cape Breton, Canada). – *Spec. Publ. – Geol. Soc. London*, **182**: 35-52, 7 fig., 2 tab.; London.
- YOUNGER, P. L. & ROBINS, N. S. (2002): Challenges in the characterisation and prediction of the hydrogeology and geochemistry of mined ground. – *Spec. Publ. – Geol. Soc. London*, **198**: 1-16, 1 fig., 2 tab.; London.
- YOUNGER, P. L. & SAPSFORD, D. J. (2006): Acid Drainage Prevention Guidelines for Scottish Opencast Coal Mining: the Primacy of the Conceptual Model: ICARD 2006. – p. 1-20 [CD-ROM], 1 fig., 1 tab.; St. Louis (Proceedings, International Conference of Acid Rock Drainage ICARD).
- YOUNGER, P. L., BANWART, S. A. & HEDIN, R. S. (2002): *Mine Water – Hydrology, Pollution, Remediation*. – 464 p.; Dordrecht (Kluwer).

- YOUNGER, P. L., JAYAWEEERA, A., ELLIOT, A., WOOD, R., AMOS, P., DAUGHERTY, A. J., MARTIN, A., BOWDEN, L., APLIN, A. C. & JOHNSON, D. B. (2003): Passive treatment of acidic mine waters in subsurface-flow systems – exploring RAPS and permeable reactive barriers. – *Land Contam. Reclam.*, **11** (2): 127-135, 3 fig., 2 tab.; Richmond.
- YOUNGER, P. L., JAYAWEEERA, A. & WOOD, R. (2004): A full-scale reducing and alkalinity-producing system (RAPS) for the passive treatment of acidic, aluminium-rich mine site drainage at Bowden Close, County Durham. – p. 1-13, 5 fig., 3 tab.; London (Proceedings of the CL:AIRE Annual Project Conference, Imperial College).
- ZAMORA, A. (1999): Small-scale mining. – In: NATURAL RESOURCES MANAGEMENT UNIT DIVISION „PROTECTION OF THE ENVIRONMENT AND NATURAL RESOURCES“: Report on the International Round Table on Mining and the Environment (“Berlin II Declaration”). – p. 47-49; Köln (Carl Duisberg Gesellschaft).
- ZEISSLER, K.-O., NINDEL, K. & HERTWIG, T. (2005): Hydrochemical Aspects of the Flooding of the Mine Königstein – A Water Mixing Model for Recognizing the Influence of Groundwater by Contaminated Water. – In: MERKEL, B. J. & HASCHKE-BERGER, A.: Uranium in the Environment. – p. 713-719, 4 fig.; Heidelberg (Springer).
- ZENTRALINSTITUT FÜR PHYSIK DER ERDE (1989): Silicic Magmatism and Metallogenesis of the Erzgebirge. – *Veröffentlichungen des Zentralinstituts für Physik der Erde*, **107**: 1-316; Berlin.
- ZHOU, J., SUN, J. & ZHANG, F. (1993): Origin of Mine Water by Using Environmental Isotope Techniques in Water-Logged Coal Mines, Pingdingshan, China. – *Mine Water Env.*, **12** (1-4): 63-69, 2 fig., 6 tab.; Wollongong.
- ZHOU, W. F. (1997): The formation of sinkholes in karst mining areas in China and some methods of prevention. – *Environ. Geol.*, **31** (1-2): 50-58, 7 fig., 1 tab.; New York, Berlin.
- ZHU, Y., MERKEL, B. J., STÖBER, I. & BUCHER, K. (2003): The Hydrogeochemistry of Arsenic in the Clara Mine, Germany. – *Mine Water Env.*, **22** (3): 110-117, 6 fig.; Berlin.
- ZIEGENBALG, G. (1999): In-situ remediation of heavy metal contaminated soil or rock formations and sealing of water inflows by directed and controlled crystallization of natural occurring minerals. – In: FERNÁNDEZ RUBIO, R.: *Mine, Water & Environment II*. – p. 667-672, 11 fig.; Sevilla (International Mine Water Association).
- ZIEGENBALG, G. (2005): A novel technology for sealing and immobilization – the use of precipitation processes from supersaturated solutions. – In: MERKEL, B. J. & HASCHKE-BERGER, A.: *Uranium in the Environment*. – p. 449-458, 8 fig.; Heidelberg (Springer).
- ZIEHR, H. (1983): Zur Geschichte des Steinkohlenbergbaues in Stockheim/Nordostbayern. – *Der Aufschluss*, **34** (12): 541-547, 3 fig.; Heidelberg.
- ZIEMKIEWICZ, P. F., SKOUSEN, J. G. & SIMMONS, J. (2003): Long-term Performance of Passive Acid Mine Drainage Treatment Systems. – *Mine Water Env.*, **22** (3): 118-129, 1 fig., 7 tab.; Berlin.
- ZITTMAN, P., RAVINGER, R. & KOVÁČ, L. (1990): Radioaktive Tracer-Meßgeräte und ihre Anwendung unter extremen Bedingungen. – *Freiberger Forsch.-H.*, **C 442**: 97-104, 5 fig.; Freiberg.
- ZOSEL, J. (1997): Funktion und Anwendung elektrochemischer Sensoren in stömenden Medien unter Einbeziehung hydrodynamischer Messungen mittels Laser-Doppler-Anemometrie. – 111 p., 54 fig., 7 tab., 1 app.; Freiberg (unpubl. Diss. TU Bergakademie Freiberg).
- ZÖTL, J. G. (1995): Badgastein spa – Austrian central Alps. – *Environ. Geol.*, **26** (4): 240-245, 6 fig., 2 tab.; New York-Berlin.
- ZÖTL, J. G. (2001): Badgastein Spa of Austria. – In: LAMOREAUX, P. E. & TANNER, J. T.: *Springs and Bottled Waters of the World – Ancient History, Source, Occurrence, Quality and Use* (8.25). – p. 230-233, 6 fig., 1 tab.; Berlin (Springer).

- ZOUMIS, T. (2003): Entwicklung aktiver Barrieren für die Entfernung von Schwermetallen aus Grubenwässern am Beispiel der Freiburger Grube, Sachsen. – 159 p., 112 fig., 69 tab.; Düsseldorf (VDI-Verl.).
- ZOUMIS, T. & CALMANO, W. (2000): Development of fly ash granules for heavy metal removal from mine waters. – Proceedings, 7th International Mine Water Association Congress: 492-504, 13 fig., 3 tab.; Ustron.
- ZOUMIS, T., CALMANO, W. & FÖRSTNER, U. (1999): Naturnahe Demobilisierung von Schwermetallen aus Grubenwässern. – In: FACHGRUPPE WASSERCHEMIE IN DER GESELLSCHAFT DEUTSCHER CHEMIKER: Kurzreferate und Teilnehmerverzeichnis – Jahrestagung 1999 (V27). – p. 138-142, 2 fig., 1 tab.; Regensburg (Eigenverlag Gesellschaft Deutscher Chemiker).
- ZOUMIS, T., CALMANO, W. & FÖRSTNER, U. (2000): Demobilization of heavy metals from mine waters. – Acta Hydrochim. Hydrobiol., **28** (4): 212-218, 7 fig., 4 tab.; Weinheim.
- ZSCHOCKE, K., PREUSCHEN, E., PITTIONI, R., FIRBAS, F., KISSER, J. & NETOLITZKY, F. (1932): Das urzeitliche Bergbauggebiet von Mühlbach-Bischofshofen. – Materialien zur Urgeschichte Österreichs, **6**: 1-287, 19 fig., 28 plates, 6 maps; Wien.
- ZUBER, A., GRABCZAK, J. & GARLICKI, A. (2000): Catastrophic and dangerous inflows to salt mines in Poland as related to the origin of water determined by isotope methods. – Environ. Geol., **39** (3-4): 299-311, 10 fig., 5 tab.; Heidelberg.
- ZURL, R., ECKART, D., VOGEL, D. & KUHL, J. (1992): Sanierung des Tagebau Lichtenberg im Thüringer Bergbauggebiet. – 9 p., 1 fig., 1 tab.; Chemnitz (Vortrag Workshop Wismut).

14 Abbreviations, List of Tables and Figures

14.1 Abbreviations

ABA	acid base accounting	ERMITE	Environmental Regulation of Mine Waters in the European Union
AD	<i>anno domini</i>	EU	European Union
ADP	Adenosindiphosphate	EURATOM	European Atomic Energy Community
ALD	Anoxic Limestone Drain	FDEM	frequency domain electromagnetic
AMD	acid mine drainage	FEFLOW	Finite Element Subsurface FLOW System
AMLIS	Abandoned Mine Lands Inventory System	FEM	finite element method
ANOVA	analyses of variance	FIZ-GEO	Fachinformationszentrum Geowissenschaften (geoscientific information centre)
ARD	acid rock drainage	GALILEO	European Global Satellite Navigation System
ARD/ML	acid rock drainage/metalliferous leachate	GaRD	Gas Redox and Displacement
ASCII	American Standard Code for Information Interchange	GARD	Global Acid Rock Drainage Guide
a.s.l.	above sea level	GeoRef	Geoscientific References
ATP	Adenosintriphosphate	GIS	geographical information system
BATNEEC	best available technology not entailing excessive costs	GLONASS	Global'naya Navigatsionnaya Sputnikovaya Sistema (global navigation satellite system)
BRGM	Bureau de Recherches Géologiques et Minières	GmbH	Gesellschaft mit beschränkter Haftung (Inc.)
BS	British Standard	GPM	U.S. gallons per minute
CAVE	Carbonate Aquifer Voids Evolution (computer software)	GPS	global positioning system
CFD	computational fluid dynamics	GRAM	Groundwater Rebound in Abandoned Mineworkings
DESD	UN Department of Economic and Social Development	GSF	Gesellschaft für Strahlen und Umweltforschung
DIN	<i>Das ist Norm</i> / Deutsches Institut für Normung	GVV	Gesellschaft zur Verwahrung und Verwertung von stillgelegten Bergwerksbetrieben mbH
DMT	Deutsche Montan Technologie GmbH	HDPE	high density polyethylene
DNA	deoxyribonucleic acid	HDS	high density sludge
DO	dissolved oxygen	HERO	Hydrogeochemical Engineering Research and Outreach
DQO	data quality objective	HLNRA	high level natural radiation area
DVWK	Deutscher Verband für Wasserwirtschaft und Kulturbau e.V.	IC	ion chromatography
EAI	environmental impact assessment	ICMM	International Council on Mining & Metals
EHTD	Efficient Hydrologic Tracer-Test Design	ICP-AES	Inductively-Coupled-Plasma Atom Emission Spectroscopy
EN	European Norm		
ENRA	elevated natural radiation areas		
EPA	Environmental Protection Agency		
eq. / eqs.	Equation / equations		

ICP-MS	Inductively-Coupled-Plasma Mass-Spectrometry	NW	northwest
ICRP	International Commission on Radiological Protection	OPAC	Online Public Access Catalogue
IMP	Inorganic Media Passive Systems	PADRE	Partnership for Acid Drainage Remediation in Europe
IMWA	International Mine Water Association	PC	Personal Computer
INAP	International Network of Acid Prevention	PCB	poly chlorinated bi-phenyls
IR	infra red	PDA	Personal Digital Assistant
ISI	Institute of Science Information	pers. comm.	personal communication
ISL	in-situ leaching	PE	polyethylene
ISO	International Organization for Standardization	PIR	Passive in-situ remediation
JPEG	Joint Photographic Experts Group	PIRAMID	Passive In Situ Remediation of acid Mine/Industrial Waters
LAWA	Länderarbeitsgemeinschaft Wasser	PTFE	polytetrafluorethylen
LDO	luminescent dissolved oxygen	PUF	polyurethane foam
LDS	low density sludge	PVC	polyvinyl chloride
LED	light emitting diode	QA	quality assurance
LLNRA	low or normal level natural radiation area	QC	quality control
LydiA	Lycopodium Apparatus	RAPS	Reducing and Alkalinity Producing Systems
m a.s.l.	meters above sea level	REE	rare earth elements
MAC	Maximum Allowable Concentration	SAPS	successive alkalinity producing system
MDM	multi-dispersion-model	SC	side calculation
MeFiSTo	Multiple Filter Storage Tool	SCI	Science Citation Index
MEND	Mine Environment Neutral Drainage	SDAG	Sowjetisch Deutsche Aktiengesellschaft (Soviet-German-Corporation)
MIFIM	Mine Water Filling Model	SFB	Subsurface-Flow Bacterial Sulphate Reducing Systems
MIRO	Mineral Industry Research Organisation	SFDM	Single Fissure Dispersion Model
MLNRA	medium level natural radiation area	SL	sampling location
MMSD	Mining, Minerals and Sustainable Development	SME	small and medium enterprises
MOTIF	model of transport in	TDS	total dissolved solids
MPC	Mineral Policy Center	TIC	total inorganic carbon
MS	Microsoft™	TinA	Tracer injection Apparatus
Mt	mount	TOC	total organic carbon
NGO	non governmental organisation	TU	Technical University
NNP	net neutralization potential	UK	United Kingdom
NORM	naturally occurring radioactive materials	UNCED	United Nations Conference on Environment and Development
NR	Nebenrechnung (side calculation)	UNDESA	United Nations Department of Economic and Social Affairs
NURBS	Non uniform rationale B-splines	UNEP	United Nations Environment Programme Industry and Environment
		UNSCERA	United Nations Scientific Committee on the effects of Atomic Radiation

US/U.S./USA	United States of America	WGS84	World Geodetic System 1984
UTM	Universal Transversal Mercator System	WHG	Wasserhaushaltsgesetz (water management legislation)
VEB	Volkseigener Betrieb (nationally owned enterprise)	WTP	Wetland-Type Passive Systems
VHLNRA	very high level natural radiation area	Zl.	Zahl/Zähler (number)
		ZPC	zero point of charge

Abbreviations not listed here are either the names of companies, of computer software, or standard abbreviations. The authors of computer software are usually not referred in the text.

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15 Appendices

15.1 Appendix I: Electrical Conductivity Calculations

In section 9.3.2 the principles of comparing chemical analyses by the use of the electrical conductivity and the limiting ionic molar conductivities are described. There is a simple forward method based on the theoretical investigations of Debye and Hückel for calculating the electrical conductivity of a solution with weak and strong electrolytes. Appendix I extracts the most important parts of the publications of Rossum (1975) as well as the compiled table of limiting ionic molar conductivities in Coury (1999). Tab. 30 lists only those ions which might be important for mine water analyses. Other molar conductivities can be found in the literature cited by Rossum (1975) and Coury (1999) – (according to a pers. comm. by Peter T. Kissinger, Chairman and CEO Bioanalytical Systems Inc., the second part of Coury's article never was published).

Tab. 30: Limiting ionic molar conductivities for common mine water constituents at 25 °C. λ^0 has to be divided by the ionic charge when used in equation 90.

Cation	$\lambda^0_+, \text{ S} \times \text{cm}^2 \times \text{mol}^{-1}$	Anion	$\lambda^0_-, \text{ S} \times \text{cm}^2 \times \text{mol}^{-1}$
H ⁺	349.6	OH ⁻	199.1
Li ⁺	38.7	F ⁻	55.4
Na ⁺	50.10	Cl ⁻	76.35
K ⁺	73.50	Br ⁻	78.1
Rb ⁺	77.8	I ⁻	76.8
Cs ⁺	77.2	NO ₂ ⁻	71.8
Ag ⁺	61.9	NO ₃ ⁻	71.46
NH ₄ ⁺	73.5	HCO ₃ ⁻	44.5
Be ²⁺	90	H ₂ PO ₄ ⁻	57
Mg ²⁺	106.0	HSO ₄ ⁻	50
Ca ²⁺	119.0	CO ₃ ²⁻	138.6
Sr ²⁺	118.9	HPO ₄ ²⁻	66
Ba ²⁺	127.2	SO ₄ ²⁻	160.0
Fe ²⁺	108.0	C ₂ O ₄ ²⁻	148.2
Cu ²⁺	107.2	PO ₄ ³⁻	207
Zn ²⁺	105.6		
Pb ²⁺	42.0		
UO ₂ ²⁺	64		
Al ³⁺	183		
Fe ³⁺	204		
La ³⁺	209.1		
Ce ³⁺	209.4		

The first equation gives the calculation of the electrical conductivity based on the physical and electrical properties of the ions analysed and yields the electrical conductivity in $\mu\text{S cm}^{-1}$. All the other terms within the equation are described in the cited literature.

$$\kappa = \kappa_+^0 + \kappa_-^0 - \left[\frac{\Lambda^0 \times Z_+ \times Z_-}{115.2 \times (Z_+ + Z_-)} \times \left(\frac{2 \times Q}{1 + \sqrt{Q}} \right) + 0.668 \right] \times [(Z_+ + Z_-) \times C]^{1.5} \quad (86)$$

$$Q = \frac{\Lambda_0 \times Z_+ \times Z_-}{(Z_+ + Z_-) \times (Z_+ \times \lambda_+ + Z_- \times \lambda_-)} \quad (87)$$

$$Z = \frac{\sum (c \times z^2)}{\sum (c \times z)} \quad (88)$$

$$\lambda = \frac{\kappa^0}{\sum c} \quad (89)$$

$$\kappa^0 = \sum (c \times \lambda^0) \quad (90)$$

$$\Lambda^0 = \lambda_+ + \lambda_- \quad (91)$$

$$C = \frac{\sum c_+ + \sum c_-}{2} \quad (92)$$

The use of those equations shall be exemplified by a simple calculation for a given water analyses from the abandoned Schwaz/Austria silver mine (analyses SWZ-2511-MB09).

Tab. 31: Results of the SWZ-2511-MB09 mine water analyses and the necessary calculations for the electrical conductivity. Differences are due to rounding.

Parameter / element	mass concentration c_m , mg L ⁻¹	concentration c , meq L ⁻¹	$c \cdot \lambda^0$	$c \cdot z^2$	$c \cdot z$
pH	7.75				
electrical conductivity	1086 $\mu\text{S cm}^{-1}$				
Ca ²⁺	66	3.29	195.97	13.174	6.587
Mg ²⁺	80.1	6.59	349.99	26.365	13.182
Na ⁺	20.2	0.88	44.02	0.879	0.879
K ⁺	8.61	0.22	16.19	0.220	0.220
H ⁺			0.01	0.000	0.000
	sum	10.98	606.17	40.63	20.87
HCO ₃ ⁻	174	2.85	126.90	2.852	2.852
SO ₄ ²⁻	432	8.99	717.72	35.976	17.988
Cl ⁻	2.77	0.08	5.96	0.078	0.078
NO ₃ ⁻	2.84	0.05	3.27	0.046	0.046
OH ⁻			0.11	0.001	0.001
	sum	11.97	853.96	38.96	20.97

Calculation based on the tabulated values in Tab. 30 und Tab. 31:

$$Z_+ = \frac{40.63}{20.87} = 1.947$$

$$Z_- = \frac{38.96}{20.97} = 1.858$$

$$\lambda_+ = \frac{606.17}{10.98} = 55.207$$

$$\lambda_- = \frac{853.96}{11.97} = 71.342$$

$$\Lambda^0 = 55.207 + 71.759 = 126.548$$

$$C = \frac{10.98 + 11.97}{2} = 11.475$$

$$Q = \frac{126.548 \times 1.947 \times 1.858}{(1.947 + 1.858) \times (1.947 \times 55.207 + 1.858 \times 71.342)} = 0.498$$

$$\kappa = 606.17 + 853.96 - \left[\frac{126.548 \times 1.947 \times 1.858}{115.2 \times (1.947 + 1.858)} \times \left(\frac{2 \times 0.498}{1 + \sqrt{0.498}} \right) + 0.668 \right] \times \\ \left[(1.947 + 1.858) \times 11.475 \right]^{1.5} = 1091.45$$

As can be seen the result from the calculation of the electrical conductivity, $1091 \mu\text{S cm}^{-1}$, is very close to the measured electrical conductivity of $1086 \mu\text{S cm}^{-1}$, the total error being only 0.5%.

15.2 Appendix II: Weir Equations

15.2.1 Introductory Remarks

This appendix shall not add another solution to the several hundreds of solutions already published but shall give mine water researchers a quick tool for measuring flows over thin plated weirs without going into the depths of discharge measurement. Section 9.2.1.4 gives an overview of several prerequisites and limitations of weir constructions that are commonly used in mine water research. Usually, those simplifications might be satisfactory but if a higher accuracy is needed, the simplified equations should be substituted by the following set of equations. All of the equations are based on the published literature (e.g. Ackers et al. 1978; Herschy 1995) but the tables and graphs for getting the relevant coefficients have been updated, compared with other published data and brought into equations that can be used in spreadsheets or routines for flow calculation. For coefficient fitting the software package TableCurve has been used and all of the equations have been selected to get a minimum error from the reported data which is usually below 0.5%. Those equations can only be used in cases where the limitations of the original researchers are met, and therefore they are listed here as well. They can only be used for “normal” mine water under “normal” temperature conditions (usually +4—+29 °C). Sludge flows or mine waters with extreme contents of solids cannot be measured without calibrating the weir to the special conditions on site.

Besides those limitations, the crest thickness, the roughness of the weir plate, the rounding of the crest, the fluids viscosity, and – to a smaller extend – the velocity distribution of the flow approaching the weir crest influence the discharge (Ackers et al. 1978). Furthermore, weir plates must be installed vertical, they regularly have to be cleaned from bio films and precipitates and the overflowing water must be ventilated, as the flow depends on the pressure conditions around the weir crest. Finally, the surface tension and the viscosity must be taken into

account, for which analytical solutions for thin-plated rectangular weirs have been discussed as well (e.g. equation 47 in Sarpkaya et al. 1959). And – keep in mind – even if thin plate weirs “look” simple, measuring flows with such devices is other than that!

15.2.2 Symbols and Notation for Appendix II

A	Cross-sectional area of stream bed (Cipolletti weir only), m^2
B	Bed width, m
b	Crest breath, m
b_e	Effective crest breath, m
C_d	Coefficient of discharge, –
C_v	Coefficient of velocity of approach
f	Factor
g	Gravitational acceleration ($\approx 9.807\text{—}9.815 \text{ m s}^{-2}$), m s^{-2}
h	Gauged head related to weir crest, m
h_e	Effective head related to weir crest, m
$h_{\max}$	Maximum expected head related to weir crest, m
k_b	Width correction factor, m
k_v	Head correction factor, m
L_h	Distance from weir to position of head measurement, m
m	Coefficients 1—5 (Thomson weir)
n	Coefficients 1—5 (Thomson weir)
o	Coefficients 1—4 (Thomson weir, simplified discharge equation)
P	Crest height of weir above mean bed level, m
Q	Discharge, $\text{m}^3 \text{ s}^{-1}$
r	Coefficients 1—2 for calculating the coefficient of discharge
s	Coefficients 1—4 (contracted Poncelet weir)
t	Coefficients 1—9 (contracted Poncelet weir)
u	Coefficients 1—7 (contracted Poncelet weir)
v	Coefficients 1—3 (Cipolletti weir)
α	Angle of V-notch weir (full opening), $^\circ$ (degrees)

15.2.3 V-notch (Thomson) Weirs (90° , $\frac{1}{2} 90^\circ$, $\frac{1}{4} 90^\circ$)

Nowadays, most researches use the Kindsvater-Shen equation for calculating the flow over a V-notch weir. It has been simplified by many authors to a form as given in equation 93. Compared to a 90° V-notch weir, the $\frac{1}{2} 90^\circ$ and $\frac{1}{4} 90^\circ$ V-notch weirs have a flow which is half and a quarter of the flow over a 90° V-notch weir at the same gauged head, respectively. All the calculations for C_d and k_v given here, are based on the originally published graphical relations by Shen (1981) and were used for the curve fitting (Tab. 32). They are only relevant if the following limitations are met (compare inset in Fig. 126):

- $h \geq 0.05 \text{ m}$
- $P > 0.45 \text{ m}$
- $\frac{h}{P} \geq 0.4$
- $B > 0.9$
- $4 \leq \frac{L_h}{h_{\max}} \leq 5$
- $\tan \frac{\alpha}{2} \times \frac{h}{B} > 0.2$
- $20^\circ \leq \alpha \leq 100^\circ$
- $4^\circ \text{C} \leq \theta \leq 29^\circ \text{C}$

$$Q = \frac{8}{15} \times \sqrt{2 \times g} \times C_d \times \tan \frac{\alpha}{2} \times h_e^{2.5} \quad (93)$$

$$h_e = h + k_v \quad (94)$$

$$C_d = m_1 + m_2 \times h + m_3 \times h^2 + m_4 \times h^3 + m_5 \times h^4 \quad (95)$$

$$k_v = n_1 + n_2 \times \alpha^{1.5} + n_3 \times \alpha^2 + n_4 \times \alpha^{-1.5} \quad (96)$$

There is also a calculation for the simplified equations given in section 9.2.1.4. If the simplified equations have the general form of

$$Q = f(\alpha) \times h^{2.5} \quad (97)$$

where α is given in degrees then $f(\alpha)$ can be written as

$$f(\alpha) = \frac{o_1 + o_2 \times \alpha}{1 + o_3 \times \alpha + o_4 \times \alpha^2} \quad (98)$$

with

$$20^\circ \leq \alpha \leq 100^\circ$$

Tab. 32: Coefficients for equations 95, 96, and 98.

Coefficient	m	n	o
1	$6.057 \cdot 10^{-1}$	$1.900 \cdot 10^{-3}$	$6.886 \cdot 10^{-3}$
2	$-7.524 \cdot 10^{-4}$	$-4.757 \cdot 10^{-6}$	$1.211 \cdot 10^{-2}$
3	$2,809 \cdot 10^{-6}$	$3.575 \cdot 10^{-7}$	$9.483 \cdot 10^{-4}$
4	$4,082 \cdot 10^{-8}$	$1.198 \cdot 10^{-1}$	$-3.506 \cdot 10^{-5}$
5	$-1,831 \cdot 10^{-10}$	—	—

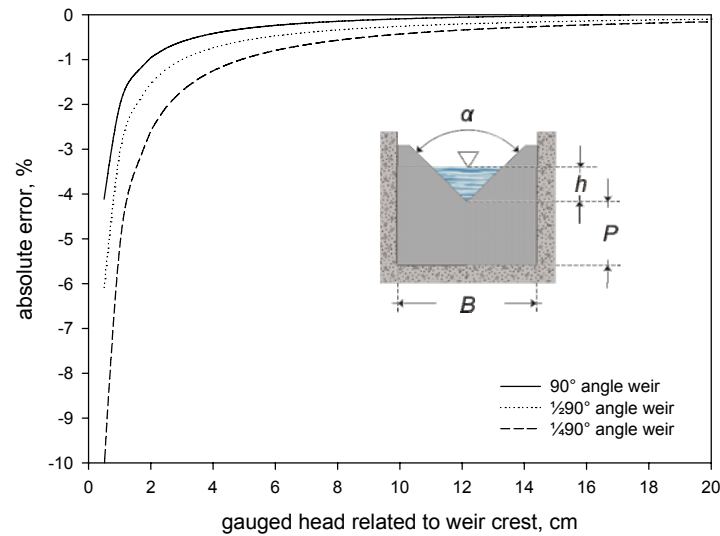


Fig. 126: Absolute errors of discharge measurements by using the simplified equations 54—56 for V-notch weirs. Inset: typical layout of a V-notch weir.

15.2.4 Rectangular, full Width Notch (Poncelet) Weirs

Though several hundreds of publications about weirs are available it is still not possible to give *one* equation which can be recommended for all possible boundary conditions. Therefore, Ackers et al. (1978), based on investigations conducted by the Hydraulics Research Station Wallingford England suggested to use equation 99 for full width weirs with the following limits of application:

- $h \geq 0.02 \text{ m}$
- $b \geq 0.20 \text{ m}$
- $P \geq 0.15 \text{ m}$
- $\frac{h}{P} \leq 2.2$
- $2.5 \leq \frac{L_h}{P} \leq 3$

$$Q = 0.564 \times \sqrt{g} \times \left(1 + 0.150 \times \frac{h}{P} \right) \times b \times h_e^{1.5} \quad (99)$$

$$h_e = h + 0.001 \quad (100)$$

For weirs with $\frac{h}{P} > 1.8$ and in cases where h is smaller than 0.075 m the above equation loses its accuracy.

15.2.5 Rectangular, Contracted Notch (Poncelet) Weirs

As in the case of full width weirs, different researchers found different experimental solutions for the general discharge equation. The first investigations for flows over rectangular weirs were conducted in Munich, Dresden, Leipzig, and the Walchensee/Germany hydraulic laboratory by Kirchhoff, Kötter, Ott, Zschiedrich, Kirschmer, and Esterer (Kolupaila in Sarpkaya et al. 1959). Yet, frequently, the Kindsvater equation is used and therefore will be given here,

too. Ackers et al. (1978) showed that the flow over a contracted rectangular weir is a function of many variables defining the geometry of such a flow measurement installation. Therefore, the number of coefficients and functions to calculate the flow through a contracted rectangular weir is far higher than for the other cases. All efforts to reduce the number of parameters that resulted from the curve fitting code were unsuccessful, therefore, 20 coefficients are necessary, to calculate the flow through such a type of weir (Tab. 33). Sarpkaya (in Sarpkaya et al. 1959) derived an equation for calculating C_d which includes the Reynolds and Weber numbers Re and We for taking into account the surface tension and viscosity of the fluid. They are not taken into account here, because viscosity and surface tension become only important if the head above the weir crest and the weir width are comparatively small. In most mine water measurement cases such a situation will be reached rather seldom and is therefore not considered here.

- $h \geq 0.03$ m
- $b \geq 0.15$ m
- $P > 0.10$ m
- $\frac{h}{P} \leq 2$
- $\frac{B-b}{2} \geq 0.1$ m
- $4 \leq \frac{L_h}{h_{\max}} \leq 5$

$$Q = \frac{2}{3} \times \sqrt{2 \times g} \times C_d \times b_e \times h_e^{1.5} \quad (101)$$

$$C_d = r_1 + r_2 \times \frac{h}{P} \quad (102)$$

$$b_e = b + k_b \quad (103)$$

$$h_e = h + 0.001 \quad (104)$$

$$r_1 = \frac{s_1 + s_2 \times f}{1 + s_3 \times f + s_4 \times f^2} \quad (105)$$

$$r_2 = \frac{t_1 + t_2 \times f^2 + t_3 \times f^4 + t_4 \times f^6 + t_5 \times f^8}{1 + t_6 \times f^2 + t_7 \times f^4 + t_8 \times f^6 + t_9 \times f^8} \quad (106)$$

$$k_b = \frac{u_1 + u_2 \times f^2 + u_3 \times f^4 + u_4 \times f^6}{1 + u_5 \times f^2 + u_6 \times f^4 + u_7 \times f^6} \quad (107)$$

$$f = \frac{b}{B} \quad (108)$$

Tab. 33: Coefficients for equations 105, 106, and 107.

Coefficient	s	t	u
1	$5.877 \cdot 10^{-1}$	$-3.260 \cdot 10^{-3}$	$2.390 \cdot 10^{-3}$
2	$-5.279 \cdot 10^{-1}$	$-2.131 \cdot 10^{-1}$	$-8.640 \cdot 10^{-3}$
3	$-9.108 \cdot 10^{-1}$	$2.545 \cdot 10^{-0}$	$1.766 \cdot 10^{-2}$
4	$1.005 \cdot 10^{-2}$	$1.227 \cdot 10^1$	$-1.148 \cdot 10^{-2}$
5	–	$-5.907 \cdot 10^{-0}$	$-3.893 \cdot 10^{-0}$
6	–	$20.15 \cdot 10^1$	$6.560 \cdot 10^{-0}$
7	–	$4.783 \cdot 10^2$	$-3.596 \cdot 10^{-0}$
8	–	$-7.995 \cdot 10^2$	–
9	–	$4.165 \cdot 10^2$	–

15.2.6 Trapezoidal (Cipolletti) Weirs

Both side slopes of a Cipolletti weir have to be at an angle of 1:4 and the intention was to reduce the side contraction effects (Herschy 1995). Though Ackers et al. (1978) concludes that Cipolletti weirs do not have an advantage in practice, the simpler empirical weir equation compared to Poncelet weirs seems to prove something else (Tab. 34). Yet, it is a matter of habit if one uses a Cipolletti weir or a rectangular weir.

- $0.06 \text{ m} \leq h \leq 0.6 \text{ m}$
- $P < 2 h_{\max}$
- $P \geq 0.30 \text{ m}$
- $\frac{h}{b} \leq 0.5$
- $4 \leq \frac{L_h}{h_{\max}} \leq 5$

$$Q = \frac{2}{3} \times \sqrt{2 \times g} \times C_v \times C_d \times b \times h^{1.5} \quad (109)$$

$$C_d = 0.63 \quad (110)$$

$$C_v = v_1 + v_2 \times e^{\left(\frac{-f}{v_3}\right)} \quad (111)$$

$$f = \frac{C_d \times b \times h}{A} \quad (112)$$

Tab. 34: Coefficients for equation 111.

Coefficient	v
1	$9.846 \cdot 10^{-1}$
2	$1.261 \cdot 10^{-2}$
3	$-2.743 \cdot 10^{-1}$

15.3 Appendix III: The Berlin I and II Guidelines

Introduction

In 1991 the first International Round Table on Mining and the Environment (Berlin I) was organised by the UN Department of Economic and Social Development (DESD) and the Development Policy Forum of the German Foundation for International Development. Mainly meant for developing countries, the Berlin I Guidelines became also adopted by mining companies in developed countries and some of the changes in the mining industry can clearly be related to those guidelines. Because those guidelines are important for mine planning and advance mine closure planning they shall be cited here in its central parts (United Nations Department of Technical Co-operation for Development and Development Policy Forum of the German Foundation for International Development 1992). In the meantime a second round table was organised in 1999 and the recommendations been published (Natural Resources Management Unit Division „Protection of the Environment and Natural Resources“ 1999; United Nations Department of Economic and Social Affairs [UNDESA] and United Nations Environment Programme Industry and Environment [UNEP] 2002). During the Berlin II meeting the delegates realised:

Since the publication of the first edition of the Guidelines, continuing changes have occurred within the mining sector – particularly in the evolution of legal, fiscal and regulatory policies, accompanying the trend of liberalization and privatization of the industry. Among other things, environmental conditions attached to credit and insurance have raised the profile of environmental planning and management of sustainable development. There have also been major advances in voluntary actions by industry and the standardization of environmental management systems.

During this period, significant advances have been made in defining and promoting best practices in the sustainable management and operation of mining projects. These advances have often been stimulated by the attention of pressure groups, the media, and public opinion. Many of the major mining companies have now started to work in partnership with regulators and the local communities to maximise corporate, national and community benefits, while minimising the social and physical impacts.

Yet, the delegates of the Berlin II round table realised that a mine's life has also to deal with the abandonment and therefore put forward guidelines for mine closure and abandoned mines (Balkau 1999).

Environmental Management Guidelines for Mining

World-wide long-term economic development can best be achieved through the pursuit of sustainable development policies comprising a balance of economic, socio-cultural, and environmental protection measures. While taking into account global environmental concern, each country should apply this concept to meet the needs of their environmental and economic circumstances. Sustainable mining activities require good environmental stewardship in all activities, from exploration and processing to decommissioning and reclamation. It acknowledges the importance of integrating environmental and economic considerations in the decision-making process and the fact that the mineral deposits are unique in their occurrence. It recognises the importance of mining to the social, economic, and material needs of society, in particular for developing countries, and that minerals, notably metals, offer great potential for the use of future generations through increased recycling programs. Sustainable mining under appropriate environmental guidelines is based on interaction be-

tween industry, governments, non-governmental organizations and the public, directed towards optimising economic development while minimising environmental degradation. The need for such guidelines is recognised by industry, governments, and international agencies. It is also recognised that the political will of governments, together with the commitment of industry management and of the community, are the essential conditions needed to enforce environmental legislation and more importantly, to ensure compliance with all applicable laws for the protection of the environment, employees and the public.

Environmental Guidelines for Action Addressed to the Mineral Sector (Berlin I Guidelines, revised in 2000; United Nations Department of Economic and Social Affairs [UNDESA] and United Nations Environment Programme Industry and Environment [UNEP] 2002)

Governments, mining companies and the minerals industries should as a minimum:

1. Recognise environmental management as a high priority, notably during the licencing process and through the development and implementation of environmental management systems. These should include early and comprehensive environmental impact assessments, pollution control and other preventive and mitigative measures, monitoring and auditing activities, and emergency response procedures.
2. Recognise the importance of socio-economic impact assessments and social planning in mining operations. Social-economic impacts should be taken into account at the earliest stages of project development. Gender issues should also be considered at a policy and project level.
3. Establish environmental accountability in industry and government at the highest management and policy-making levels.
4. Encourage employees at all levels to recognise their responsibility for environmental management and ensure that adequate resources, staff and requisite training are available to implement environmental plans.
5. Ensure the participation of and dialogue with the affected community and other directly interested parties on the environmental and social aspects of all phases of mining activities and include the full participation of women and other marginalised groups.
6. Adopt best practices to minimise environmental degradation, notably in the absence of specific environmental regulations.
7. Adopt environmentally sound technologies in all phases of mining activities and increase the emphasis on the transfer of appropriate technologies that mitigate environmental impacts, including those from small-scale mining operations.
8. Seek to provide additional funds and innovative financial arrangements to improve environmental performance of existing mining operations.
9. Adopt risk analysis and risk management in the development of regulation and in the design, operation, and decommissioning of mining activities, including the handling and disposal of hazardous mining and other wastes.
10. Reinforce the infrastructure, information systems service, training and skills in environmental management in relation to mining activities.
11. Avoid the use of such environmental regulations that act as unnecessary barriers to trade and investment.

12. Recognise the linkages between ecology, socio-cultural conditions and human health and safety, the local community and the natural environment.
13. Evaluate and adopt, wherever appropriate, economic and administrative instruments such as tax incentive policies to encourage the reduction of pollutant emissions and the introduction of innovative technology.
14. Explore the feasibility of reciprocal agreements to reduce transboundary pollution.
15. Encourage long-term mining investment by having clear environmental standards with stable and predictable environmental criteria and procedures.