

Contemporary Reviews of Mine Water Studies in Europe

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Abstract. Europe was once the most important mining region in the world and nearly every European country has remnants of historic and even pre-historic mining sites. Though the importance of mining activities in most European countries declines, the abandoned sites are still there and can cause environmental dangers as well as technological challenges. On the basis of selected European countries and case studies, these dangers and challenges are described and potential solutions are illustrated.

Key words: Abandoned mine; Austria; Estonia; Europe; Hungary; Italy; mine water; Netherlands; Slovakia; United Kingdom; Ireland; Belgium; Czech Republic; Portugal; Switzerland; Bosnia and Herzegovina; Norway; Finland; Sweden; Germany; Poland; France; Serbia and Montenegro; Macedonia; active treatment; passive treatment; mine water microbiology; PADRE (Partnership for Acid Drainage Remediation in Europe); policy

Introduction

Europe has a long tradition of mining activity, dating back as far as records of human settlement of Europe with the development of flint mines in France over 10,000 years ago. By the Bronze Age, copper, silver and gold were being mined by the Phoenicians in Spain. During the same time gold placers were worked in the Black Sea area and probably formed the basis of the legend of the golden fleece for Jason and the Argonauts.

The first gold coins were produced in the area of Bulgaria-Greece-Turkey around 700 BC and from around 560 BC the earliest records of metal refining are preserved. The Roman empire expanded throughout Europe with a great thirst for metals and the Romans advanced techniques of underground mining, processing and even dewatering. In AD 79 Pliny the Elder produced the oldest preserved accounts of mine dewatering based on his experiences in the metal mines of southern Italy (Temple 1972).

The birthplace of modern mining or at least the recording of modern mining occurred during the 16th century in the Erzgebirge (Ore Mountains) area of Germany and the Eastern Alps. Georg Bauer, or as you preferred to be known by his Latin name *Georgius Agricola*, a local physician recorded methods of mining including dewatering and the environmental impacts associated with mining and mineral dressing in his classic thesis, *De Re Metallica* (Agricola 1556).

During the following three centuries mining in Europe was set to reach its pinnacle and formed the basis for scientific and engineering development that would spread throughout the world as European colonization

occurred. By the middle of the nineteenth century the mature ore fields of Europe started to be abandoned in preference to the rich untapped wealth of the "New World".

This trend has continued with minor resurgence during the twentieth century. At the start of the third millennium, some mining has survived in Europe although the majority of orefields are now abandoned and the emphasis is now on control of environmental impact and remediation of the historic legacy of Europe's mining heritage.

We hope that the reviews will stimulate further research on the complex and multifaceted issues associated with mine waters in Europe and result in further collaboration from within and outside Europe amongst *Mine Water and the Environment* readers.

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European Union Policies and Mine Water Management

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Mining in European Environmental Policy

Mining is one of the oldest industrial sectors in Europe. European mining policies have been shaped by the historical importance of mining for industrial development and the relatively recent introduction of environmental concerns in public policy. As a result, the emphasis of mining policies has been on the industrial (including safety and health) and economic aspects. Within the European Commission, mining interests reside in two Directorates: DG Transport and Energy, which deals with the energy extractive industry, and DG Enterprise, which deals with the non-energy extractive industry.

Mining had been specifically excluded from much of the environmental policy developed by DG Environment. Recent reviews of relevant legislation show how the mining industry has been favourably treated compared to other industrial sectors (Hámor 2002). Mining was excluded from the Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC) and the Seveso II Directive (96/82/EC). It was included, but with greater freedom of interpretation, in the Environmental Impact Assessment Directive (97/11/EC). Whether or not it should be included in waste legislation has been a contentious issue due to the clause of the Waste Framework Directive (75/442/EEC) stating that mining waste would be excluded where it is already covered by other legislation, interpreted by the European Commission (EC) as referring exclusively to European legislation (see later the AvestaPolarit ruling). Water legislation has much less direct references to mining than waste. The Water Framework Directive (WFD) (2000/60/EC) applies to mining activities in a generic sense but there is no specific water legislation addressing the specific requirements of this sector.

In the aftermath of the Aznalcóllar (April 1998) and Baia Mare (January 2000) accidents, the EC created the Baia Mare Task Force (March 2000) to propose a plan of action. In less than one year, the EC published three communications on environmental aspects of mining emanating from two different Directorates: Enterprise, [COM (2000) 265f] and Environment, [COM (2000) 664f] and [COM (2000) 593f]. In partic-

ular, the Baia Mare Task Force recommended three key actions discussed in more detail below: amendment of the Seveso II Directive, a document on Best Available Techniques (BAT) similar to those produced under the IPPC Directive, and an initiative on the management of mining waste. Conspicuously, the Task Force with its narrow focus on tailings dam safety failed to identify the need for a water-related initiative (Kroll et al. 2001). In another independent policy initiative, mine water management will also be affected by the new Environmental Liability Directive adopted in March 2004, with national implementation three years later. This Directive will make mining operators liable for the clean up of contaminated sites.

Seveso II Directive

The Seveso II Directive (96/82/EC) has two aims: the prevention of major accident hazards involving dangerous substances and limiting the consequences of such accidents not only for man but also for the environment. The scope of the Directive is related to the presence of dangerous substances in establishments. However, article 4 (e) excluded the activities of the extractive industries. Following the recommendations of the Baia Mare Task Force, the EC [COM (2001) 624f] proposed to amend the Directive to include chemical and thermal processing of minerals and related storage operations and tailings disposal facilities if they involve dangerous substances regulated by the Directive. The European Parliament introduced new amendments calling for an extension of the scope to all mining activities. Under the final agreement, the Directive covers chemical and thermal processing operations and storage related to those operations plus operational tailings disposal facilities containing dangerous substances, when used in connection with both chemical/thermal and mechanical/physical processing of minerals. The Directive 2003/105/EC of 16 December 2003, amending Directive 96/82/EC, published in the Official Journal on 31 December 2003 requires that Member States bring into force the laws, regulations, and administrative provisions necessary to comply with this Directive before 1 July 2005.

BAT for Management of Tailings and Waste-Rock

The IPPC Bureau established a Technical Working Group (TWG) in June 2001 to develop a technical document that would contribute to the knowledge available to prevent accidents from tailings and waste-rock disposal facilities and provide technical support for legislative activities of the EC (e.g. proposed Directive on mine waste). Usually, TWGs are organised to facilitate the exchange of information between the European Union's Member States and industry under the IPPC Directive but this TWG was set up based only on the Communication of the Commission after the

Baia Mare Task Force [COM (2000) 664f]. The TWG decided that the scope of its activities related to mineral processing, tailings, and the waste-rock management of ores that have the potential for a significant environmental impact or that can be considered as examples of good practice. The document covers 14 metals, 10 industrial minerals, coal only if processed (lignite is not covered), and oil shales. The issue of abandoned mines was not addressed. The final draft reference document on BAT for management of tailings and waste-rock from mining activities was published in March 2004 (<http://eippcb.jrc.es>).

Proposed Directive on the Management of Waste from the Extractive Industry

The most important initiative triggered by the Baia Mare report has been the development of a proposal for a new Directive on the management of waste from the extractive industry as a daughter directive from the Waste Framework Directive. The first proposal coming from the Commission relied heavily on the recent Landfill Directive (1999/31/EC). This was highly contested by interested parties. An open process of consultation helped to craft two more draft versions that were more adequate as a license-based system for the management of waste facilities in mining operations. The scope was also clarified covering now almost all mine and quarry wastes. However, the final proposal that came out of the European Commission [COM (2003) 319f] softened many provisions, excluding some waste, reducing the attention to mining voids and scrapping the duty of Member States to remediate closed waste facilities. The position of the Commission has been challenged on two fronts. First, the European Court of Justice on AvestaPolarit Chrome Oy (Case C-114-01, 11/09/2003) determined that left over rock and ore-dressing sand used directly for infilling of underground galleries would be regarded as a by-product and not as waste. It also ruled that the “other legislation” referred to in the Waste Framework Directive can also include national legislation.

Secondly, the amendments approved by the European Parliament in its first reading (A5-0177/2004) on 31 March 2004, strengthen many provisions of the proposal, in particular, the management of closed sites. The proposal follows the complicated steps of the co-decision procedure with several iterations between the European Parliament, European Council, and the EC. The Council reached a Political Agreement on a Common Position on 14 October 2004. The text approved by the Council is quite similar to the original Commission proposal. The provisions for dealing with closed sites within the Council document are less prescriptive than the European Parliament version. The final text of the Directive could be ready by the end of 2005.

Mine Water Management

The EC Framework 5 project (EVK1-CT-2000-00078) “Environmental Regulation of Mine Waters in the European Union” (ERMITE), which finished in January 2004, has produced a thorough evaluation of European legislation for mine water management (<http://www.minewater.net/ermite>). The ERMITE policy briefs emphasise the remediation of abandoned sites, the links between mine wastes and mine voids, and the management of so-called “inert” waste. ERMITE has also produced guidelines for the management of mine waters at the catchment scale (ERMITE Consortium 2004). It is recommended that the EC produce similar official guidelines for the implementation of the Water Framework Directive.

Disclaimer

The views expressed are purely those of the authors and may not in any circumstances be regarded as stating an official position of the EC.

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Acid Rock Drainage from Norwegian Mines

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Mining in Norway

The oldest registered mine in Norway was the

Akersberg silver mine, located beneath the Aker Church in Oslo. This mine was mentioned in “*Historiae Norwegia*”, published in 1170. However, the industry has been a cornerstone of the Norwegian economy since the early 1600's. The early mines were primarily mined for copper, silver, and iron. Kongsberg (1623), Løkken (1633), and Røros (1644), to mention a few, all operated for more than 300 years. The latter two are massive sulfide deposits, while the Kongsberg Mine is an epithermal vein deposit.

Mining was, for a long time, focused at massive sulfide deposits where primarily copper and sulfur were

mined, with lead, zinc, silver, and gold as by-products. Of the many such deposits mined during the last 400 years, the Røros and Sulitjelma mining districts are the largest. As these massive sulfide deposits commonly have little neutralizing minerals within the ore, they have had a large impact on the surrounding environment. Other massive sulfide deposits have been mined until relatively recently: Bleikvassli, Bidjovagge, Grong, Skorovass, Løkken, Hjerkin, Vigsnes and Mofjell. Since 1850, these massive sulfide deposits have produced some 54 million t of pyrite, 2 million t of copper (Table 1), and an unknown amount of acid generating or potentially acid generating tailings and waste rocks.

The Røros district, located some 100 km south of Trondheim, has several dozen mines. The deposit was first discovered in the mid 1600's and the last operation closed in 1977. The main products from Røros were sulfur and copper, with zinc and silver as by-products. There has been some exploratory interest in the district again over the last few years.

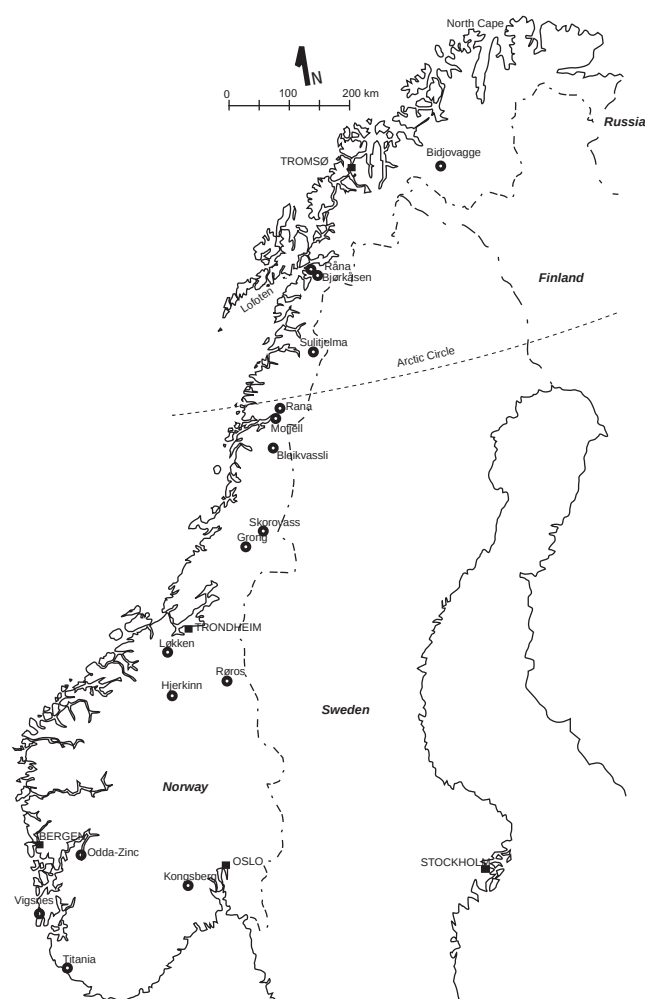


Figure 1. Location map

The Sulitjelma mining district was once Norway's second largest industrial enterprise with underground train tracks for 880 km. Mining started in 1887 and ceased in 1991 after 104 years of operation. Until the late 1920's, copper ore was roasted in large piles to remove sulfur as sulfur dioxide, but this method was later replaced by flotation and electric copper smelter (Segalstad 2002).

The Kongsberg deposit, which is high in carbonate, once hosted one of Europe's larger silver mines, and contains many small waste dumps and numerous adits. Water draining from the waste material is alkaline.

The Titania and Rana Mines are the only two metal mines operating in Norway today. Titania Mine extracts ilmenite, and the ore is relatively low in sulfides. Rana Mine extracts hematite and magnetite, and is a high grade iron mine with no sulfides.

Environmental Aspects of the Mining

The mining of massive sulfide deposits has been especially harsh to the local environment. This was documented in the 1700's: Johan Falkberget described a lake (Orvsjøen) in the Røros district as a "dead sea, without life, and poisoned. No fish moving in the red-pink water in the evening ..." Today, this lake is still without fish (Norsk Institutt for Water Research 1991).

Water analyses were started in the early 1970's as a response to the first Norwegian Water Quality Act. Some rivers and fjords have been monitored for almost 30 years. Several of the mines in Norway discharged tailings into nearby lakes (Bleikvassli, Grong, Røros Kobberværk, Skorovatn, Sulitjelma). The contamination from these deposits is considerably less than from tailings deposited in tailings ponds (Arnesen et al. 1997).

Remediation and reclamation efforts have been performed at many mines in Norway. The Norwegian government has taken responsibility for reclaiming abandoned mines. The Commissioner of Mines, assisted by the Norwegian Institute for Water Research (NIVA), oversaw most of the reclamation and remediation between 1985 and 1995; however, some mitiga-

Table 1. Production numbers for the largest mines in Norway (in 1000 t)

Mine	Pyrite	Copper	Zinc
Orkla (Løkken)	19,649	511	46
Foldal (Hjerkin)	8,897	204	174
Sulitjelma	7,266	421	66
Røros	553	126	17
Grong	—	148	129

tion work is still ongoing. The following are a few examples of these efforts (Figure 1).

Røros Mining District

The Røros mining district includes many massive sulfide deposits that have a very high potential to generate low pH, highly metalliferous waste streams. Prior to mining, many of the deposits were exposed at the surface and generated metal-laden streams. Copper, together with lead and zinc, have been leaching from mine waste material and contaminating streams and rivers in the area. Some of the tailings and waste rocks have been covered with local till/moraine material; this has reduced metal leaching into the streams and rivers by approximately 80%.

However, there are still many smaller point sources that are contaminating streams. One of these is the Kongen's Gruve (the King's Mine) adit, where several different approaches have been tried to improve the stream water quality. An attempt was made to extract copper and iron from the water in a pilot-scale experiment, but the cost of chemicals exceeded the value of the metals recovered. Pilot-scale wetland experiments, funded by the Commission of Mines, have also been performed for several years, with promising results; a full-scale wetland is being designed. However, currently, the adit water is still discharging into the Orvsjøen.

Reclamation activities have been further hindered by the fact that Røros and some mining areas in the district are listed as World Heritage Sites. Therefore, many waste rock piles and tailings inside and around Røros are protected, limiting characterization and reclamation efforts.

Løkken Mine

Like most of the metal sulfide mines, Løkken Mine has had contaminated water discharging from tailings, waste rocks, and adits (pH 2–2.2, 500 mg/L of Cu, and 700 mg/L of Zn). The mine produced approximately 250,000 t of tailings a year for about 13 years until closure in 1987. The tailings were deposited behind a watertight dam. After closure, the tailings were flooded to reduce sulfide oxidation. The water quality of the effluent stream from the dam improved immediately afterwards; however, both zinc and copper concentrations (approximately 0.5 and 2 mg/L, respectively) have since been increasing (Arnesen et al. 1997).

Underground mine water quality improved considerably when the mine was flooded after partial closure: pH increased from 2 to 4; Cu decreased from 530 to 1 mg/L; though Zn increased from 1,500 to 4,000 mg/L (Arnesen and Iversen 1997). However,

with time, the water in the underground workings stratified, with much lower Zn concentrations (30–50 mg/L) in the shallower zone, while Cu stabilized at 5–10 mg/L. Seeps from the waste rocks are being pumped into the flooded mine workings. During operation, water pumped from the mines were the main source of contamination in the nearby salmon-bearing Orkla River; metal loading to the Orkla River has since been reduced by approximately 90% due to these efforts.

Odda Zinc Smelter

The zinc smelter in Odda lies along the Sørfjord, which is considered to be the most polluted fjord in the world due to several smelters (zinc, aluminum, titanium, carbide) discharging their effluent into it. The zinc smelter contributed the greatest contaminant load, discharging its effluent to a shallow bank along the fjord. The sulfate-laden effluent contained high concentrations of Cd, As, and Se, in addition to Zn. Solids settled on the embankment, but slow leaching over the years has so contaminated fish and bottom fauna that they are hazardous to eat. In the early 1990's, the company had the shallow deposit covered with a geo-textile and sediment to avoid further leaching. Rock caverns were constructed in the mountain along the fjord for any further effluent deposition.

Sulitjelma Mining District

The Sulitjelma Mining District is another massive sulfide deposit where copper and zinc were the main metals extracted. More than 10 mines in the district drained into the same lake, Langvatn. Tailings from the flotation plant fed by several of the mines were deposited over the years along the shoreline. There were discharges from several of the mine adits and leaching from numerous waste rock dumps. Prior to reclamation, drainage pH ranged from 2.4–3.2, with Cu and Zn concentrations up to 500 and 250 mg/L, respectively. Groundwater from wells in the area were also acidic (pH 2.8–5), but with metal concentrations an order of magnitude lower (Norsk Institutt for Water Research 1991).

The tailings have since been covered with soil and revegetated. Several of the adits have been plugged and the main acidic drainage is piped into underground workings. This has reduced metal loading of surface water by 80–90%.

Hjerkinn Mine

The Hjerkinn Mine is a massive sulfide deposit situated above the timber line. Principal minerals are pyrite, pyrrhotite, chalcopyrite, sphalerite, and magnetite. Copper, zinc, and sulfur was extracted from the ore.

The tailings were discharged into a tailings pond and were always kept underwater. Copper concentrations are around 0.05 mg/L and Zn between 0.1 and 0.2 mg/L. There are currently trout living in the tailings pond.

Råna Mine

Råna Mine was a nickel mine located in northern Norway. It operated for approximately 10 years and closed in 2002. Tailings were deposited in flood plains along the stream, and on top of older tailings from the Bjørkåsen Mines. These older tailings were seeping Cu-laden effluent. The newer tailings have reduced the seepage considerably. The recipient fjord has been on the Norwegian Pollution Agency (SFT) watch list for many years. The tailings from the Råna operation have been covered with 10 cm of borrow material and revegetated. The gangue minerals (olivine and anorthite) have a high buffering capacity. However, olivine and nickel sulfide minerals have the potential to oxidize and release nickel into the environment. Streams around the mine are being monitored continually. Small amounts of waste rock were put back into the mine opening. The mine is still being flooded and the final effluent water quality, though unknown, is expected to be relatively good.

Titania Mine

Titania Mine has extracted ilmenite ore since 1902, with ilmenite, pyroxene, amphibole, biotite, magnetite,

and minor pyrite as the main minerals. Coarse tailings were deposited onshore from 1918 to 1965. Fine tailings were discharged in the nearby river and transported into the fjord. From 1960 to 1984, all tailings were deposited into the fjord. Due to an environmental push, the mine was required to deposit its tailings onshore behind tailings dams. Within two years after fjord deposition ceased, the bottom fauna had recovered. There was no metal leaching detected from the fjord deposits, which were monitored frequently in the 1980's and 90's (Norsk Institutt for Water Research 1985, 1991, 1994). The mine is looking into restarting fjord deposition due to a lack of surface space for tailings.

Conclusions

The Norwegian government has taken responsibility for reclamation and remediation of abandoned mines, which has considerably reduced metal loading from these sites. Monitoring programs continue at most of the mitigated sites. The regulatory agencies are awaiting new EU regulations on tailings and waste rocks before developing specific mining regulations that will also cover industrial mineral mines. Characterization of acid rock drainage has focused very much on water quality in the draining and recipient streams, while geochemical and geotechnical characterization of the waste material has been sparse. For this reason, many of the reclaimed sites will have to be revisited in the near future when pollution loading starts to increase.

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Finland – Mine Water Quality in some Abandoned and Active Finnish Metal Sulphide Mines

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Introduction

Finland's mining history began with small pits and continues in large underground mines today. From 1530 to 1995, at least 280 metallic ore mines were in

operation. Most of these mines have exploited iron or copper ores (Puustinen 1997). All 33 metal sulphide mines or Au mines were worked in the 20th century and some of them are still in operation. The production of metallic ore reached its peak in 1979, when ore output totalled 10.3 Mt (Puustinen 1997). At present, there are only four active metal sulphide mines, and ore output in 2003 was 2.14 Mt (Söderholm 2004).

This paper presents some data on surface and ground-water quality at six abandoned metal sulphide mines and one mine currently in operation. Since 2000, the Geological Survey of Finland has investigated environmental impacts of mines wastes and criteria for the mitigation in the old Hammaslahti Cu-Zn mine (1974–1986), Luikonlahti Cu-Zn-Co mine (1968–1983), Enonkoski Ni-Cu mine (1985–1994) and Otravaara pyrite mine (1919–1923; Räisänen 2003; Räisänen and Juntunen 2004). The Department of Geology at the University of Turku has done studies in the old Aijala (1949–58, 1964–1974) and Orijärvi Cu-Zn-Pb-Ag mines (1758–1954) and in a joint project

Table 2. The pH and mean concentrations ($\mu\text{g/L}$) of Al, Co, Cu, Fe, Mn, Ni, S and Zn in surface waters in areas of the old Hammaslahti Cu-Zn mine, Luikonlahti Cu-Zn-Co mine, Enonkoski Ni-Cu mine and Otravaara pyrite mine and their neighbouring, uncontaminated (background) areas in eastern Finland; metal concentrations were determined with ICP-AES and ICP-MS techniques using filtered water samples (Räsänen et al. 2001)

	pH	Al	Co	Cu	Fe	Mn	Ni	S	Zn
Hammaslahti Cu-Zn Mine (2001)									
Pit, discharge ($n = 4$)	5.1	108	21	2.1	26	8880	46	432	297
Tailings heap leachate ($n = 4$)	3.3	8940	64	34	35	8518	98	451	936
Iiksenjoki River* ($n = 4$)	5.9	345	3.7	5.7	1.1	332	10	27.5	73
Background surface waters ($n = 25$)	5.7	533	4.9	6.7	5.21	476	12	28.6	51
Luikonlahti Cu-Zn-Co Mine (2003)									
Tailings impoundment leachate ($n = 10$)	3.6	10,000	348	513	228	20,000	822	580	4135
Tailings impoundment discharge** ($n = 6$)	3.7	4680	191	121	12.5	14,100	654	280	1600
Background surface waters ($n = 4$)	6	224	0.9	2.6	0.38	24	5.8	3.08	12
Enonkoski Ni-Cu Mine, tailings heap surroundings (2002)									
Leachate ($n = 1$)	6.4	10	35	<0.5	12.9	3040	1220	297	15
Discharge*** ($n = 1$)	7.5	8.5	0.5	0.7	0.68	108	142	76.5	8.6
Otravaara pyrite mine, rock pile surroundings									
Seepage water ($n = 6$)	2.3	194,000	652	1940	1400	8330	960	1740	6180
Discharge*** ($n = 5$)	3.7	2480	11	7.6	3.40	386	15	28.7	42
Background surface waters	5.8	231	0.9	1.8	0.46	33	1.9	3.47	9.6

n refers to number of samples; *Below the discharge from tailings heap and pit; **Discharge is pumped into settling pool for purification (liming); ***Discharge from natural wetland into lake

Table 3. The pH and mean concentrations (mg/L) of sulphate, Cu, Fe and Zn in ground and surface water from the Orijärvi and Aijala Cu-Zn-Pb-Ag mine areas in southern Finland. Sulphate concentrations were determined with ion spectrophotometry; Cu, Fe and Zn concentrations were determined in filtered samples with AAS

	pH	SO_4^{2-}	Cu	Fe	Zn
Orijärvi Cu-Zn-Pb-Ag Mine (1998-1999)					
Groundwater (tailings) ($n = 1$)	2.5	3800	0.58	1.32	11.1
Surface water quality					
Pit, discharge ($n = 2$)	4.9	56.5	1.67	0.47	17.0
Leachate from the tailings area ($n = 4$)	5.1	217	0.64	0.71	14.3
Lake Orijärvi ($n = 8$)	6.1	22.5	0.02	0.09	0.24
Uncontaminated waters					
Lake Määrjärvi ($n = 3$)	6.4	18.7	0.02	0.07	0.27
Brook ($n = 1$)	6.3	1.0	0.03	0.14	0.01
Aijala Cu-Zn-Pb-Ag Mine (2001-2004)					
Groundwater quality					
Tailings area ($n = 2$)	5.8	17,400	0.19	7114	82.0
Influence area ($n = 2$)	6.5	1400	0.04	47.3	0.56
Uncontaminated ($n = 1$)	6.4	14.0	<0.01	0.25	0.11
Surface water quality					
Tailings water ($n = 3$)	4.3	20,000	0.56	43.8	13.3
Brook from the tailings area (< 500m, $n = 7$)	5.3	434	0.17	18.1	4.59
Brook from the tailings area (> 500m, $n = 5$)	5.8	194	0.20	4.48	1.58
Kiskonjoki river* ($n = 2$)	6.2	29.5	0.02	1.15	0.17
Uncontaminated waters					
Kiskonjoki river ($n = 2$)	6.8	12.5	<0.01	0.76	0.11
Brook ($n = 1$)	6.2	16.0	0.04	0.29	0.04

n refers to the number of samples, < concentration below the detection value

Table 4. The pH and concentrations (mg/L) of sulphate, Cu, Fe and Ni in the Hitura Ni-Cu mine area, western Finland. Mine water and surface water results were calculated mean values from annual monitoring data; the groundwater mean values were reported by Heikkinen et al. (2002).

Hitura Ni-Cu Mine	pH	SO ₄ ²⁻	Cu	Fe	Ni
Mine water, 1998–2000 and 2002–2004	7.5	499	–	0.40	0.15
Settling pond in the facility, 1989–2001	6.1	4134	0.04	1.80	2.33
Discharge*, 1989–2001	6.2	235	0.02	1.17	0.17
Kalajoki river**, 1989–2001	6.4	17.0	0.02	1.19	0.01
Kalajoki river (upstream, uncontaminated), 1989–2001	6.5	7.93	0.02	1.17	0.01
Contaminated groundwater, 1999*** (<i>n</i> = 31)	5.7	4320	0.04	51.7	3.65
Uncontaminated groundwater, 1999*** (<i>n</i> = 42)	6.3	55.5	0.01	5.90	0.02

*Discharge to the Ainasoja brook after settling; **Below the discharge from the Ainasoja brook; ***Heikkinen et al. 2002

with Outokumpu Mining Oy at the Hitura Ni-Cu mine (1970–today), which is still in operation (Heikkinen et al. 2002).

Surface Water and Groundwater Quality in Abandoned Mines

In Finland, tailings from several abandoned metal sulphide mines have been engineered in bog and pond basins. In both cases, substrata are organic rich sediments underlying glaciolacustrine silt sediments that are compressed under load. This leads to a waterproof basement (Sipilä and Salminen 1997; Räisänen 2003). Water seeps through the dams and sites where the tailings rest to the water-permeable moraine hills (Räisänen 2003). This describes the situation at the tailings impoundments of the Hammaslahti Cu-Zn mine and the Luikonlahti Cu-Zn-Co mine previously used in talc operations (Räisänen 2003; Räisänen and Juntunen 2004). There, contamination of seepage waters is mainly limited to surface waters outside the facilities (Table 2), whereas groundwater under the facility is less contaminated due to the waterproof basement. Constructed wetlands have been planned to treat the contaminated surface waters in both areas.

Tailings from the Enonkoski Ni-Cu mine were dammed on compact till overlying a thin organic sediment layer in a topographic depression. Water from the facility mainly seeps into a bog, which acted as a settling pond during mining, and which has naturally transformed into a wetland after the mine was closed. As seen in Table 2, metal and sulphur concentrations in leachate from the tailings are markedly decreased before discharging into the lake. A similar trend is seen in the wetland complex in the Otravaara pyrite mine area (Table 2). The wetland that originally formed has increased in size via sedimentation of Fe precipitates and organic matter in the creek valley downstream of the rock pile.

Tailings at the Aijala and Orijärvi Cu-Zn-Pb-Ag mines are dammed in topographic depressions. In both cases, the tailings are oxidized in the upper parts, raising metal and sulphate concentrations in tailings waters and groundwaters close to the facility. In contrast to the Aijala mine, the tailings water in the older Orijärvi facility is very acidic (Table 3). Surface water discharging from both mine areas is slightly acid and contains less sulphate and metals than the tailings water due to precipitation in brooks.

Surface and Groundwater Quality in the Hitura Mine Area

The Hitura nickel mine has been operating since 1970 in Nivala, western Finland (63° 50' N, 25° 02' E). Ni-Cu ore was originally extracted from an ultramafic complex by open pit mining until 1991, when underground production began. The wastewater and tailings from the milling process are pumped as slurry into an impoundment located a few hundred meters south of the mill. After settling and clarification, wastewater is recycled back to the milling process for reuse. Additional water for the process is received from the mine dewatering and from a local water supply. During wet seasons, part of the wastewater and the pumped mine water is discharged through settling ponds, ditches and the Ainasoja brook into the nearby Kalajoki River. The environmental permit of the mine defines the maximum allowed concentrations for nickel (< 2.5 mg/L and 200 kg/a) and sets limits for pH (between 6 and 7) and the amount of discharge water (maximum of 0.2% of the Kalajoki river streamflow). As Table 4 shows, the present impact of the mine on the Kalajoki river water quality has recently been almost negligible.

Initial indications of groundwater contamination around the tailings area were detected in the early 1980's (Korkka-Niemi et al. 1999). Protective measures were taken to prevent further pollution, but the measures have proven to be insufficient due to the rais-

ing of the tailings area. Groundwater contamination (elevated sulphate, nickel, and chloride) has been reported close to the tailings facility (Table 4; Heikkinen et al. 2002; Korkka-Niemi et al. 1999). Recently, the Hitura mine has sealed the leaking tailings dams and deepened the seepage ditches around the tailings area (Artimo et al. 2004; Heikkinen et al.

2001). A closure plan, including technical solutions to inhibit future loadings from waste areas, is under preparation by the Hitura mine.

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Sweden

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Historical Situation

Mining of metals has been carried out in Sweden for more than 1000 years, primarily in three provinces: the Norrbotten and Skellefte fields in the north and a region known as Bergslagen in the middle of the country (Figure 2). In addition, other ore bodies are spread throughout the country, with mining currently concentrated in the middle and northern parts. Historically, mining also occurred in the south. Mining has played a key role in Sweden's socio-economic development, and is still socio-economically important in the northern part of the country.

The total number of mines operating in Sweden has decreased from approximately 500 in the 1920s to 16 in 1999 (2 iron ore mines, 14 non-ferrous ore mines; Salmon and Destouni 2001); despite this, total ore production has increased, from approximately 4 million t in 1900 to about 45 million t in 1999 (SGU 2000). The increasing profitability of mining lower grade ores, due to improved extraction techniques, as well as the use of open cut mines, has increased production of mining waste (SEPA 1998). Today, the mining industry generates approximately half of all waste produced in Sweden (SCB 2000), most of which is deposited above land.

Overview of Current Working Mines

Sweden has substantial base metal, gold and iron ore deposits that are being actively mined and developed, as well as peat mines and quarries, yielding Europe's largest mining industry (Table 5), which in 2003 produced around 2% of the world's iron ore input and 1%, 3.3%, and 3.7% of the western world's production of copper, zinc, and lead, respectively. Mining accounted for 1% of the market value of Sweden's total industrial production and employed 0.5% of the total industri-

al labour force (Swedish Institute 2003). The two major types of metal ore deposits being mined today are iron ore (mined at two sites), which usually contain little or no sulphide minerals, and non-ferrous ores (mined at 14 sites), where the extractable element is often found as a sulphide; the non-ferrous ores in Sweden are mined for copper, zinc, lead, gold, silver, and graphite.

Three large mining companies dominate Sweden's mineral production. In terms of production volumes, Boliden Mineral AB (sulphide ores) is the largest company, followed by LKAB (iron ore), and the much smaller Zinkgruvan Mining AB. Boliden Mineral AB operates several polymetallic mines, as well a copper mine and a lead zinc mine; the company's open cut copper mine at Aitik in northern Sweden (Figure 2) is responsible for 75% of the annual Swedish sulphide ore production, and also for 85% of the annual production of waste (SEPA 1998).

Mine Water Management and Problems

The greatest long-term source of water pollution associated with mining is waste from the mining of sulphide ores (SEPA 1995); it is estimated that in 1995, approximately 60% of all lead, cadmium, zinc, and copper discharges to water in Sweden were related to mining and primarily to mine waste. Even though major water quality problems have been observed elsewhere to also originate from water-filled voids of

Table 5. Swedish metal production and its position in EU in 2003 (according to statistics available from the Swedish Mining Association at <http://www.mining.se/statistics.html> in January 2004)

Metal	Production*, kt	EU production	EU rank
iron	21,500	90%	1
silver	0.341	60%	1
copper	83.1	47%	1
gold	0.006	29%	1
lead	51.0	30%	1
zinc	186	25%	2

*weight of metal content in ore, with the exception of iron, which is given as weight of ore

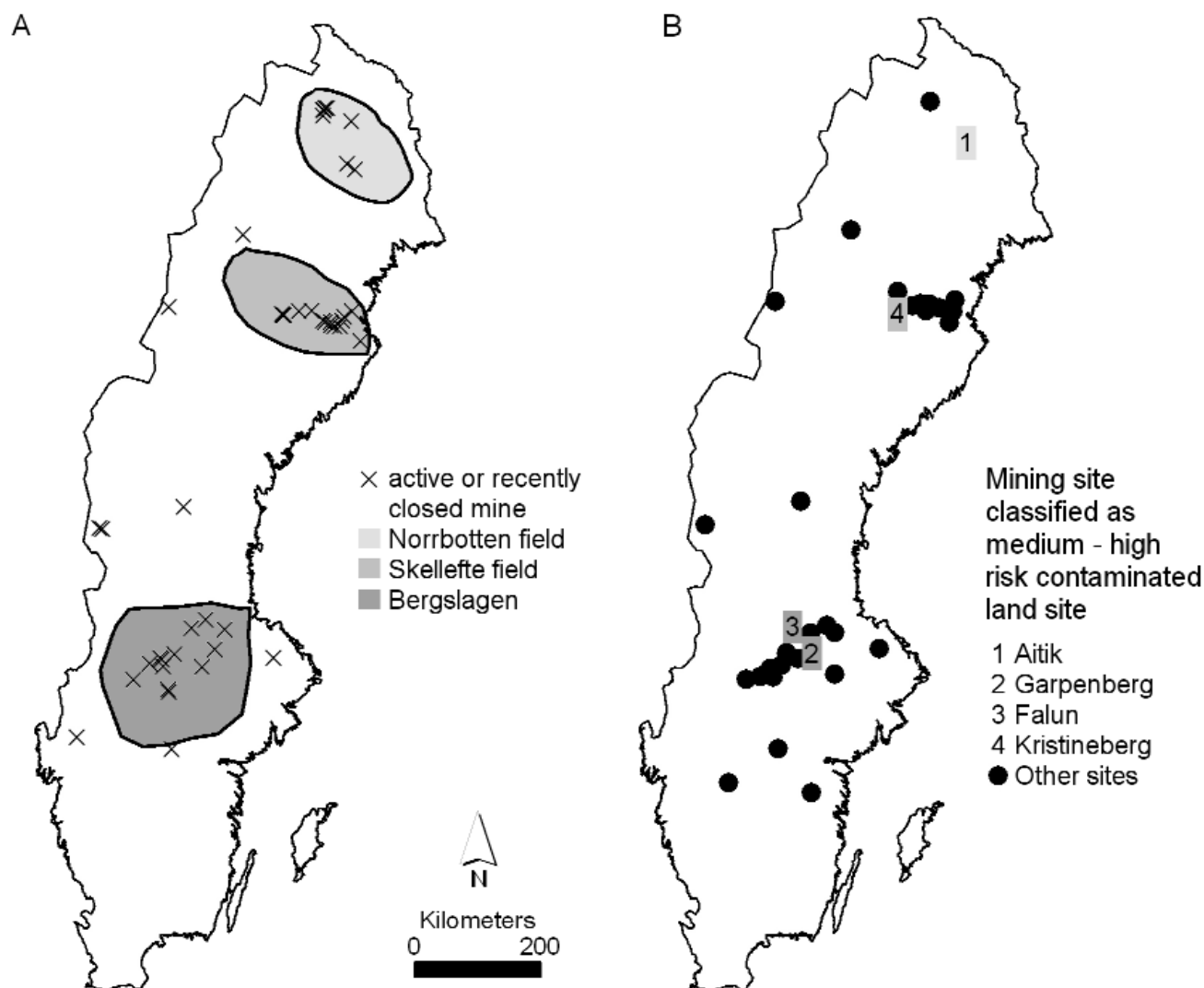


Figure 2. Mineral deposits in Sweden and mine sites with high and medium environmental risks (from Salmon and Destouni 2001): A) Mineral deposits in Sweden and active or recently closed mines; the demarcated regions are, from top to bottom: the Norrbotten field, the Skellefte field, and Bergslagen field (after SGU, with permission); B) the approximate location of 42 of the 50 mining sites that were classified by the Swedish Environmental Protection Agency as high to medium risk to the environment (coordinates were not available for the remaining 8 sites; modified from SEPA 1995); mine sites discussed in this paper: 1) Aitik, 2) Garpenberg, 3) Falun, and 4) Kristineberg.

closed mines (Woods et al. 1999; Younger 1997; Younger 2003), such problems are generally not considered important in Sweden (Destouni 2003; Salmon and Destouni 2001; SEPA 1998). The scientific and empirical basis for such a view is questionable, as demonstrated by a recent environmental assessment report on mining activities by Boliden Mineral AB in Garpenberg (Karlsson 2005; see Figure 2B for location), which states that diffuse emissions of metals to groundwater are not possible to assess due to the severe lack of groundwater quality information in the area. It appears that Swedish authorities rely on a single groundwater sampling well for the whole area to determine if and which mine water sources are affecting groundwater quality. In addition, the Swedish

Geological Survey recently reported a general lack of groundwater quality data relative to complying with the EU Water Framework Directive (Åsman and Ojala 2005). With such limited information, the amount of mining-related copper, zinc, lead, and cadmium leaching from Swedish mine waste deposits are estimated to be approximately $400 \cdot 10^3$ kg/year (SEPA 1998), with the Falun (estimated at $200 \cdot 10^3$ kg/year) and Saxberget (estimated at $20 \cdot 10^3$ kg/year) mining areas constituting two of the main metal sources.

In several places elevated concentrations of metals downstream of historic mining sites have been found to seriously affect water quality and thereby also aquatic biota (Lithner 1998). Some mines are located

upstream of valuable nature reserves. One example is the Färnebofjärden national park, located in the lower part of the Dalälven catchment, where Bergslagen, one of the largest mining regions in Sweden, is also located (Figure 2). Accidents involving failure of mine water containment structures have occurred, with the most recent example being the Aitik tailings dam failure in 2000, which resulted in a release of 1.6 million m³ of tailings impoundment water into surrounding waterways, though it is not judged to have caused environmental harm (Göransson et al. 2001).

In 1995, the Swedish Environmental Protection Agency inventoried known contaminated land sites, with the aim of identifying the need for remediation measures. Of the 1700 sites investigated, 70 potentially contaminated sites were associated with the mining industry, and 50 of these sites were classified as posing a large to medium risk for health and the environment (SEPA 1995, 1998; Figure 2).

Exploration, Remediation and Monitoring

Despite a global downturn in exploration expenditure, Sweden continues to attract exploration investment, increasing by more than 8% from 1999 to 2001 (Salmon and Destouni 2001). Total ore production increased to approximately 45 million metric tons in 1999 (SGU 2000). Several foreign companies are actively exploring Sweden for base metals, gold, and diamonds, and Sweden's Skellefte District (Figure 2) has been recognised as a gold province, with similarities to deposits in Australia (Hannerz and Destouni 2002).

Of the 50 mine waste sites that pose large to medium risk for health and the environment (SEPA 1995, 1998; Figure 2), as of 1998, 8 sites were still in active use, and remediation had been planned or completed for 23 of the sites, leaving 19 for which decisions on remediation have not yet made. In addition, there are approximately 900 mostly small sites with sulphide waste rock, where remedial actions have not been taken (Salmon and Destouni 2001).

Water and soil covers are the commonly chosen and planned mining waste remediation methods in Sweden; the main aim of both is to create a barrier to oxygen diffusion into mine waste deposits. Water cover remediation projects to date have been less expensive than application of soil covers (10–20 kr/m² compared to 80–200 kr/m²; SEPA 1998). Research studies for identifying catchment-scale cost-effective mine waste remediation measures under various conditions, using the Swedish catchment of the Dalälven River as a case study for model quantification, have therefore considered soil and water covers as typical

Swedish mine waste remediation measures and arrived at optimal catchment-scale remediation solutions that include water covers wherever hydrological conditions are favourable, and soil covers only under unfavourable hydrological conditions and/or for very high emission reduction requirements. The possibility of downstream metal load abatement measures, such as wetland construction, as potential complementary approaches to mine water remediation, even though such downstream approaches are not normally considered in Sweden; results showed that such measures are economically viable compared to both water and soil covers at mine waste sources, in particular given the uncertainties caused by the limited availability of groundwater quality data (Baresel et al. 2003, 2005; ERMITE Consortium 2004).

Supervision of ongoing mining in Sweden is generally based on the self-check principle (Holmgren 2000), in which the mining company is responsible for ascertaining whether its emissions and recipient impact are in accordance with permit conditions, and the supervising authorities carry out checks to verify company reports, with obligation to report to the public prosecutor findings of permit conditions not being fulfilled (Lindgren and Destouni 2002). However, the limitations in groundwater quality data discussed above imply a major gap in Swedish water quality monitoring and management and put into question interpretations of point and diffuse pollutant source apportionment drawn primarily from surface water observations. An associated institutional gap has been pointed out (Destouni 2003; Lindgren and Destouni 2002): no Swedish institution has the responsibility (and required competence and resources) for coordinated monitoring, supervision, and management of diffuse water pollution sources over the wide range of spatial-temporal scales that are generally implied by pollutant transport through coupled surface-subsurface pathways (Kirchner et al. 2000; Lindgren et al. 2004), especially when various sources of mine water pollution are involved (Malmström et al. 2004). This institutional gap has not been resolved by the five Swedish water authorities, each with a staff of a few individuals, recently established in order to fulfil the formal requirements of the EU Water Framework Directive.

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Mine Water Issues in the United Kingdom

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The UK has a rich history of both coal and metal mining. However, with a few notable exceptions, the coal and ore-fields of the UK are now abandoned. Mine water issues are therefore a major concern across former mining areas of the UK. Problems encountered include rising mine waters (which sometimes intercept important aquifers), and surface water pollution arising from discharges of acidic and/or metalliferous mine waters from abandoned mine and spoil heaps. This summary article provides an overview of the type of problems encountered in the UK, the measures taken to address them to date, and the likely direction of future research.

In the United Kingdom (UK), owners of mines abandoned prior to 1 January 2000 cannot be held liable for pollution from their mine unless causation of pollution can be proved. Due to the complex hydrological setting of the underground mines, it is extremely difficult to prove causation; only one successful prosecution was ever brought prior to 1 January 2000. Since most mines in the UK were abandoned prior to this date, the vast majority effectively fall outside legislative control. Further details on the legislative loophole that precipitated this situation can be found in Fry (1997).

The inability to apportion liability for mine water pollution in the UK is a serious issue given the extent of

the problem and the requirements of the Water Framework Directive (WFD). Mine water pollution is currently one of the most severe and most widespread forms of aquatic pollution in the UK (NRA 1994; Younger 2001; Potter et al. 2004). The England & Wales Environment Agency has recently produced its draft pressure assessment maps, as the first stage of meeting the objectives of the European Union Water Framework Directive. The map for mining impacted waters suggests that approximately 2,500 km of streams in England and Wales, and an area of some 9,000 km² of groundwater bodies, are “at risk” of failure of WFD objectives due to mine water pollution.

Volumetrically, the majority of mine water issues in the UK relate to discharges from abandoned coal mines (Figure 3). Since the legal structure in the UK does not allow liability for pollution to be apportioned, the government-funded Coal Authority has adopted responsibility for addressing discharges from abandoned underground coal mines. Since its formation in 1994, the Coal Authority has implemented more than 20 full-scale treatment schemes, and oversees the monitoring of rising mine waters across the UK. The method the Coal Authority uses to prioritise and select the discharges for treatment was developed by the England & Wales Environment Agency in the mid 1990s, based on the scale and severity of the chemical and biological impacts of mine waters on the receiving watercourses. These lists, initially produced on a regional basis, have now been combined into a national priority list. For the interested reader, Davies et al. (1997) describe the methodology, and Jarvis and Younger (2000) provide a critique of mine water impact assessment techniques in general.

The majority of these treatment systems remediate net-alkaline, ferruginous mine waters, and typically comprise aeration, settlement lagoons, and aerobic wetlands. A few include chemical dosing, either for raising pH (usually using sodium hydroxide), or for rapid oxidation of ferrous iron (usually using hydrogen peroxide). Only one Coal Authority system, at Deerplay, Lancashire, includes a reducing and alkalinity producing system (RAPS) for elevation of alkalinity (see Jarvis and England 2002). The RAPS at Deerplay treats spoil tip leachate, and is located on the same site as a larger system that treats net-alkaline deep coal mine water (settlement lagoons and aerobic wetland). Settlement lagoons and aerobic wetlands are not particularly a focus of treatment research in the UK at present. A more pressing issue at coal mine water treatment sites is the large volumes of ochreous sludge that are generated and, specifically, how to re-use or dispose of them.

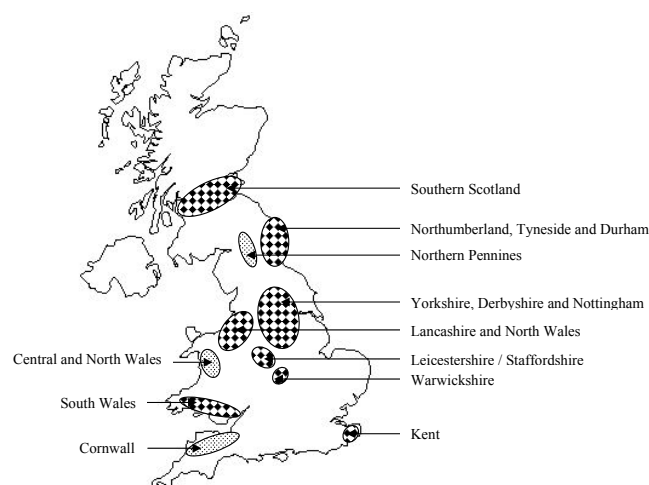


Figure 3. Main mining regions of the UK (diamonds indicate coal mining; dots indicate metal mining)

With the exception of those that it owns, coal mine spoil heap discharges do not fall within the remit of the

Coal Authority. Responsibility for treatment of such discharges ordinarily lies with the landowner. Novel passive treatment systems have been installed at some of these sites due to their more acidic nature (Rees et al. 2002). For example, the first compost wetland in Europe was installed at Quaking Houses, County Durham, in 1997 (see Jarvis and Younger, 1999). At Shilbottle, Northumberland, a permeable reactive barrier (PRB) is now operational, and at Bowden Close, County Durham, a RAPS unit has recently been commissioned (Younger et al. 2003). These systems are currently the main focus of passive treatment research activities in the UK. Particular areas of interest include the behaviour of sulphate reducing bacteria in aiding remediation, sulphur and carbon cycling in general, and calculating the longevity of carbon supply and its impact on the lifespan of such passive systems.

An area that has received little attention in the UK is the issue of diffuse mine water pollution. Diffuse pollution certainly arises from spoil heaps, but may also arise as seepage faces at the surface in areas of former shallow mines. Accurate quantification of diffuse sources, and designing effective approaches to remediation, is a particular area of current interest to the England & Wales Environment Agency.

There is no equivalent body to the Coal Authority that has responsibility for addressing metal mine discharges in the UK, and only one major metal mine discharge, at Wheal Jane, Cornwall, has a treatment system associated with it. There is currently no sign of any immediate resolution to this problem. Metal mine discharges affect Wales and Cornwall in particular, as well as many parts of northern England (Figure 3). Discharges can be highly acidic or circumneutral, depending on the host mineralization, and contami-

nants may include zinc, copper, lead, and iron. The treatment challenge is further exacerbated by typically remote locations, difficult topography, and the proximity of many discharges to the receiving watercourse. The England & Wales Environment Agency has begun assessing the severity and potential for treatment of such discharges (Environment Agency Wales 2002; Johnston 2004; Rees et al. 2004), but it remains unclear who will take responsibility for financing the construction of, and then operating, metal mine water treatment systems.

Consistent removal of metals such as zinc and copper from acidic and circumneutral mine water using passive treatment technologies is a subject of ongoing research in the UK. Efforts to date have included the removal of zinc as carbonate minerals in anoxic limestone drains (e.g., Nuttall and Younger 2000). More recently, the potential of dried and pelletised ochreous sludge from coal drainage treatment, already demonstrated as having potential for phosphorus removal (Heal et al. 2004), has been investigated as a media for metal sorption (unpublished data). Bacterially-mediated systems for zinc and copper removal are also being investigated, although there are currently no systems operating at field-scale (to the authors' knowledge).

In summary, in terms of coal mine drainage treatment, the UK has an excellent track record in both research and application. The challenges for the future will undoubtedly be in areas such as ochreous sludge management, addressing rising mine water issues, and remediating acidic and circumneutral metal mine water discharges.

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Estonia

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Introduction

Estonia is a small country and generally not known for its mining, as its specific mineral resource, oil shale, is often not included in overviews and comparisons. Estonian oil shale is called kukersite. It is a light to dark brown (calcareous and terrigenous) sedimentary rock, the main components of which are organic mat-

ter of sapropel origin (kerogen, 20–60% by mass) and mineral matter (40–80%). Basically, all Estonian power generation and a large part of the chemical industry are based on burning and processing oil shale. As the calorific value of mined oil shale is low (between 10.2–11.5 MJ/kg), Estonia is Europe's leader in per capita production of solid fossil fuels (about 10 t per capita per year). Other mineral resources include phosphate (no longer mined), limestone, dolomite, clay, sand, gravel, peat, and curative mud (Figure 4). The problems related to active mine waters are almost exclusively related to oil shale mining, though from a historical perspective, water problems related to phosphate mining and uranium production are also important.

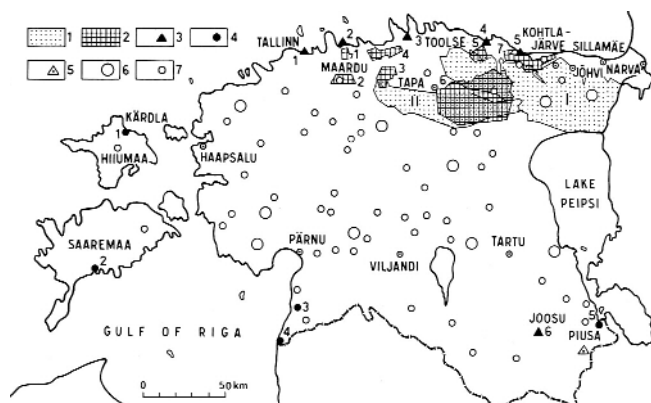


Figure 4. Mineral resources of Estonia: 1. oil shale (I – Estonia deposit, II – Tapa deposit); 2. phosphorite deposits (2.1 – Maardu, exhausted; 2.2 – Raasiku; 2.3 – Kehra; 2.4 – Tsitre; 2.5 – Toolse; 2.6 – Rakvere; 2.7 – Aseri); 3. clay deposits (3.1 – Kopli, exhausted; 3.2 – Kallavere; 3.3 – Kolgaküla; 3.4 – Kunda; 3.5 – Aseri; 3.6 – Joosu); 4. mineral water wells (4.1 – Kärkla; 4.2 – Kuressaare; 4.3 – Häädemeeste; 4.4 – Ikla; 4.5 – Värskä); 5. sand for glass: Piusa deposit; 6. peat bogs with area over 100 km²; and 7. peat bogs with area between 50 and 100 km² (Puura and Raukas 1997)

Historical situation

Oil shale

Oil shale has been extracted and used for 88 years; 900 million t have been extracted and either burnt in power plants (ca 85%) or chemically processed (ca 15%). The black areas on Figure 5 represent deposit boundaries and mined-out areas. Down to a productive layer depth of 30 m and at places covered with forests and bogs, opencast mining is preferred because it is cheaper and leads to quicker production (due to highly productive machinery), with safer and healthier working conditions than underground mining. Approximately half of the shale has been and is produced from opencast mines, while the other half is produced underground.

Phosphate

The opencast mining of sandstone that contains phosphate was carried out in Maardu from 1964 to 1991. Altogether, an area of 10.6 km² was mined out and covered by waste rock plateaus with heights of 5 to 25 m. Large trenches separate plateaus from each other and from the surrounding areas.

Uranium

In 1948, a secret company was started to produce uranium in Sillamäe. Local dictyonema shale was mined and used for a few years, but later uranium ore was

imported from Eastern Europe. In 1959, a tailings pond was established on the seashore. Incorporating pulp and wastewater from the production of tantalum and niobium, the tailings dam developed to a height of 25 m, with a tailings pond covering about 40 ha. A special remediation project is on-going and is scheduled to be completed by the end of 2006. As the share of local mine waste in the tailings pond is very small, the location is not considered as a mine waste site in Estonia.

Overview of Currently Working Mines

During the last fiscal year, which ended on 31 March 2004, the Estonian Oil Shale Co. produced 14 million t of oil shale. Narva Pitch Ltd and Aidu Pitch Ltd operate opencast mines, while Estonia Mine Ltd and Viru Mine Ltd operate underground mines. At other active mines in Estonia, 1.51 mill m³ of limestone, 0.339 mill m³ of dolomite, 0.168 mill m³ clay, 0.652 mill m³ gravel, and 1.402 mill m³ sand was extracted in 2002, as well as 1.508 mill t of peat and 1.7 thousand t of curative mud. Relative to mine waters, however, these activities had minimal effect on the environment.

Mine Water Management and Problems

Oil Shale Mines

During the last several decades, 200–240 million m³ of water have been pumped annually from oil shale mines and discharged into surface waters. In 2002, after several mines had been closed and flooded, 183 mill m³ of water was pumped out, which amounts to 77% of Estonia's groundwater use (Narusk and Nittim 2003). Approximately 175 million m³ were discharged from the operating mines. Groundwater levels surrounding the mining areas have dropped and 300 deeper wells have had to be drilled in the countryside to supply drinking water. The issue of drinking water supply in the central mining town, Kohtla-Järve, is very important at the moment. The possible alternatives are utilisation of deep groundwater (Cambrian-Vendian aquifer) that has naturally high concentrations of some elements (e.g. barium) or construction of a pipeline that would be several tens of kilometres long.

Closing of mines after decades of operation has led to other impacts. For example, towns have been established or expanded onto areas that were wetlands before mining started; with the water table rising after closure, there have been problems with flooding.

The pumped-out water is near-neutral, containing elevated concentrations of sulphate (200–500 mg/L). There are no problems with acidification, as the oil shale contains approximately 40% calcite (CaCO₃), buffering pyrite and acidic organic-matter oxidation products. In 2002, 86% of the waters were passed

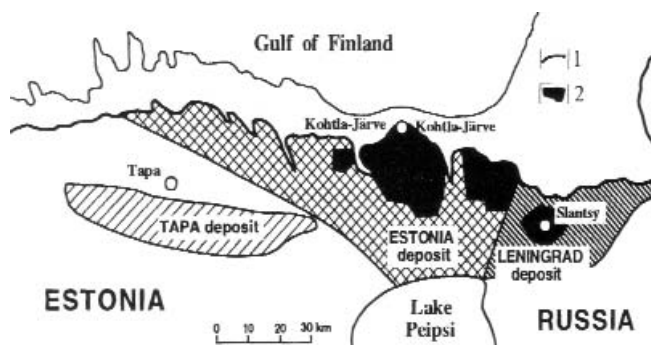


Figure 5. Location of oil shale deposits in the Baltic Oil Shale Basin: 1. Recent erosion boundary of kukersite oil shale; 2. mined out areas and fields of active mines (Raukas and Teedumäe (1997), with the permission of the Institute of Geology of Tallinn Technical University)

through sedimentation ponds to remove suspended solids (Narusk and Nittim 2003).

Regarding ions and complexes other than sulphate, the mine waters do not have a major impact on the chemistry of natural waters: total Fe, NH_4^+ , and BHT20 levels are decreased, and Cl^- , HCO_3^- , K^+ , and Na^+ levels somewhat increased. The concentrations of heavy metals in mining waters do not differ significantly from natural background values and are often even lower.

To avoid impacting the lakes in the Kurtina Landscape Preservation Area, the Viivikonna department of Narva Pitch Ltd use a filtration dam and infiltration pools so that they can excavate the oil-shale resources deposited on the border of the Preservation Area.

South of there, Puhatu National Park was established in 1999 to preserve the Puhatu bog and the species of the area. In the southern part of the oil-shale region of Sirgala, the mining area partly falls within the borders of this national park. It is anticipated that the excavation will reach the problematic territory in 15 years. The impact of mining on the level of subsoil water is regularly monitored at 3 observation wells in the southern part of the Narva oil-shale region.

Maardu phosphate mine

The Maardu phosphate mine site, located in northern Estonia, 10 km east of Tallinn, is rather unique; the range of major environmental problems is not caused by the commodity itself, but by hazardous compounds in the overburden. Ordovician and Cambrian sedimentary rocks in Scandinavia and Estonia contain layers of pyritic, metalliferous and organic-rich alum shale. Exposed to the atmosphere, pyrite in the shale tends to oxidise at high rates and release acidity and sulphates.

The leachate is often rich in heavy metals that were initially present as sulphides or were adsorbed/structurally contained in other solid phases of the dump. The total amount of the shale opened to oxidative weathering is 71.7 million t, with the potential to produce more than 5 million t of sulphuric acid during low-temperature oxidative weathering; assuming that each m^2 of the waste rock contains, on average, 7 t of shale containing 4–6% pyrite and 9–12% of kerogen-type organic matter (Puura 1998).

Key environmental issues related to mine waters at the site are:

- Low-temperature oxidation of alum shale, leading to pollution of groundwater and surface water with sulphates (up to 1 g/L);
- A risk for breakthrough of acidity and heavy metals. At the moment, limestone still buffers all acidity formed, but especially in the north-eastern part of the site, where the limestone layer was thinnest, there is a possibility for breakthrough of contaminants; and a
- Modified surface and groundwater pattern.

Future Mining, Treatment and Remediation Activities

Oil shale remains the main mineral resource of Estonia. Present national plans have not identified any other major power generation source for Estonia for the next several decades. Because of high carbonate content, acidification and metal mobilisation are not problematic in the mining region. The main unknown factor regarding mine waters is how elevated concentrations of sulphates (200–500 mg/L) in pumped-out waters will be assessed and approached by the EU Water Framework Directive implementation, as the quantities are very large. The Estonian Oil Shale Company has been successfully demonstrating the quality of the water after purification in sedimentation ponds by trout farming.

Presently, a model for hydrological changes after mine closure in oil shale mining region is being prepared by Tallinn Technical University's Mining Institute. Maardu phosphate waste heaps continue to discharge sulphates and the risk of elevated heavy metals remains. The mixed mining waste and processing tailings pond in Sillamäe will be remediated by the end of 2006.

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Ireland – How Green is The Emerald Isle? Consequences of Mining on Receiving Waters

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Introduction

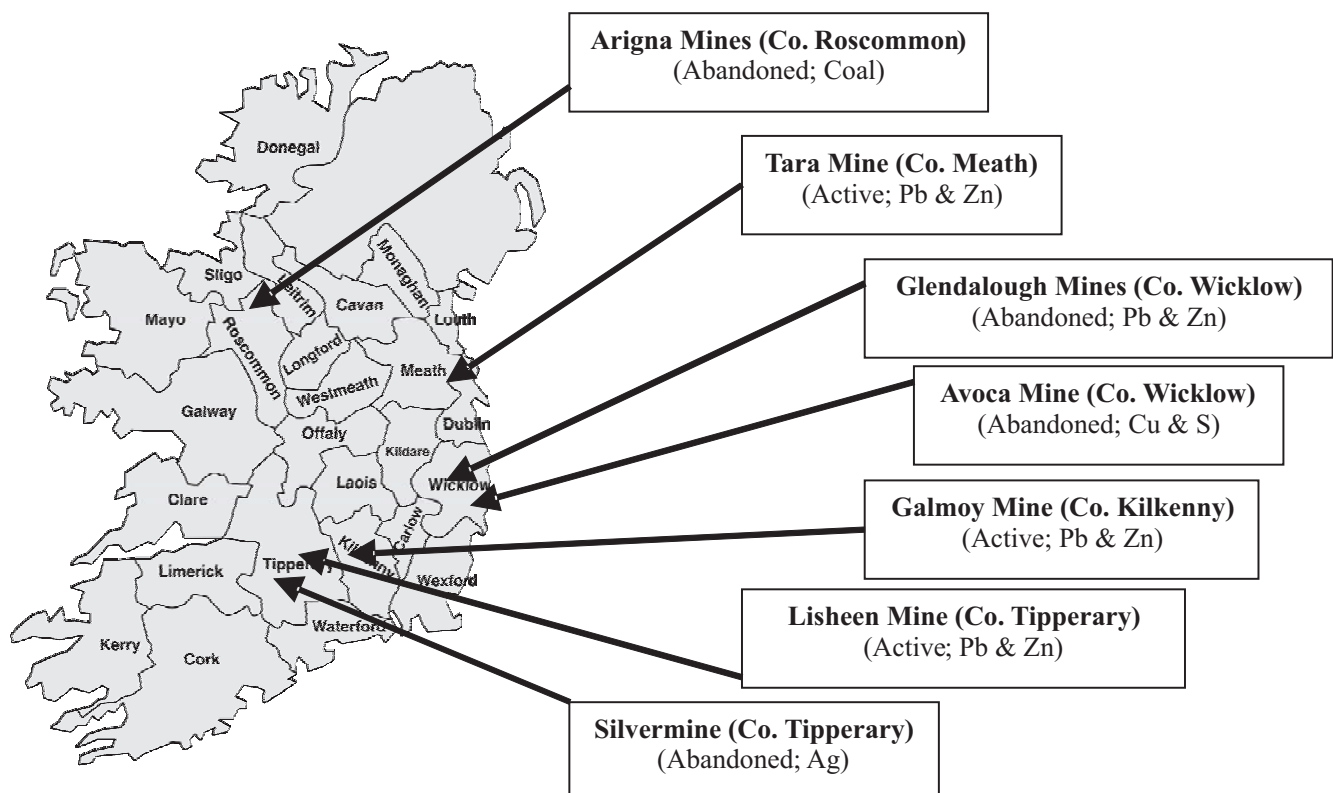
Ireland boasts a wealth of mining activity, possibly dating back to the Bronze Age, when copper was reputedly mined at Mount Gabriel near Schull, County Cork (Gallagher and O'Connor 1999). Over time, natural resources from Ireland such as silver, gold, iron, lead, zinc, barite, coal, and peat have supplied the materials processing and energy industries, both at home and overseas. In recent years, Ireland has played a major role in supplying lead and zinc for world markets. These deposits are primarily derived from mines located (Figure 6) at Tara (County Meath), Lisheen and Mogul (County Tipperary), and Galmoy (County Kilkenny). While resource extraction is an important driver in the national economy, mining industries cause significant environmental disturbance. Before strict legislation regarding resource extraction and associated waste management came into force in Ireland in the 1970s, many mining activities caused ecosystem dev-

astation. Some of these sites continue, unabated, to impact neighbouring watersheds because of acid mine drainage (AMD). One notable case is that of abandoned copper mines at Avoca, County Wicklow, where adits discharge AMD directly into the Avoca River, which is classified as severely biologically impaired (Gray 1998). Historically, silver mining at Silvermines, County Tipperary, has resulted in acid discharges to the Yellow River; approximately 60% of the catchment is considered directly impacted (Aslibekian et al. 1999). Abandoned lead-zinc mine tailings at the monastic site of Glendalough, County Wicklow has resulted in moderately-acidic metal-laden runoff into the neighbouring lake. While some of this mine drainage has inadvertently been "treated" by the strategic position of a natural marsh (Beining and Otte 1996), neighbouring stockpiles of tailings remain unvegetated and thus are potential sources for metal contamination. The purpose of this paper is to present a synopsis of the major metal mining activities in Ireland, both historical and current, that have inherent mine water issues. Most of this information has already been documented in some form but has been previously disparate. It is hoped that this synopsis will aid continued rehabilitation strategies for treating abandoned mine discharges in Ireland.

Mining History of Ireland

Historically, copper mining at sites in Cork and Wicklow formed an important industry (Gallagher and

Figure 6. Map of Éire (Republic of Ireland) with counties; major mining activities are indicated



O'Connor 1999). Coal mining at Arigna, County Roscommon, following initial mining for pyrite, contributed a major energy resource along with peat for Ireland and elsewhere. In later years, off shore gas harvesting on the southwest coast replaced these resources for fuel and energy supplies. More recently, most mining exploration has focussed on the carboniferous lead and zinc deposits in the Midlands region, in addition to the deposits of the Tara Mines. While current lead and zinc mining activities are constrained to operate within strict EU (and national) legislation, responsibility assumed for abandoned mine impacts is unclear. In these instances, mine water issues can be severe and their degree of impact(s) and rehabilitation strategies have only been addressed relatively recently (Aslibekian et al. 1999; Beining and Otte 1996; Gray 1998). Table 6 summarises publications that have addressed mine wastewater in Ireland and provides accompanying citations where they can be examined in more detail.

The local geology of the mines in Ireland influences the associated mine wastewaters. For instance, the calcite and dolomites of the Lower Carboniferous period at Tara Mines buffer the spent water to alkaline levels (O'Leary 1996). By contrast, acidic waters emanating from the abandoned copper mines in Avoca are a consequence of the volcanic (granitic) rock formations (Bowell et al. 1999).

Future Developments

Most of the reported work addressing mine waters in Ireland is from university research (National University of Ireland at Dublin, Trinity College

Dublin, University of Limerick, Sligo Institute of Technology and the University of Newcastle-Upon-Tyne in the UK). Rehabilitation of abandoned mining districts has been initiated with the assessment of Avoca, Gortdrum, and Silvermines. Continued funding is essential if sites such as these are to be adequately rehabilitated.

The successful rehabilitation of ecosystems impacted by abandoned mine drainage will also depend on applying sustainable treatment technologies, which can simulate natural processes to mitigate past activities. Passive treatment options developed and implemented overseas may serve as appropriate technologies in Ireland and are being piloted at Lisheen and in the Silvermines district. Most of these technologies have incorporated a reducing and/or oxidising strategy in their design depending on the waste being treated (Watzlaf et al. 2003). While passive treatment systems can offer many ancillary benefits, such as ecological niches, it is important to recognise that they have limitations; their space requirements may prove inhibitory. Before any passive technology is proposed, the mine waste itself must be reliably characterised and the chemistry (and biology) behind the treatment strategy clearly identified and understood.

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Table 6. Major reported wastewater issues resulting from metal mining activities in Ireland

Mine	County	Resource(s)	Wastewater Issue	Reference(s)
Avoca	Wicklow	Cu, Fe, S	AMD impacts and rehabilitation Abandoned AMD impacts AMD Rehabilitation Metal contaminated soils	Bowell et al. (1999) Gray (1998) Gallagher and O'Connor (1999) Herr and Gray (1997)
Silvermine	Tipperary	Ag	Abandoned AMD sites Groundwater contamination Mine rehabilitation	Aslibekian et al. (1999) Aslibekian and Moles (2001) Rees et al. (2004)
Tara	Meath	Pb, Zn	Recycling of spent water Passive treatment technology	O'Leary (1996) O'Sullivan et al. (1999)
Lisheen	Tipperary	Pb, Zn	Lined tailings facility Passive treatment technology	Dillon et al. (2004) Treacy and Timpson (1999)
Glendalough	Wicklow	Pb, Zn	Abandoned metal runoff	Beining and Otte (1996)
Ballisodare	Sligo	Pb, Zn	Colonisation of estuary after mine tailings discharge	Tierney and Timpson (1999)
Gortdrum	Tipperary	Cu, Hg	Rehabilitation with topsoil	Dallas et al. (1999)

Germany

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Historical Situation

The earliest mines in Germany were for flint, in the Bavarian Mountains, Arnhofen/Kelheim, and in Northern Germany during the Neolithic and Bronze Ages (Binsteiner and Pleyer 1987; Boroffka 2003; Weisgerber 1993). Several of the numerous lead, silver, and copper deposits in the southern part of the Black Forest, lead deposits near Gommern and Mechernich, as well as smithsonite mineralization at the Altenberg near Aachen have been mined since Roman times. Recorded mining in Germany dates back to the 10th century (in the Harz Mountains and in the Ruhr area). At the Goslar Rammelsberg mine in the Harz Mountains, ore was mined since 930 (Clement 1996; Treptow 1900; Goldenberg et al. 2003).

Mining reached its peak in Medieval Times between the 12th and 14th century and during the Renaissance in

the 16th century (Steuer and Zimmermann 1993). Due to the introduction of innovative mining methods and improved smelting techniques, exploration and exploitation work focused on the following raw materials (Figure 7):

- ferrous metals (e.g. pyrite, iron ore)
- non-ferrous metals (e.g. lead, copper, zinc, silver, uranium)
- coal
- industrial minerals (e.g. barite, fluorspar, potash and mineral salt)

Overview of Currently Working Mines

After 1990, in the wake of the German reunification, a large number of mining companies had to shut down as a result of the economic changes in the new federal states. Most of the still active mines in Germany are coal (both hard coal and lignite), potash and mineral salt, industrial minerals, and hydrocarbon mining (BMWI 2003; Table 7). In addition, the Wohlverwahrt-Nammen/North Rhine-Westphalia Mine extracts ferruginous limestone (Table 7), fluorspar is mined as by-product in the Clara mine, and the Kropfmühl/

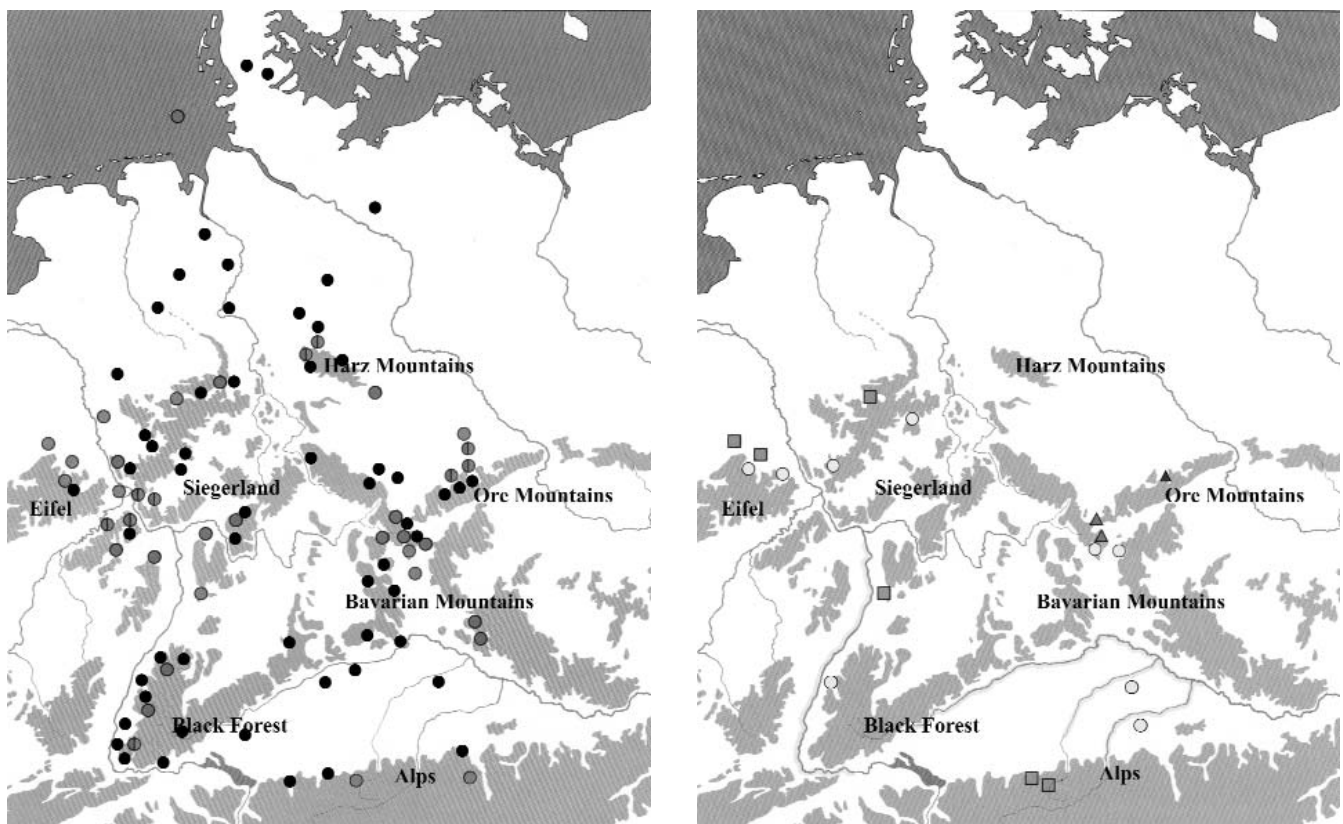


Figure 7. Overview of historical metal ore deposits (modified after Steuer and Zimmermann 1993); ■ Cu; ● Cu/Ag; ● Pb/Ag; ● Fe; ▲ Sn; ■ Zn; ● Au

Bavaria graphite mine is in the process of closing.

Mine Water Management and Problems

Mine water in Germany is legally not treated as waste water. Generally, most of it is summarily referred to as 'mine water'. However, in Germany, depending on the actual kind of use, different terms exist for the water that is generated during raw material production and processing (Wolkersdorfer and Hasche 2001).

The legal basis for all mining activities in Germany is the Federal Mining Law, Bundesberggesetz (BBergG), which came into force on 13 Aug. 1980 and was last amended 21 Aug. 2002. It comprises the acts and directives that apply to the mining industry, taking into account the dynamics and site specific situations of mining. The law covers exploration activities, production and processing of mineral resources as well as the after-care phase and the reutilisation process for areas that are no longer used for mining. Additional legal provisions, related to water and environmental issues, also apply to the mining industry. The most important permits for using water during the different phases of mining are regulated by the German Water Resource Act, Wasserhaushaltsgesetz (WHG) as of 27 July 1957 (last amended 3 May 2005), by the State Water Resources Acts, and by the Waste Water Directive, Abwasserverordnung (AbwV), which went into effect 9 Feb. 1999 (last amended 15 Oct. 2002). All German

water legislation has already been amended or is being amended to include the demands of the Water Framework Directive (Directive 2000/60/EC). Since 13 July 1990, environmental issues in the mining sector are regulated by the Environmental Impact Assessment Act (Umweltverträglichkeitsprüfung bergbaulicher Vorhaben). This environmental impact assessment is carried out as early as possible as part of the general mine management plan procedure (according to BBergG). Regional mining authorities are responsible for all mining issues, though decisions are made in cooperation with the authorities responsible for water and the environment at the local and regional levels. For water that is used during the different mining phases, the authorities make their permitting decisions on the basis of the water law and the relevant regulations (Reinhardt 1999).

Some of the greatest mine water problems occur in the former Eastern German lignite mining areas in the Lausitz and around Leipzig. Due to the high potential for AMD production and a general water deficit in the area, many investigations have been conducted there and innovative remediation techniques are being developed (e.g. Kohfahl and Pekdeger 2004).

Notwithstanding comprehensive legislation referring to mining and mine water management, there are large untreated mine water discharges and drainage water effluents at German mine sites that in some cases pollute surface water on a local or regional scale. Examp-

Table 7. Selected German mine working sites in 2003 (BMW 2003, Wirtschaftsvereinigung Bergbau 2004)

Mining sector	Mining site/area	Mining company	Production rate (tons/year)
Hard coal	Ruhr Area	Deutsche Steinkohle AG	18,900,000
	Saar Area	Deutsche Steinkohle AG, Merchweiler Mining Company	5,400,000
Lignite	Ibbenbüren	Deutsche Steinkohle AG	1,800,000
	Rhine Area	RWE-Rheinbraun	97,500,000
	Helmstedt	BKB	2,100,000
	Vereinigtes Schleenhain	MIBRAG	22,000,000
	Lausitz Area	LAUBAG	57,400,000
Iron ore	Mine Wohlverwahrt-Nammen	Barbara Rohstoffbetriebe GmbH	419,000
Fluorspar	Mine Clara	Sachtleben Bergbauservice GmbH	70,400
Barite	Mine Clara	Sachtleben Bergbauservice GmbH	64,500
	Mine Wolkenhügel	Deutsche Barytindustrie Alberti	40,000
	Mine Dreislar	Sachtleben Bergbauservice GmbH	45,300
Potash salt	Kaliwerk Werra	Kali + Salz GmbH	24,100,000
	Kaliwerk Neuhoof-Ellers		
	Kaliwerk Sigmundshall	Kali + Salz GmbH	2,600,000
	Bergmannsseggen-Hugo		
	Kaliwerk Zielitz	Kali + Salz GmbH	11,100,000
Mineral salt	Mines in Northern Germany	ESCO	12,100,000
	Mines in Southern Germany	Südwestdeutsche Salzwerke	6,400,000

les include polysulphide underground mines in the Freiberg/Saxony area, mercury mines in the Saar-Nahe-Basin, the Silberberg mine/Bavaria area, and the Mechernich/Eifel lead-zinc area; these were abandoned before the German Federal Mining Law came into force (Bundesverwaltungsgericht 1995; Schalich et al. 1986; Vornehm et al. 2002; Wieber 2001, 2002).

In the Mechernich/Eifel mining area, slightly mineralized sandstone was mined (Schalich et al. 1986). The Burgfey Adit dewateres the mine area and discharges high amounts of zinc, cadmium, and nickel into the Veybach and Erft Rivers, with large total annual loads of zinc (25 t) and nickel (12 t; Christoffels and Schindler 1996; Landesumweltamt Nordrhein-Westfalen 2002).

The environmental impact in the Freiberg region is caused by heavy metal leaching of heaps and tailings and acid mine drainage (Baacke 2000). The mine water is characterised by low pH and elevated concentrations of heavy metals, arsenic, and aluminium. In recent years, several research projects have investigated geochemical processes and heavy metal transport within the underground mine.

Another example for a mine not governed by the mining control authority is the Hohenwarte fluorspar mine near Gernrode in the Harz Mountains, which was closed in 1985 due to economic reasons. Despite poor water quality and the existence of the BBergG, the mine was closed with the requirement that the mine's drainage adit be used as a water reservoir. As a result of the German reunification, the former mine owner has disappeared; the mine now belongs to the Federal State Saxony-Anhalt. The discharged mine water is characterised by high conductivity and high sulphur, iron, manganese, and zinc concentrations, and degrades the water quality of the neighbouring river

(Tamme 2001). Despite several years of discussions among the authorities and the current owners, no solution has emerged.

To date, no mine water inventory has been conducted in Germany and therefore no data is available on total quantities of mine water or water quality for abandoned or closed mines. However, at active mines, under the mining law, mine water quantities and qualities are generally well recorded and, if the quality does not meet the guidelines, then the water has to be treated.

Future Mining, Treatment, and Remediation

Mine water in Germany is usually treated by conventional methods involving LDS and HDS techniques. In view of the potentially long timescales over which water pollution from abandoned mines can persist (Younger 1997; Wood et al. 1999; ERMITE Consortium 2004), there has been increased interest in Germany to develop cost-effective water treatment strategies that can operate over decades or even centuries with little maintenance, and low investment and operational costs (e.g. Eger and Wagner 2002; Wolkersdorfer and Younger 2002; Younger 2002). Therefore, in recent years, passive mine water treatment systems have been constructed in Germany. These include small test systems, such as the RAPS-System at the Hohe Warte fluorspar mine near Gernrode/Harz Mountains (Hasche and Wolkersdorfer 2004) and large pilot plants, such as the Pöhla/Saxony uranium mine (Kießig et al. 2004). Furthermore, extraction methods for the separation of particular contaminants are currently being researched (e.g. Schmeide et al. 2004).

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Poland

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Introduction

Poland's mining tradition goes back to prehistoric times. The first evidence of mining are traces of flint mining from 3000 BC (the flint pits of Opatów) and the bog iron ore mines from the 4th century BC. Salt was extracted in Poland as early as the 9th century. Zinc,

lead, silver, iron, and copper ores have been extracted since the 12th century. The most intensive development of mining in Poland took place during the first half of the 20th century. Copper, iron, zinc and lead, nickel, hard coal, lignite, salt, sulphur, gypsum and anhydrite, barite, phosphorite, rock materials, crude oil, and earth gas were extracted during that time. Since the 1970s, mining activity in Poland has declined. After the great political changes in Poland in 1989, there was intensive abandonment of mines (Table 8). All iron ore mines, open-cast sulphur extraction, all collieries of the Lower Silesian basin, many collieries in Upper Silesia, and all zinc-lead ore mines were abandoned. Currently, there are over 2400 active open-cast mines (Szczepiński 2003) and 48 active underground mines.

Table 8. Changes in mining activity in Poland from 1989 to 2002

Mineral	Exploitation of (minerals in 10 ⁶ tons)			
	1989	1994	1999	2002
<i>Minerals for energy production</i>				
Oil	0.16	0.17	0.19	0.73
Coal	176.00	134.10	109.99	103.7
Lignite	72.00	66.80	60.86	58.24
<i>Metal minerals</i>				
Zn-Pb ores	4.20	4.82	4.92	4.73
Cu ores	26.50	23.74	26.93	29.00
<i>Non metal minerals</i>				
Sulphur	5.00	2.13	1.18	0.76
Salt	4.00	3.18	3.16	3.56

About $140 \cdot 10^6$ t of various useful minerals and circa $60 \cdot 10^6$ t of lignite are extracted in open-cast mines (Grzesiak and Sieradzki 2002). The Polish underground mines extract coal, copper ore, zinc-and-lead ore, and salt. Sulphur is extracted by the Frasch method.

Coal

Poland is one of the best-known coal producers in the world. Coal production in Poland in 2002 totalled 103.7 million t, which is about 48% less than the maximum output of 201 million t that was reached in 1979. In 1996, mines pumped out circa 355 million m³ of mine water. An average water inflow to a single mine ranged from 200 m³/d (in the Barbara Experimental Mine) to 70,000 m³/d (in the Jaworzno Coal Mine). In 2002, the total volume of drainage water decreased to 255.67 million m³, i.e. circa 700,500 m³/d (Table 9). Almost 60% of the mine water is discharged to the rivers, which constitutes a fundamental hazard for the aquatic environment because much of the water is saline, with total dissolved solids (TDS) ranging from several to 110 g/L (Rożkowski et al. 2004).

The hard coal deposits in Poland occur in the three upper Carboniferous coal basins: the Lower Silesian, the Upper Silesian, and the Lublin. The origins of mining in Lower Silesia (the Wałbrzych and the Nowa Ruda area) date back to the 15th century. In the 1970s, four collieries extracted about 2.5 million t of coal. The process of abandoning the collieries was started in 1972. All mines in the town-of-Wałbrzych area were closed from 1991 to 1998 and have been flooded since 1994. In the Nowa Ruda area, gradual closing of collieries started in 1972 and the last one was flooded in 2000.

The first coal extraction in the Lublin coal basin began in 1982 and is still active. There is one colliery there with an output of 3.8 million t of raw coal. The water inflow to this mine ranges from 8 to 10 m³/minute.

Table 9. Mine water pumped from Poland's mines, 1966–2002

Mineral	Mine water pumped out in 10 ³ m ³ /d			
	1966	1988	1996	2002
Coal	1,028	1,033	973	700
Lignite	317	1,044	561	989
Zn-Pb ores	220	444	412	551
Cu ores	121	81	85	71
Fe ores	255	—	—	—
Sulphur	43	51	50	41

The Upper Silesian coal basin holds about 85% of Poland's coal reserves. The coal mining industry in the region was developed in the 17th century and reached its height in the 1970s. In the 1980s, 75 collieries were operating there. After the political changes caused by the Solidarity movement, i.e. since 1989, unprofitable collieries have systematically been abandoned. At the end of the 2002, there were only 40 coal mines in operation. Many years of intensive mining drainage have produced widespread cones of depression, with a surface area of circa 1750 km² and depth ranging from 300 to 700 m, locally up to 1160 m below the ground surface (Wilk 2003). It is estimated that the volume of the drained Carboniferous ground amounts to circa 100 km³ (Wilk et al. 1990).

Currently Tertiary lignite deposits are exploited in four large mines: Adamów (three open pits), Bełchatów (two open pits), Konin (2 open pits) and Turów (one open pit) and one small Sieniawa (mixed underground and stripping operations). The total lignite production in 2002 amounted to 58.2 million t (Ney and Smakowski 2003). More than 58% of that amount is produced by the largest mine, Bełchatów. In 2002, about 361 million m³ of water was pumped from the lignite open pits, producing cones of depression with a reach estimated at over 1200 km² (Wilk 2003). The well drainage (well barrier) method is prevalent. In general, the drainage water is of good quality and is not environmentally hazardous. An exception is the water from the Tertiary aquifer in the Turów mine and the water draining the salt diapir in the Bełchatów mine. Locally, these waters have a salinity of over 5 g/L. From 1991 to 2002, four open pits were abandoned and filled with overburden or water.

Metals

Poland has one of the largest copper deposits in the world (Permian). They are located in the Lower Silesia area. The copper ore is extracted in the Lublin-Głogów copper district in three mines (Lubin, Polkowice-Sieroszowice and Rudna). The Konrad Mine is the oldest; it was closed in 1989 and has been flooded since 2001. Currently, 27–29 million t of ore (almost

570,000 t of copper) are extracted. 26.5 million m³ of water were pumped out of the mines in 2002, which produced a cone of depression in the Permian aquifer with a reach of around 850 km². The mine water's salinity varies between 2 and 112 g/L, but water with total dissolved solids less than 10 g/L constituted over 96% of the total volume. All of the mine water is used for various technological purposes. The processes of ore preparation produce a vast volume of tailings (26–27 million t per year). The tailings are discharged to two huge tailing ponds: Gilów (6 km², 64 million m³) and Żelazny Most (12.6 km², 350 million m³; Wilk and Bocheńska 2003).

Poland has also one of the largest European reserves of zinc-lead ores, located in the Silesian-Cracow region (Triassic deposits). Over 800 years of mining activity in four major areas (Tarnowskie Góry, Bytom, Olkusz, and Chrzanów) have caused huge environmental damages. There are still three active mines in the Olkusz and Chrzanów areas. The water inflows to these mines range from circa 30 m³/minute in the Trzebionka mine to over 300 m³/minute in the Pomorzany mine. Intensive drainage by these mines has caused a significant lowering of the groundwater table (from 100 m to 260 m), and created extensive regional cones of depression covering about 500 km² in the Olkusz area and over 170 km² in the Chrzanów area. In 2002, over 201 million m³ of water of various quality were pumped out of the mines. There is an observable decrease in water quality; very high concentrations of sulphate (over 5 g/L) are locally affirmed (Motyka and Witkowski 1999).

Tailings from these mines are dumped into a few tailing ponds that cover an area of about 1.1 km² near Olkusz and hold about 40 million t of tailings.

Other Minerals

Deposits of native sulphur (in Miocene limestones) occur in Poland in the Tarnobrzeg area. Sulphur exploitation started in this region in 1960. Two sulphur open pits and three mines exploiting sulphur with the Frash method (underground smelting by hot water) were active there in the past. Both of the open pits are currently closed. The Piaseczno mine has been partly flooded. Flooding of the Machów mine will start in 2005. Currently, native sulphur is extracted only at the Osiek mine (Frash method).

Poland also has large deposits of rock salt. The salt deposits are exploited in the Subcarpathian (Miocene) and the Kujawy (Permian) regions. There were 10 mines: 6 underground and 4 borehole mines. The Wieliczka salt mine is the oldest; it is located near Cracow and is of great historical value. It is 700 years old and visually very attractive; it is one of the most

popular Polish monuments and was registered in UNESCO's 1st World List of Cultural and Natural Heritage. Now there is only one active underground salt mine, Kłodawa, and two bore-hole mines. Since 1993, salt has been extracted in the Sieroszowice-Polkowice copper mine as a by-product.

After World War II, 21 deep mines were built in the Częstochowa region that extracted about 2 million t of iron ore per year till the end of the 1960s. About 255,000 m³ per day of mine water were pumped out of the mines in that time. From 1970 to 1984, all the iron ore mines were abandoned. Since then, all the mines have been flooded. The negative impact of that mining activity on groundwater quality, mainly within the middle Jurassic aquifer, is still observable (SO₄²⁻ concentrations up to 1100 mg/L, Mn up to 6.6 mg/L, and Fe up to 271 mg/L; Razowska 2000).

Many rock minerals are also extracted in Poland, among them building stone, limestone, dolomites, gypsum and anhydrite, clay materials, and sand for back-filling. The total quantity of water pumped out of these open-pit mines is estimated at 700,000 m³/day. These are mainly waters of good quality.

Such intensive, often strongly concentrated, multi-year mining activities has to have an influence on the natural environment. The negative influence of these activities mainly results from the intensive dewatering, mine water discharges into surface water, waste storage, and surface subsidence (Kowalski 2000; Rogoż and Posyłek 2000; Wilk 2003; Wilk and Bocheńska 2003; Wilk et al. 1990).

Mining and Mine Dewatering

Intensive drainage lasting for many years has produced huge cones of depression with a total area of about 4600 km² and a maximum depth exceeding 1100 m beneath the ground. Mining has profoundly transformed the hydrodynamic conditions of groundwater, resulting in changes in flow direction, increased hydraulic gradients, changes in hydrogeochemical conditions, and groundwater contamination on a regional scale.

In 2001, over 1039 million m³ of mine water were pumped out of Polish mines. Over 162 million m³ of this mine water was saline, and of that, about 153 million m³ (94%) were discharged to surface waters. Thus, a total of about 2.5 million t of Cl and SO₄²⁻-ions were discharged to surface waters.

In 2001, Polish mining and preparation plants together produced about 70 million t of wastes, of which about 10 million t were placed on mine waste dumps. At the end of 2001, the total quantity of wastes in these

dumps amounted to over 1280 million t, containing about 558 million m³ of tailings (Bolewski et al. 2002). The area of the tailings ponds totalled 20 km².

Intensive mining generated widespread subsidence troughs with depressions that locally exceed 30 m. The ground surface depressions caused changes in hydrographic conditions; it is estimated that at the end of the 20th century, the area of land subject to flooding exceeded 8 km² in Upper Silesia alone (Wilk 2003). In some areas, the water table was situated up to 7 m (Bierawka) above the surrounding terrain, causing rivers to flow through waste rock heaps.

Outlook

Currently, the fundamental challenges for Polish mining, relative to mine water, are associated with abandoned mines, their influence on the natural environment, and the need to utilise mine waters (Kulma et al. 1998; Kulma and Haładus 2001; Rogoż and Posyłek 2000; Szczepański 2003; Szczepiński 2003; Wilk 2003; Wilk and Bocheńska 2003). The abandoned Upper Silesian collieries are especially problematic because they are largely hydraulically connected to each another. In this situation, closed collieries still

have to be partially drained. Currently, closed collieries are drained by underground stationary pumping systems and by submersible pump systems. In the future, the stationary pumping systems are to be replaced by submersible pumping systems, which are more cost-effective. Moreover, a need for transferring water to active mines and application of high-pressure dams are anticipated. Re-evaluation of hydraulic connections will also be necessary. The next problem to be dealt with may be acid mine processes, which are currently under investigations (Pluta and Jackowicz-Korczyński 2003).

An equally important problem is utilisation of saline mine waters. At present, efforts are being made to decrease the quantity of pumped-out water, use hydrotechnical methods (retention reservoirs with controlled discharges), and, on a limited scale, desalination. The pilot Dębieńsko desalination plant treats over 14,000 m³/day of mine water with a mineralisation ranging from 8 to 115 g/L, producing 300 t of salt (>99.8% NaCl) per day. Experimental work is also underway on injecting saline water into the ground.

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The Netherlands – Rising Mine Water in the Former Dutch Coal Mines

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Introduction

Large-scale coal mining took place in the Netherlands between 1900 and 1975, when 600 million tons of coal was extracted from 12 mines. Coal extraction was primarily performed using longwall mining systems, with an average face length of 200 m and an average opening of 1 m. The extraction depth varied in the east from 0 to 500 m and in the west from 400 to almost 1000 m. Per ton of coal mined, 2 m³ of water needed to be pumped out.

The coal mines in South Limburg were closed during the period of 1967 to 1974, and the pumping of mine water ceased, except at the Beerenbosch pump shaft, where it was maintained to protect the neighbouring operating German mines. The rising of these mine waters has gradually increased since then. There is concern that the rising mine water could reach into the overburden sediments. This could be a threat for existing drinking water wells in the Cretaceous limestone

because of the high concentrations of salts and heavy metals in the mine water.

Possible surface damage is also to be expected, on the one hand by caving as an effect of erosion of historical mineshaft fillings, and on the other hand by ground level uplift. For an overview of the Limburg Coal District, see Figure 8. The Dutch coalfield has an elongated shape and measures roughly 7 by 30 km; it is situated on the north-eastern edge of a structural high and separated from the Central Graben by a system of faults along which the Carboniferous surface sank to considerable depths. The Upper Carboniferous strata yielding the coal belong to the Westphalian A, B and the lower part of C. Important for the occurrence of mine water is the lithological alternation of shale and sandstone.

Measures Taken

During the period 1967–1974 dams have been built into the galleries to protect the neighbouring, not yet closed mines in Limburg from water influx. In this way a few underground compartments (basins) were created with water overflows on different depths (Figure 9).

To protect the then still-active mines in Germany, the Beerenbosch pump shaft near Kerkrade was installed, which kept the mine water rising level at –220 m NAP

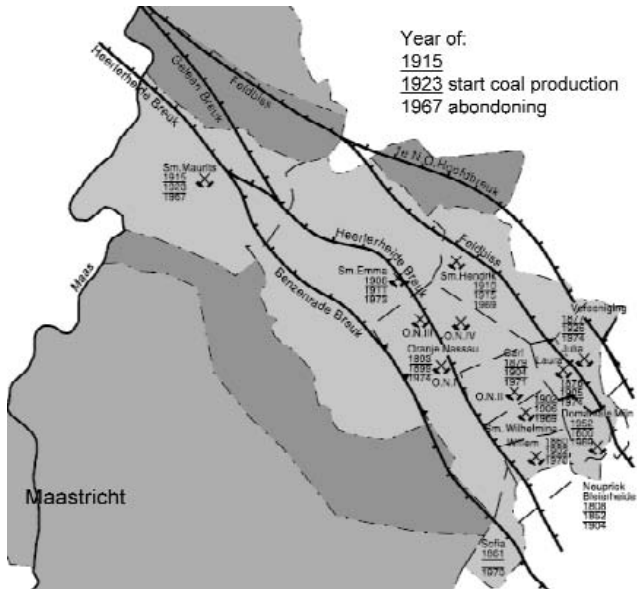


Figure 8. Limburg Coal District, tectonic faults and mining licenses

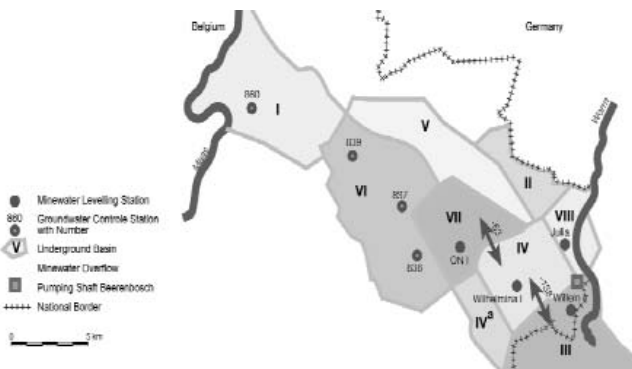


Figure 9. Mine water basins – levelling stations and groundwater

(National Ordnance Datum). This pump shaft was dismantled early in 1994 when the German mines in the Aachen mine district closed. Since then, the mine water has also risen in that part of the coalfield.

To monitor the mine water rise, water level monitoring stations were installed in four former mineshafts and, in addition, four observation wells were constructed over the mine area to monitor the water quality in the overlying limestone aquifer.

Effects of the Mine Water Rise

Water Quality

The level of the mine water is at present above the Carboniferous sediments and will continue to rise in the future. However, infiltration of rainwater maintains a groundwater level in the overburden that is above the NAP. It is anticipated that this potential difference will

cause a permanent downward groundwater flow from the overburden into the Carboniferous sediments. This is being confirmed through quality measurements of the groundwater, which shows no worsening in quality. The water level in the limestone did not drop noticeably during coal extraction, which indicates that the Cretaceous limestone and the Carboniferous sediments are still poorly connected hydrologically. The implication is that the rising mine water will be stopped by the relative isolation of the strata and the downward flow of the groundwater in the overburden.

It is to be expected that eventually the original density gradient will establish itself again and that stratification of the groundwater will ultimately occur. This will not differ significantly from the original situation, where the fresher groundwater floated atop the deeper and saltier mine water. The permeability of the lower strata of the overburden is limited, and vertical permeability relative to horizontal permeability is minimal. This contributes to the effective protection of drinking water reservoirs in the overburden from the influence of rising mine water.

Ground Movements

Ground movements as a result of mine water rise can only cause damage in places where geological faults are manifested on surface and only then when uneven ground level uplift takes place on both sides of these faults. This means that on both sides the mining method must have been different. Possible caving of shallow mineshafts of before the 19th century, by erosion of the shaft fillings, can only occur on a narrow strip along the German border.

Measured Rise of the Mine Water

Currently, the water level in the eastern mining district is at about 10 m NAP, about 140 m below ground level. It continues to rise about 6 m per year. It is expected that the water table in the southeast will, at the most, increase up to the level of the river Worm. From subsidence research, it can be calculated that the level in the western mining district has increased up to -6 m NAP. The average rate of mine water rise in recent years is currently 2 to 3 cm per day in the Limburg mining district.

Measured Ground Level Uplift

Detailed levelling measurements have been carried out in the Aalbeek-Schinnen area. It turns out that the ground level uplift that occurred as a result of the rising mine water is proportional to the land subsidence that occurred as a result of the extraction of coal. The measured ground level uplift since 1974 is 23 cm. The maximum ground level uplift expected will be between

2 and 5% of the subsidence that occurred. The amount of the surface subsidence in the coal mining area of South Limburg varies between 2.5 and 10 m.

Conclusions

In the Limburg mining area, ground level uplift due to rising mine waters has, up till now, not led to new dam-

age. It is anticipated that the rising mine water will not mix with the fresher groundwater, and does not pose a risk to the drinking water supply. Finally, there are currently no indications that mine water is being transported to the overburden.

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Belgium – Mining and Non-Ferrous Metal Processing Activities: Environmental Impact and Remediation Measures

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Introduction

Belgium has no operating metal or coal mines anymore. The last coal mine was closed in 1992 and the only mining operations left in Belgium are extracting clay, sand, and gravel in Flanders, and quarrying limestone, dolomite, slates, porphyries, and marble in the Walloon area. Belgium is a world leader in silica sand and flours, and Belgian marble quarrying has a tradition of more than 2000 years.

The Walloon steel industry was already famous in the 16th century. Mining of calamine, a zinc silicate, started in 1805, and formed the basis of new developments in mineral processing (patents on calamine extraction and zinc refining date back to 1809). Coal mining, which started later, delivered the energy for steel production. In addition, due to the rich Cu-Co deposits in its former colony Congo, the country continued to be an important producer of non-ferrous metals and materials. The refining and processing of non-ferrous metals currently employs some 8,900 people; in addition, the production of steel (11.1 million t in 2003) and stainless steel (900,000 t in 2003) also continues to be an important component of Belgian industry (Franceschi 2004; Newman 2000; Rombouts 1999).

Former metal and coal mines still cause environmental problems and the environmental impact of the old steel and non-ferrous industry is also significant. Impacts are both source linked (e.g. waste heaps, landfills), or non-point and diffuse (including emissions to air and water, fugitive dust, and dissolved metals from the use of slags to improve road quality). The distribution of the metal mines and the diffuse pollution by the old

steel and non-ferrous metals industry led to very large areas of impact. Cleaning of these sites is technically and economically unlikely so rehabilitative measures are based on risk management, i.e., the source – path – receptor approach.

Legislation

Environmental legislation depends on the three regional environmental authorities of Wallonia, Flanders, and Brussels. Generally, environmental policy is the responsibility of the Federal Ministry of the Environment of Brussels and its comparable Flemish and Walloon ministries. Besides this, the industrial companies are responsible for environmental protection of the area influenced by their activities. Consequently, these companies are also involved in environmental investment programmes (Newman 2000).

Besides the conventional legislation related to air and water pollution, the cleaning up of contaminated sites (including the soil and groundwater) in Flanders is regulated by the Soil Remediation Decree (1995). The operator or owner of the land is responsible for the remediation of contaminated sites where the pollution entered the soil, but only if the pollution was caused by carelessness and negligence or was committed knowingly and intentionally. In the case of pollution that occurred before the Soil Remediation Decree became operational, the operator or owner is only responsible for the remediation if the contamination represents a risk to the health of people or the ecology, or has the potential to affect larger areas. In Brussels, the recently enacted Ordonnance sol à Bruxelles (RBC/05/2005) is based on risk assessment. This risk assessment can lead to a remediation plan based on control and risk management of the pollution and in some cases remediation measures. In Wallonia, the soil decree, '*Décret relatif à l'assainissement des sols pollués et aux sites d'activités économiques à réhabiliter* (MB 07/06/2004),' will soon apply. It requires a preliminary site evaluation, followed by a more detailed assessment, if justified. This can finally lead to a full remediation plan, if the source represents a risk to the health of people or the ecology, or has the potential to affect larger areas.

Metal and Coal Mining

Lead and zinc mining took place in south-eastern Belgium. At the moment, we only know the location of about 3000 of the estimated 8500 mine shafts. Most of these mines were rather small, no large problems have been identified, and risks are thought to be generally limited. In most cases, the groundwater flows through limestone aquifers, which leads to immediate neutralisation and precipitation of dissolved metals. However, waste rock has been used to make road foundations and other applications, leading, in many cases, to an uncontrolled dispersion of metal contaminants. At some sites, these problems have been recognized, and waste heaps have been removed in a controlled way (e.g. some wastes of the La Plombière site were removed and landfilled).

However, coal mining was performed at rather large scales. The risks related to these mines (e.g. subsidence, movement of waste heaps, burning waste heaps, contaminated drinking water) are significant. Also, randomly located waste and large waste heaps cause environmental problems due to the leaching of metal chloride and sulphate salts and organic contaminants. Risk management measures are based on removal of contaminated soil and abatement of further leaching and erosion by isolating the waste from contact with rainwater and by revegetation of the heaps (pers. comm. B. Hendrickx). In the Kempen area, the heaps have been remodelled (shallowing of the steeper slopes) and superficially compacted by heavy trucks, decreasing both the amount of rainwater infiltration and runoff.

Non-ferrous Metals Industry

Two large regions are affected by the metal processing industry, the Kempen area in Flanders and the Liège - Charleroi area in Wallonia. The dispersion of pollutants has been caused by:

- emissions by the old thermal processes (diffuse soil contamination);
- landfill (e.g. containing goethite, slags, sludge) leaching;
- use of lead and zinc slags to stabilise roads (even to fill bomb craters);
- use of contaminated compost to improve the organic content of garden and agricultural soil;
- irrigation with contaminated surface water; and
- contaminated sediment deposited on river banks

There has been widespread pollution of soil and some infiltration of metals (especially Cd and Zn) into groundwater. Landfill leakage has also led to significant contamination of certain groundwater aquifers. In Flanders, the aquifer is a very deep sandy aquifer. Two famous epidemiological studies were performed in Flanders for 1980–1990, on the impact of cadmium on human health (Cadmiel and Phecard). Current efforts are focusing on exposure routes and their relative contributions of Cd and Zn to humans. It is anticipated that:

- Cultivation of food crops will be forbidden, or limited to certain crops (with restricted metal uptake characteristics). Soils can also be treated by mixing with certain additives (e.g. based on silicates, organics, zeolites, iron shots, hydroxyapatite, lime) in order to immobilize the toxic metals and to reduce plant uptake and infiltration to the groundwater by rainwater. Large agricultural areas will have to change from food crops to industrial use.
- Use of contaminated groundwater will be forbidden.
- Large, highly contaminated areas (so called zinc deserts) are being cultivated with special Zn-resistant grasses and soil additives; the plant cover should prevent the dispersion of metal-laden dust.
- Groundwater contamination will be reduced by removal of slags from roads. The inventory of the roads contaminated with metal slags has started.
- Contaminated sediments will need to be dredged and removed to a controlled landfill.

As the problem affects both sides of the Belgium-Holland border, the Flemish and the Dutch Ministries of Environment are cooperating, within the framework of an INTERREG III project, to make a complete inventory of the problems, to assess the related risks, and to come up with an integrated management system to manage this large risk management zone (a megasite of > 300 km²).

In Wallonia, the aquifer is sandy clay, limestone, and gravel and most of the industries are located very near to the Meuse River. The areas affected by diffuse spreading of metals (Zn and Cd) in Wallonia (originating from the non-ferrous industry) are thought to be more restricted in area. Groundwater is known to be affected near some non-ferrous industries in the alluvial zones of the Meuse and Vesdre Rivers due to the historical regular dumping of Zn- and Cd-rich slags. Other areas may also be contaminated, but the precise extent of the affected zones has not been investigated yet; preliminary studies are going on to identify the

intensity of the problem and related risks. In a general way, the higher pH and loamy textures (compared to the affected zones in the Kempen region in Flanders) could prevent the contamination from infiltrating deep into the soil (Scokart and Meeus-Verdinne 1985), but the possible effects of diffuse Cd and Zn sources on groundwater quality is not known yet.

Groundwater contamination at the industrial sites will be controlled by pump and treat technology; in some cases, the effluent from this treatment will be used as industrial process water. In addition, several tests are underway on a new process of in situ immobilization of the metals in the groundwater aquifer, based on injecting electron donors (e.g. molasses, lactate, HRC[®], MRC[®]) via wells to induce sulphate reducing bacteria (SRBs). Under appropriate conditions (pH between 4 and 8 and Eh lower than -200 mV), the SRBs will use the electron donors to oxidize the organics, reduce the sulphates to form sulphide, which will precipitate the metals as insoluble metal sulphides. This method has been shown to work in sand and clay aquifers and is now being tested on the clay-sand-gravel aquifers of Wallonia. Special focus is now on optimization of this method by selecting the right electron donor, injection conditions, and controlling the stability of the precipitates (Diels et al. 2002; Geets et al. 2003; Van Roy et al. 2004).

In addition, special attention is being paid to the influence of contaminated soil on surface water contamination (transport via groundwater), within the framework of the European Project AQUATERRA (505428). In this project, several groups are examining the fate of

heavy metal soil contamination on surface water quality. Special attention is paid to the natural processes of transport and immobilization (retardation) of metals on their way to the surface water by looking at the influence of plants, groundwater, the interface between groundwater and surface water, and sediments.

Conclusions

Belgium has no real metal or coal mining activities anymore and the impact on the environment is rather restricted. However, the steel and non-ferrous metals industrial activities from the past still heavily affect the environment. Several risk-based measures are under study and implementation. It is anticipated that natural processes, combined with some of the remediation measures being undertaken, will lead to a reduced future exposure of the receptor (e.g. humans or surface water). All these measures are being taken within the framework of the European Integrated Management System in the framework of the Welcome project (EVK1-CT-2001-00103).

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France

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Introduction

France has a long history of mineral resource exploitation. However, at the end of the 20th century, the collapse in metal and mineral prices, coupled with a depletion of the principal mineral deposits caused a decrease in French mining activity. Most of the ore exploited in the country occurs in old "crystalline basement rocks" located in Brittany, massif-Central, Alpes, Pyrénées and Ardennes. In addition, there has been exploitation of coal (Carboniferous and Tertiary Age), iron, salts (potassium hydroxide, sodium), and aluminium (bauxite) from sedimentary formations

History

In France, mining started during the Neolithic Period with flint being mined by means of shafts and galleries, sometimes requiring roof support (pillars). During the Gallo-Roman age, gold, silver, lead, copper, tin, and iron were mined. In massif-Central and Brittany, gold and tin exploitation has been dated back to the first millennium A.D. The most intense period of mining in France was from the 17th until the 19th century, initiating the industrial revolution. Mining decreased at the beginning of the 20th century (Anonymous 1996b), during the colonial period, and then resumed after the Second World War as the nation rebuilt. At the end of the 20th century, a collapse in metal and mineral prices coupled with the depletion of the principal mineral deposits caused a decrease in French mining activity.

Recent mining activities have focused on:

- coal (last mine closed in 2004)

- iron (exploitation ended in 1997)
- potassium hydroxide (exploitation ended in 2003)
- uranium (exploitation ended in 2003)
- fluorite
- gold and silver

Mine Water Characteristics and Issues

Coal

Coal mines are localised in the North Field, the Lorraine Field, and the Centre-Midi Field (in southern France). These three fields are very different in terms of their hydrogeology and associated water quality.

In the North Field, mining took place at great depth, which has always limited infiltration. The end of exploitation has resulted in a slow recovery of groundwater, which will go on for 100 to 150 years. Initially, water quality is rich in iron and sulphates but with high pH, but it is expected to improve with time. Coupled to the slow recovery is the problem of mining subsidence, which has taken place on a land surface that is already very flat, creating low zones in which there are no outlets. These zones are located in urban districts and thus require constant dewatering. In total, 6,000 ha are susceptible to flooding and are drained by 133 pumping stations (Tubiana 2002).

In the Lorraine Field, the end of the last underground exploitation was in May 2004. Coal mines are localised under Triassic sandstone overburden. The sandstone formation is an important regional aquifer, exploited for industrial and drinking water. Mining exploitation under this aquifer has created fractures, which drain the Triassic aquifer into the mine. The piezometric swallow-holes in the aquifer produce high inflows. In addition, there are several regional problems that are a legacy of mining:

- Dewatered (pumped) mine waters were used as a regional resource for drinking and industrial applications. This resource disappeared when mining ceased, at least while the mines refill, and strong mineralization will exclude most uses for a long time. Figure 10 shows the observed diminution of iron content in the Messeix Mine and gives an idea of the possible evolution of water quality in coal mines after flooding.
- The mineralised mine water will likely contaminate Triassic sandstone aquifer.
- Low zones above the piezometric swallow-holes in

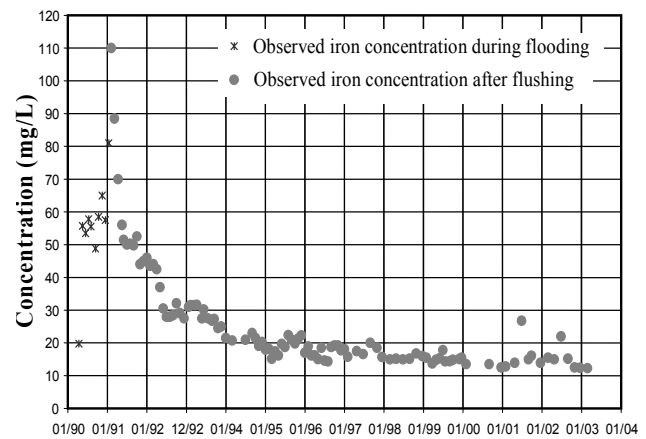


Figure 10. Decline in iron concentration in the Messeix Mine (central France)

the sandstone risk being flooded when the whole system finds its piezometrical balance.

A number of studies are in progress to resolve these problems. In the Centre-Midi Coal Field, there were 100 mining concessions distributed among 10 different mining areas, and so mine water problems vary. Mining ceased in 2004; the last ones were the Gardanne underground mine and the Gard, Aveyron (Decazeville), Tarn (Carmaux) and Saône et Loire (Blanzay) open cast mines.

The Gardanne Mine discharges water through a 14 km long gallery into the Mediterranean Sea. The mine water flow is topographically low so there is minimal risk of flooding; there are, however, water quality problems: the water channels through a karst aquifer that provides drinking water for the city of Marseille and for the cooling circuits of a sugar refinery; the outlet of the gallery flows into the commercial harbour of Marseille. Iron is a significant issue and has discoloured water in Marseille's harbour. The problem is believed to be short term (10 to 20 years) as the residual iron content decreases, corresponding to the peak of mineralization after mining ceased.

The other underground mines are closing or are already flooded. As a general rule, mine water in French coal mines is not acidic due to the presence of alkalinity in the surrounding country rock. However, it is common to observe high iron and manganese content in initial mine water discharges, which justifies the use of passive treatment. Two passive systems were constructed about 10 years ago (Messeix and Alès), and two others are being constructed (Carmaux and Buxières-les-mines).

At open cast mine sites, it is common to flood the excavations; this ensures that iron will decrease and limits

the impact of the mine water on the natural environment.

Iron

The last iron mine exploited was located in Lorraine. The end of the dewatering in the 1990's caused a problem for the nearby towns and villages, since the pumped mine waters were used to supply drinking water. The flooding of the mine increased sulphate and iron concentrations, so it was necessary to find an alternative resource. However, as in the coalfields, the water of Lorraine's iron mines are not acidic.

In 1996 and 1997, several sudden collapses above mine works caused significant damage and frightened people. These very localised collapses were caused by the disintegration of residual pillars of oolitic Aalénians limestone and accompanied the rise of the water in the former underground exploitations. The mined deposit extends over an area of 1700 km² and the exploitation of the iron developed from the outcrop to reach depths of 250 m. In 1996–1998, zones with a risk of collapse were systematically identified. Fortunately, there are very few zones that were already urbanised among the most fragile sectors; ongoing studies on these sectors will determine whether it is more feasible to fill the mine voids or evacuate the threatened places.

Salt

From 1910 until 2002, a deposit of sylvinit in Alsace was exploited over an area of 20,000 ha. The ore contained 60% Na, 24% KCl, and 16% insoluble material. The extraction of potassium left solid waste which, at the beginning of the exploitation (1910–1933) were put in stock piles, then afterwards were discharged as brine to the river Rhine within discharge standards according to international treaties. Fifteen stock piles rich in salt have been built during salt exploitation. When it was raining, these deposits produced leachates that infiltrated into and formed salty plumes in the groundwater beneath the plain of Alsace. The concentration of chlorides was above drinking water standards and represented a regional resource problem.

To address this problem, the former operators of the mines have taken two steps (Anonymous 1996a):

- Since 1980, chlorides have been extracted from the groundwater, using 59 pumping wells downstream of the mining coal tips; and
- Since 1999, the stock piles have been treated, either by accelerated dissolution (extensive irrigation of the stock piles and the recovery of the salt solutions before they infiltrate into the groundwater), or by

capping and then revegetating the surface of stock piles. This treatment will end in 2009.

These operations have allowed the extraction of half a million tons of salt per year, are very important investments, and are publicly financed.

Other ores: a rapid overview

The exploitation of the two last gold mines (Salsigne/Aude, Bourneix/Haute-Vienne) stopped in 2001. Salsigne was one of the most important mining centres in the south of France, and dates back to the 2nd millennium B.C. Many ores have been exploited (e.g. iron, copper, lead, silver). Gold was detected at the end of 19th century and, after the First World War, Salsigne became one of the first arsenic producers in the world. Also sulphuric acid and bismuth have been extracted there. Arsenic pollution of the Orbiel River is a problem, due to weathering of the ancient mines and the millions of tons of mine wastes.

The last exploitation of uranium occurred in Jouac (Haute-Vienne), and the site has been rehabilitated. However, some residual questions exist near other French uranium mines due to potentially radiologically-contaminated sediments settled in dammed lakes down-river from the mines (at Lac de Vassilières/Haute-Vienne and Lac de Saint-Priest-la-Prugne/Allier).

Many small abandoned metal mines induce local acid mine drainage problems (e.g. Saint Bel, Chessy, Largentiere; Schmitt 2000). In all cases, restoration is required by the last operator (if the company still exists) or by the French state, under the regional directorate of industry, research and environment (DRIRE).

Conclusion

Mining is mainly finished in France, but many environmental problems remain (e.g. stability, gas, water). There are less acid water problems than in many other countries thanks to the presence of carbonates, but some mine water discharges have to be treated because of the presence of iron or other metals.

In all cases, restoration is required by the last operator (if the company still exist), or French state under the regional direction of industry, research and environment (DRIRE). Soon a national agency for “after mining” will be created by the government, probably under the direction of the BRGM (Bureau de Recherche Géologique et Minière).

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Mine Water Issues in the Czech Republic

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Historical Introduction

From the early Middle Ages, the Czech lands have played an important role in the development of mining sciences worldwide. Deposits of gold, silver, copper, tin, tungsten, and other metals in localities such as Kutná Hora, Příbram, and Jáchymov are particularly known. In Jihlava, two of the world's first mining laws, *Jura Seu Statuta Illaviae Civitatis* and *Ius Regale Montanorum*, were issued in 1249 and 1250, respectively. The Saxonian author, Georgius Agricola (1494–1555), wrote *De Re Metallica Libri XII* in Jáchymov. That is also where the uranium ore samples used by Marie Skłodowska-Curie (1867–1934), in her research on radioactivity, originated. In the year 1875,

a depth of 1000 m (i.e. the greatest depth worldwide at the time) was achieved in the Vojtěch Mine. The Annenská Shaft (1455 m) was the deepest ore mine in Europe in the year 1940; Jindřich Shaft II (1453 m) at Zbýšov u Brna was the deepest coal shaft in Central Europe. In the 17th century, the Czech lands became a significant producer of hard and brown coal; after understanding the importance of radioactive raw materials, they also became a significant European mining locality for these ores. Nowadays, all of the ore mines in the Czech Republic are closed, and this year the last underground uranium mine is to be shut down. Coal mining is also being reduced; underground coal mining only proceeds in a small part of the Upper Silesian Coal Basin and open pit mining only continues in two brown coal basins in the Podkrušnohoří area in West Bohemia. In addition to these raw materials, non-metallic raw materials, such as kaolin, perlite, raw materials for cement, brick and glass making, and aggregate is still mined in the Czech Republic.

There are hundreds of old underground metallic and non-metallic mines in the Czech Republic that are sources of mine water. Oil and gas exploitation in

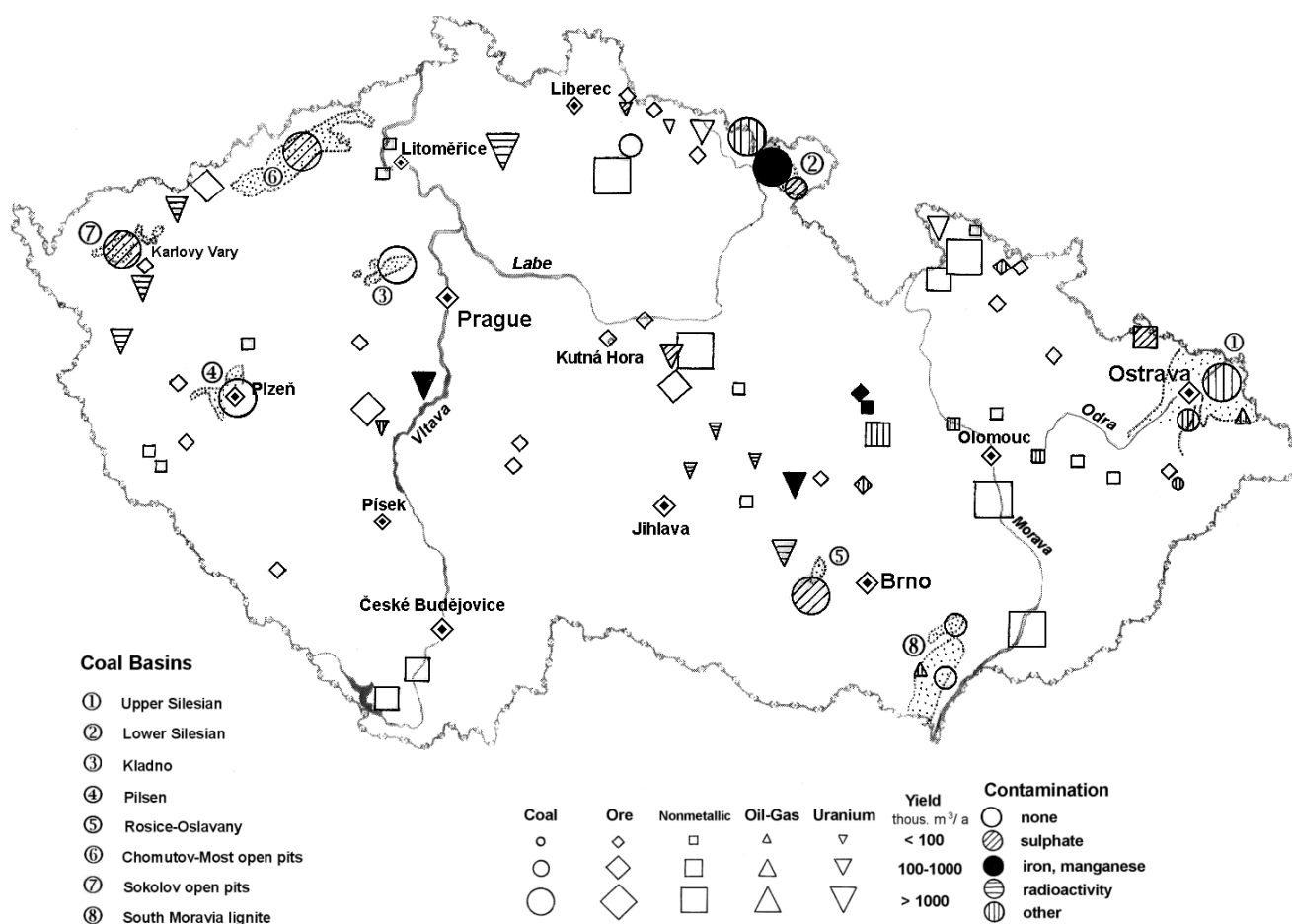


Figure 11. Mine water sources in the Czech Republic (revised from Grmela 1998)

Table 10. Legislative limits for mine water discharges (Explanatory notes: PAH – polyaromatic hydrocarbons, TDS – total dissolved solids, TPH – total petroleum hydrocarbons, Undiss. solids – undissolved solids).

Exploitation from deposit	Surface stream quality	
	Exception for mine water discharges	Limiting value according to 61/2003 S.B.
Coal exploitation		
pH	6 – 9	6 – 8
Undiss. solids	40 mg/L	25 mg/L
PAH	0.01 mg/L	0.0002 mg/L
Iron	3 mg/L	2.0 mg/L
Manganese	1 mg/L	0.5 mg/L
Uranium exploitation		
pH	6 – 9	6 – 8
Undiss. solids	30 mg/L	25 mg/L
Ore deposits exploitation		
pH	6 – 9	6 – 8
TDS	40 mg/L	25 mg/L
TPH	3 mg/L	0.1 mg/L
Iron	5 mg/L	2.0 mg/L
Zinc	3 mg/L	0.2 mg/L
Lead	0.05 mg/L	0.015 mg/L
Copper	1 mg/L	0.03 mg/L
Arsenic	0.5 mg/L	0.02 mg/L
Aggregate exploitation		
Undiss. solids	40 mg/L	25 mg/L

South Moravia is also associated with the release of mine water.

Mine Water Legislation

Legislation dealing with mine water is not uniform in the European states, which often results in non-uniform economic procedures for mine water disposal. In the Czech Republic, the Water Act No. 254/2001 Coll. classifies mine water as “special water,” which is excluded from the direct effect of the Act. According to

Mining Act No. 44/1988 Coll. (Section 40), the term mine water is defined as: “... all ground, surface and precipitated waters that penetrated into underground and surface mine spaces, until they are mixed with other permanent surface and ground waters”. The Czech Mining Office draws the conclusion that: “... if mining activity was completed, the mining claim was cancelled, the licence for the mining company to mine was abolished ... worked-out spaces cannot be taken as underground/surface mine spaces any longer, and thus any water penetrating into these spaces cannot be considered as mine water either.” Mine water in the Czech Republic can be discharged into surface watercourses by permission of the water management authorities based on limited quality and quantity parameters. However, in Decree No. 61/2003 Coll., the Government of the Czech Republic imposed water quality requirements for watercourses; this imposes limits on the quality of discharged mine water by monitoring water quality in the river, after mixing has occurred. However, mine water affected streams are allowed more latitude than those affected by other sources of contamination (Table 10). Mine water cannot be legally discharged into groundwater in the Czech Republic (Grmela and Rapantová 2004).

Mine Water Management

Hydrologically, the Czech Republic affects several of Europe’s watersheds, so that mine water discharged into surface watercourses affects the quality of international streams; management of mine water disposal can therefore be complicated. Mine water production for the characteristic years 1975, 1998, and 2004 is given in Table 11.

In Table 11, the effects of phasing out of mining is evident, as is the effect of legislating the exclusion of discharged groundwater from mines if the mining claim of the old mine is cancelled administratively.

Mine water sources (in a broader sense than given by the legislative definition presented above) are illustrat-

Table 11. Production of mine water in the Czech Republic (in thousands of m³ annually; Grmela 1999 updated).

Production of mine water		Labe watershed	Odra watershed	Morava watershed	Total
1998	Total Czech Republic 1975	122,000	31,000	5,000	158,000
	Non-metallic deposits	5,652	953	6,390	12,995
	Radioactive materials	19,219	185	1,906	21,310
	Crude oil	—	—	83	83
	Ores	1,740	2,838	19	4,597
	Coal-underground exploitation	20,739	16,849	1,932	39,520
	Coal-open-pit exploitation	17,750	—	—	17,750
	Total Czech Republic 1998	65,100	20,825	10,330	96,255
	Total Czech Republic 2004	45,024	8,891	8,379	62,294

ed by deposit type, yield and contamination in the map of the Czech Republic in Figure 11.

Mine Water Issues in the Czech Republic

The greatest problems associated with water related to mining in the Czech Republic are:

- The disposal of mine water from the hard coal deposit of the Upper Silesian Coal Basin (Odra watershed – pollutants: Na^+ , Cl^- , SO_4^{2-}) and the brown coal deposits of West Bohemia (Labe watershed – pollutants: SO_4^{2-}).
- The disposal of acid solutions from chemical uranium mining by leaching in North Bohemia (Labe watershed – pollutants: U, Ra, SO_4^{2-} , and a low pH). In situ leach uranium mining was operated from 1967–1990. Uranium was extracted from Cenomanian sandstones at a depth of more than 200 m via about 9300 surface boreholes in an area of 6.3 km² at Stráž pod Ralskem. Leaching agents were H_2SO_4 (used $3.7 \cdot 10^6$ t), H_2NO_3 ($0.27 \cdot 10^6$ t), HCl ($0.025 \cdot 10^6$ t), and NH_3 ($0.1 \cdot 10^6$ t). Used acid leaching solutions is still being pumped and discharged from these mines.
- The disposal of excess water from tailing ponds and the disposal of water from water treatment plants of former ore mines (Labe watershed – pollutants: heavy metals, U, Ra, SO_4^{2-}).
- The remediation of environmental loads of old ore mining. In the Labe watershed in the Czech Republic, 3011 old environmental loads of ore (including uranium) mining have been recorded, of which in 230 cases (8%), the water outflows from old mine workings to the surface.
- In 135 cases (59%), the water downstream of these sites meets the appropriate water quality standards; in the remaining 95 cases, the water is polluted by inorganic pollutants (Figure 12).
- The disposal of water accompanying oil reservoirs in South Moravia (Danube watershed – pollutants: total petroleum hydrocarbons).

Mine Water Issues Connected with Mine Closure

Recently, the underground uranium mines in the Czech Republic have been intentionally flooded (the last uranium mine in the deposit of Rožná will be closed in this way; the commencement of closure operations is expected in 2006). After underground mining was completed, the mines were left to be flooded naturally with controlled discharging of the mine water that exceeds a prescribed water level in the mine. This

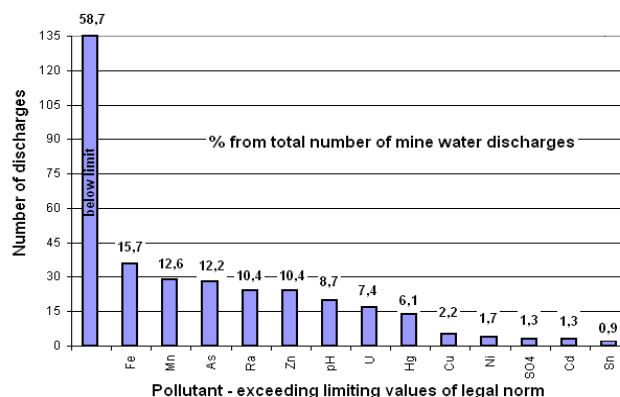


Figure 12. Discharges of gauged waters from abandoned ore mines (including uranium) in the Czech Republic – Labe watershed

water level was determined for each deposit based on surrounding terrain morphology to avoid uncontrollable releases of polluted mine water. Stratified mine water in the lowest levels can be a potential source of uranium in the future.

Two methods of coal mine closure were applied in the Czech Republic. Most abandoned coal mines were simply flooded. Monitoring of mine water discharges has shown growing problems associated with contamination by iron (in addition to other pollutants, such as sulphates). In the case of the former J. Šverma Mine at Žacléř, closure was performed by filling the mine workings with a self-solidifying ash mixture with an admixture of secondary raw materials in the quasi-closed hydrogeological structure of the mine. Long-term monitoring has proven that, in this case, environmental load limits have not been exceeded (Rapantová and Grmela 2002). In the Upper Silesian Coal Basin, the closure of the Ostrava part of the coalfield is complicated by continuing mining operations in the Karviná Basin, which means that the level of flooded mine workings must be kept below the level of these mining operations (Grmela and Rapantová 2002).

Conclusion

As mining activity in the Czech Republic has diminished, mine water production will be decreasing along with problems of their disposal. Furthermore, mine water discharges (quantity and quality) will be stabilised and a decreasing trend in environmental load can be presumed. Even at present, it is evident that the quality of surface streams leaving the territory of the Czech Republic is much better than it was 15 years ago. Nevertheless, mine water issues remain one of the country's primary environmental concerns.

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Mine Water Issues in Slovakia

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Introduction

The complicated geology of the Slovak territory, which is part of the West Carpatians, exhibits substantial variability and hosts a high number of mineral deposits (Figure 13). Underground mining of these deposits has caused numerous environmental changes, the most important being the drainage of natural waters and the generation of contaminated mine water.

Mining in Slovakia dates back to prehistoric times and was especially significant during medieval times. Gold, silver, copper, iron, and polymetallic (Pb, Zn, Cu) ore mining was especially noteworthy. Subsequently, there was a gradual decrease in metal ore exploitation, which was accelerated by the social-economic changes that occurred in Slovakia after 1989; at present, only a single iron ore mine is operating. Coal, magnesite, and gypsum mining operations developed during the 20th century and are still going on.

Characterization of Mine Water Issues

Table 12 shows mineral deposit types that were or are exploited by underground mines in the Slovak territory, together with the type of host rock. The frequency and total average yield of mine water discharges are also shown for the individual mineral deposit types. Most of the mine discharges are of low yield, and are connected with ore veins in Paleozoic rocks. They are concentrated mainly in the Slovenské rudohorie Mts, but also occur in others mountain areas. There are also many discharges of mine water from coal deposits in Neogene sediments, ore deposits in neovolcanic rocks, and magnesite deposits. The mine water yield of approximately 460 mine discharges totals about 1.5 m³/s. However, the presented data should be considered preliminary since not all discharges have been registered yet.

Individual ore mines in near-surface metamorphosed Paleozoic rocks, which are of low permeability, produce relatively small amounts of mine waters, rarely exceeding 1 L/s (maximum up to 20 L/s). The average mine water yield and its regime is controlled by the extent of the mine, the possibility of atmospheric water infiltration (via openings, fractures, and faults) and the climatic conditions of the locality (Bajtoš 2000). However, different hydrogeological conditions occur in Paleozoic carbonate bodies, containing metasomatic



Figure 13. Regional distribution of mine water issues in Slovakia; symbols represent mineral deposit types, numbered to correlate with Tables 1 and 2

siderite, magnesite, or talc deposits, due to karst phenomena. As the extent of such carbonate bodies is not very large (maximum length of 4.5 km, in the Dúbravský masív carbonate structure), they are not regionally important and mine waters yield of individual mines do not exceed 20 L/s.

The chemical composition of mine waters in the Paleozoic rock environment is controlled mainly by the sulphide: carbonate mineral ratio and by the extent of the mine and the mineral deposit. The chemical composition of mine water from ore deposits without sulphides and carbonates (quartzite veins) is hydrogeochemically like background water. Dispersed sulphidic mineralization generates slightly acidic mine waters of low TDS content, enhanced SO_4^{2-} concentrations, and elevated levels of some metals (Table 13) when carbonate minerals are absent (quartz-sulphidic veins), or Mg-Ca- SO_4 - HCO_3 type waters when carbonate minerals are present (carbonate-sulphidic veins). If only carbonate minerals are present (metasomatic siderite, magnesite and talc deposits, carbonate veins) and sulphide content is equivalent to the geochemical background, alkaline Ca-Mg- HCO_3 or Mg-Ca- HCO_3 type mine waters are produced. Stratiform sulphide deposits produce metal-sulfate type acid mine waters with a high total dissolved solids (TDS) content.

Higher mine water yields occur in ore mines in neovolcanic rocks due to the high permeability of the area due to regional faults. Large faults drain phreatic waters over large areas, but also enable ascent of geothermal waters from underlying aquifers (mainly Triassic carbonates). Mine water yields of main drainage galleries can exceed 100 L/s (Table 13). In shallow galleries, Ca-Mg- HCO_3 -type mine waters (hydrogeochemical background) prevail, but in larger mines, the water is dominantly of the Ca- SO_4 -type, due to the influence of sulphide oxidation. Coal mines drain large amounts of groundwater from neogene aquifers (Tables 12, 13). Complex hydrogeological conditions results from the location of the deposits under an erosion basin and the

presence of both overburden and footwall aquifers. Groundwater sometimes bursts into the mines; the sand and andesite layers are particularly dangerous (Kováč 1999). The natural chemical composition of the groundwater changes with depth from a Ca-Mg-HCO₃ to Na-HCO₃ type, with a TDS content up to 0.6 g/L, but is then altered by the oxidation of pyrite, realgar, arsenopyrite, and auripigment in the coal workings to Ca-SO₄ type water; the TDS increases to 4 g/L, the SO₄²⁻ content increases to 2.4 g/L, and metal concentrations increase (Fe to 25 mg/L, As to 1.2 mg/L) in the abandoned mines. However, these high concentrations are decreased by dilution by the natural groundwater. Mine waters of the Modrý Kameň coal region contain gaseous carbon dioxide.

Mine Water Exploitation

In the past, mine waters were mainly used in mining technology, but nowadays, many abandoned mine water discharges are used for drinking purposes, mainly in the Slovenské rudohorie Mts. and Central

Slovakia neovolcanities region (Cicmanová et al. 1999). Utilisation of other mine water discharges is limited mainly by elevated metal concentrations (most often Fe, Mn, Sb, and As), or SO₄²⁻, and TDS.

The thermal energy potential (TEP) of the largest mine water discharges reaches values of some MW, so exploitation of their geothermal energy is possible using heat pumps (Bajtoš 2001). A specific problem is the need to isolate the most concentrated TEP resources; this can be done by pumping the groundwater to the surface through the mine workings or a borehole. The Kremnica thermal swimming baths, which are supplied through an underground borehole outflow ($Q = 39 \text{ L/s}$, $t_{\text{water}} = 56.8 \text{ °C}$) from the Ľudovít shaft, is an example.

Mine Waters and Environment

From an ecological point of view, the most important problems are the long-term effects of mining on groundwater circulation and the contamination of sur-

Table 12. Quantity and frequency of mine water discharges in Slovakia, classified by mineral deposit types; Q = total yield, n: number of issues, PM: Paleozoic metamorphities, MS: Mesozoic sediments, PS: Paleogene sediments, NR: neovolcanic rocks, NS: Neogene sediments

Deposit type	Raw minerals	Host rocks	Main mine districts/deposits	Q L/s	n
1. hydrothermal quartz-sulphidic veins	antimonite, gold	PM	Pezinok deposit (Malé Karpaty Mts.) Dúbrava deposit (Nízke Tatry Mts.), Betliar, Čučma, Zlatá Idka, Popoč ore districts (Slovenské rudohorie Mts.)	30	37
2. hydrothermal siderite-sulphidic veins	siderite (chalkopyrite, cinnabar, barite)	PM	Rudňany ore field, Slovinky- Gelnica ore field, Rožňava ore field, (Slovenské rudohorie Mts.)	450	36
3. metasomatic siderite	siderite	PM	Nižná Slaná - Kobeliarovo mine, Železník deposit (Slovenské rudohorie mts.)	27	4
4. metasomatic magnesite deposits	magnesite	PM	Dúbravský massif deposit, Bankov deposit (Slovenské rudohorie Mts.)	91	5
5. metasomatic talc deposits	talc	PM	Mútnik, Samo deposits (Veporské vrchy Mts.)	8.4	7
6. stratiform sulphidic deposits	pyrite, chalkopyrite	PM	Smolník deposit (Slovenské rudohorie Mts.)	6	1
7. stratiform uranium deposits	uranite, molybdenite	PM	Novoveská Huta deposit (Slovenské rudohorie Mts.)	10	5
8. gypsum and anhydrite sedimentary deposits	gypsum and anhydrite	MS	Novoveská Huta – Tollstein deposit (Slovenské rudohorie Mts.)	12	1
9. sedimentary Mn deposits	pyrolusite, manganocalcite	PS	Švábovce-Kišovce mine (Hornád basin)	5	1
10. polymetallic ore veins, Au and Ag stockworks	galenite, sphalerite, gold, silver	NR	Štiavnica-Hodruša mining district, Kremnica (Central-Slovakia neovolcanites Mts.), Zlatá Baňa deposit (Slanské vrchy Mts.)	400	30
11. veinlet-disseminated hydrothermal mineralization	cinnabar	NR	Dubník Hg deposit, Merník deposit (Slanské vrchy Mts.), Malachov deposit (Central-Slovakia neovolcanites Mts.)	2	3
12. veinlet-disseminated hydrothermal mineralization	noble opal	NR	Dubník opal mine (Slanské vrchy Mts.)	0.5	1
13. coal deposits	brown coal, lignite	NS	Nováky, Cígeľ, Handlová (Horná Nitra basin), Modrý Kameň coal mines	450	5
14. salt deposits	rock salt	NS	Prešov-Solivar mine (Košice basin)	0.5	1

Table 13. Hydrological and hydrochemical parameters of chosen mine water discharges. Asterisk indicate data linked to mines operation, others data represent abandoned mines

	Q (L/s)	Measured period	T_{water} (°C)	Chemical type of mine water	TDS (g/L)	Contaminants
1. Quartz-sulphidic ore veins in Paleozoic rocks						
Zlatá Idka Hauser adit	1.6–2	1997	8.4–9.6	Ca-Mg-HCO ₃	1.9–2.2	As, Fe, Mn
Čučma, Gabriela adit	0.5–38	1991–1992	7–10	Ca-Mg-HCO ₃	0.3–1	As, Sb
2. Carbonate-sulphidic ore veins in Paleozoic rocks						
Rudňany, Mier shaft*	16–25	1988–1997	8–17	Mg-Ca-HCO ₃ -SO ₄	1.4	Sb, As, SO ₄
Bindt, František adit	3.4–18	1985–1997	6.6–8	Mg-Ca-HCO ₃	0.4–0.8	–
Slovinky, Alžbeta adit*	13–20	2000	8–9	Mg-Ca-SO ₄ -HCO ₃	1.4–1.9	Fe, Mn, SO ₄ , As, Sb
Rožňava *	8.7–10	1991–1999	9–16	Ca-Mg-Na-HCO ₃ -SO ₄	0.9–1.2	Fe, Mn
3. Metasomatic siderite deposits in Paleozoic rocks						
Nižná Slaná mine*	19	2000	9	Mg-Ca-SO ₄ -HCO ₃	0.5–8	Fe, Mn, SO ₄
4. Metasomatic magnesite deposit in Paleozoic rocks						
Dúbravský masív deposit*	35–45	1989–1990	15	Mg-Ca-HCO ₃	0.4–0.5	–
Bankov*	12–25	1981–1988	18.5–21	Mg-Ca-HCO ₃	–	–
Podrečany*	18–20	1980	12–14	Ca-Mg-HCO ₃	0.2–0.4	–
Burda*	12–13	1959	12.5–13	Ca-Mg-HCO ₃	0.9–1	–
5. Metasomatic talc deposit in Paleozoic rocks						
Mútnik*	3.3–9.4	1995		Ca-Mg-HCO ₃	–	–
6. Stratiform sulphidic (pyrite) deposits in Paleozoic rocks						
Smolník, Pech shacht	6	1999	15	Mg-Fe-SO ₄	4.9	pH, SO ₄ , Al, Cu, Zn
7. Stratiform uranium deposits in Paleozoic rocks						
Novoveská Huta, Vodná adit	2.5–10	1994–1998	9–10	Ca-Mg-HCO ₃ -SO ₄	0.7	Ra, U, Rn, SO ₄ , Fe, Mn
8. Gypsum and anhydrite sedimentary deposits						
Novoveská Huta Tollstein	10–15	1987–1993	8–9	Ca-SO ₄	1.9–2.2	SO ₄
9. Sedimentary Mn deposits in Paleogene sediments						
Kišovce-Švábovce	4–20	1973–1995	13–16	Na-Ca-HCO ₃ + CO ₂	2.5–4.1	Fe, Mn, As
10. Au-Ag and polymetallic ore veins in neovolcanic rocks						
B. Štiavnica-Hodruša district VDŠ adit	100–130	1993	16–18	Ca-SO ₄	1.2–1.3	SO ₄ , Fe, Mn, Zn,
Kremnica district, KDŠ adit	70–100	1987–1988	19–20	Ca-SO ₄	1.5	SO ₄
11. Hg mineralization in neovolcanic rocks						
Dubník Hg	1.2–2.5	2002	9–9.7	Al-Ca-Fe-SO ₄	0.9–1.2	pH, Al, Fe, Mn, Zn, Co
12. Noble opal mineralisation in neovolcanic rocks						
Dubník opal	0.5–1	2002	7	Al-SO ₄	0.8	pH, Al, Fe, Mn, Zn, Co
13. Coal deposits in neogene sediments						
Nováky*	112–156	1992–1998	15–25	Na-HCO ₃	0.3–0.8	SO ₄
Handlová*	116–156	1992–1998	15–28	Ca-Mg-HCO ₃	0.5–1	SO ₄ , Fe, Mn, As
Cígel*	128–160	1992–1998	10–19	Ca-Mg-HCO ₃	0.4–1.3	SO ₄
Modrý Kameň* (mine / boreholes)	48–77 97–164	1979–1983	18	Ca-Mg-HCO ₃	2–4	SO ₄
Gbely* (mine / boreholes)	16–24 2.6–10.1	1990–1996	15–22	Ca-Mg-HCO ₃	0.75	Fe, Mn, NH ₄
14. Salt deposits in neogene sediments						
Prešov-Solivar*	0.5–0.6	1800–1979		Na-Cl	310	Na, Cl, SO ₄

face waters. Groundwater levels in Palaeozoic rocks have been lowered, affecting nearby domestic wells. Similarly, the irreversible drying up of the exploited “Markus” spring (the Spišská Nová Ves town water supply) by gypsum mine drainage in Lower Triassic sediments has been well documented.

The “Teplá voda” spring yield (the Jelšava town water supply) has been gradually decreased by the “Dúbravský masív” magnesite mine dewatering but it is anticipated that after future mine flooding, it will probably be renewed. The dewatering of the Kišovce-Švábovce Mn mine depressed groundwater levels in the surrounding paleogene sediments and the level of the mineral waters in the underlying Triassic carbonate structure. The exploited “Tatra” mineral spring was irreversibly destroyed during the mining; the mineral water now flows out of the mine.

Mine water issues can contaminate surface water bodies. An extreme example is the contamination of the Smolník stream by acidic metal-sulphate mine waters from the Smolník pyrite deposit (Slovenské rudohorie Mts.), with concentrations of 0.4–1 g/L Fe, 0.03–0.14 mg/L Mn, 0.1–1.3 g/L Al, 3–90 mg/L Cu, 5–140 mg/L Zn, and 2.7–4.2 g/L SO_4^{2-} . Acid metal-sulfate mine waters also contaminate streams in the Dubník district (the Slanské vrchy Mts.). Mine waters of quartz-antimony veins have high Sb and As content (tens of mg/L). Mine waters of U-Mo mineralisation in the Novoveská Huta, occurring together with carbonate-sulphidic veins, introduce a special type of contamination (tenths of Bq/L ^{226}Ra , hundreds of Bq/L ^{222}Rn).

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Austria

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Historical Situation

Austria has a long tradition in mining. The first underground mining (for chert) commenced about 4000 B.C., and salt was mined from approximately 1000 B.C. From the 11th to the 15th centuries, mining played an important role in Austria's economy.

Water can be of benefit in mining, for example, in the solution mining of salt. However, it can also cause problems. In the 16th century, the Röhrebühel copper

mine was amongst the world's deepest underground operations (with an 800 m deep vertical shaft), and suffered significant problems with water ingress. The Schwaz copper-silver mine also had problems with water ingress and developed an elaborate water haulage system to deal with it. In 1361, a terrible water inrush stopped operation of the Oberzeiring silver mine (Weber 1997).

Current Mining

Today, mining contributes only 0.5% to the Austrian Gross Domestic Product. However, the remaining 99.5% of the GDP would not be possible without the availability of those mining products. Indeed, despite the fact that some metals (iron ore, tungsten), industrial minerals (graphite, gypsum, kaolin, magnesite, salt, sulphur, talc, oilshale) and energy fuels (lignite, natural gas, oil) are mined in Austria, most of the minerals



Figure 14. Water outlet before water outburst in 1998



Figure 15. Same water outlet after water outburst

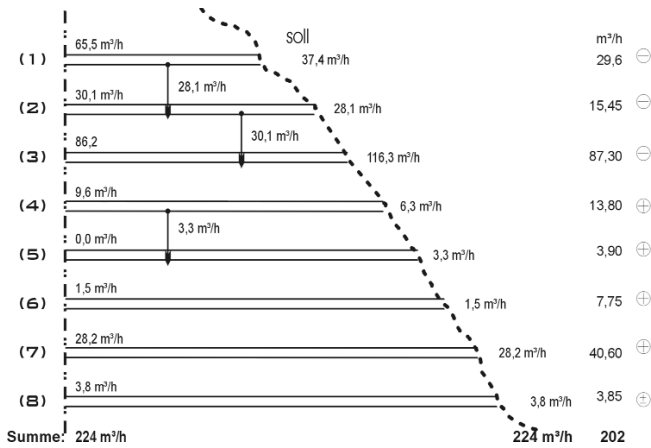


Figure 16. Quantitative water balance

and metal commodities (such as ferro-alloy metals) have to be imported.

The exploration, exploitation and (underground) storage of mineral raw materials, as well as their beneficiation, are governed by the Mineral Act (MinroG), which was enacted 1 January 1999 (Mineralrohstoffgesetz 1999). Key articles regulate miner safety, protection of the environment, protection of mineral deposits during production, and usability of the surface after the mine closure. The utilisation of mine water is regulated in § 106 of MinroG.

Mine Water Management and Risk Assessment

There is a strong need for mine water risk assessment. As a matter of fact, controlled drainage of mine waters is of crucial importance for the entire mining period as well as post-closure. A careful observation of changes in the inflow of water into mine voids allows the timely recognition of risks and the development of methods to minimize their impact. The chemical composition of mine waters can also greatly impact the environment. In addition, geotechnical problems can result from water in alpine areas. Most problems arise at old, abandoned mines, as most of those mines were not closed “*lege artis*”.

The Austrian mining authorities require quantitative and qualitative mine water balances for underground operations. A mine water balance may be defined as the difference between the total of inflowing water into the mine openings and the total amount of natural drained or pumped water at the main mine entrance.

Quantitative Balances

Sufficient knowledge of the hydrogeological situation is essential to define necessary safety measures during or after mining. Such information includes: where the

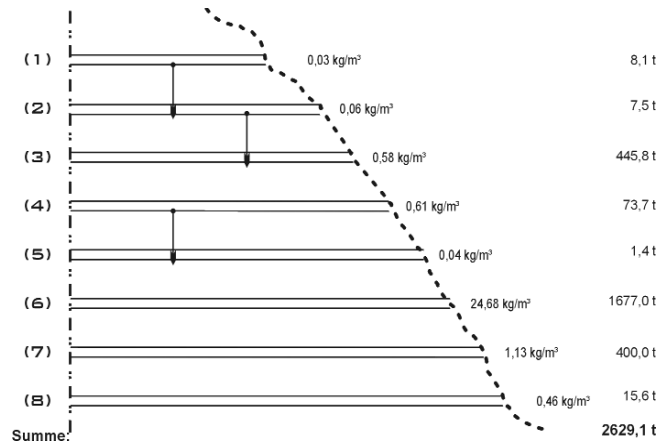


Figure 17. Qualitative water balance: mass balance of total dissolved solids.

main water inflows occur; the (maximum) variation of the inflow with respect to precipitation; the amount of mine water that can be freely drained; and the amount to be pumped. Insufficient knowledge about the hydrogeological situation can result in incorrect conclusions and cause serious problems.

An example of this is the blockage and collapse of some old lead-zinc mine adits that had operated in the 19th century and were closed about 50 years ago (Figures 14 and 15). The main host rock for the mineralization is a karstic limestone. At most times, there was no significant water inflow, but in wet periods (such as after snowmelt), water drained out of the deepest adit, which was about 100 m above the bottom of a valley. As the deepest adit was water-bearing only periodically, concrete tubes had been inserted to drain the mine when necessary. No-one noticed when some tens of meters of the part of the adit near the surface became completely blocked with fine-grained material. There was no evidence of any malfunction of the drainage system until 1999, when due to a very wet autumn, lots of snow that winter, and an extreme wet spring, the inner adit became completely flooded. At midnight, 26 June 1999, a huge explosive water outburst occurred, producing a mudflow that flooded the valley. Fortunately, nobody was hurt, but this example shows clearly that sufficient knowledge of the hydrogeological regime is essential. Additionally, underground mines should be drained properly. In particular, adits in water-bearing carbonate rocks should remain open to allow water to freely drain.

Qualitative Water Balance

Knowledge of the chemistry of mine waters is of crucial importance for exploration and for the utilization of mine waters. However, changes in the chemistry of mine waters, especially those draining evaporite deposits, may indicate possible problems in the stabil-

ity of mining openings.

Solution mining makes use of the natural solubility of salt by controlled solutions. However, uncontrolled dissolution can adversely influence the stability of pillars between the mining voids. If uncontrolled water contacts underground workings, then parts of the underground system may destabilize. The resultant subsidence can induce more unplanned infiltration and increased instability. Due to this, any water inflow into an evaporite mine has to be drained properly, so that contact between water and soluble materials is avoided.

A detailed quantitative water balance (Figure 16), comparing ingress water into the mine system and drained water at the portals, should be done at least twice a year (during dry and wet periods). In evaporite mines, either operating or not, there is also a need for qualitative balances (Figure 17), to check whether salinity of the water is stable or not. An increase in total dissolved or suspended solids may indicate uncontrolled dissolution. For example, the quantitative balance in Figure 16 clearly shows negative balances in the upper levels and positive balances in the lower levels, due to malfunctions in the drainage system.

Hungary

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Mining History

Mining that occurred in Hungary during prehistoric times (flint) through the medieval ages (e.g. precious metals, salt) did not significantly impact ground water. Mine water pumping only became significant at the beginning of the 20th century.

Classic Issues

Sizable mineral reserves such as brown coal, manganese, and bauxite are located in basin areas of a karstic massif called the Transdanubian Mountain Range. As these reserves are situated near or beneath the water-table, security of mining operations required continuous mine dewatering.

After World War II, three different methods (passive, preventive, and active) of mine dewatering were applied in Hungary, depending on the risk to mining. In the passive case, only the amount of water that flowed

Though most Austrian mine waters are known to be of good environmental quality, there are some localized areas where abandoned mines cause affect the receiving streams. One example is the above-mentioned abandoned medieval silver mines near Schwaz in Tyrol, where elevated antimony concentrations have been found recently (e.g. Wolkersdorfer and Wackwitz 2004). Another source of antimony, as well as some other heavy metals, is the former Brunnalm mining area in Tyrol, where elevated concentrations were found close to the old mine workings (Schaffer et al. 2001).

Conclusion

Quantitative and qualitative mine water balances, which should be tailored to the specific type of operation, are a simple but effective tool for an early diagnosis of risks during operation of mines. They are also an important prerequisite for mine closure. Therefore, this method of risk assessment is essential for any mining operation and is part of the approval procedure for exploitation permits.

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spontaneously into the galleries was pumped out. The preventive case was enforced when there was a pending danger of an unexpected water in-rush from the surrounding network of large underground cavities; this required that controllable pathways be opened by horizontal or oblique fore-drilling to direct the water into pre-arranged dewatering facilities. When considerable depression of the water table was required, active dewatering was implemented prior to shaft-sinking; this required the drilling of large diameter (up to 5 m) pumping wells.

Between the mid 1970's and mid 1980's, approximately 860,000 m³ per day (600 m³/min) of karst water was

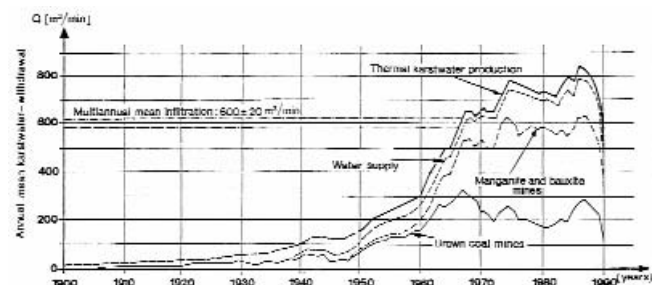


Figure 18. Annual mean karstwater withdrawals from the main karst of the Transdanubian Central Mountains Range from 1900 to 1991

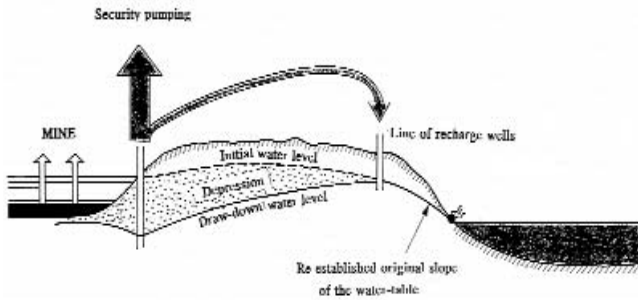


Figure 19. Artificial recharge to restore the initial gradient of the water table and the discharge of the damaged thermal springs

discharged from the mining operations of the Transdanubian Range (Figure 18).

Adding in the water supply needs of the region (200 m³/min) meant that about 800 m³ were being removed every minute. However, the long-term mean groundwater recharge for the region is only about 600 m³/min. As a result, a significant depression in the karst water table occurred over a total aquifer region of 8,000 km². The average depression was 30–50 m, but near mines, depressions of 100–150 m were observed. By the end of this period, previously existing water supplies had been drained. Moreover, the famous thermal water springs, which had been used for centuries as health spas and then as important tourist attractions, were significantly affected.

To replace the damaged local provisions, new, large-scale water supply projects were installed using the existing mine dewatering structures (Figure 19). Unfortunately, 5 to 10 years after this was done, the extraction of underwater mineral deposits became less profitable. Combined with political considerations, the authorities and the mining companies mutually agreed to close the mines.

Current Status

Prior to the decommissioning of the mines, the state authorities required the relevant companies to perform a clean-up operation. This operation involved bringing to the surface those substances which would have harmful effects when mixed with groundwater (Figure 20). However, only partial clean-up was accomplished before the companies declared bankruptcy. The security-equipment (i.e. roof supports, ventilation, light, pumps, transport facilities and related electrical network) had to be left in place as they could not be retrieved safely (collapse and water intrusion). Consequently, several thousand tonnes of iron structures, 10–20,000 m³ of wooden structures, hundreds of tonnes of hydraulic oil, and considerable amount of various chemical substances (fire extinguisher materi-

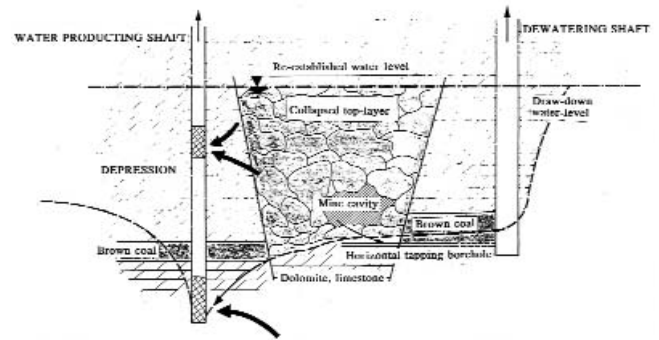


Figure 20. Closed brown coal mine as underground pollution source

al, plastics used for sealing, etc.) remained there.

Analysis of water samples taken from the subsequent pumpings, which were intended to purify the affected areas, revealed that the rock-material itself was the source of a number of contaminants. The most prominent natural pollutant was pyrite, which was dispersed throughout the host-rock. During the mining operation, pyrite in the fractured rock was oxidized as a result of the forced ventilation. Once inundated by the recovering water table, the acid salts dissolved; the acid solution in turn dissolved other materials left in the mines.

Pollutants originating from the altered host-rock are relatively inexhaustible. Their overall contribution was found to be at least 1,000 times that of the man-made pollutants in the pumped water during the control period, which lasted several months. This has serious implications on the potential use of groundwater for water-supply purposes.

Several techniques were applied to protect the water-supply facilities against these underground pollution sources: Artificial barriers (injected concrete screens) were created to seal up various mine cavities in order to reduce circulation, creating dead areas of water. Pumping the water to the surface, treating it and then injecting it was also tried but was found to be of limited success. Initial results were promising but later it was shown that the pumping only created “short circuits” in the tunnels, allowing uncontaminated water to arrive at the surface. Once the pumping ceased, pollution resumed. The most successful method was a hydraulic defence. This involved continuous long-term pumping to create small-scale local depressions in order to keep the contaminated water in place (i.e. away from the water-supply area).

The costs of these remedial pumping operations, which commenced in 1991, had to be paid by the state (the tax-payers). The mining companies responsible for causing these environmental problems could not be held liable as they had dissolved without any legal suc-

cessor. The environmental impact from the mining and particularly from the mine decommissioning is significant and will continue for at least several decades.

Focus for Future Research

The lesson to be learned from this experience is that adequate planning for mine decommissioning (which is a complicated procedure) must be formulated in detail prior to licensing mining projects. On-going

reviews of the decommissioning procedures should be carried out during the life of the mine, as new problems and information arise. Furthermore, a budget for the mine decommissioning must be retained in the care of the licensing authority, and the mining company must be required to contribute to it on a regular basis.

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Switzerland

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Introduction

There are about 200 small ore deposits in the Swiss Alps and the Jura mountains (Figure 21, based on data of Kündig et al. 1998), most of which were mined, mostly underground, from the middle ages to 1945 (end of World War II). The last two iron mines (Gonzen and Herznach) closed around 1965 for economic reasons. A few small brown coal mines were situated in Tertiary Molasse or Quaternary gravel beds

and closed down as well in 1945. Recently, the Swiss Geotechnical Commission, financed by the Swiss Academy of Science, has compiled a comprehensive data bank on Swiss ore deposits and has started to publish condensed versions on various regions (Kündig et al. 1998, Cavalli et al. 1998; Kündig et al. 1990). Between 1956 and 1984, a comprehensive survey was carried out for U, with the hope of finding enough raw material to secure Swiss needs (Gilliéron 1988). From 1980 to 1989, within the frame of a National Research project, which had the aim of inventorying natural resources, a large scale geochemical prospecting campaign was carried out in the Wallis and Graubünden areas, looking especially for W and Au (Della Valle 1991; Della Valle and Haldemann 1991; Woodtli et al. 1985, 1987). During this project, about 6,000 river sediments and soil samples were analyzed for Mo, Pb, Zn, W, Cu, Ni, Cr, Co, Ag, Th, Bi, U, As, Sb, S, Ba, Sn, and Au, but no water samples.

Environmental Data on Tailings and Mine Waters from Switzerland

The first environmental assessments of former Swiss mining activities were made starting in 1992 (Bondietti et al. 1994; Pfeifer et al. 1994, 1997; Pfeifer et al. 1999a,b). The discovery of elevated concentrations of As in surface and ground waters resulted in a new assessment of several former gold mines and their contribution to arsenic contamination (Greppin 1997; Häussermann 2000; Pfeifer et al. 2000, 2002, 2004).

The relevant environmental data is summarized in Table 14. In most cases, contamination is restricted to the immediate surroundings of the former mining site (typically, no more than 200 m away) and usually small waste rock repositories.

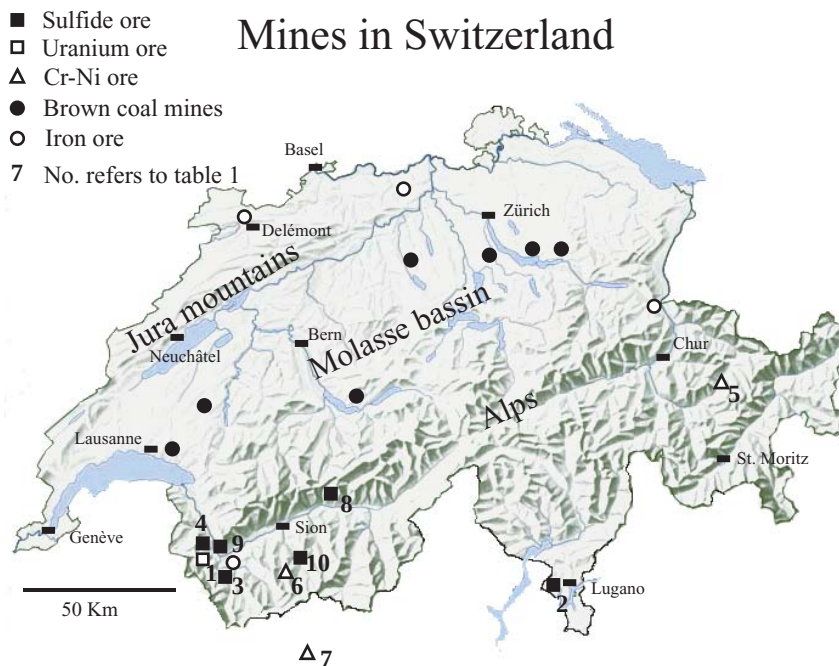


Figure 21. Selected mines in Switzerland; numbers refer to the mines mentioned in Table 14

Table 14. Overview of the available environmental studies of mining related contamination in Switzerland

Metal and locality	Environmental data available	References
U La Creusaz-Les Marécottes /VS [1]	soil (dump zone): 10–2000 ppm (3 ppm) vegetation: 51 ppm (6 ppm) water: 21 µg/L (0.5µg/L)	Dominik et al. 1992; Pfeifer et al. 1994, 2000
As Astano-Costa/TI [2]	soil (dump zone): 0.2–10% (20–100 ppm) vegetation (birch leaves): 2–10 ppm (< 0.1 ppm) water: 5–200 µg/L (<1 µg/L)	Bondietti et al. 1994; Pfeifer et al. 2000
As La Payanne-Bagnes /VS [3]	soil (natural downhill creep): 20–1250 ppm (0 ppm) water: 2–4 µg/L (<1 µg/L)	Greppin 1997
As Salanfe-Finhaut/VS [4]	soil: 10–8,400 ppm (< 3 ppm) water in mine: 750–4,000 µg/L water in local ponds: 7–750 µg/L	Häussermann 2000
Ni Davos [5]; Val d'Hérens [6], Baldissero/It [7]	soil: 2,000–25,000 ppm (30 ppm) vegetation (birch leaves): 4.7 ppm (3,000 ppm) spring water: pH 8.5–10, 5–17µg/L (< 3 µg/L)	Juchler 1988; Pfeifer et al. 2000; Scheder and Streiff 1997
Cr Davos [5]; Val d'Hérens [6], Baldissero/It [7]	soil: 1,300–1,800 ppm (40 ppm) vegetation (birch leaves): 2.1 ppm (1.6 ppm) spring water: pH 8.5–10, 1.3–10 µg/L (< 2µg/L)	Juchler 1988; Pfeifer et al. 2000; Scheder and Streiff 1997
Pb Astano-Costa/TI [2]	soil (dump zone): 0.1–1.7% (20 ppm) vegetation (birch leaves): 1–1.6 % (0.6%) water: 0.2–9 µg/L (0.1 µg/L)	Bondietti et al. 1994; Pfeifer et al. 2000
Pb Goppenstein/VS [8]	soil (dump zone) : no data soil naturally enriched: 200–5,000 ppm (50 ppm) vegetation: no data water: no data	Woodtli et al. 1985
Pb Alesse/VS [9]	soil (dump zone): 0.05–12% soil naturally enriched: 1,000–9,000 ppm (50 ppm) vegetation: 52 ppm (<i>Taraxum off.</i>) water: 11–22 µg/L (0.1 µg/L)	Kufrin 2001
Zn Astano-Costa/TI [2]	soil (dump zone): 200–4,000 ppm (70 ppm) vegetation (birch leaves): 10–16 ppm (6 ppm) water: 20–170 µg/L (8 µg/L)	Bondietti et al. 1994; Pfeifer et al. 2000
Zn Goppenstein/VS [8]	soil (dump zone): no data soil naturally enriched: 800–2,100 ppm (300 ppm) vegetation: no data water: no data	Woodtli et al. 1985
Cu Baicolliou-Tsirouc-Biolec- Anniers/VS [10]	soil naturally enriched: 75–4,000 ppm (30 ppm) vegetation: no data water: no data	Woodtli et al. 1985

Locality number [column 1] refers to Figure 30. Legend for the cited concentrations in parentheses: 1 (...ppm): uncontaminated local reference value, 2 (...µg/L): local uncontaminated water.

Whereas the composition of the waste material and adjacent soils is relatively well known, there is a paucity of data on metal contents in waters and plants. The mine waters are rarely very acid (the lowest pH values measured were 3.1 [Meisser 2003], but most are between 5 and 6). In the case of U and As, the ore deposits are at the origin of an important natural dispersion extending from the headwater areas of the watersheds down to the Mediterranean, and are responsible for elevated metal concentrations in some drinking waters. In western Switzerland, dissolved and particulate U flows through the Rhone catchment, with up to 15 t of U entering Geneva lake every year (Dominik et al. 1992; Pfeifer et al. 1994, 2000). In southern Switzerland and adjacent areas in northern Italy, the presence of till and river sediments stemming from the

well known Permian sulfide veins in the basement of the Southern Alps is at the origin of elevated As-concentrations in surface and ground waters, which in many cases exceed legal thresholds for drinking water (Camusso et al. 2002; Pfeifer and Rey 1998, Pfeifer et al. 2000, 2002, 2004).

Mine Water Related Initiatives

The Mine-Water Interdisciplinary Network Europe (M-WINE) was initiated by a European workshop (70 participants) in May 2002 at the University of Lausanne to bring together European scientists to improve the knowledge and exchange experiences on mining related contamination. The second one (60 participants) was organized in June–July 2003 in Lisbon

by the Geological Survey of Portugal (IGM). The 3rd meeting took place in August 20th 2004 in Florence (Italy) during the 32nd International Geological Congress (IGC). The network groups currently include about 250 European researchers.

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Italy – Some Examples of Mine Water Problems in Tuscany

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Historical Situation

The mining history in the Tuscany region (Central Italy) is representative of mining activities in Italy, including the stone sector. Tin, copper, lead, zinc, and iron deposits are present in central and southwestern Tuscany in the Metalliferous Hills and on Elba Island (Figure 22). Today, almost all of the mines are closed because of mineral exhaustion or because of unfavourable economic and environmental conditions;

the exceptions are the numerous marble and sandstones quarries of Carrara and the rock salt mines of Volterra.

Many sites were exploited from at least 1000 B.C. up to the 1970s and 1980s. Evidence attests to exploitation during the Etruscan, medieval and modern periods (in particular, the historical literature by Vannoccio Biringuccio, 1480–1538). Some historical mines in northwestern Tuscany (Apuan Alps) exhibited great mineralogical variety, but ore quantities were limited (with the exception of barite and possibly silver). More recently, mining since the end of the 19th century up to the 1980s was based on pyrite, iron, barite, cinnabar, and lignite deposits (Tanelli 1983). The cinnabar mines were situated around Mt. Amiata. In this ancient volcanic area and in the northern area of the Metalliferous Hills, geothermal resources are very important. Lignite was exploited from a large open mine in Saint Barbara, closed some years ago (Upper Valdarno, near Florence), and from Ribolla underground mines, near Grosseto, closed down after a serious accident which resulted in 45 fatalities (4 May 1954). Pyrite mining mainly supplied the chemical industry, and subordinate iron production, together with Elba mines (limonite and hematite). The pyrite mines are concentrated in a restricted area in the northern Grosseto district; of these, Gavorrano, Fenice Capanne, Niccioleta, and Campiano were the principal mines. The Gavorrano was one of the largest pyrite mines in Europe and is famous because it represented the historical evolution of mining techniques in small drifts. In contrast, Campiano represents modern mining based on wide rooms.

The main geological context of the Tuscan mines, in particular in the Colline Metallifere area, is represented by the contact between the carbonate units and the granite, granodiorite, and quartz-monzonite intrusions. The area is characterized by NNW–SSE elongated post-orogenic basins developed over an antecedent extensional horst and graben structure consequent to the Tyrrhenian sea opening. Typical of this tectonic province are intrusive bodies, with decreasing age from west (7–8 My) to east (4 My), the emplacement of which was followed by more extension. The activity of the province is attested by important geothermal fields (Larderello, Amiata) located within a major mining district (Campiglia, Elba Island, Amiata, Abbadia, in Figure 22).



Figure 22. Location of the principal mining areas in Tuscany: 1) mines of the southwestern Apuan Alps; 2) Metalliferous Hills; 3) Campiglia mines; 4) Elba mines; 5) Santa Barbara Upper Valdarno lignite; 6) Mt. Amiata (the most important mine is in Abbadia S. Salvatore village)

Table 15. Characterization of some Tuscan mine water issues

Mine	Minerals extracted	Mining method	Volume mined (mill. m ³)	Filling (mill. m ³)	Theoretical voids (mill. m ³)	Mine waters	Discharge (L/s)	Acid water
Gavorrano	Pyrite	Horizontal cut and fill	10.12	6.5	3.62	Pumped	60–120*	No
Niccioleta	Pyrite	Sub-level	5.06	0	5.06	Flooded	230	Yes/No
Campiano	Pyrite	Filled sub-level	1.5	0.64	0.86	Flooded	16*	Yes
Abbadia San Salvatore	Cinnabar	Horizontal cut and fill	6	3.18	2.82	Flooded	-	No

* Presence of thermal waters

Mine Water Management and Problems

At present, except for rock quarries, the only working mines are for rock salt (near Volterra, by Solvay S.P.A.). Large quantities of fresh water from the Cecina river and from wells are utilized (10^6 m³/year) to extract, by dissolution, about $2 \cdot 10^6$ ton/year of salts, for soda and chlorine producing. This activity results in hydrologic and hydrogeologic problems, in particular ground water pollution and subsidence.

Although all other mines have been closed, some for more than 30 years, water pollution and acid waters can still be observed, especially around small dumps. This is due to deficient environmental practices. The most serious problems are linked to the presence of large tailings ponds (at Gavorrano, Campiano, and Fenice Capanne) and above all, the interruption of dewatering, leading to water table rebounds and flooding in the pyrite mines.

Table 15 characterises some of the large Tuscan mines that have groundwater problems. In particular, the mine pool in the Campiano mine, which closed down in 1995, feeds a spring with a discharge of about 16 L/s



Figure 23. Water pumping system at the Gavorrano underground mine (at present, about 80 L/s)

of acid mine waters (in this case, mixed with geothermal waters). The polluted water has a pH of 4 and elevated concentrations of sulphate and metals (iron, copper, zinc, and arsenic), and produces serious environmental problems in the Merse river, which happens to be an area of particular natural and touristic interest.

The land above the decommissioned Gavorrano mine is affected by failure phenomena (subsidence and sink-holes) caused by the failure of overburden strata above mine voids or the collapse of drifts and shafts (Crosta and Garzonio 1998). It is anticipated that these problems will worsen due to the future water table rebound and the rebirth of hot springs (interrupted in 1957) in the Bagno di Gavorrano Village, where the ancient thermal springs site are now surrounded by a large urbanized area. At present, about 80 L/sec are being pumped out of the mine (Figure 23); when dewatering stops, the flow leaving the ancient thermal zone could be about 30–40 L/sec, which could provoke subsi-



Figure 24. Water sampling at the -110 m level of the Gavorrano mine

dence events and soil stability problems affecting the foundations of numerous buildings.

Ongoing Activities in Mining Areas

Over the last decade in Tuscany, the heritage of mining has been recognized as culturally important; the Elba mining area is included in a UNESCO provisional list of the world's prominent tourist parks.

Many of the mine sites are historically and geologically interesting. Others are part of a natural landscape, with archaeological, monumental sites, karst or woody areas, etc., and with important thermal springs or circulations. Recently, the mine park of Metalliferous Hills was instituted by a national law and, consequently, is sustained by national and European funds. In this area, the study of mine water quality, discharge and reutilization is an important research topic (Garzonio 2000) because it is important to a profitable rehabilitation of the mining areas and its water resources (fresh and thermal waters; at present, the temperature of the mixed waters is 38 °C). The results of these studies and recent situations concerning the circulation, quality, and discharge of the mine water have highlighted the effects of the controlled water rebound and the com-

plex measures necessary to stop dewatering safely. Water rebound is being analysed on the basis of available historical and recent data, both for its hydrodynamic influence and the consequences it will have on restoration of water resources. Different uses and possible actions are proposed, taking into account thermal, chemical, and discharge characteristics, as well as local planning objectives. A recent study (Garzonio and Affuso 2004) described all these aspects and highlights the great importance of continuously monitoring the water level and water chemistry changes, both during and after completion of mining (Figure 24). It stresses the need to install monitoring wells and automatically monitor the water levels, to perform new geophysical tests when the water table rebounds, and new chemical and isotopic analyses, pumping separately and contemporaneously in different pumping systems. This geogostic campaign started during the summer of 2004.

Many rehabilitation projects will be supported by Italy's national "watershed planning" law and the regional environmental departments that are now responsible for water resource management planning.

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Serbia and Montenegro

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Historical Situation

The exploitation of mineral resources, including copper, lead, zinc, nickel, bauxite, gold, silver, various decorative stones, lignite, coal, oil, and natural gas, is one of the oldest industries in Serbia and Montenegro. The diverse mineral reserve, especially in the territories of Eastern Serbia, Central Serbia, and Northern Montenegro, has resulted in the mining of a number of minerals, and manufacturing their primary products has increased several fold in the past few decades (Sarajevo Center for Environmentally Sustainable Development 2003).

Copper ore deposits occur as porphyry copper and massive sulphide types, predominantly in the East Serbian sector of the Carpatho-Balkanides (the Bor metallogenic zone). Traces of very old mining and metallurgy are still visible at many locations; the oldest copper mine, from the Early Neolithic Period (about 4500 B.C.), is located in Rudna Glava. This

mine was exploited until 1960 (Mining and Metallurgy Museum Bor 2004).

Archaeological research in Lepenski Vir, Vinca, Rudna Glava, and elsewhere have also revealed evidence of prehistoric mining of lead and silver at the Crveni Breg mine and mercury in the Šuplja Stena mine. Roman period mining took place in many locations and almost all lead and zinc deposits were exploited during the 13th and 14th centuries. In the 14th century, some mining settlements in Serbia and Montenegro were among the largest towns in Europe, e.g. Novo Brdo, which had 28,000 inhabitants at that time (Jović 2002).

Apart from significant concentrations of gold in the Bor metallogenic zone, other potential areas for gold have been found in Serbia, such as the volcanic complex of Lece, where gold is associated with hydrothermal vein type lead-zinc-copper deposits and several prospective areas with volcanic hosted gold mineralization. The most significant area for lead-zinc ore is the Kopaonik metallogenic district. Mineral deposits are of skarn, volcanic replacement, and vein type. Additional lead and zinc deposits and occurrences are located in the Ljubisnje and Bjelasica regions, with associated gold, silver, copper, bismuth and cadmium. Deposits of bauxite are located in the Niksic area. Red bauxite, together with coal, is a strategic mineral raw material in Montenegro. Silica, with mineral potential

of quartz sands, is located in the Sava, Danube, Morava and other riverbeds in the Serbia and Montenegro (Jović 2002).

The activities and livelihoods of villagers settled along the riverbanks are limited in mining zones. Local communities have rebelled in the past, such as in Bor in 1934, 1935, and in 1936, when the residents living around the Bor copper mine protested the degradation of the environment, vegetation, and crops in the area (Mining and Metallurgy Museum Bor 2004). Today's mines are also accompanied by numerous ecological problems, both current and those inherited from past exploitation.

Overview of Currently Working Mines

Copper mines are located in the Bor basin (Bor, Breznik, Novo Okno, Veliki Krivelj, and Majdanpek). Bor and Majdanpek mines annually discharge 395,000 t of sulphide concentrate and 24,000,000 t of flotation waste, seriously polluting the water and soil. Annually, 22 million m³ of acid mine water is discharged into local rivers, elevating levels of Cu, S, As, Fe, Mn, Pb, Ni, and Co. Mine water flows uncontrolled from abandoned exploitation fields and associated tailing dams and flotation tailing lakes on the riverbanks into ground and surface waters. As a consequence of flooding of the Bor and Veliki Timok rivers, over 20,000 ha of very fertile agricultural soils are degraded and contaminated.

The annual production of lead-zinc ore in the Rudnik, Blagodat (Vranje), Trepca, Ajvalija, Kižnica and Šuplja Stena (Kosovo), Veliki Majdan – Ljubovija and Mojkovac (Crna Gora) mines is about 130,000 t; more than 2,500,000 t of tailing waste are produced at the same time. Acid mine drainage containing SO₄²⁻, Pb, Zn, Cd, In, Fe, As, Sb, and Bi threatens the Ibar River and tributaries of Morava, Drina Cehotina, and Tara Rivers (part of the Durmitor National Park). Near the Trepća industrial complex in Mitrovice, the soil concentration of soluble lead was found to be 57–720 mg/kg; the level normally considered acceptable for crops is 20 mg/kg of soluble lead. Soluble zinc in the same soil ranged from 100–3,500 mg/kg; limit values for soluble zinc are in the range of 250–400 mg/kg.

Bauxite ore is produced in the Niksic surface mines where ore processing wastewater ponds and tailing dumps endanger the environment. The red mud, a by-product of aluminum production, is discharged in large ponds with inadequate liners, allowing residual sodium to leach into the soil and groundwater.

Pressure on the environment posed by about 150 coal mines, is reflected in soil and surface and ground water

pollution, hydrogeological changes over a wide area, soil degradation, air pollution, etc. The most important locations are the Kosovo coal basin (1,080 ha), Kolubara basin (open pits), Kostolac basin, Berane and Pljevlja, and small scale coal mines with underground exploitation in Eastern Serbia. Kostolac (1,479 ha) and Kolubara mines (3,481 ha) are the most critical as their open pits and waste dumps, including ash deposits from thermal power plants, are major sources of air pollution (flying ash, SO₂, Co, Ni, As, Cd, Pb, Cu, and Zn).

Despite the abundant resources, the mining industry is in a bad state, due mainly to outdated and obsolete technology, the poor economic situation in the country, and low market prices for metals. The last decade of the past century was marked by decreased production and a consequent drop in economical development. Without sufficient funds for maintenance, the mining infrastructure started to deteriorate, unemployment rate rapidly increased, and poverty has increased. Many lead and zinc mines in Serbia and Montenegro, with the exception of the Rudnik Gornji Milanovac and the Grot Vranje mines, are not in operation. During 2001 and 2002, production practically ceased in the Lece-Medveda, Veliki Majdan-Ljubovija and Suva Ruda-Raška mines; the Šuplja Stijena mine stopped operating even earlier. The zinc and lead mines in the Kosovo and Metohija territory ended their production after the bombing campaign in 1999, with the exception of the Leposavicev mine, which is barely operating. Some of these mines are partly flooded (Jović 2002).

Mine Water Management and Problems

The fact that Serbia and Montenegro is very rich with sulphide minerals shows the potential for mine water pollution of the environment, especially with heavy metals. Mining areas are significantly degraded and contaminated. The pollution is the combined result of mining and natural leaching from ore bodies, and has a negative effect on health and the quality of life of local residents. The most difficult situation is in the vicinity of the Bor, Krivelj, Majdanpek, Mojkovac, Trepća mines, where most of the pollution is caused by ore processing plants and release of wastewater from flotation plants into nearby rivers and/or tailing lakes with inadequately constructed protection dams. Several dam failure incidents have occurred at the Majdanpek, Veliki Majdan, and Brskovo mines, directly affecting the Pek, Drina, and Tara rivers (Jović 2002). Mine water from ten mines (the Kižnica and Novo Brdo mines, Kosovo coal basin, Trepća lead and zinc mines, Suva Ruda magnetite, lead and zinc mines, Bijela Stijena magnesite mines, Korlace asbestos mines, Ibar coal mines, boron ore deposits, and the Bogutovac magnesite mines) degrades the quality of the Ibar

River watershed to “out of class”. After heavy rainfall, the Kraljevo water supply system downstream is contaminated with high concentrations of phenols from the Obilići power plant, and with lead and zinc drained from Ajvalija and Kišnica lead and zinc mine. Due to the 33 million t of pyritic wastes disposed along the Ibar bank in Kosovska Mitrovica and Zvečane, groundwater in this area has been contaminated with high levels of lead and zinc for decades.

Many of the historical and new mines in Serbia and Montenegro represent a regional problem as well, posing a threat to the environment of neighbouring countries and international watercourses. The mine water generated from the Bor mine flows into the Bor, Krivelj, Brestovac and Ravna rivers and endangers all communities downstream, including the Timok river bordering with Bulgaria. The characteristics of discharged mining waters include a high content of suspended substances, such as copper and other heavy metals, and low pH. Similarly, metallurgical waters formed in the sulphuric acid plant and tank house are acid and contain harmful elements (Cu, As, Pb, Zn). These wastewaters are released into a reception lake and further to the Borska, Timok, and Danube rivers, without treatment. About 300–500 t of sulphuric acid, 300–350 t of As, 30–100 t of Pb, and 10–35 t of Zn are released annually from the RTB Bor operations (Jović 2002). Therefore, the Danube River is constantly threatened, since it is the final recipient of water contaminated with large quantities of arsenic, heavy metals, and sulphates (Sarajevo Center for Environmentally Sustainable Development 2003).

Unclear and incomplete regulations are an important aspect of mine water management. Since only water

generated during ore separation is legally recognized as mine water, prevention of environmental pollution is generally limited to end-of-pipe treatment of process wastewaters. All other wastewaters generated during the mining cycle can be and generally are discharged into the environment without treatment. In addition, process wastewater treatment requirements are limited, emphasizing removal of suspended matter, without involving any of the advanced methodologies of active or passive treatment.

Compounding this issue are inadequate control over legal provisions and overlapping jurisdiction of different state sectors e.g. for mining, environment, water, soil, health. Serbia and Montenegro needs to adopt systematic environmental management regulations, and should also address the environmental consequences of pollution caused by mining activities. This law should contain measures for spatial and economic development of a wide area around the mine.

Future Mining, Treatment, or Remediation

Bearing in mind the present situation in Serbia and Montenegro and the scope and complexity of steps needed to clean up the polluted areas and remediate the degraded land, financial and technical support from the international community is needed. There is a lack of knowledge on advanced methods for mine water treatment (such as passive treatment); closer cooperation and joint research projects with international institutions and universities that would enable transfer of know-how are crucial.

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Bosnia and Herzegovina

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This section is based on the European Union's R&D ERMITE project (Kupusović et al. 2001).

Historical Background

In 1844, an expedition of French geologists visited Bosnia and Herzegovina, which at that time was a European component of the Ottoman Empire. With their work and geological maps, they located various mineral deposits. The area is richly endowed with natural resources, including extensive and varied mineral-

ization, some of which can generate acid rock drainage (ARD), causing contamination of nearby vegetation, soil and water.

There are 110 sites where coal is deposited in the territory of Bosnia and Herzegovina. Bauxite mines are located at four different locations: in the Herzegovina region, in the North-Bosanska Krajina region, in the Central-Jajce-Banja Luka region, and in eastern Bosnia-Milići-Zvornik. In northeast Bosnia, from Srebrenica to Maglaj, Zavidovici, and Teslić, there are breakthroughs of tertiary igneous rocks that are associated with occurrences of Zn, Pb, Sb, Ag Bi, As, Fe, and other minerals. This area, rich with iron ore, zinc, sulphide minerals, and non-metal minerals has attracted geologists for many years. Iron ore production was concentrated in Jablanica and Vares, while production of manganese ore took place in Bosanska Krupa. The Vares iron ore mine in central Bosnia is one of the old-

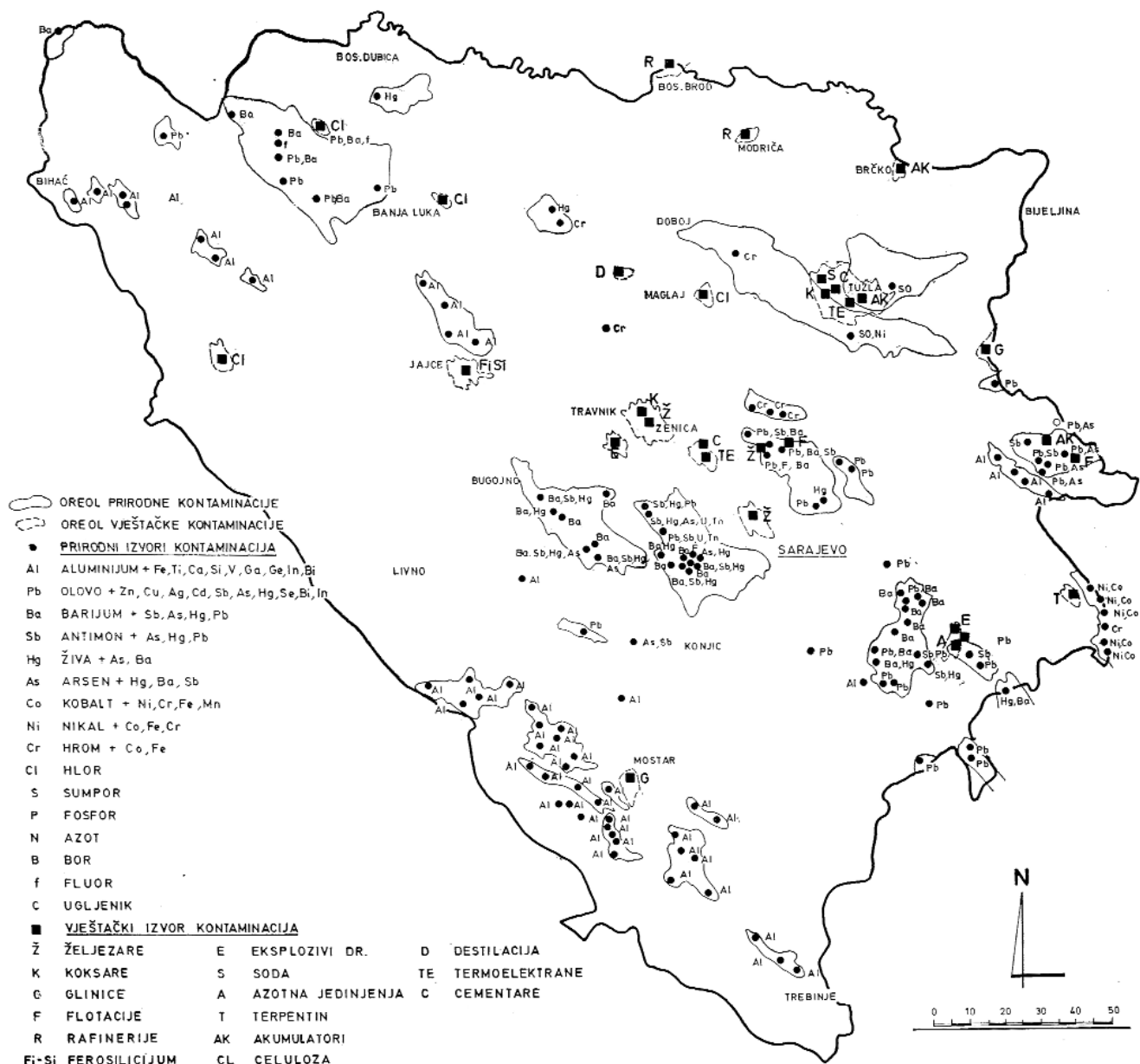


Figure 25. Areas of natural or industrial contamination in Bosnia and Herzegovina (after Kubat 1987)

est mines in Bosnia and Herzegovina, and dates back to at least 1692. Among the non-metal mines, the largest and most important for the Bosnia and Herzegovina economy are salt deposits, which are located in the Tuzla region.

Although most of the mining areas are associated with sulphide minerals that have the potential to generate ARD, little attention has been given to contaminated or acidic mine water in Bosnia and Herzegovina. The environmental aspects of mine water have been neglected relative to the attention paid to mine safety.

Overview of Currently Working Mines

A poor post-war economy and the complex political situation in the Balkan region have affected the mining

industry in Bosnia and Herzegovina, causing it to operate at minimal capacity. Mines in Bosnia and Herzegovina cannot expect to be competitive on the foreign markets due to outdated technology and an inadequate transportation infrastructure. At present, and for the near future, ore utilization will be dependent on the domestic market.

Generation of electrical energy in thermal power plants consumes about 80% of the coal produced and thus drives the industry. Mining is needed to support the state energy system, and therefore the state itself, but the state invests little in the development of the mining sector.

Nearly all metal mines (e.g. iron, copper, lead, zinc, silver) have been closed, but the majority of industrial-



Figure 26. The Srebrenica Spring, Veliki Guber, and the Kiseljica River (Midžić 2002)

mineral mines (e.g. sand, gravel, stone, calcium carbonate, slate, clays, gypsum, salt, barite, shale) are still active. Only the Srebrenica lead and zinc mine and Posušje and Milići bauxite mines remain in production, and they are being operated at approximately 10% of their capacities. None of the closed sites have been rehabilitated. Some of the abandoned bauxite sites are now used as uncontrolled landfill sites for disposal of communal wastes. A naturally formed lake at the Vares abandoned open pit iron mine is now used for fish farming and recreational purposes. It is the same with the abandoned bauxite mine at Mrkonjic.

Mine Water Management and Problems

Environmental problems related to active and abandoned mines and their mine waters are serious threats to the aquatic environment and require special attention. Figure 25 indicates areas of natural (acid rock drainage) and industrial (e.g. heavy metal factories, thermal-power plants, mineral processing) contamination in Bosnia and Herzegovina contaminated with lead, mercury, arsenic, antimony, barium, aluminium, and compounds such as sulphuric acid. The following areas of sulphide mineralization are associated with ARD:

- Mrkonjić Grad – Jajce-Prozor – Konjic: Fe, Mn, Ba, Cu, As, Hg

- V. Kladuša – Sanski Most: Mn-Pb, Ba
- Borovica-Vareš – Čevljanovići – Srednje: Pb, Zn, Ba, Cu, Au, Ag, Hg
- Olovo – Kaljina – Podromanija: Pb, Mn
- Jugoistična Bosna (Foča-Prača): Sb, Cu, Pb, Zn, Ba, Mn
- Šekovići – Drinjača – Kravica: Fe, Mn, Cu, Pb
- Srebrenica: Pb, Zn, Cu, Ag, Au, Sn, V, Bi, In, Hg, Sb, Mn

One of the most interesting is the Srebrenica lead and zinc mine, which is located in permeable porous media; part of the deposit lies above and part below the piezometric ground water level. In an area of 100 km², there are 120 mineral springs, of which 50 are highly mineralised. Most of these springs are near old mine shafts. The high precipitation, high porosity of the watershed, and number of old mine shafts that accumulate precipitation constantly recharge the capacity of these mineral springs.

Unfortunately, few studies have been done to assess the impact of the mining industry on water resources in Bosnia and Herzegovina. Due to this lack of systematic monitoring of mine water quality and quantity, the exact type and degree of mine water pollution is unknown. There has been very little research on the effect of mine drainage on the environment or potential pollution abatement measures.

The nation's legal framework mainly addresses wastewaters generated during ore beneficiation, including leachate from tailings (Midžić 2001). There are no laws, strategies, or activities addressing the environmental problems of abandoned mine sites. In very few cases, legal obligations related to land recultivation are being implemented. There are no other legal obligations on the mining companies concerning mine water pollution control or decontamination of closed mine sites, and no institution responsible for control of the environmental impact of abandoned mines. The dispersion of responsibilities in the field of water and environment in different ministries and sectors is an additional problem when attempting to address issues of mine waters and their impact on the environment.

Future Mining, Treatment or Remediation Activities

Current market conditions only favour coal extraction for thermal power plants. Since private investors have not shown interest in metal mining, it is evident that the state, as the owner of the abandoned mines, should

Figure 27. Major abandoned mines in Portugal (adapted from Oliveira et al. 2002). ★: uranium mines.

of which are seriously degraded and contaminated (Figure 27), and large volumes of old mining residues (Table 16), some of which have significant environmental impact. Of 85 abandoned mines (not including the uranium mines nor the most problematic ones), studied by Oliveira et al. (2002), 14% were found to generate acid mine drainage (AMD) and/or to pose a high degree of environmental risk.

Tailings and waste dumps can have a chemical and radiological impact on a local or regional scale. The fact that much of Portugal's mineralization is associated with sulphide minerals, along with in situ acid leaching of uranium, provides great potential for the production of AMD. Examples include polymetallic sulphides mines in the Iberian sulphide province (e.g. S. Domingos, Aljustrel, Caveira, Lousal), uranium mines (e.g. Urgeiriça, Cunha Baixa, Quinta Bispo, Vale de Abrutiga), W and/or Sn mines (e.g. Covas, Montesinho, Vale das Gatas, Tuela, Murços, etc), Au and/or Ag mines (e.g. Castromil, Jales, Penedono, Freixedo, Terramonte), and Cu mines (e.g. Miguel Vacas). The drainage of waste piles and fine grained tailings can be particularly problematic (e.g. Montesinho, Jales, Argoselo, Vale das Gatas, Terramonte, Adoria, Lousal, Caveira).

The main geochemical signatures with environmental importance are: W ores: As-Cu-Mo-Pb; Sn ores: As-Be-(Cu); W-Sn ores: As-Zn-Cd; Au-Ag ores: Pb-As-Zn-Cd-(Ag); Pb-Zn ores: Pb-Zn-Cd-(As)-(Ag); Fe ores: Fe-(P)-(V)-(Mn); coal: (As)-(Fe); sulphide ores: Cu-Pb-Zn-Fe-As-Sb; Mn ores: Mn-Ba-Fe; and U ores: U, ^{226}Ra , Cl, F, SO_4^{2-} , Ca, Mn, Fe (Oliveira et al. 1999)

In terms of environmental risk, the mines are ordered as follows: uranium, massive sulphides (Iberian sulphide province), tin and/or wolfram and/or gold (often accompanied by non-dominant sulphides), and coal (Costa and Leite 2000). Mines that had used chemical methods for ore treatment were more likely to have extensive (in terms of impact or duration) pollution problems (Oliveira et al. 1999; Costa and Leite 2000).

Portugal was particularly rich in uranium mines (Table 16). The U-mineralization was generally associated with sulphide minerals. Up to 2000, approximately 4370 t of U_3O_8 were produced, in addition to the production of radium salts, resulting in about $13 \cdot 10^6$ t of mining residues (Nero et al. 2003). At many old mining sites, low grade ores, waste rock, tailings dams, leaching pads, and sludge from effluent treatment still exist. In the Vale de Abrutiga U-mine, located close to the Aguieira dam reservoir, a lake was formed in the open pit. Acid water from the low grade ore, waste rock, and pit lake flows directly to the reservoir. Surface waters draining the mine site have a $\text{pH} \approx 2.6$,

high conductivity (up to $6000 \mu\text{S cm}^{-1}$), high concentrations of U, SO_4^{2-} , Zn, Fe, Mn, Ra, Cu, Th, and Pb; the contaminant concentrations are higher in winter than in summer (Pinto et al. 2004). The surface water from the reservoir, close to the confluence of the contaminated streams, and the groundwater cannot be used for human consumption or irrigation. Stream sediments have high geoaccumulation indices for U, Fe, Ag, Zn, Cr, Co, and Pb (Pinto et al. 2004). The Urgeiriça U-mine was mined by in situ acid leaching to recover U from the low grade ore. This greatly increased the level of U and Ra contamination, especially of groundwater (Pereira et al. 2004). Surface and ground water have Ra contents above maximum permitted values, according to Portuguese law. Mine drainage water has high levels of U, ^{226}Ra , SO_4^{2-} , Al, Mn, Ni, Zn, Be, Co, Cu, and Cr. The soils and stream sediments from the main water-course show high average radionuclide activity (Pereira et al. 2004). However, the uranium mining complex of Urgeiriça does not greatly affect radon concentrations (Neves et al. 2003). In the Cunha Baixa U-mine, acid leaching was also used. Surface water has high levels of U, Mn, Zn, Al, and Sr. Elevated levels of U were found in alluvium and stream sediments up to 10 km downstream (Oliveira et al. 2002).

The Caveira and Lousal mines are located in the Iberian sulphide province. The mineralization consists of pyrite, chalcopyrite, galena, sphalerite, and other sulphides, with Au as one of the secondary minerals. In the Caveira mine, the tailings, waste dump, and an acid lake continue to pollute an area extending 15 km downstream from the mine site. Surface water has high concentrations of Cu, Pb, Zn, As, Mo, Se, Cd, and Fe; stream sediments show anomalies of Pb, As, Sb, Hg, Cu, and Zn; soils are contaminated with Mo, Pb, Ag, As, Hg, Cd, Sb, Bi, and S (Silva et al. 2003a). The S. Domingos pyrite mine, inactive since 1966, exploited Cu, Pb, Zn, S, Fe, Ag, and Au. The open pit is flooded with acidic water ($\text{pH} \approx 1.7$). The interaction of meteoric water with the tailings and permeable slag material also generates AMD (Quental 2001). Water, sediments, and soils samples from this mine site should be considered as hazardous materials (Macalady 2003).

Castromil is an abandoned gold mine in NW Portugal that left a large amount of tailings and sulphide-containing (mainly pyrite and arsenopyrite) wastes. The soils have high Pb and As content (Reis et al. 2003; Silva et al. 2003b), and a pH of about 4 was found in a well in the vicinity of the mine (Pinto 2001). The Penedono Au-mine occurs in a similar situation to that of Castromil. The waste rock, ≈ 1 million tons (Matias et al. 2003), has high levels of As, Pb, Zn, Cu, Mn, and Bi; the mine waters are acid; the surface waters have high contents of As, Cu, Zn, Cd, Mn, Al, and sulphate, but the groundwater is not polluted (Matias et al. 2003).

Argoselo was mined for Sn-W in the north of Portugal. The mine tailings (≈ 1 million tons) have significant levels of As, Cu, Zn, W, Fe, and Cd, and primary sulphides (arsenopyrite, pyrite, chalcopyrite, sphalerite), and supergenic minerals (arsenates, sulphates, and oxides; Oliveira and Ávila 2003). The mine waters are acidic ($\text{pH} \approx -3.7$) and rich in Mn, Al, Cu, Pb, Zn, Cd, Ni, Co, and Be. Stream sediments and soils near the old mine site are contaminated with As, Cu, and Zn (Oliveira and Ávila 2003). At the Segura mining area (44 small mines), Sn, W, Ba, and Pb were exploited until 1953. No significant AMD was associated with the old mine workings, but the waters associated with mineralized veins have high levels of As, Fe, and Mn and should not be used for human consumption or agriculture activities. Soils must not be used for agriculture or residences due to their Sn, B, As, and Ba concentrations, while stream sediments show Sn, W, B, As, Cu, Ba, Pb, and Zn anomalies (Antunes et al. 2002).

The Pejão mine site is a recently abandoned coal mine, with many waste dumps dispersed nearby. Mine water and waste drainage is acidic, with higher levels of water contamination in winter than in summer. However, the contamination is not strong and is localized, because the wastes are poor in heavy metals and coarse-grained, which prevents the acid leaching of metals. Only Fe occurs at truly anomalous levels (Oliveira et al. 1999).

Remediation Legislation and Activities

Decree-Law 198-A/2001 establishes the most important principles and objectives concerning the remediation and monitoring of old degraded mine sites. Its fundamental purpose is the environmental rehabilitation of mining areas and its specific objectives are the following:

- To eliminate the risk factors that constitute a threat to public health and safety, and which result from water pollution, soil contamination and possible existence of unstable waste dumps or unprotected cavities;
- To rehabilitate the landscape environment and the natural development conditions of the local fauna and flora, with reference to the respective habitats previous to the explorations;
- Identify the patrimony abandoned by the old explorations, whenever this represents a significant economic relevance or a testimony of industrial archaeology;
- To create conditions for the economic valuation of the recovered areas based on their specific potential

Table 16. Portugal's abandoned mines: numbers (No) and notable examples of each type and volume of resultant residues (extracted from Nero 2004a with some modifications).

Type	Nº of mines	More important examples	Wastes ($1 \cdot 10^6 \text{ m}^3$)
Radioactive Ore	61	Urgeiriça, Quinta Bispo, Cunha Baixa, Vale de Abrutiga, Castelejo, Bica	8.52
Polymetallic	10	São Domingos, Aljustrel, Lousal, Caveira	6.57
Sn and W	40	Argoselo, Covas, Montesinho	1.44
Basic metals	28	Terramonte, Coval da Mó, Miguel Vacas	0.99
Fe and Mn	16	Orada, Cercal/Rosalgar, Ferragudo	0.61
Coal	3	São Pedro da Cova, Pejão	2.2
Au	12	Jales, Castromil, Penedono, Freixeda	1.27
Other	5	Gouveia de Baixo, Cortes Pereira	0.67
Totals	175		≈ 22

in each particular case, namely for agricultural and forestal use, promotion of tourism and culture, besides other kinds of uses found adequate and convenient;

- To ensure a good application of the financial resources to use in the Programme, through the maximization of the binomial social benefits/costs, namely in terms of the economy and efficiency of the solutions to adopt.

This Decree-Law also assigns the responsibility for the reclamation of abandoned mine sites for a period of ten years to the EXMIN – Industrial Co. of Environmental and Mining Services, S.A. Priority has been given to the radioactive and sulphide mines. Some old mines (Jales, Au and Argoselo, Sn-W) are already remediated and many others (e.g. Vale de Abrutiga, U, Urgeiriça, U, Cunha Baixa, U) are now being reclaimed, using methods and techniques such as slope stabilisation, confinements, agronomic restitution, compaction of residues, surface impermeabilization, inertization or inhibition using natural or biochemical materials and removal and storing in open pits or underground cavities (under aerobic or anaerobic conditions; Nero et al. 2003; Nero 2004 a, b). Tailings and sludge from several U mines are being removed and confined to a few existent open pits, reducing the contaminated areas to about 50%, and their acid waters are treated with strong bases. Post remediation monitoring is being carried out on slopes stability and subsidence levels,

waters, soils, and air in the neighbourhood of the mining areas.

Impact evaluation of water is done according to Decree-Law No. 236/98. The limits for radioactive waste exposures are described in Regulamentar Decree No. 34/92. In Portugal, there is no legislation to evaluate environmental impacts on soils and stream sediments; it is therefore common to estimate the impacts on soils by using the Canadian Norms (Canadian Council of Ministers of the Environment 1991) or Dutch Norms (Swartjes 1999).

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Macedonia

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Historical Situation

Mining is one of the most important industries in Macedonia, employing a large number of workers



Figure 28. River Kiselica polluted by acid mine waters (source: Sarajevo Centre for Environmentally Sustainable Development 2003).

throughout the country. The former Yugoslav Republic is well endowed with mineral deposits necessary for the production of copper, iron, lead, precious metals, and zinc. Industrial minerals such as bentonite, feldspar, gypsum, sand and gravel, and stone (carbonate and silicate), as well as cement and other construction materials that are based on quarried products, are produced mainly for export. The most significant mineral deposits are lead and zinc ore, the exploitation of which is carried out in extraction, smelting, and metal-processing industrial plants. Lead and zinc mines (Zletovo in Probistip, Toranica in Kriva Palanka, and Sasa in Makedonska Kamenica) are located in the northeastern part of Macedonia, close to the Bulgarian border (Privatisation Agency of the Republic of Macedonia 2003). Contamination of water and soil by heavy metals has been observed near the Probistip lead and zinc mine since the area was first mined during the Roman period. Mine water has polluted the Kiselica (Acid; Figure 28) and Koritnica Rivers, which flow into the Zletovica River, which flows through Bregalnica and Vardar to the Aegean Sea, thus creating transboundary pollution problems (Sarajevo Center for Environmentally Sustainable Development 2003).

Although these issues have been identified, little has been done to prevent pollution and/or decontaminate the sites. Mine safety has been taken very seriously, but environmental aspects have largely been ignored.

Overview of Currently Working Mines

Most of the mines in Macedonia have been permanently closed. This was due to a lack of financial resources, which limited the possibility of marketing the mineral resources, a high level of amortization of equipment and facilities, inadequately equipped and outdated maintenance capacities, and the practice of hiring a surplus work force for the level of production. Although the country's extraction capacity was high, the industry's output dropped due to the loss of former USSR and ex-Yugoslavian markets.

Three Macedonian lead and zinc mining companies (Zletovo, Toranica, and Sasa) are currently in liquidation (Privatisation Agency of the Republic of Macedonia 2003). During their operation and later, after closure, these mines have had significant environmental impact. Lack of care recently resulted in the collapse of the drainage culvert beneath the Sasa mine tailing dump. The culvert was supposed to keep the unpolluted river water from interacting with the metal-rich waste dump. However, the collapse left a crater-like hole, 170 m wide and nearly 50 m deep. Approximately 4 million t of waste were carried downstream in the steep-sided valleys, eventually settling 12 km away in a reservoir (a local source of fish). Preliminary work has shown that the grey waste deposited all the way down the valley contains 0.5% lead and 0.5% zinc (Alderton 2003).

Coal mining is only carried out at four mines: Suvodol, Oslomej, Brik, and Priskupstina. The mineral deposits at the Suvodol mine, from which three blocks of the REK Bitola thermal power plants are supplied, are estimated at approximately 120 million t. Exploitation there has already disturbed the natural balance and landscape in this area. In general, due to neglect, very little or no remediation is done after exploitation is completed (Regional Environmental Center for Central and Eastern Europe 2000).

Mine Water Management and Problems

While there is a lack of mine water management, Macedonia systematically monitors water bodies in the mining areas in a manner that exceeds that in other more developed Balkan countries. Thus, it is well documented that the water quality is particularly unsatisfactory in the central and lower Vardar River, and the Pcinja, Bregalnica, and Crna Rivers. Analytical data from regularly monitored streams near the Veles lead and zinc mine indicate that its effluent into the Vardar River consistently exceeds maximum concentration levels for lead, zinc, and cadmium. The National Environmental Action Plan reported that cadmium, lead and zinc levels were 10–15 times higher in vegetables grown in Veles than in control regions. Poor ground water quality is observed near Skopje, and especially in Veles. Data collected through regular monitoring of wells in the valley downstream of the Radovis copper mine tailings dam indicate that copper concentrations are in the range of 50–200 mg/L (Center for Environmentally Sustainable Development Sarajevo 2003). Unfortunately, there are many such examples of environmental issues caused by mining activities.

The metals in Rudnici Zletovo mines in Probištip are extracted from mines 2.5 km north of the company's concentration plant. Approximately $1 \cdot 10^6$ m³/yr of

wastewater contaminated with heavy metals and cyanide was pumped from the concentration plant into the Koritnico and Kiselica Rivers without cleaning or neutralization. According to the company's observations and analyses, there is little life in the rivers and high levels of heavy metals have been found in fish and other biological samples. The mining waste contains zinc, lead, cadmium, and cyanide, and is deposited at two different sites. The first is an old hydro-tailings area built on top of the Kiselica River on the outskirts of Probištip. The second is a new hydro-tailings sedimentation basin and dam built in a valley close to Probištip. This valley also drains to the Kiselica River, which, in turn, flows into the Zletovska River, a tributary of the Vardar (UNEP 2000).

The closure of many mines did not improve the environmental situation in the country, since natural drainage of mine waters has continued and will continue to pollute surrounding water and land resources (Privatisation Agency of the Republic of Macedonia 2003; Regional Environmental Center for Central and Eastern Europe 2000). Macedonian legislation does not provide a regulatory framework for addressing these issues and assigning responsibilities. Only the management of wastewaters generated in the process of mineral ore separation and tailing wastewater in active mines is regulated. There are no specific provisions concerning mine water pollution control. There are also no legal provisions or policies addressing environmental issues at abandoned mining sites. In very few cases are legal obligations being implemented relative to land recultivation.

The care for the environment that has been degraded by mining activities is left to the Macedonian Government, which currently does not implement any program for control of pollution from the mining activities or abandoned sites. The bankrupt mining companies, which will be sold on the international market, will have new owners who will most probably be considered responsible for the environmental impacts of mine water and mining activities in general. However, for the time being and the near future, there is a lack of ownership and responsibility for issues related to pollution from mine waters.

Future Mining, Treatment, or Remediation

The Macedonian Government has initiated a program of reform for the enterprise sector, which includes a commitment to privatise 22 loss-making enterprises in which the State has significant interest. The Government is offering a number of incentives to potential purchasers to encourage them to invest in these companies. Of these 22 loss-making companies, four are mining or smelting businesses.

Much has to be done in the field of institutional strengthening and education/training:

1. The existing legislation must be improved and upgraded to the European level and their consequent enforcement through regular compliance monitoring should be ensured.
2. An integrated river basin management approach should be applied to efficiently address water pollution problems caused by mine waters; transboundary and multi-sector cooperation will be necessary to overcome the numerous existing environmental problems.
3. An education/training curriculum should be designed for all levels of public authorities, focusing on capacity building for reporting, hazards and emergencies response, regulations, risks, etc.
4. The mining industry needs to be encouraged to use the best available technologies for ore processing,

wastewater and mine water treatment, and remediation of pollution caused by mining activities. In order for the mining industry to be able to apply these technologies and achieve environmentally sustainable mining, training workshops, sharing of good (and bad) practices, site visits, and technology transfer are necessary.

Apart from the above mentioned activities, special attention also needs to be paid to raising public awareness on environmental and health risks resulting from mining activities and ensuring public participation in environmental impact analyses for future investments in the mining industry, and in drafting regulations, plans, policies related to regulation of mine water and mining activities, as well as their impact on the environment.

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Active Treatment of Mine Water: A European Perspective

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Introduction

The selection of the most appropriate method for treating mine water is normally made on the basis of a combination of technical and economic considerations, but social considerations can also be important. The final decision is normally based on the robustness of the treatment process and whole life cost (capital, opera-

tional, and decommissioning costs). For projects with a relatively low metal load (flow × concentration) passive treatment is the preferred option and has been extensively used throughout Europe (for a review of the processes available and the associated design considerations, see the Final Report of the PIRAMID project, PIRAMID Consortium 2003). Active treatment is generally only adopted for temporary schemes, for waters that are difficult to treat passively, or where other considerations, such as land availability, prevent the use of passive treatment.

Most active mine water treatment processes remove metals by rendering them insoluble using either a commonly available chemical (e.g. Ca(OH)₂) or by making use of the chemicals contained in the mine water (e.g. sulphate). In addition, there are a number of proprietary reagents available, but these are generally not widely used due to their relatively high unit costs, con-

Table 17. Commonly used precipitation reagents, theoretical doses and costs in British pounds

Reagent		Unit Cost	Theoretical consumption		Actual consumption		
		£/tonne	kg per kg Fe		kg per kg Fe		
			Dose	Cost	Efficiency	Dose	Cost
Calcium Oxide	CaO	100	1.00	10p	65%	1.54	15p
Calcium Hydroxide	Ca(OH) ₂	100	1.33	13p	65%	2.05	21p
Magnesia	MgO	220	0.72	16p	80%	0.9	20p
Magnesium hydroxide	Mg(OH) ₂	260	1.04	27p	80%	1.3	34p
Sodium hydroxide	NaOH	260	1.433	37p	95%	1.50	39p
Sodium Carbonate	Na ₂ CO ₃	150	1.89	28p	95%	2.00	30p

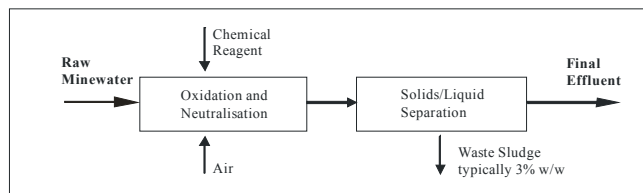


Figure 29. Conventional precipitation plant

cerns about sole source availability or lack of acceptance by the technical community. In Europe, unlike more arid areas such as South Africa and Arizona for example, the focus is on metal removal; less attention is paid to more complex considerations such as sulphate removal discussed by Maree et al. (2004).

Conventional Precipitation

The best available technique for the treatment of mine waters with a high metal loading is chemical precipitation. Although a number of different reagents are readily available, precipitation is most frequently achieved by alkali addition. Typically treatment is undertaken in two stages (Figure 29), with the metals rendered insoluble and then separated from the treated water in a clarifier, or even in large lagoons if space and planning regulations permit. The optimum pH required for minimum metal solubility varies with the metal species and type of precipitation process and the designer is therefore faced with either adopting multi-stage treatment, with each metal removed at its optimum pH, or by compromising on a single pH, which enables the effluent concentrations of all the target metals to be achieved. For most mine waters, a satisfactory effluent quality can be achieved by operating at a single pH; therefore, multistage removal is not frequently used.

Table 17 lists the commonly used neutralising reagents together with their estimated dosage costs at typical unit costs of the raw materials. Although alkali precipitation is effective at removing dissolved metals from solution, the resultant precipitate is gelatinous and difficult to dewater. Low settled sludge concentrations of between 2 and 5% solids are frequently achieved on separating the solids from the treated water using a conventional clarifier/thickener or lagoons. This sludge density can be increased to between 20% and 35% (wt/wt) by dewatering via a filter press or centrifuge.

Iron is removed by oxidation from the ferrous to the ferric state, followed by precipitation from solution as a metal hydroxide. The primary benefits of iron oxidation are: ferric iron is insoluble at a relatively low pH (whereas the optimum pH for ferrous removal is 10.5), ferric hydroxide sludge tends to be more stable than ferrous sludge, and iron oxidation and precipitation

Table 18. Typical influent and discharge qualities at the Wheal Jane HDS plant

Parameter	Influent	Discharge
pH	3.8	8.6
Total suspended solids (mg/L)	4.5	7.0
Dissolved Cu (mg/L)	0.1	<0.01
Dissolved Fe (mg/L)	230	0.05
Dissolved Zn (mg/L)	46	<0.01
Dissolved Al (mg/L)	15	0.45
Dissolved Mn (mg/L)	6	0.05
Dissolved Cd (mg/L)	0.01	0.001
Dissolved Ni (mg/L)	0.4	<0.01
Dissolved As (mg/L)	10	<0.01

tends to encourage co-precipitation of other metals at moderate pH. Oxidation is most commonly achieved by the introduction of air into the reaction tanks through a sparge pipe. Hydrogen peroxide, chlorine, sodium hypochlorite, potassium permanganate, and ozone can all be used as alternative oxidizing reagents. The principal advantage of these alternatives is the speed of reaction and the ease of application. This can eliminate the need for a large reaction tank, and hence reduce capital costs. In comparison with conventional aeration, however, the reagent costs are significant and these reagents only tend to be used for applications with low metal loads.

There are numerous examples of such conventional plants across Europe. For example, at the abandoned St Salvy mine near Castres in southwest France (commissioned in 1997), lime neutralisation is used to remove Zn, Mn, Fe, and minor amounts of Cd and Pb from a mixture of mine water and ARD at the rate of 60 m³/hr. The plant feed concentrations at commissioning were typically 100 mg/L of Zn, 60 mg/L of Mn, and 15 mg/L of Fe at a pH just below neutral and with an alkalinity of about 100 mg/L as CaCO₃. It is interesting to note that the feed was aerated for 20 minutes prior to addition of lime as the neutralisation agent, a practice that is now being pursued at other active plants. After addition of milk of lime to a pH of 9.5 and reaction for 20 minutes, the resultant slurry was flocculated and fed to a lamella clarifier. The overflow meets all discharge consents (typically pH 9 and soluble metal concentrations below 1 mg/L); the underflow is stored prior to batch dewatering in a filter press. The consistency of the filter cake allows disposal via conventional skips.

At the abandoned Ynysarwed/Blaenant coal mine in South Wales, contaminated mine water was discharged to the Neath canal for several years, causing significant environmental damage, threatening an industrial water supply, and limiting recreational use of the canal. A

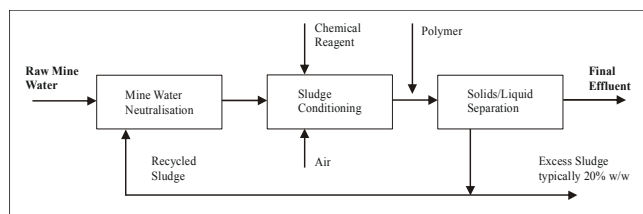


Figure 30. Typical high density sludge process configuration

consortium comprising the County Borough Council, the Environment Agency, the Coal Authority, the Welsh Office, and Neath Canal Navigation financed the construction of an active treatment plant and a passive wetland for final polishing at a total cost of £1.5 million. The plant was commissioned in 1998 and treats 100 m³/hr of water at pH 5.5 containing 240 mg/L of Fe and 6.5 mg/L of Mn. The plant consists of a combined aeration neutralisation tank with a residence time of 100 minutes, a lamella clarifier, and a decanting centrifuge for sludge dewatering to a consistency (typically 30% wt/wt) suitable for skip disposal. Lime (a calcium hydroxide slurry at 18% wt/wt) is used to control the neutralising pH at 8.9; the liquid discharged to the wetland typically contains 20 mg/L of suspended solids, 1 mg/L of dissolved Fe (about 2 mg/L of total Fe), and 1.3 mg/L of dissolved Mn.

High Density Sludge Process

The performance of a hydroxide precipitation plant can be improved by the introduction of sludge recirculation to produce high density sludge (HDS). This process encourages nucleation, with the metals precipitated onto the surface of previously created sludge particles. Solids concentrations of 15 to 25% (wt/wt) can be obtained from clarifier-thickener units operating at HDS plants. This sludge can be further dewatered to about 50% in a drying bed and to between 50 and 80% solids in a centrifuge or filter press. Although a number of HDS process schemes have been developed, they all rely on the use of multistage neutralisation and

the recirculation of a portion of the sludge. HDS can be formed by mixing the recirculated sludge with either the lime prior to introducing the mine water, or with the mine water prior to adding the lime. Figure 30 shows the HDS plant in the latter configuration and demonstrates the dramatic increase in solids concentration obtained in the thickener underflow compared to that obtained in the conventional process.

In addition to the reduction in sludge disposal costs, HDS settles at significantly greater velocities and is more easily thickened than conventional hydroxide precipitates. The enhanced sludge settling characteristics allows solid-liquid separation to be undertaken in a smaller thickener/clarifier resulting in a significant saving in capital cost. For example at Wheal Jane in Cornwall, UK, the use of a HDS process together with a lamella clarifier has allowed the foot print of the solids/liquid separation unit to be reduced to about 13% of that required by a conventional hydroxide precipitation plant. Although HDS plants have traditionally been employed for treating mine waters containing high iron, the process is also applicable to waters with high concentrations of zinc, copper, or aluminium. A recent study by Coulton et al. (2004) has demonstrated that it is not necessary for the mine water to contain any iron in order to achieve the high solids concentrations normally reported for HDS with high Fe contents.

The Wheal Jane water treatment plant is commonly regarded as the classical example of the HDS process in Europe and has been the subject of several publications (e.g. Coulton et al. 2002; Hallett 1999). Production at the Wheal Jane Mine was discontinued in 1991 and the mine was allowed to flood. This resulted in an uncontrolled release of mine water containing up to 5000 t of metalliferous sludge to the local estuary with disastrous environmental impacts. Following this release, a temporary conventional lime dosing plant was installed and studies were undertaken to determine the most appropriate long term solution. These studies concluded that using a lime-based HDS plant was the preferred option.

Table 19. Other technologies/proprietary chemicals

Process/Chemical	Description
Keeco KB1	Micro-silica encapsulation technology
Aquafix	Water power solid reagent dosing plant for use in remote locations without power
Virotec Bauxsol	Modified 'Red Mud' residues from bauxite refineries, providing alkaline solids capable of neutralising acidity and removing metals from minewater
Savmin	Hydroxide precipitation of metals followed by precipitation of calcium sulphate using gypsum crystals as a catalyst and significant additions of acid and alkali reagents.
Ion Exchange	An adsorption process that uses the reversible interchange of ions of the same charge between a solid ion-exchange medium and a solution, requiring reagents for resin regeneration as well as a conventional precipitation process.



Figure 31. Wheal Jane HDS plant showing the reactors and lamella clarifiers

The Wheal Jane plant (see Figure 31 for a panoramic view) is designed to treat 1000 m³/h of mine water with the typical composition given in Table 18. Clearly the plant is producing an aqueous discharge of high quality. The thickened sludge is pumped for deposition in the tailings lagoon of the abandoned mine and does not have to be further dewatered. It is typically between 15 and 25% (wt/wt) solids.

During the first 22 months of operation, the plant treated in excess of 12.3 million m³ of mine water at an average rate of 720 m³/h removing 3,200 t of metals with an overall removal efficiency of 99.2%.

Sulphide Precipitation

The sulphide precipitation process for metals removal relies on the generation of sulphide activity, either through reagent addition or by the biological reduction of sulphate to hydrogen sulphide in a specially engineered reactor. The sulphide produced reacts with the dissolved metals, which precipitate as insoluble sulphides. The principal advantages of the process are: the

lower residual concentrations of dissolved metals achieved by sulphide precipitation; the reduction in sulphate concentrations in the effluent; and the precipitation of the metals as a dense, relatively easily dewatered sludge. The capital and operating costs associated with sulphide precipitation can be higher than the equivalent costs for either conventional or HDS precipitation. Applications therefore tend to be limited to the treatment of waters containing sufficient high metal concentrations to allow the higher treatment costs to be partially off-set by commercial metal recovery on site or where sulphate removal is critical. For example, at Budelco in Holland, an SRB process is used to recover zinc from contaminated groundwater and at Kennecott Utah, the process is used to remove and recover copper from contaminated groundwater.

Other Technologies and Proprietary Chemicals

A number of proprietary chemical processes are available for the treatment of mine water as summarised in Table 19. Unfortunately, there are no publicly available data to allow the reporting of these technologies as European case studies.

Summary and Process Selection

Although the selection of the appropriate active mine water treatment process depends on the particular circumstances of each project, most active treatment plants use hydroxide reagents to precipitate dissolved metals either conventionally or in HDS form. The most common oxidising agent is air (oxygen); lime is the most frequently used alkali source.

Sulphide precipitation is most commonly used where sulphate removal is required and/or the recovered metals are of economic value.

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An Overview of Passive Mine Water Treatment in Europe

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Background

Passive treatment has been defined by the PIRAMID consortium as: “improvement of water quality using only naturally available energy sources, in gravity-flow treatment systems designed to require only infre-

quent but regular maintenance to operate successfully over its design life” (PIRAMID Consortium 2003). The approach was initially based on published observations at sphagnum bogs in the USA (Huntsman et al. 1978; Wieder 1982), which led to the concept that aerobic wetlands could be used for mine drainage treatment (Kleinmann et al. 1983).

Various technologies have evolved for passive treatment of mine waters in Europe through a combination of trial and error, building on pioneering work undertaken in the USA (Cohen 1992; Hedin et al. 1994a, 1994b). However, earlier work also clearly recognised that passive processes could be used beneficially:

- At the Cwmrheidol abandoned lead-zinc mine in West Wales, a concrete lagoon filled with limestone was constructed to treat two acidic mine discharges in the 1960's (Rees et al 2004). The system eventually failed when the limestone became coated with iron precipitates due to insufficient maintenance.
- Tuttle et al. (1969) demonstrated partial treatment of acidic mine water using a sawdust dam.
- Starting in the late 19th century and continuing until recent times, miners at Parys Mountain on the Isle of Anglesey used brick-lined precipitation ponds to extract copper from water that was purposefully drained through the mine workings. The acidic, copper-rich water was collected in these ponds and scrap iron was added. As a result, the iron passively dissolved while the copper was deposited as a black powder, which was removed for processing. The dissolved iron was subsequently recovered as ochre (Younger 2004)

Some of the earliest attempts at applying passive treatment technologies in Europe began in the UK about 10 years ago (Younger 1998; Rees 1999, 2002; Wiseman 2002). The technology at the time was largely based on constructing aerobic surface flow wetlands using USA guidelines for the treatment of circumneutral coal mine drainage that contained elevated iron (Hedin et al 1994a). This technology proved suitable to the temperate UK climate; as a result, there are now over 15 surface wetland schemes in the UK and over 20 in Europe (PIRAMID 2003).

Current Situation

Based on these early successes, passive treatment began to emerge as a practical alternative for mine water management, particularly at abandoned mine sites where financial resources were limited. The technology was perceived as sustainable, involved a minimal use of capital and financial resources, and added to the biodiversity of a site. In response to these benefits,

increased funding and research (private and public) has enabled the technology to be trialled and adopted at an increasingly diverse range of mine sites under various European climates and for various contaminants. The technologies currently operating in Europe are summarised in Table 20.

Other technologies are being tested, including: phytoremediation, algal mats, use of zeolites and other reactive media for metal encapsulation, and using other alkaline agents in addition to limestone e.g. steel slag. The promotion of all these technologies and guidance for their use in Europe has been published by the European Commission (PIRAMID 2003). This publication was developed alongside the ERMITE programme, which provides guidelines for the management of mine water on a catchment scale (ERMITE Consortium 2004).

UK Best Practice Case Study

Like many parts of Europe, there is relatively little active mining in the UK, although a legacy of over 3000 abandoned coal and metal mines exists dating back to the Bronze Age (Rees 2004; Younger et al. 2004). In this sense, the UK case study is an example of what can be achieved through a proactive approach to management of abandoned mines. In response to the detrimental effects (primarily on surface water) of abandoned coal mine discharges, the UK Coal Authority, supported by the Environment Agency (the Agency), have operated a rolling programme of feasibility study and mine water treatment since 1997. Since this time, over 20 passive treatment schemes have been constructed. Based on this successful rolling programme of coal mine treatment, the Environment Agency published a Metal Mine Strategy for Wales in 2002. This document set out a foundation for a programme of scoping and feasibility studies aimed at improving water quality on a catchment scale in accordance with the Water Framework Directive (Johnston 2004; Potter et al. 2004). To date, feasibility designs have been produced for two mining complexes encom-

Table 20. Operating passive treatment schemes in Europe (adapted from PIRAMID 2003)

Passive Technology	European Country	Contaminants
Aerobic wetlands	England, France, Sweden, Scotland, Slovenia, Spain, Wales, Germany, Poland	Fe, U, Ma, As, CN
Anaerobic wetlands	England, Germany, Norway, Sweden, Spain, Wales	SO ₄ , acidity
Reducing and alkalinity producing system ¹	England, Scotland, Spain, Wales	SO ₄ , acidity
Permeable reactive barriers	England, Sweden, Spain, Hungary	SO ₄ , acidity
Anoxic limestone drains	Germany, England (in removal), France	Acidity, Zn, Cd, As, Mg, Ca
Oxic limestone drains	France, England	

¹Also sometimes referred to as SAPS (Kepler and McCleary 1994)

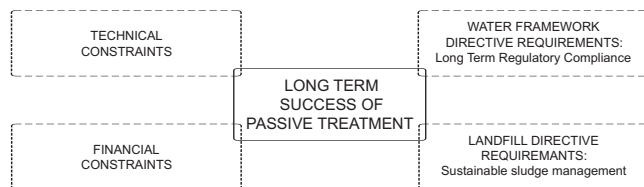


Figure 32. Illustration of inter-dependence of factors affecting long-term success of passive treatment

passing seven mine sites. Some of the mine water discharges are the most acidic and metal rich in the UK and yet may eventually be treated passively following pilot studies and laboratory treatability test work (Rees 2004; Rees et al. 2004; Younger et al. 2004).

As a result of the relatively large and increasing number of passive schemes in the UK, and to encourage knowledge transfer throughout Europe, a research facility referred to as CoSTaR (Coal Mine Sites for Targeted Remediation Research) has been established that encompasses six passive treatment schemes (see www.ncl.ac.uk/minewater/CoSTaR).

Research Needs

Coupled to the requirements of the WFD and more recently the EU Landfill Directive, passive treatment faces several challenges:

- How long will the treatment systems keep operating at sufficient levels?
- Can the bacteria responsible for several removal processes be provided with more suitable substrates?

- What will be done with the metal-rich sludge that is accumulating at treatment sites?

The inter-dependence of such factors on the long-term demonstrable success of passive treatment is illustrated in Figure 32.

These challenges are recognised internationally and measures are being put in place to provide solutions, including research on:

- iron ochre recovery and utilization
- sludge dewatering and utilization (as opposed to landfill)
- substrate selection and potential ongoing supplementation
- passive remediation of alkaline waters to neutral pH

The continued application and long-term success of passive technologies in all suitable parts of Europe will be determined by the results of this research. Ultimately, there may be a redefinition of passive treatment that includes reference to the sustainable long term management of the metal sludge produced.

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Mine Water Microbiology

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Introduction

Depending on the mineralogy, mine water can be alkaline ($\text{pH} > 7$), circum-neutral, or acidic ($\text{pH} < 7$). Water from coal mines can have a wide range of pH, while those draining metal mines, where acid-generating sulfide minerals (chiefly pyrite, FeS_2) often greatly exceed alkali-generating minerals (chiefly carbonates), tend to be acidic. Such mine water, often referred to as acid mine drainage (AMD) or acid rock drainage (ARD), usually contains large concentrations of toxic

metals and metalloids (e.g. As), which are generally more soluble at low pH. The Richmond mine at Iron Mountain, California, has the lowest natural pH recorded (as low as -3 ; Nordstrom et al. 2000).

Due to its acidic nature, as well as its elevated concentrations of dissolved metals, AMD is often considered to be devoid of life. While this is generally true for higher life forms, it has been known for nearly half of a century that microorganisms inhabit AMD (Colmer and Hinkle 1947). The first microbe isolated from AMD (Colmer et al. 1950) was an iron-oxidizing microbe called *Thiobacillus ferrooxidans* (now known as *Acidithiobacillus ferrooxidans*). Long before the discovery of microbes in AMD, a related microbe (*Acidithiobacillus thiooxidans*) that is also found in AMD was isolated from acidic, sulfur rich soils (Waksman and Joffe 1921). These two microorganisms belong to a class of extremophilic microorganisms

(those that thrive in extreme conditions) called acidophiles, meaning “acid loving”.

This brief review will outline what is currently known about the biodiversity and physiologies of microorganisms that live in acidic waters that flow from deep mines and surface mine spoils and tailings. Reference will also be made to “moderate acidophiles” that live in less acidic (pH 3–6) mine waters. More in depth reviews are referred to, should the reader require more detailed accounts of the subject matter.

While much of this review focuses in the main on the microbiology of mine waters in Europe, there is no strong evidence that any significant differences in microbial populations occur in mine waters in different parts of the world. One of the main influences on microbial populations appears to be physico-chemical properties of the mine water, and hence ultimately the mineralogy of the mine where the water arises.

Acidophilic Bacteria and Archaea

Prokaryotes (organisms lacking a membrane bound nucleus) that are found in AMD are widely distributed in the domains Bacteria and Archaea (for information on these two domains see Woese 1987). The physiological and phylogenetic diversity of the acidophiles has been thoroughly reviewed elsewhere (Hallberg and Johnson 2001), as has their distribution in mine waters (Baker and Banfield 2003; Johnson and Hallberg 2003). Acidophiles can be categorized on the basis of phenotypic traits, most typically temperature (as mesophiles, moderate thermophiles, and thermophiles) and pH optima for growth, and also on the basis of cellular carbon acquisition; autotrophs assimilate CO₂, heterotrophs assimilate organic carbon and “mixotrophs” use both. The various species of acidophilic microorganisms that can grow in acidic waters are listed in Table 21. Not all of these organisms are officially recognized species (as indicted by names in quotation marks) and others require re-classification, such as *Thiobacillus ferrooxidans* m-1, which is phylogenetically distinct from *bona fide* *Thiobacillus* and *Acidithiobacillus* spp as determined by 16S ribosomal RNA gene sequenceing. Thermo-acidophilic prokaryotes are particularly abundant in geothermal areas (e.g. Brock 1978) and in anthropogenic environments, such as self-heating coal spoils, and will not be discussed here.

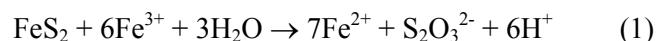
The most well studied acidophile is probably *Acidithiobacillus ferrooxidans*. Originally known as *Thiobacillus ferrooxidans*, phylogenetic analysis has revealed that it is not related to the original *Thiobacillus* species described, and hence it has been renamed (Kelly and Wood 2000). Aside from oxidizing ferrous iron, this microbe shares the ability to oxidize

sulfur and reduced sulfur-containing compounds with the closely related *Acidithiobacillus thiooxidans*, and can also use ferric iron as an electron acceptor (reducing it back to ferrous iron) in anoxic environments. *At. ferrooxidans* has historically been described as the major microbe in acidic environments, but this may be due to experimental artefacts and not reflect the true importance of this microbe in mine water. In many studies, enrichment culture with ferrous iron has been used to enumerate microbes, and *At. ferrooxidans* usually dominates such cultures (Johnson 1998). In other cases, the culture media used to determine the importance of iron-oxidizing microbes do not reflect the environment being studied (e.g. Kirby et al. 1999).

Another iron-oxidizing acidophile, *Leptospirillum ferrooxidans*, is increasingly being recognized as more important in generating acidic mine waters. The Afon Goch (a stream draining the former Parys Mountain copper mine in north Wales; pH 2.2–2.8) was found to contain about 10³/mL of both *At. ferrooxidans* and *L. ferrooxidans* (Walton and Johnson 1992). The relative numbers of these iron-oxidizers changed with distance from the discharge adit and appeared to correlate with changing concentrations of ferrous iron; *L. ferrooxidans* has a higher affinity for Fe(II) and were more numerous than *At. ferrooxidans* when ferrous iron concentrations were < 10 mg/L. In contrast, *At. ferrooxidans* was noted to be the most numerous iron-oxidizer present in AMD draining the abandoned Cae Coch pyrite mine in north Wales, where ferrous iron concentrations were often > 500 mg/L and the pH was 2.3–2.5 (McGinness and Johnson 1993). As at Parys Mountain, *L. ferrooxidans* was dominant over *At. ferrooxidans* in mine waters at Iron Mountain (Schrenk et al. 1998). Further work at Iron Mountain revealed the presence of a microbe related to *L. ferrooxidans*, which on occasions represented nearly 100% of the total microbial population (Bond et al. 2000a; Bond et al. 2000b).

Role of Acidophiles in AMD Generation

The role that microbes play in the formation of AMD has been described in detail elsewhere (e.g. Edwards et al. 2000; Johnson 2003; Nordstrom 2000). Briefly, sulfidic minerals such as pyrite (FeS₂; the most abundant of all sulfide minerals) are stable in situations where both oxygen and water are excluded. Upon exposure to both moisture and air by mining, sulfide minerals will oxidize spontaneously, with either molecular oxygen or ferric iron acting as the oxidant (Equation 1).



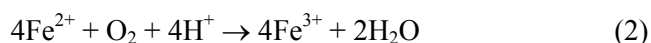
Sulfide mineral oxidation may be an abiotic process, but the rate of reaction is far greater, by many orders of magnitude, in the presence of iron-oxidizing acidophiles (Figure 33). For acid-stable sulfides, such as

Table 21. Acidophilic prokaryotic microorganisms associated with mine water

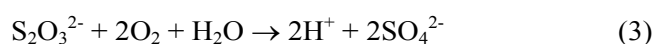
Mineral-degrading acidophiles	Thermal Classification*	Phylogenetic affiliation
Iron-oxidizers		
<i>Leptospirillum ferrooxidans</i>	Meso	Nitrospira
<i>L. ferriphilum</i>	Meso	Nitrospira
<i>L. thermoferrooxidans</i>	Mod Thermo	Nitrospira
" <i>Thiobacillus ferrooxidans</i> " m-1	Meso	β -Proteobacteria
" <i>Ferrimicrobium acidiphilum</i> "	Meso	Actinobacteria
<i>Ferroplasma acidiphilum</i>	Meso	Thermoplasmatales
" <i>Fp. acidarmanus</i> "	Meso	Thermoplasmatales
Sulfur-oxidizers		
<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>Hydrogenobacter acidophilus</i>	Mod Thermo	Aquificales**
Iron-reducers		
<i>Acidiphilium</i> spp.	Meso	α -Proteobacteria
Iron-oxidizers/reducers		
<i>Acidimicrobium ferrooxidans</i>	Meso	Actinobacteria
Iron-oxidizers/reducers and sulfur-oxidizers		
<i>Acidithiobacillus ferrooxidans</i>	Meso	β/γ -Proteobacteria
<i>Sulfobacillus</i> spp.	Meso and Mod Thermo	Firmicutes
Heterotrophic acidophiles (non mineral-degrading)		
<i>Acidocella</i> spp.	Meso	α -Proteobacteria
<i>Acidisphaera rubrifaciens</i>	Meso	α -Proteobacteria
<i>Acidobacterium capsulatum</i>	Meso	Acidobacterium
<i>Acidomonas methanolica</i>	Meso	α -Proteobacteria
<i>Alicyclobacillus</i> spp.	Mod Thermo	Firmicutes
<i>Picrophilus</i> spp.	Mod Thermo	Thermoplasmatales
<i>Thermoplasma</i> spp.	Mod Thermo	Thermoplasmatales

* Meso – mesophiles ($T_{\text{optimum}} < 40^{\circ}\text{C}$); Mod Thermo – moderate thermophiles ($T_{\text{optimum}} 40\text{--}60^{\circ}\text{C}$); ** inferred ability to oxidize minerals (via production of sulfuric acid)

pyrite, these microorganisms accelerate the oxidative dissolution of the mineral by re-generating ferric iron (Equation 2).



Ferric iron oxidation of pyrite is abiotic and may occur in anaerobic as well as aerobic environments, while oxidation of ferrous iron (Fe^{2+}) usually requires oxygen. At pH above 4, molecular oxygen will spontaneously react with ferrous iron, but at pH < 4, the abiotic oxidation of Fe^{2+} is insignificant (Stumm and Morgan 1981) and therefore ferrous iron oxidation will only occur in the presence of iron-oxidizing acidophiles. Some bacteria and archaea can also oxidize thiosulfate, the initial sulfur by-product of pyrite oxidation (Equation 1), and other reduced inorganic sulfur compounds and elemental sulfur, producing sulfuric acid (Equation 3).



While not having a direct role in the dissolution of sulfidic ores, other acidophiles certainly play an important

role through commensal interactions (Figure 33). Many of the autotrophic iron- and sulfur-oxidizing acidophiles are sensitive to trace amounts of organic matter, specifically organic acids (Alexander et al. 1987), and acidophilic heterotrophs such as *Acidiphilium* spp. will detoxify the environment by consuming the organic material (Harrison Jr. 1984). Conversely, some heterotrophs can catalyze the oxidation of iron (and hence sulfidic ores) only when supplied with a source of organic carbon, which can be supplied by the autotrophic acidophiles. This has been shown to be the case for *Ferrimicrobium acidiphilum*, which oxidized pyrite in the presence of *At. thiooxidans* but not in its absence (Bacelar-Nicolau and Johnson 1999).

Neutrophiles and Moderate Acidophiles

In contrast to the microbiology of acidic mine waters, relatively little is known about the microbiology of neutral mine waters. Due to their conspicuous appearance, neutrophilic iron-oxidizing and iron-depositing bacteria were amongst the first prokaryotes to be observed and described. These include the stalked *Gallionella ferruginea* and the filamentous *Leptothrix*

Moreno-Paz 2003). The application of these new techniques will lead to great advances in understanding the microbes that inhabit mine waters and how they influence the geochemistry of these environments.

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PADRE – Partnership for Acid Drainage Remediation in Europe

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Introduction

Acid drainage is the single greatest environmental challenge in the mining sector. Within Europe, the problem is manifest at both active and abandoned mine sites, from as far north as Spitzbergen (only 640 km from the North Pole) down to the arid regions near the Mediterranean coast. Given the present status of mining in Europe, it is not surprising that most European mine water problems are associated with abandoned mines, thousands of which can be found in nearly all European countries. Similar problems have been recorded worldwide, though in most other continents, a greater proportion of the problems are being tackled at active mining sites.

When policy makers within the European Commission began the ongoing process of preparing a European Union Directive on mining wastes, they had little awareness of the nature and scale of acid drainage problems associated with mine voids and bodies of mine waste. Fortunately, a process of awareness-raising soon ensued, facilitated in part by two very timely research projects of the European Commission's 5th R&D Framework Programme (PIRAMID and ERMITE), in which the authors were heavily involved. The experience of engaging with policy making made it abundantly clear that future mine drainage research within Europe needs to be coordinated in order to avoid unnecessary duplication of effort, both within and beyond Europe. A similar realisation at the global scale led to the formation first of INAP, and then of the

Global Alliance, in which INAP is partnering with relevant organisations from several continents, most notably ACMER (The Australian Centre for Mining Environmental Research, Australia), MEND (Mine Environment Neutral Drainage, Canada), and ADTI (Acid Drainage Technology Initiative, USA). The latest member of the Global Alliance is PADRE (Partnership for Acid Drainage Remediation in Europe).



What is PADRE?

PADRE is a “Permanent Commission” of IMWA, the International Mine Water Association. It was established by a formal vote by the IMWA General Assembly in October 2003. PADRE now has its own statutes and bye-laws, which are based on (and therefore consistent with) those of IMWA as a whole. PADRE's five objectives are:

- To promote international best practice in the stewardship of water and soil at European sites subject to the generation and migration of acidic drainage.
- To foster collaborative, international research and development into techniques for characterisation and abatement of acidic drainage in Europe.
- To promote dissemination of knowledge of current best-practice and innovations relating to acidic drainage prevention and remediation, with particular reference to European conditions, including the evolving framework of relevant EU legislation.
- To advance the training of present and future generations of European professionals who will engage in the art and science of acidic drainage prevention and remediation.
- To actively collaborate with a Global Alliance of organisations based in other continents which share similar objectives.

Membership is open to all individuals and institutions working in acid mine drainage remediation and research, and all European IMWA members are automatically members of PADRE, unless they request not to be. Currently PADRE has 140 members, of whom the majority are working in acid mine drainage remediation or research.

What has PADRE done so far?

PADRE was publicly launched during IMWA's Mine Water 2004 conference, in Newcastle Upon Tyne (UK) in September 2004. In the run-up to that Conference, founding members of PADRE were engaged in the production of the ERMITE and PIRAMID guidelines, as well as a passive treatment database, which are all available through the internet. Furthermore, course notes of a mine water remediation course within the EU IMAGE-TRAIN network have been produced and engagement with several symposia and congresses were initiated.

Planned Activities

To educate junior researchers and practitioners in the acid mine drainage field, PADRE plans to seek funding from European Commission sources (including the EU objective 4 "Know-How Fund") to organise training courses at various locations in Europe.

PADRE will actively contribute to the IMWA 2005 Congress in Oviedo/Spain and the IMWA 2007 Symposium in Portugal. The founder-Chairman of PADRE serves on the Scientific Committee of the 2005 "Securing the Future" conference to be held in Skellefteå (Sweden). Furthermore, PADRE will co-organize the Bergakademie Freiberg Meeting on "Mine Water Treatment" in June 2005.

All PADRE information will be disseminated through the IMWA journal "Mine Water and the Environment" and the PADRE web page www.PADRE.IMWA.info. Furthermore, the European knowledge gathered within PADRE will be disseminated through the Global Alliance, which was launched in July 2003 for collaboration between various institutions world-wide. Until the next General Assembly of IMWA, at which the membership of the PADRE Council will be put to a vote, the provisional Executive Council of PADRE will consist of Paul Younger (UK; Chair), Rob Bowell (UK; Treasurer and Vice-Chair), Christian Wolkersdorfer (Germany; Secretary), Ingar Walder (Norway), and Ludo Diehls (Belgium).

If you are interested in obtaining up-to-date information about acid mine drainage issues in Europe, or if you want to contribute actively to the partnership, feel free to contact the PADRE Councillors through PADRE@IMWA.info.

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