Environmental Science

Series editors: R. Allan · U. Förstner · W. Salomons

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José M. Azcue (Ed.)

Environmental Impacts of Mining Activities

Emphasis on Mitigation and Remedial Measures

With 76 Figures and 54 Tables

Springer

Editor

Dr. José M. Azcue

National Laboratory of Civil Engineering (LNEC)
Department of Hydraulics
Av. Brasil 101
1799 Lisbon
Portugal
E-mail: jmazcue@lnec.pt

ISBN-13: 978-3-642-64169-5 e-ISBN-13: 978-3-642-59891-3
DOI:10.1007/13: 978-3-642-59891-3

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Library of Congress Cataloging-in-Publication Data
Environmental impacts of mining activities: emphasis on mitigation and remedial measures / José M. Azcue (ed.)
p. cm. -- (Environmental science)
Includes bibliographical references and index.
ISBN 978-3-642-64169-5 (softcover)
1. Mineral industries--Environmental aspects. I. Azcue, José M.
II. Series: Environmental science (Berlin, Germany)
TD195.M5E517 1999

Softcover reprint of the hardcover 1st edition 1999

Production: ProduServ GmbH Verlagsservice, Berlin
Typesetting: Büro Stasch, Bayreuth
Cover layout: Struve & Partner, Heidelberg
SPIN: 10558835 32/3020 – 5 4 3 2 1 0 – Printed on acid-free paper

Preface

> *"It is mined from deep mines,*
> *for it is material that nature hides from us,*
> *teaching us to leave it alone as harmful,*
> *but this does not cause the arrogant miners to leave it"*
> *(Birringucio 1540)*

Mining has gone on for thousands of years and abandoned mines and active mine operations are found throughout the world. Mining has played a key role in the development of humanity and is a major asset to the economy of many countries. Our ancestors were already aware of the potential toxic risks of some of the mining activities, as shown in the above citation regarding arsenic mining (V. Birringucio, 1540 *The Pirotechnia.* The MIT Press, Massachusetts, 1943). The concern for the health of miners has evolved in parallel with mining development. However, only in the last decades did the environmental impacts of mining activities become a social concern.

Multidisciplinary approach is a prerequisite to tackle mining problems. This book was designed to serve a wide international audience, including those working in government agencies, universities, and the private sector, evaluating environmental impacts and suggesting remedial measures. Special attention was given to selecting an eclectic group of problems and approaches. The information is divided in to three sections, namely, Section A '*Evaluation of Effects of Mining in the Environment*', Section B '*Treatment Methods for Mine Effluents and Rehabilitation of Mine Tailings and Overburden Materials*', and Section C '*Study Cases*'. Care has been taken to provide suitable literature sources for those desiring more specific information. Due to the magnitude of the problem and the vast range of solutions no attempt has been made to cover all aspects and this book does not pretend to be a final answer but a step in a new approach.

Section A (Chapters 2 to 7) covers a wide range of relevant topics for the evaluation of environmental effects of mining activities. Mining inevitably represents an alteration in the environmently of its area of influence. Nevertheless, its environmental impacts can be significantly minimized by using environmental compatible mining and ore-dressing techniques and by developing new technologies for treating ores, effluents, and waste materials. An understanding of the causes originating some of the environmental problems led to the application of preventive measures. The technologies for treatments of mining effluents and rehabilitation of mine tailings are under continuous development. Section B (Chapters 8 to 12) presents several physico-chemical and biological treatment methods for mine effluents and overburden materials. There are numerous studies in the literature showing severe environmental effects of mining activities. For instance, the environmental effects of the uncontrolled and intensive development of gold mining activities in the Brazilian Amazon during the past 15 years have shown to have global significance (Lacerda and Salomons 1997 *Mercury from gold and silver mining: a chemical time bomb?* Springer-Verlag). Fortunately, there are as well several positive examples of improved industrial technologies and land and water reclamation techniques such as the Sudbury case in Ontario, Canada (Gunn 1995 *Restoration and recovery of an industrial region.* Springer-Verlag). In Section C four different study cases are presented.

Nowadays large amounts of money and efforts are being devoted by many countries and industries to minimize environmental mining problems. However, many of the mineral resources are located in less-developed countries where environmental protection measures are a low priority. This represents a major challenge and responsibility in order to approach the environmental effects of mining activities on a global scale. We should make sure that the problems that industrialized countries are facing today will not be repeated in new mining developments.

José M. Azcue
Lisbon, 1998

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Acknowledgments

I wish to thank all the co-authors of this book for their efforts in the preparation of the chapters and for their excellent cooperation during editing of the manuscripts. I would like to give special recognition to the reviewers who contributed to this book: M. Brown, E.C.G. Couto, M.A. Cunha e Sa, A. Duarte, J.G. Farmer, A. Fortunato, D. Ibana, T. Jackson, M. Langedal, A.B. Mackenzie, J. Maskall, J.R. Miller, E. Pereira, R. Reuther, R. Shown, W. Tin-Chee, C. Vale, and A. Zeman. Most of the authors also assisted with manuscript reviews. Finally, I gratefully acknowledge Rod Allan and Ulrich Förstner for their valuable support and encouragement throughout this project.

During my professional career I had the privilege to be disciple of four outstanding professors: Wolfgang C. Pfeiffer, Jerome O. Nriagu, Alena Mudroch, and Treffor Reynoldson. Each of them, in his or her distinctive way, has significantly contributed to fostering my scientific interest. I feel honored and grateful for their advice and inspiration. This book is dedicated to the four of them.

Contributors

Gert Asmund
National Environmental Research Institute Department of Arctic Environment
Tagensvej 135, 4. sal, 2200 Copenhagen N, Denmark

Sukumar Bandopadhyay
University of Alaska Fairbanks, Dept. of Mining & Geological Engineering,
Fairbanks, Alaska 99709-5800, USA

Ragula Bhaskar
Department of Mining Engineering, University of Utah, Salt Lake City, Utah 4112
USA

Per G. Broman
Boliden AB, Boliden 936 81, Sweden

Donald R. Clark Jr.
US Geological Survey, Brazos Field Station, Environmental and Contaminents Research Center, Department of Wildlife and Fisheries Sciences,
Texas A&M University, College Station, Texas 77843-2258, USA

Bernd Eccarius
Darmstadt Technical University, Institute of Geology and Paleontology, Darmstadt,
Germany

Ronald Eisler
US Geological Survey, Patuxent Wildlife Research Center,
11510 American Holly Drive, Laurel, Maryland 20708-4017, USA

Derek V. Ellis
Biology Department, University of Victoria, British Columbia Victoria, V8W 2Y2
Canada

Nils Eriksson
Water Resources Engineering, Royal Institute of Technology, Stockholm 100 44,
Sweden

Ulrich Förstner
Technical University Hamburg-Harburg,
Eissendorfer Str. 40, 021071 Hamburg, Germany

Rene Frömmichen
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Walter Glässer
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Torbjörn Göransson
Boliden AB, Boliden 936 81, Sweden

Sivaraman Govindarajalu
Curtin University of Technology, G.P.O. Box U1987, Perth, Western Australia 6845,
Australia

R. Larry Grayson
National Institute for Occupational Safety and Health,
200 Independence Av., S.W., Washington, DC 20201, USA

Jean R. D. Guimarães
Federal University of Rio de Janeiro, Institute of Biophysics,
Ilha do Fundão, 21941-900-Rio de Janeiro, Brazil

Hans E. Gustafsson
Swedish Environmental Protection Agency,
Blekholmsterrassen 36, Stockholm 106 48, Sweden

Charles Hagedorn
Virginia Polytechnic Institute and State University, Department of Crop and Soil
Environmental Sciences, Blacksburg, Virginia 24061-0404, USA

Autumn-Lynn Harrison
Virginia Polytechnic Institute and State University, Department of Crop and Soil
Environmental Sciences,
Blacksburg, Virginia 24061-0404, USA

William W. Hellier
Pennsylvania Department of Environmental Protection,
P.O. Box 209, Hawk Run, Pennsylvania 16840- 0209, USA

Charles J. Henny
US Geological Survey, Forest and Rangeland Ecosystem Science Center,
3200 SW Jefferson Way, Corvallis, Oregon 97333, USA

Poul Johansen
National Environmental Research Institute, Department of Arctic Environment,
Tagensvej 135, 2200 Copenhagen N, Denmark

Hans Jönsson
Boliden AB, Boliden 936 81, Sweden

Christopher Kennedy
University of Toronto, Department of Chemical Engineering, Toronto, Ontario, Canada

Helmut Klapper
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Manfred Lindvall
Applied Geology, Luleå University of Technology, Luleå 97187, Sweden

Lars-Åke Lindahl
Boliden AB, Boliden 936 81, Sweden

Tom Lundgren
Swedish Geological AB, Repslagaregatan 19, Linköping 582 22, Sweden

Olaf Malm
Federal University of Rio de Janeiro, Institute of Biophysics, Ilha do Fundão,
21941-9000 Rio de Janeiro, Brazil

Philip Maxwell
Curtin University of Technology, G.P.O. Box U1987, Perth, Western Australia 6845,
Australia

John A. Meech
University of British Columbia, Department of Mining and Mineral Process Engineering,
6350 Stores Rd., Vancouver, British Columbia, V6T 1Z4, Canada

Piotr Michalski
Agricultural University in Krakow, Department of Soil Mechanics and Earth Structures, Al. Mickiewicza 24/28, 30-059 Kraków, Poland

A. Dogan Paktunc
Mineral and Sciences Laboratories–CANMET, 555 Booth St., Ottawa, K1A 0G1 Canada

Raymond B. Reneau Jr.
Virginia Polytechnic Institute and State University, Department of Crop and Soil Environmental Sciences,
Blacksburg, Virginia 24061-0404, USA

James D. Robertson
Placer Dome Inc., P.O. Box 49330, Vancouver, British Columbia, V7X 1P1, Canada

Martin Schultze
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Peter Schreck
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Krystyna M. Skarzynska
Department of Soil Mechanics and Earth Structures, Agricultural University in Krakow, Al. Mickiewicza 24, 30-059 Kraków, Poland

Ulrich Stottmeister
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Gerhard Strauch
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Navin Tandon
Trinity Consultants, 12801 N. Central Expressway, Suite 1200, Dallas, Texas 75243, USA

Marcello M. Veiga
The University of British Columbia, Department Mining and Mineral Process Engineering, 6350 Stores Rd., Vancouver, British Columbia, V6T 1Z4, Canada

Erika Weißbrodt
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Katrin Wendt-Potthoff
UFZ, Permoserstraße 15, 04318 Leipzig, Germany

Stanley N. Wiemeyer
US Fish and Wildlife Service, Nevada State Office,
4600 Kietzke Lane, Suite 125C, Reno, Nevada 89502-5055, USA

Chapter 1

Introduction

Ulrich Förstner

In the five decades since World War II, the volume of nonfuel minerals consumed has exceeded the total extracted from the Earth during the rest of human history. While the world population doubled between 1959 and 1990, the production of six major base metals (aluminum, copper, lead, nickel, tin, and zinc) increased more than eightfold. Contrary to predictions between the 1950s and the mid-1980s, persistent shortages of nonfuel minerals have not occurred. A significant factor affecting mineral availability today, however, is the environmental conscience that has arisen over the last three decades throughout the industrial world and, increasingly, in developing countries. Carroll Ann Hodges, summarizing her review article "Mineral Resources, Environmental Issues and Land Use" (Science, Vol. 268: 2 June 1995), depicts a priority scenario, where mineral-rich developing nations, plagued by inefficient state-owned mining enterprises, high population growth rates, and environmental degradation could realize substantial benefits by reforming government policies to encourage foreign investment in resources and by the appropriate allocation of mineral rents. Here, transfer of technology, training, education, and environmental stewardship are major steps toward the mitigation of both poverty and pollution.

Mining and smelting have been the cause of major pollution problems since their beginning. In some instances, little progress has occurred between the Middle Ages, when metallic mining industries of Europe started extracting ore from greater and greater depths (described by Georgius Agricola in his De re metallica, published in 1556), and the primitive uses of mercury during the present gold rush in Brazil (see Chap. 4, this Vol.). Presently, from an environmental point of view the main interest is directed towards sulfidic mining waste. The sulfidic ore bodies have remained practically unchanged in the bedrock for very long periods. However, when brought in contact with atmospheric oxygen and water, traces of metal sulfides in the waste material are oxidized to produce an acidic leachate containing heavy metals. Depending on the buffering capacity of the waste material and various chemical retention factors, not all deposits produce the same environmental concern. An undesirable future scenario (which is described among others in Chapts. 3 and 13, this Vol.) would be that, in the long-term, the buffering capacity of waste deposits might be consumed, resulting in gradually increasing heavy metals discharges.

Priority problems are associated with the enormous masses of mine tailings. Their global quantity of approximately 18 billion m^3 per year is in the same order of magnitude as the actual sediment discharge to the oceans. A simplified estimation suggests that due to the future extraction of lower grade ores, the production of mine wastes will be doubled within a period of 20–30 years. On the other hand, it should be considered that the trend in use of minerals in industrialized countries is down; mature economies became increasingly tied to high technology and service industries that require less intensive use of raw materials, thanks to miniaturization, economies of scale, and substitution. Future developments will strongly depend on the extent and quality of recycling. The sum of the arguments, as outlined by the United States National Academy of Engineering in its book *Technology and Environment* (National Academy Press, Washington, 1989), suggests a long-term imperative favoring "dematerialization", that results in reduced extraction of virgin materials, reduced loss of waste materials, and increased recycling of useful ones. With regard to the quantity problem of wastes, such a perspective will provide the greatest benefits at the beginning of the product lifecycle of the mining and smelting residues.

The current situation relevant to policy decisions in most countries is characterized by a dramatically expanding world trade, including that in nonfuel mineral commodities. Technological improvements in mining and metal recovery have enabled massive earth-moving operations and greater efficiency in production, with an associated reduction in costs. Justification for environmental protection now often exceeds that for mining a specific ore body. Traditional mining regions may become noncompetitive, when social and remediation costs are incorporated in the price of the product. An example is the uranium mining in the German States of Saxony and Thuringia, with a post-war production of 220 000 metric tons of uranium exceeding by far the post-war production of Australia, South Africa, and probably the former Soviet Union ("owner" of the Wismut Company until 1990, who carried out uranium mining in the early post-war years without practically considering the damage to the environment or humans). With the opening of the Iron Curtain it became clear that uranium mining in Germany was far from profitable by Western standards. In 1991 a huge decommissioning and rehabilitation program – approximately 8 billion US$ over a project period of 10–15 years – was initiated with the aim of minimizing ecological hazards from the 45 years of uranium mining (D. Mager, German Federal Ministry of Economy, Geotechnica, Cologne, 1993).

In Western countries with vast mineral resources, for example Australia, Canada, and the USA, there has been an early awareness of the environmental problems associated with mining activities. Australia's uranium mining industry strictly follows the perspective of an "ecologically sustainable development", which does not allow for long-term pollution and requires the placement of bonds to cover all the costs of rehabilitation according to the designated future use of the land. Ranger Uranium has already placed 2 billion A$ in the bank to cover final closure. Furthermore, the experience gained by Ranger has now been transferred to nonuranium mining activities in the Northern Territory. The Northern Territory is considered a model for other developing countries located in the tropical zone, e.g. Papua New Guinea, Indonesia, the Philippines, Ghana, Guinea, etc. (B.N. Noller, Department of Mines and Energy, Northern Territory, Australia, pers. comm.). The most extensive coordinated research program in the area of modeling and characterization of geochemical processes in tailing impoundments and groundwater influenced by acid mine seepage is the Canadian Mine Environment Neutral Drainage (MEND) project, which was conducted from 1988 to 1997. It covers topics such as prediction methods, prevention and control, effluent treatment, and field monitoring (more details of the MEND program are presented in Chap. 9 of this Vol.). The full-scale implementation and quality control of remediation methods have been described as the "flagships" of the program (MEND annual report, 1995). Based on these experiences, Swedish researchers from six universities and two mining companies have started a project titled "Mitigation of the Environmental Impact of Mining Waste", focusing on environmental issues associated with mining operations and the disposal of waste rock and tailings, while at the same time finding practical methods to ensure a competitive mining industry in Sweden in the future. One of the major aims is to develop strategies to find efficient combinations of different prevention and control methods so that a high efficiency is achieved in each part of the near-field system, e.g. combinations of physical and hydraulic barriers (covers), geochemical control measures and suitable biochemical conditions. Each project's economic viability should be evaluated throughout the sub-programs (Swedish Foundation for Strategic Environmental Research, pers. comm.).

These are only a few examples of social, economic, and environmental implications in former, contemporary, and future mining activities. They have been selected to give an impression of the broad spectrum of knowledge, forming today's basis of – mostly multidisciplinary – projects on the mitigation of and remedial measures on environmental impacts during and after mining. Much more detailed information on any of these and other important aspects can be found in the present book, which has been written by leading experts in this rapidly developing field of environmental science and technology. "In terms of effluents and wastes, mining is largely a local issue but one of global importance" (R.J. Allan 1997 Mining and Metals in the Environment. J. Geochem. Explor. 58). Jose M. Azcue, who is intensively involved with local researchers and practitioners, and has experience of typical projects on mitigation of and remedial measures on mine effluents and wastes in all parts of the world, can be congratulated for his skillful and comprehensive edition, which provides guidance to both global thinking and local action on this important issue.

Section A

Evaluation of Effects of Mining in the Environment

Chapter 2

Do Australian Mining Companies Pay Too Much? Reflections on the Burden of Meeting Environmental Standards in the Late Twentieth Century

Philip Maxwell · Sivaraman Govindarajalu

2.1
Introduction

Australia has been a major mining nation for almost 150 years. This status seems certain to continue for another fifty years and perhaps longer. In 1995–1996 the estimated contribution of the minerals and energy sector to the nation's Gross Domestic Product was about $ US 15 billion (just over 4% of GDP). When basic metal processing is added to mining, this percentage almost doubled.

A feature of Australian mining today is the domination of large companies. In 1996 there were fourteen mining companies in the largest fifty corporations listed on the Australian Stock Exchange. Over the past 30 years these companies have generally performed well. As visible and successful businesses they attract considerable attention and as such are expected to behave as good corporate and environmental citizens.

The exploitation of minerals and energy can cause considerable environmental damage. This has been recognized for many centuries. Authors such as Eggert (1994) show that even writers such as Agricola were aware of the negative environmental externalities which mining activities can generate. Warhurst (1994) is one of a number of commentators who provide a useful classification of the sources of these environmental costs. Her list includes land degradation, ecosystem disruption, acid mine drainage, chemical leakages, slope failures, toxic dusts arising from the mining process, and pollution from compounds of carbon, sulfur and nitrogen, metal particulates, as well as chemical leakages, in the smelting process. The emergence of stricter environmental protection legislation in Australia and elsewhere since about 1960 represents a typical regulatory response to endeavor to internalize these costs. Petroleum and mining companies have responded by placing environmental management issues on a much higher plane than previously.

As well as mining, there are many other sources of environmental degradation. In a nation like Australia agriculture and urban development have been major contributors in this regard. Agricultural establishments cover more than 60% of the land mass (Australian Bureau of Statistics 1997) and it is widely recognized that farming and pastoral activities have significantly degraded at least 10% of this area, i.e. more than 6% of the land mass. Significant urban populations are confined to a much smaller area. Three quarters of the country's population of 18.5 million live in an area which covers around 1% the geographical area. Although Australian cities and major towns are relatively clean they have also changed and degraded the natural environment.

Mining's contribution to environmental damage seems to pale by comparison. Its activities occupy only about 0.2% of the land area. Much is located in remote and arid areas where waste materials and emissions can be dissipated without interfering with nearby populations. The absence of flowing rivers and lakes also contains this waste threat in many parts of the continent. Yet, mining companies are apparently prosperous and they seldom appear to exert the same political influence as the urban masses. Also, they have not been facing the difficult times of the agricultural sector with depressed prices and barriers to international trade. In this scenario it seems conceivable that, together with the forestry industry, the minerals and energy sector may have become a major target for the emerging conservation lobby. Indeed it might be argued that, because of their visibility, both of these industries may be receiving too much attention and as a result are being forced to incur excessive environmental costs to do business.

The focus of this paper is on considering the validity of this view. In developing our argument we reflect initially on the historical importance of mining to the Australian economy before moving to reflect more generally on the environmental effects of mining in the past and the present. This discussion provides the foundation for a more theoretical discussion of the economics of environmental protection in mining and the presentation of data.

2.2
Importance of Mining to the Australian Economy

The minerals and energy sector has played an important role in the economic development of Australia since European settlement began in 1788. Although Lieutenant Shortland found coal in Newcastle in 1794, it was not until the Burra copper finds in South Australia in the early 1840s that mining developed in any significant way. This was closely followed by major gold rushes in New South Wales and Victoria in the early 1850s in the wake of the California gold rushes. One impact of these first Australian gold rushes was to stimulate immigration and thereby to propel the colonial economy into a major new development phase. Whereas Australia's non-aboriginal population stood at 190 000 in 1840 it had risen to 405 000 in 1850 and 1 146 000 in 1860.

Sinclair (1976) has noted that: "...the discovery of alluvial gold must be regarded as a major discontinuity. The significance of this for the course of Australian economic development was heightened by the rapidity with which gold was extracted from the ground".

Since 1860 there have been many major mineral finds which have in turn stimulated population growth. These have included copper in Tasmania in the 1870s, silver, lead, and zinc at Broken Hill in New South Wales in 1883, gold in Western Australia in the late 1880s and 1890s, copper at Mt. Isa in 1924, oil and gas in the Bass Strait, the North West Shelf and elsewhere from the 1950s onwards, bauxite in Queensland and Western Australia in the 1960s, iron ore in Western Australia in the 1960s, coal in Queensland in the 1960s, nickel and mineral sands in Western Australia and Queensland from the mid-1960s, and diamonds in Western Australia from the early 1970s.

At the height of the first gold rushes in the 1850s mining accounted for more than 50% of Australia's Gross Domestic Product and for perhaps 80% of her exports. These high percentages quickly subsided but the industry continued to make a major contribution to the wealth of the emerging nation for the remainder of the nineteenth century. This contribution was so significant that during the 1870–1900 period, Maddison (1982) has estimated that Australians had the highest income per capita of any of the world's economies.

In 1900 mining accounted for around 10% of GDP and the country's non-aboriginal population had reached 3.7 million. The contribution of mining fell away after World War I and, by the mid-1950s it contributed only about 1% of GDP and perhaps 5% of exports. The major new resources boom which began in the 1960s has again propelled the minerals industry (mining and minerals processing) to the forefront of the Australian economy. By 1990 the industry again accounted for almost 10% of GDP and more than 50% of merchandise exports. In an era when the country's internationally competitive agricultural sector struggled, manufacturing faced considerable structural adjustment as tariff barriers fell, and the services sector began a slow emergence, the strong performance of the mineral and energy sector has been a key to the nation's continuing prosperity.

As an empty "developed" Western nation Australia has attracted many immigrants since 1945. The nation usually ranks in the top twenty nations in terms of GDP per capita.

2.3
Mining and the Environment in Australia

Australia is the world's driest continent. Almost half of its land mass is in arid zones which experience an average of less than 350 mm of rain per annum. Excluding Antarctica, it is also the world's least populated continent. The combination of these characteristics and the location of mineable mineral and energy deposits throughout the land mass, well represented in remote and arid areas, has meant that mining has not contributed in the past to influencing environmental quality in such an adverse way as it has in the other continents. In this regard Australia seems to have been quite fortunate.

Yet as elsewhere there are cases where mining activities have adversely affected local physical environments either directly or indirectly. Farrell and Kratzing (1996) describe some of the well known historical examples in Australia. Their discussion covers the Rum Jungle uranium and copper mine in the Northern Territory, the gold mines at Captain's Flat in New South Wales and more generally the earlier impacts of gold mining in Victoria and Western Australia and of mineral sands mining in Western Australia. The long-term effects of acid mine drainage were a major problem at Rum Jungle and at Captain's Flat. The Federal government in combination with relevant state and territory administrations spent 17 million A$ of cleaning up the South Alligator River downstream from Rum Jungle. A further 3 million A$ was devoted to restoration of the environment around Captain's Flat.

The strong development of the minerals and energy sector since the mid-1960s has also coincided with the era of greater environmental awareness and the emerging sustainable development debate in which conservationists have been prominent and often critical of mining interests. A notable event in the recent past in this debate involved the commissioning by the Hawke Federal Labor Government in 1990 of a series of studies on ecologically sustainable development. In its report, the mining working group (Ecologically Sustainable Development Working Groups 1991) noted that: "The mining industry generally has substantially improved its level of performance in environmental protection, pollution control and minesite rehabilitation in recent years. The degree of improvement has not always been recognized widely in the community". Some commentators, e.g. Hore-Lacey (1991), hold the view that this report was particularly favorable to conservation interests. It is therefore notable that the above comments appeared.

As part of its deliberations the above mining working group considered a series of 42 case studies which examined past and present environmental management practices for mining, exploration and petroleum operations throughout Australia.[1] From these studies (Ecologically Sustainable Development Working Groups, 1991, p. 25) reported that:

> "On the evidence available, environmental management technology and techniques exist to adequately control all these impacts under most operating conditions encountered in Australia. However, it is also apparent that the best available technology is not applied consistently throughout the industry. As would be expected, operations which commenced before about 1970 generally operate to lower environmental standards than more modern operations. It also appears that operations close to population centres or in or near areas of conservation significance operate to higher standards than more remote regions less open to public scrutiny".

More recently mining companies have been keen to illustrate to the public their improved credentials as protectors of the environment. In Mulligan (1996) there are 21 papers which describe recent environmental management practice by companies in the coal, gold, bauxite, alumina and aluminum, iron ore and steel, base metals and uranium, heavy mineral sands, construction materials, and petroleum sectors.

It is clear that mining companies operating in Australia now pay a lot of attention to environmental matters and that they devote considerable resources to this area. This is recognized officially by the Federal government in its Environment Protection Agency's *Overview of Best Practice Environmental Management in Mining* booklet published in June, 1995. This notes that: "The mining and energy industries in Australia have a great deal of expertise in environmental planning, management and rehabilitation".

Suggesting that best practice environmental management is expensive in the short run, the authors note industry estimates that up to five per cent of capital and operating costs for new mining projects are necessary for this to be achieved. Personal discussions with senior environmental advisers for mining companies support this view, though in some environmentally sensitive areas close either to major population centers or well known national parks the percentage may reach higher percentage levels than this. Specific cases which come to mind are the Ranger and Nabarlek uranium mines in the Northern Territory, the bauxite mines in the Darling Ranges close to Perth and mineral sands mines on Fraser and Stradbroke islands in Queensland, and in the south west of Western Australia.

2.4
Economics of Environmental Protection in Mining

The environment has recently attracted a lot of interest from economists. A widely held view in the associated literature is that markets do not allocate environmental resources efficiently. This market failure arises in part from the widespread existence of exter-

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[1] These case studies came from a consultancy undertaken for the working group by Professor Ralph Buckley and an earlier study conducted for the Australian Mining Industry Council (now the Minerals Council of Australia) by Dr. Graham Webb.

nalities and because many environmental resources are public goods. There is therefore a good case for appropriate public regulation to achieve an optimum economic outcome. The impact of mining, as one of the many human activities which influences the environment, can be integrated into this framework.

In an unregulated market environment, mining will generate a series of negative externalities. In a reasonably standard definition Perman *et al.* (1996) define an externality as occurring: "when the production or consumption decisions of one agent affect the utility of another agent in an unintended way, and when no compensation is made by the producer of the externality to the affected party". Public regulation undertaken with the assistance of environmental protection legislation is generally needed to assist in the process of internalizing these negative externalities.

A rudimentary diagrammatic explanation of the effect of externalities on the allocation of resources can be seen in terms of the simple demand/supply curve analysis. Without appropriate environmental regulation a typical mining company might have a supply curve such as S_1 in Fig. 2.1. Facing a demand curve, D_1 it will produce an output level of q_1 at a price p_1. If, however, the producer pays the optimal level of environmental damage costs, the supply curve will shift back to S_2. This will lead to a lower equilibrium level of mineral output, q_2, and a higher price p_2. A similar conclusion arises when the activities of individual mining operations are aggregated to provide a broader industry view. The absence of market regulation would lead to an overallocation of resources to mining in both situations.

There is clearly a role in this world for appropriate environmental protection legislation. From a practical viewpoint, however, there is an important issue of how zealous this legislation should be. If it is too demanding, the mining operation in Fig. 2.1 may face a supply curve of S_3 which would result in a less than optimal level of output q_3 at a price p_3. The difficulty of meeting environmental requirements in the more densely populated Australian states might lead an observer to conclude that this is occurring at present.

A useful further insight into the issue of the best environmental protection policy for the minerals industry is gained by considering some of the thinking which has followed on from the seminal article by Coase (1960).[2] A simple way of viewing the environmental impacts of the mine is in terms of a diagram which incorporates marginal damage and marginal abatement cost (or marginal benefit) curves which arise from air, water and noise pollution and aesthetic damage.

The most efficient level of output of mineral production will be that which maximizes the net benefit to society, assuming that this is positive. This net benefit is the difference between the benefit from the output with which the environmental degradation resulting from the various forms of pollution is associated, and the damages resulting from it. The extent of damage to the environment in a broad sense resulting from mining and mineral processing might takes a form such as:

$$\text{Environmental damage} = a\,(\text{level of mineral output})^k,$$

where a is a constant and $k > 1$.

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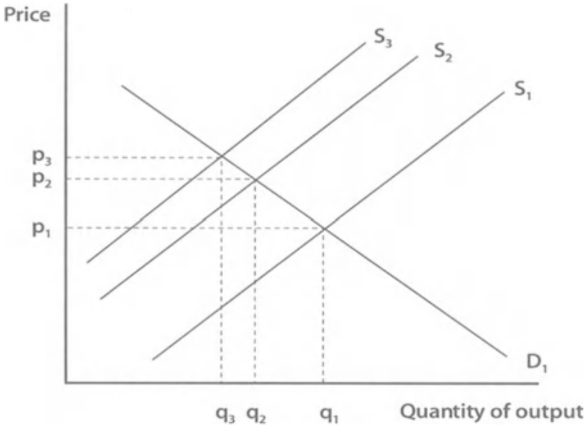
[2] A well developed discussion of the points made here appears in Perman, Ma, and McGilvray (1996, p. 197–250). Our exposition is derived from this analysis.

Fig. 2.1. A simple view of effect of negative externalities on resource allocation



One view of this model can be shown in Fig. 2.2 where marginal damage and marginal benefit curves are plotted. The vertical axis of the diagram measures cost and the level of pollution per period is on the horizontal axis. Marginal damage rises as output increases. The marginal benefit of pollution refers to the benefits from the output less the costs of production. This falls as output (and pollution levels) rise. The most economically efficient level of environmental damage will occur where the marginal damage and marginal benefit curves cut one another. This occurs at the pollution emission level of ψ^* which corresponds to a marginal level of damage of λ^*.

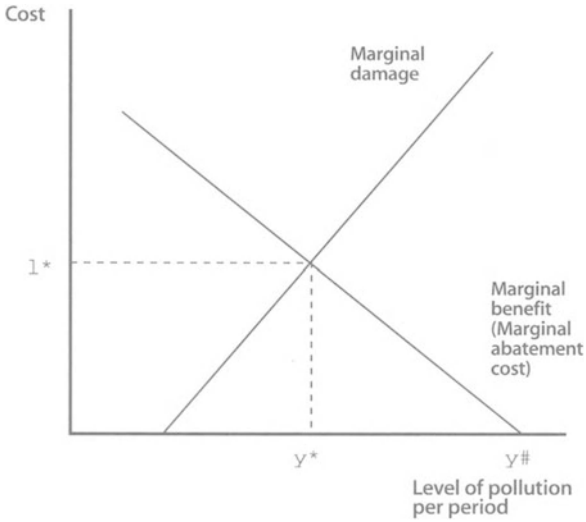
The essence of the Coasian argument is that an economically efficient solution is possible through bargaining between the affected parties in a situation such as that shown in Fig. 2.2 as long as enforceable property rights exist. The way in which property rights are allocated does, however, come into play when distributional issues are considered. Perman *et al.* (1996) point out additionally that two other conditions are important for bargaining solutions to emerge. One is that the number of affected parties should be relatively small because otherwise the bargaining process may become too costly and difficult. The second is that all of the affected parties must be identifiable.

If one ascribes to the Coasian view, government still has a role to play in developing and maintaining institutions which enhance the opportunity for bargaining behavior. A less optimistic view of the world will require government to intervene as well with a more comprehensive regulatory system to try to ensure the optimal environmental outcome. In terms of Fig. 2.2, the economically efficient level of environmental damage will occur where the level of damage abatement is the amount $\psi\#-\psi^*$.

What we have provided is a quite simple comparative static framework for our theoretical perspective. The picture becomes more complicated when dynamic considerations are included, when distributional issues are incorporated and when issues relating to stock as well as flow pollution leading to environmental damage are considered. These issues are important but complex. The simple framework provided here provides a useful, if somewhat constrained, starting point from which to begin an applied policy analysis.

In terms of the approach in Fig. 2.2 the key issue is whether the branches of government in Australia responsible for environmental standards with respect to mineral and

Fig. 2.2. The efficient level of environmental damage from mining



energy exploitation, have placed their policy settings at the appropriate point along the horizontal axis. If they have done too little the marginal damage cost of environmental degradation will exceed the benefit it provides. If their regulations are too constraining the reverse will be the case.

2.5
State and Territory Environmental Regulations and Guidelines

Within the Commonwealth of Australia, the Federal government plays largely an advisory role through its publications on best practice in environmental management. Taking this into consideration, each state government then develops and applies its own set of environmental guidelines. Although the term "guideline" appears consistently in the official literature most of the guidelines are effectively requirements.

Even at the exploration stage operators in each state must consider "guidelines" with respect to:

- Location and layout
- Timetables for operations including the development of associated infrastructure, land clearing, extraction operations, and haulage
- Land management
- Water management and
- Land rehabilitation

With development of the extraction and beneficiation stages, plans to address matters of:

- Atmospheric emissions
- Waste rock management

- Tailings dams
- Revegetation and
- Land rehabilitation

become key issues.

The level of detail in the environmental guidelines published by state governments is extensive. In Western Australia, for example, official environmental guidelines appear in at least sixteen different publications and cover a comprehensive range of issues. These range from procedural advice regarding project approvals and annual environmental reports through to specific guidance on areas such as tailings dams, vat leaching, heap leaching, management of inorganic mercury in gold plants, cyanide and arsenic management, noise control, bound walls around abandoned open pits, ground control, minesite rehabilitation and exploration in environmentally sensitive areas.

Despite the apparent similarity and extent of environmental regulation between the six states and the Northern Territory,[3] it seems generally agreed that it is more difficult to gain environmental approvals and meet environmental standards while operating in some states than in others. New South Wales and Victoria – the states with the largest populations – are presently the most difficult. This challenge seems to arise because aggrieved parties seem capable of extending the process approval period and mounting court challenges without difficulty. Also, where potential mines are located on Commonwealth lands within a state approvals are also difficult. The strong resource states of Western Australia and Queensland are perhaps the most disposed to new mineral and energy developments. As might be expected, closeness to wilderness areas and national parks complicate matters and it is also much more difficult to obtain approvals to mine commodities such as uranium.

2.6
Estimated Spending on the Environment by Mining Companies

While environmental awareness and protection is now an integral part of the operating environment of mining companies in Australia, there is a remarkable paucity of data available about how much is being spent to meet current environmental standards and guidelines. The Australian Bureau of Statistics (ABS) began publishing official data in the area in 1994. The most recent of its publications (Australian Bureau of Statistics 1997) reports environment protection expenditures on an industry by industry basis for the 1992–1993 and 1993–1994 financial years. These data do not apparently include expenditure on minesite rehabilitation. One of the major industry bodies, the Mineral Council of Australia, has independently published estimates of spending in this area for its member companies since 1991–1992[4] (Mineral Council of Australia, various years).

In reporting its data set, the ABS presently uses the United Nations System for Integrating Environmental and Economic Accounting (SEEA). This is based largely on

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[3] We exclude the Australian Capital Territory from our discussion because the level of mining activity there is minimal.

[4] It seems generally accepted that the activities of Mineral Council member companies account for approximately 90% of the mineral and energy sector.

Eurostat's European System for the Collection of Economic Information on the Environment (SERIEE) and includes expenditures on activities such as the protection of biological diversity, soils and water, as well as activities such as waste management and protection of ambient air quality. It is not clear, however, that these data incorporate spending on minesite rehabilitation as such and some other areas of environmentally focused activity.

The national ABS estimates of mining industry expenditure on environment protection in 1992–1993 and 1993–1994 appear in Table 2.1. These percentages of operating expenditure and capital expenditure are considerably less than the 5% figure suggested previously in this paper for new mines by the federal Environmental Protection Agency. Because spending in longer term established mines is generally much less than 5%, this explains part of the discrepancy. There seems a question as well about how much of the salaries of environmental professionals in the minerals sector are included in the ABS survey.

However, it also appears legitimate to add to the data in Table 2.1 the minesite rehabilitation expenditure estimates of 118 million A$, 125 million A$ and 158 million A$ for 1991–1992, 1992–1993, and 1993–1994, respectively (Mineral Council of Australia 1994). Now the relevant percentages can be seen to approach 3% and therefore much more in line with perceived industry standards.

It is also instructive to extend the analysis as much as possible to the state and territory level. Available estimates in this regard appear in Table 2.2. Unfortunately we have been unable to report the key ratio of environmental expenditure to operating costs at this level of disaggregation. What does stand out, however, is a quite wide range of percentage values on a state by state basis for the ratio of environmentally related spending on mining to GDP at factor cost. This raises interesting issues about the extent in variation of environmental spending depending on the minerals being exploited, their location relative to population centers, and the climate and topography of the surrounding region more generally. The published data show higher than average percentages in New South Wales, Queensland, Tasmania, and the Northern Territory in both 1992–1993 and 1993–1994, and lower percentages in Victoria and South Australia. The percentages for the largest mining state – Western Australia – were lower in the first year and higher in the second.

A further area of interest relates to the extent of environmental protection spending by broad industry group. The Australian Bureau of Statistics uses a three-way classifi-

Table 2.1. Mining industry expenditure on environment protection – Australia 1992–1993 and 1993–1994 (Australian Bureau of Statistics 1997)

	1992–1993	1993–1994
Capital spending (million A$)	48.3	57.1
Current (million A$)	95.7	128.8
Total (million A$)	144.0	185.9
Mining's contribution to GDP at factor cost (million A$)	16629	16509
Environmental expenditure/ mining's GDP contribution (%)	0.86	1.12
Environmental expenditure/ operating costs (%)	1.15	1.53

Table 2.2. Mining industry expenditure on environment protection – Australian states and territories (1992–1993 and 1993–1994) (Australian Bureau of Statistics 1997)

	1992–1993	1993–1994
Environmental spending (million A$)		
New South Wales	24.1	22.4
Victoria	5.2	10.5
Queensland	59.3	55.0
South Australia	2.2	3.0
Western Australia	42.5	82.7
Tasmania	1.7	2.2
Northern Territory	9.0	10.1
Mining's contribution to GDP at factor cost (million A$)		
New South Wales	2373	2575
Victoria	2911	3181
Queensland	3370	3053
South Australia	640	578
Western Australia	5701	5993
Tasmania	134	158
Northern Territory	845	614
Environmental spending as percentage of mining GDP at factor cost		
New South Wales	1.02	0.87
Victoria	0.18	0.33
Queensland	1.76	1.80
South Australia	0.34	0.52
Western Australia	0.75	1.38
Tasmania	1.27	1.39
Northern Territory	1.07	1.64

cation – coal mining, oil and gas extraction, and metal ore extraction – in this analysis for both of the reported years. The relevant data appear in Table 2.3.

Oil and gas extraction attract a much smaller percentage of the total than their relative contributions to overall mineral production. As the upswing of the new business cycle proceeded, metal ore mining (including iron ore, bauxite, copper, gold, mineral sands, and base metals) was responsible for a growing percentage share.

2.7
Conclusions

In the late twentieth century mining companies which operate in Australia devote considerable resources to environmental protection. Answering the question of whether they pay too much, too little, or the right amount to meet their environmental commitments depends clearly on one's normative view about the way the world should be.

What is very clear is that for much of the period since the mid-1960s the industry has been both profitable and prosperous. As such it appears to have been an easier target for the environmental lobby in some states than others. In those jurisdictions those who want to ensure that the impact of mining on the environment is zero, or less than good economic theory would suggest, have been influential. To overcome a previously tainted image mining companies have also been keen to play the role of good environmental citizens. Particularly in the more densely populated states and in smaller

Table 2.3. Environmental protection expenditure by major groups within the mining industry – Australia 1992–1993 and 1993–1994 (Australian Bureau of Statistics 1997)

	1992–1993 (million A$)	(%)	1993–1994 (million A$)	(%)
Coal mining	58.2	40.4	64.4	34.6
Oil and gas extraction	13.3	9.2	12.4	6.7
Metal ore extraction	72.5	50.3	109.1	58.7
Total	144.0		185.9	

sub-state regions where mining damage has previously been significant, miners have faced greater challenges in gaining environmental approvals for new projects. It seems possible that in these areas they are paying too much to meet their environmental compliance commitments.

The situation in the pro-mining states of Western Australia and Queensland has generally been more favorable to mining interests. Our view is that the mining sectors in these states have probably operated in close to an environmentally optimal way. Elsewhere this seems to have been less likely.

An interesting additional view which has come through in this paper is that various branches of government in Australia have formally recognized the better recent environmental record of mining companies. With the passage of time and concerns with other environmental problems pressure on mining companies may begin to subside. It seems clear, however, that mining executives must continue to place environmental management issues at a very high level if they are to continue to operate profitably.

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Chapter 3

Characterization of Mine Wastes for Prediction of Acid Mine Drainage

A.Dogan Paktunc

3.1
Introduction

One of the environmental challenges the mining and minerals industry is facing today is the increased levels of metals in the environment due to mining, milling, and refining activities. Liquid effluents, mine waters, tailings, waste rocks, and dusts may contribute to the generation of potentially deleterious waste and contaminated groundwater and soil deposits in, around, and outside mine environments. Tailings which are unwanted silicate, oxide and sulfide minerals discarded during ore processing operations, are usually discharged into impoundments as a slurry with particle sizes that are predominantly in the silt to fine sand size range (1 µm–1 mm). Waste rock is essentially wall rock material removed to access and mine ore. Waste rock material, composed of particles ranging in size from silt to boulder size fragments, are often disposed of in large piles.

Mine wastes include metals, radionuclides and various soluble anions. Release of dissolved metals from mine wastes is aided by the generation of acid. Along with the dissolved contaminants, the acidic effluents are generally referred to as acid mine drainage (AMD) or acid rock drainage (ARD). AMD is caused by the oxidation of sulfides such as pyrite and pyrrhotite. AMD is one of the largest single environmental problems facing the world's metal mining industry today with Canadian liabilities ranging from 2 to 5 billion $ for the existing 12 500 ha of acid-generating tailings and 750 million tons of acid-generating waste rock (Feasby and Tremblay 1995). Weatherell *et al.* (1997) estimated that the total worldwide liability related to AMD to be in excess of 10 billion US$. Because AMD generation can result in surface and groundwater contamination that requires expensive water treatment techniques and involves potential liability in perpetuity, accurate prediction of acid producing potential through chemical and mineralogical characterization and geochemical modeling is essential (White and Jeffers 1994).

Because solubility, mobility, and retardation of contaminants are controlled by the physical and geochemical properties of waste material, prediction of contaminant releases requires a comprehensive characterization of the waste material. The ultimate goal of any waste characterization study is to provide an adequate level of information for predicting the long-term contaminant behavior and for assessing potential environmental impacts with confidence. Quality of assessments will have important implications on defining the mitigation measures or optimizing design systems to reduce potentially adverse effects. The purpose of this chapter is to review issues relevant to characterization of mine wastes for the prediction of acid mine drainage. A new approach, based on a theoretical analysis of acid producing and neutralization reactions, and their interpretation within a kinetic framework, is also provided. Although the approach has not yet been critiqued, its direct relevance to the review subject as well as its theoretical nature warrant its inclusion in this chapter as a contribution to the existing prediction techniques.

3.2
Factors Controlling Acid Mine Drainage

The quality and magnitude of metal releases accompanied by AMD from waste rock piles and tailings impoundments are influenced by many factors. These factors can be grouped under two broad categories: (1) source factors/characteristics such as types, abundance and distribution of sulfides and minerals with acid neutralizing capacity, rock type and texture, particle size and distribution, pore spaces and their interconnectivity (porosity and permeability), degree of compaction and segregation, and (2) environmental and other factors such as climate, rainfall, air temperature, hydrogeologic setting, waste volume, and disposal methods. These factors define infiltration rates, oxidation rates, temperature variations within waste material, and pore water and seepage chemistry.

Among the source factors, mineralogical composition or more specifically the abundance and reactivity of sulfides and neutralizing minerals is an important characteristic that influences the magnitude and quality of AMD. Dissolution rates of the acid neutralizing minerals are different; therefore, their overall effective contribution to the neutralization of the acidity caused by the oxidation of the contained sulfides in the rock, may be limited. Another equally important characteristic is the availability of sulfides and neutralizing minerals to oxidation and reduction reactions. In the case of waste rock, the heterogeneity in rock texture (e.g. schistosity, microfractures, brecciation) would influence preferential fracturing and weathering potential of a sulfidic rock in waste rock piles. This, in turn, effects the availability of sulfide particles to oxidation. For instance, mineral transformations resulting from oxidation reactions occurring along microfracture planes may involve volume changes. Such changes may increase the weathering rate and oxidation potential of rock masses by creating pathways for air and groundwater.

Although the source factors are somewhat similar, there are significant differences between the tailings impoundments and waste rock piles in terms of volume, nature, occurrence, and distribution of the source material. For instance, waste rock piles have much greater volumes, greater height-to-base ratio, and consist of much coarser and heterogeneous material in comparison with tailings deposits (Ritchie 1994a). Whereas the environmental factors/characteristics can vary significantly within a waste rock pile, tailings impoundments may possess relatively homogeneous characteristics. In broad terms, three distinct domains possessing different characteristics, can be identified in a waste rock pile. These are: (1) outer unsaturated zone, (2) unsaturated inner zone, and (3) saturated lower zone. The outer unsaturated zone typically has the following characteristics:

- High oxygen concentrations (up to 20%)
- Water infiltration rates determined by rainfall, permeability structure at the surface of the pile, and temperature
- It may contain open channels and high permeability
- It is usually dominated by advective transport of oxygen
- Relatively high moisture content
- Highly oxidized
- Higher rate of mechanical disintegration

The unsaturated inner zone can have the following characteristics:

- Layers of variably compacted material depending on the type of pile construction and addition of gravel for road construction
- Variable oxygen concentrations from pile to pile
- Zones of relatively high temperature defining the position of the reaction front within the pile
- Moisture content depending on grain size distribution and transport of water and oxygen

The saturated lower zone can be characterized by the following:

- Relatively well compacted
- Very low oxygen

The types and rates of reactions, and resulting products are expected to be different in these zones. For instance, the outer zone of a mature waste rock pile is expected to have low levels of sulfide minerals present in the fine fraction. It should also be depleted in readily soluble components and rich in insoluble minerals. The inner unsaturated zone, on the other hand, should be enriched in insoluble oxidation products and secondary minerals. In the inner zone, oxidation of sulfides in the fine fraction and exposed on the surfaces of rock particles, should occur along a front moving slowly with depth and time. If the distribution of sulfides in different size fractions is uniform, the waste rock pile will be the source of AMD for a prolonged period of time as more sulfides will be exposed to oxidation resulting from rock disintegration. Development of preventive and mitigation measures for environmental protection can not be properly achieved without establishing sound relationships between these source and environmental factors.

Smith *et al.* (1995) proposed four hydrostratigraphic models to characterize the hydrogeology of waste rock piles. Different hydrogeologic conditions in the unsaturated zones may have significant effects on the effluent characteristics. In the case of porous flow, the more fresh rainwater runs through the pile, the more rock particles will have a chance to react with the fresh water which will be undersaturated with respect to metals and suspended particles; therefore, it will have a greater capacity as a solvent as it percolates through the pile. On the other hand, if infiltrating rain water follows open channels or other discrete flowpaths, water-rock interactions will have a spatial limitation. The latter case is supported by the Eriksson and Destouni's (1997) work which identified preferential flow as an important hydrogeologic process operating at the Aitik waste rock pile in Sweden (Chap. 13, this Vol.). Heavy rainstorm events may cause overflow in open channels and the excess water may be forced to infiltrate through the fine particles matrix. Such events may also cause excessive erosion along the walls of the conduits.

There are difficulties in the conceptualization of complex interactions between chemical species, water and air, and coupled processes in such heterogeneous systems. For instance, the relative importance of high permeability features (open channels) and the matrix permeability on the water flow and the exchange of fluids and dissolved species between these two regions may be critical in the prediction of AMD. There is, however, some progress in this area as Smith *et al.* (1995) indicated that the kinematic wave approach is promising in terms of characterizing large-scale flow processes within waste rock piles and Eriksson and Destouni's (1997) probabilistic approach to identification of processes to which the modeling calculations are sensitive.

3.3
Prediction

Prediction involves the assessment of AMD potential of mine wastes in terms of its magnitude, duration and long term effects, and accompanying metal releases to the environment. Prediction is usually made through the use of static tests, kinetic tests, geochemical modeling, and expert interpretation of data collected in the field and laboratory. It can also be based on the empirical data derived from observations and monitoring in existing wastes (e.g. Morin *et al.* 1995; Norecol *et al.* 1996). Prediction techniques depend on: (1) the ability to determine the acid generation potential of sulfides in comparison with the neutralization potential of the wastes, (2) an understanding of the oxidation, dissolution and biochemical reactions, and (3) the consideration of the site specific conditions (Coastec Research 1991). Furthermore, an understanding of the system as a whole is required for a successful prediction.

Characterization of waste rock and tailings can be carried out in the field and in the laboratory. Laboratory tests include chemical analyses, acid-base accounting (ABA) tests, mineralogical analysis, humidity tests, and column tests. In addition to tests conducted on actual waste rock piles and tailings impoundments, field tests may involve a large amount of material placed in constructed test cells or less than 1 ton of material in barrels. Kinetic tests involve accelerated dissolution of material under laboratory and field conditions to predict long-term acid generation and neutralization potentials, and provide dissolution characteristics as a function of time. The scale and magnitude of each test and information sought from these tests differ. As effective prediction techniques require an understanding of the acid producing and neutralization reactions, robustness of geochemical calculations and modeling to simulate actual field conditions as close as possible is essential. Geochemical models must incorporate relevant acid producing and neutralization reactions and their rates as well as precipitation of secondary minerals such as oxyhydroxides, carbonates and clay minerals. A review of relevant geochemical models is given by Perkins *et al.* (1995). Simulation of physical and geochemical processes in waste rock piles is more difficult to mimic because of their heterogeneous nature.

3.4
Acid Producing and Neutralization Potentials

3.4.1
Acid Producing Reactions

Oxidation of sulfide minerals is the primary cause of acid production in mine wastes. For instance, oxidation of pyrite produces two moles of H^+ initially. Subsequent oxidation and precipitation reactions produce an additional two moles of H^+. Overall, the oxidation of one mole of pyrite produces four moles of H^+:

$$FeS_2 + 15/4\,O_2 + 7/2\,H_2O \text{ fi } Fe(OH)_3 + 2\,SO_4^{2-} + 4\,H^+.$$

Oxidation of pyrrhotite produces less acid than pyrite:

$$FeS + 9/4\,O_2 + 5/2\,H_2O \text{ fi } Fe(OH)_3 + SO_4 + 2\,H^+.$$

Precipitation of secondary minerals including carbonates, sulfates, oxyhydroxides and clay minerals may contribute to acid generation. Perkins *et al.* (1995) stressed the importance of secondary reaction products such as melanterite and ferrihydrite in controlling the acidity of solution.

Pyrite oxidation rates are in the $1.08 \cdot 10^{-15}$ to $1.8 \cdot 10^{-14}$ mol $cm^{-2}\,s^{-1}$ range (Nicholson 1994). These rates are increased by about 10^2 higher in the presence of ferric iron in solution in addition to dissolved oxygen (Nicholson 1994; Williamson and Rimstidt 1994). Oxidative dissolution of pyrite by Fe(III), in contrast to that by O_2, shows a marked pH dependence (Hering and Stumm 1990). Presence of bacteria can also increase dissolution rates by up to 100 times (Nicholson 1994). Additionally, crystallinity and type may have some influence on the oxidation rates. For instance, fine grained and poorly crystalline mineral grains oxidize faster than coarse and crystalline grains, and some coal pyrites were found to be more reactive than most non-coal pyrites. The latter was ascribed by Martello *et al.* (1994) to the density of stacking faults. Pyrrhotite is 20 to 100 times more reactive than pyrite (Nicholson 1994). Non-oxidative dissolution rate of pyrrhotite is $5 \cdot 10^{-14}$ mol $cm^{-2}\,s^{-1}$ whereas oxidation rate in the presence of ferric iron is on average $3.5 \cdot 10^{-12}$ mol $cm^{-2}\,s^{-1}$ (Nicholson *et al.* 1997).

The presence of mineral associations involving two different sulfide or oxide minerals with a mutual grain boundary may be of importance by providing a potential for galvanic reaction (Kwong 1995; Perkins *et al.* 1995). Since galvanic reactions can oxidize sulfide minerals independently or protect a sulfide mineral from oxidation, mineral associations involving pairs of sulfide and/or oxide minerals may be important in assessing the likelihood of galvanic reactions occurring and their contribution to AMD.

3.4.2
Neutralization Capacity and Reactions

Rates and mechanisms of dissolution reactions are important in terms of predicting the neutralization potentials of minerals that are commonly present in waste rock and

Table 3.1. Maximum neutralization capacity of some minerals[a]

Neutralizing mineral	Formula	Moles needed to neutralize 1 mol sulfuric acid ($n_{M,i}$)
Calcite	$CaCO_3$	1
Dolomite	$Ca_{0.5}Mg_{0.5}CO_3$	1
Ankerite	$Ca_{0.5}Fe_{0.2}\,Mg_{0.3}CO_3$	1.25
Chlorite	$Mg_5Al_2Si_3O_{10}(OH)_8$	0.125
Muscovite	$KAl_2(Si_3Al)O_{10}(OH)_2$	0.2
Kaolinite	$Al_2Si_2O_5(OH)_4$	0.67
Olivine (Fo_{100})	Mg_2SiO_4	0.5
Hornblende	$Ca_{1.7}Mg_{3.5}Fe_{1.3}Al_{1.3}Si_7O_{22}(OH)_2$	0.11
K-feldspar	$KAlSi_3O_8$	0.5
Plagioclase (An_0)	$NaAlSi_3O_8$	0.5
Plagioclase (An_{100})	$CaAl_2Si_2O_8$	0.25
Goethite	$Fe(OH)_3$	0.67
Brucite	$Mg(OH)_2$	1
Gibbsite	$Al(OH)_3$	0.67

[a] It is assumed that mineral dissolves completely in solution and not accompanied by precipitation of secondary phases.

tailings. Many rock-forming minerals have the capacity to neutralize acid upon dissolution including carbonates, silicates, oxides, oxyhydroxides, and hydroxides. In the case of carbonates, 2 mol is required to neutralize the acid generated by the oxidation of a single mole of pyrite. The situation is different for Fe-bearing carbonates. For instance, although siderite has initially the same capacity as calcite, oxidation of ferrous iron to ferric iron and its precipitation as ferric hydroxide produce acidity and eventually cancel acid neutralization.

Some rock-forming minerals commonly present in waste rock and tailings have higher buffering capacities than carbonates. For instance, plagioclase and K-feldspars have the potential buffering capacities that are 2 to 4 times greater than that of calcite (Table 3.1). However, the dissolution rates of silicates are slower than that of calcite, as discussed in a following section. Stoichiometry of the reaction between chlorite $[Mg_5Al_2Si_3O_{10}(OH)_8]$ and sulfuric acid requires 0.125 mol chlorite to neutralize 1 mol sulfuric acid:

$$Mg_5Al_2Si_3O_{10}(OH)_8 + 8\ H_2SO_4 \rightarrow 5\ Mg + 2\ Al + 3\ SiO_2 + 8\ SO_4 + 12\ H_2O.$$

Similarly, 1 mol muscovite has the potential to neutralize 5 mol sulfuric acid. Neutralization potentials of common rock-forming minerals are listed in Table 3.1. The values in the table should be considered as maximum potentials since they assume complete dissolution without the precipitation of secondary phases. Incongruent dissolution behavior of some silicate minerals will have an effect on these values. For instance, in the case of kaolinite or gibbsite precipitation during the dissolution of albite, 4 mol albite, instead of 1, will be needed to neutralize 2 mol sulfuric acid. Although both kaolinite and gibbsite will provide additional capacity to neutralize acidity (Table 3.1), their dissolution rates are much slower than their parents; therefore, their overall contribution to the neutralization potential would be insignificant.

Thus, depending upon the kinetics of the oxidation and neutralization reactions, most rock forming minerals may be effective in terms of providing additional long-term buffering capacity to the mine wastes.

3.4.3
Assessment of Acid Generation and Neutralization Potentials

Acid-base accounting tests (Coastec Research 1991) are commonly used to determine the balance between acid producing and acid consuming components of a mine waste. In these tests, acid potential (AP) of samples is determined based on whole rock sulfur analyses. The AP in tons $CaCO_3$ equivalent per 1000 tons is given by:

$$AP = C_R^S \times 31.25,$$

where C_R^S is the concentration of S in wt% in the sample. The calculation assumes that: (1) all S is coming from pyrite in the sample, (2) 4 mol H^+ results from the oxidation of 1 mol pyrite, and (3) all S is converted to sulfate. The modified acid-base accounting method makes a correction by considering sulfide sulfur only after subtracting sulfur occurring as sulfates from the total sulfur analyses. Neutralization potentials are determined by wet chemical techniques; however, the NP measurements and their interpretation are by no means straightforward. In recognition of the difficulties in NP determinations, Lawrence and Wang (1997) recommended a set of instructions and complementary tests for a better appreciation of the overall potential for AMD.

In support of the above, a new approach to static testing is proposed here to eliminate the difficulties associated with the determination of NP values by conventional chemical techniques and their interpretation. This new approach provides an unambiguous NP value for each mineral that is deemed to have a neutralization potential and a bulk NP value for the sample. In this new approach, the AP is calculated to take into consideration the presence of other sulfides and more than one type of neutralizing mineral:

$$BAP = \sum_{s=1}^{k} \frac{n_{M,a}\, \omega_a X_s\, 10}{\omega_s},$$

where $n_{M,a}$ is the stoichiometric factor (moles of sulfuric acid formed by the oxidation of 1 mol sulfide mineral; e.g. $n_{M,a}$ is 2 for pyrite and 1 for pyrrhotite), w_a is the molecular weight of sulfuric acid (g mol^{-1}), w_s is the molecular weight of sulfide mineral s, X_s is the amount of sulfide mineral s in the sample (wt%), 10 is the conversion factor for recasting in kilograms per ton and k is the number of sulfide minerals in the sample. The BAP, in kilograms of sulfuric acid equivalent per ton, is the bulk acid potential. Alternatively, if only the amount of bulk S is available, X_s can be defined as:

$$X_s = \frac{C_R 100}{C_s},$$

where C_R is the concentration of sulfide S (wt%) in the sample, C_s is the weight percentage of S in the sulfide mineral *s* (e.g. C_s is 54 wt% for pyrite and 37 wt% for pyrrhotite). Similarly, in order to take into account the presence of more than one type of neutralizing mineral in the samples, neutralization potentials of samples can be calculated based on sulfuric acid equivalent rather than $CaCO_3$ equivalent. With this approach, a bulk neutralizing potential (BNP) in kilograms of sulfuric acid equivalent per ton can be calculated:

$$BNP = \sum_{i=1}^{k} \frac{\omega_a X_i 10}{n_{M,i}\, \omega_i},$$

where X_i is the amount of mineral *i* in wt%, w_i is the molecular weight of neutralizing mineral *i* (g mol^{-1}), $n_{M,i}$ is the stoichiometric factor (moles of neutralizing mineral required to consume 1 mol sulfuric acid; see Table 3.1 for a list of values), 10 is the conversion factor for recasting in kilograms per ton and *k* is the number of neutralizing minerals in the sample.

The net neutralization potential (NNP) which is the balance between NP and AP, is used to assess the potential of mines wastes to generate an acidic drainage. Negative NNP values are suggestive of a potential acidic drainage. Additionally, NP/AP (NPR)

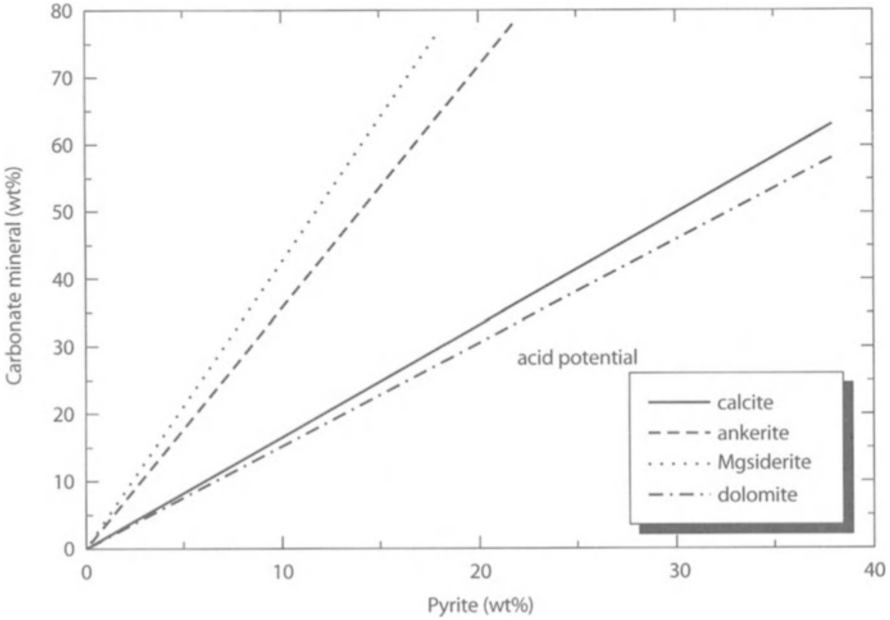


Fig. 3.1. Acid and neutralization potentials based on carbonate and pyrite abundances (wt%)

ratios are used as a supplementary criterion for assessing the potential for acid generation. In the case of the proposed new approach, the net neutralizing potential will be:

$$NNP = BNP - BAP \ .$$

Assessment of neutralization potentials can also be made on graphs such as the one given on Fig. 3.1 where one axis is represented by pyrite abundances in wt% and the other by carbonate mineral abundances. Data points plotted on the right sides of lines representing *NNP* values of 0 are considered to be acid producing. *NNP* values increase to the left away from the "0" lines. Such graphs can also be used to estimate the amount of various neutralizing minerals required to neutralize a given concentration of sulfide.

3.5
Assessment of Kinetic Factors

3.5.1
Dissolution Rates and Mechanisms

A compilation of dissolution rates of various rock-forming minerals with neutralization potentials is given in Table 3.2. Two sets of dissolution rates are compiled: one is representative of acidic environments (pH = 2) and the other near neutral (pH = 5) conditions. Some of these are approximate as they are obtained by extrapolating experimental data to pH 2 and 5 from other pH values. As also illustrated graphically on Figs 3.2 and 3.3, the dissolution rates differ by many orders of magnitude from clay minerals to plagioclase (An_{100}). With a dissolution rate of $7.08 \cdot 10^{-10}$ mol cm^{-2} s^{-1} at low pH (extrapolated from data in Blum and Stillings 1995), the calcium end-member of plagioclase appears to be one of the fast dissolving silicates. Spodumene, a Li-pyroxene and brucite, a Mg-hydroxide display similar dissolution rates. In comparison with calcite and dolomite, these minerals display dissolution rates that are 100 to 100 000 times slower.

Among the common-rock forming minerals, K-feldspars and Na-rich plagioclases have rather slow dissolution rates. In comparison with quartz which is considered to be relatively inert or slow dissolving, K-feldspar and Na-rich plagioclase rates are only about 10 to 100 times faster. Feldspar dissolution rates are increased above and below neutral pH (White and Brantley 1995). The dissolution rate of albite is generally insensitive to pH at or near neutral values but increases with the –0.5 power of pH in acidic solutions (Blum and Stillings 1995). This suggests that the weathering rates may increase by a factor of 3 from pH 4 to 5 (White and Brantley 1995). Sheet silicates, including various clay minerals also have slow dissolution rates. Muscovite is relatively stable at near neutral pH and ambient temperature conditions (e.g. $8.5 \cdot 10^{-18}$ to $3.2 \cdot 10^{-17}$ mol cm^{-2} s^{-1}). Biotite and chlorite can be considered as slightly more reactive than muscovite. The presence in the mineral structure of Fe^{2+} or other cations that may oxidize, reduces the structural stability; therefore, the rates for ferromagnesian minerals may be higher depending upon their Mg to Fe ratios.

Among the list, anorthite, wollastonite, and Fe-rich olivine and perhaps Fe-rich pyroxenes and amphiboles have dissolution rates under acidic conditions that are comparable with the oxidation rates of pyrite; therefore, these minerals could be considered as significant in terms of having potentials to neutralize acidity. With the excep-

Table 3.2. Dissolution rates of various neutralizing minerals at pH 2 and 5

Neutralizing mineral	R (mol cm^{-2}s^{-1})		Reference[a]
	@ pH = 2	@ pH = 5	
Quartz		4.07E–18	1
K-feldspar	1.58E–15	3.16E–17 - 2.51E–16	1, 2
Plagioclase (An_0)	3.16E–15 - 2.51E–16	1.00E–16 - 7.94E–17	2, 3
Plagioclase (An_{20})	4.79E–16	1.00E–16	4, 5
Plagioclase (An_{40})	1.00E–14 - 9.12E–16	1.58E–16 - 5.01E–16	2, 5
Plagioclase (An_{60})	1.74E–15		5
Plagioclase (An_{80})	2.00E–13 - 3.31E–15	1.58E–15	2, 5
Plagioclase (An_{100})	7.08E–10 - 1.00E–11	1.00E–12 - 2.82E–13	1, 2
Muscovite	5.01E–17	3.16E–17 - 8.51E–18	1
Biotite	1.70E–15 - 5.01E–16	2.75E–16 - 8.91E–17	6
Phlogopite		3.80E–17	7
Chlorite		3.00E–17	7
Olivine (Fo_0)	3.98E–11	3.80E–11 - 4.60E–12	8, 9
Olivine (Fo_{91})	1.02E–12	8.32E–14 - 1.86E–13	10
Olivine (Fo_{100})	1.00E–12 - 1.58E–13		9, 10
Augite	5.62E–14	1.00E–16 - 1.58E–15	11
Diopside	1.00E–14 - 6.31E–13	7.94E–15 - 3.16E–15	1, 11
Enstatite	6.31E–15 - 2.00E–14	1.00E–16 - 1.00E–14	1, 11
Hornblende	5.62E–13	6.31E–16 - 5.01E–15	3, 11
Glaucophane	1.58E–12	1.58E–14	11
Jadeite	1.00E–12	5.01E–14	11
Spodumene	3.16E–10	1.78E–12	11
Anthophyllite	1.00E–17	1.00E–18	11
Wollastonite	6.31E–13 - 1.26E–12	1.58E–13 - 1.00E–12	1, 11
Epidote		2.45E–17	1
Nepheline		2.82E–13	1
Kaolinite	2.24E–17 - 1.00E–16	5.25E–18 - 3.55E–17	1, 7
Montmorillonite		4.00E–18	7
Serpentine (antigorite)		1.00E–15	8
Talc		1.00E–16	7
Brucite		6.00E–10	7
Magnetite	8.00E–13		8
Gibbsite		3.00E–17 - 3.55E–16	1, 7
Dolomite	1.12E–09	3.16E–11	13
Calcite	3.16E–07	1.78E–10 - 5.62E–10	12, 14

Kinetic data at 25 °C but some measured at 20 and 22 °C; some values are extrapolations; data at pH= 2 also include rates measured at pH 3. [a] *1* Lasaga *et al.* (1994); *2* Blum and Stillings (1995); *3* White *et al.* (1996); *4* White (1995); *5* Stilings and Brantley (1995); *6* Acker and Bricker (1992); *7* Nagy (1995); *8* Heator and Engstrom (1994); *9* Sverjensky (1992); *10* Wogelius and Walther (1991); *11* Brantley and Chen (1995); *12* Morse (1983); *13* Busenberg and Plummer (1982); *14* Plummer *et al.* (1978).

Fig. 3.2. Comparative illustration of mineral dissolution rates at pH 5 in order of decreasing rates

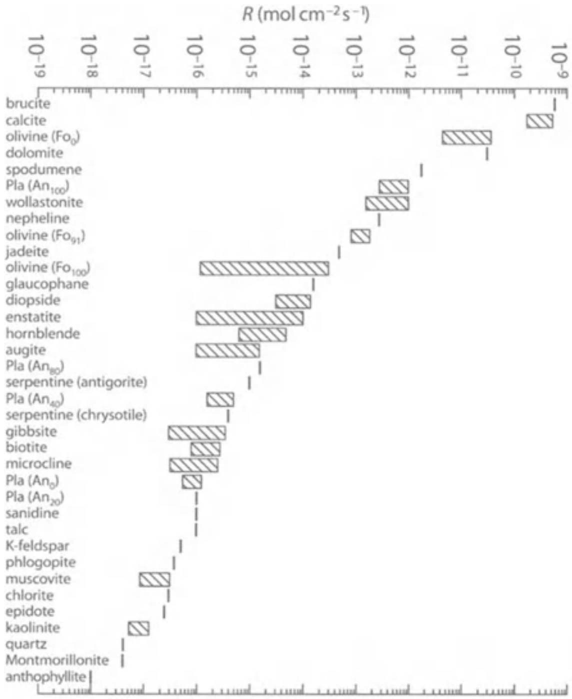
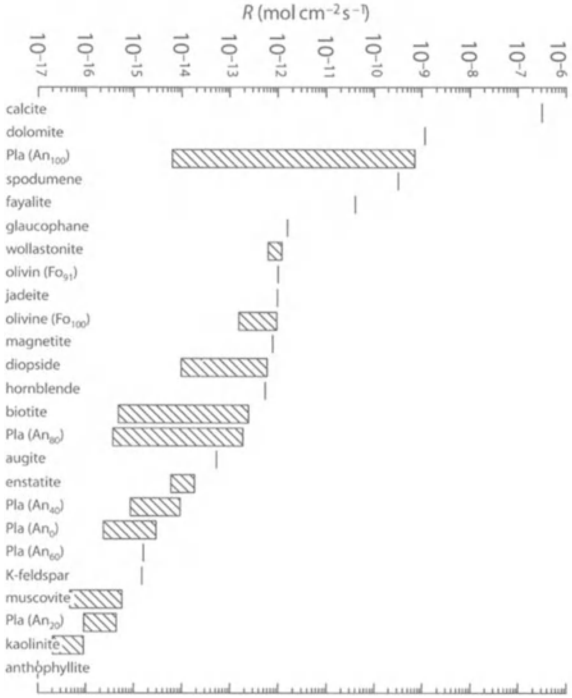


Fig. 3.3. Comparative illustration of mineral dissolution rates at pH 2 in order of decreasing rates



tion of wollastonite, others are common rock forming minerals. According to Lapakko (1988), rapid acid consumption in Duluth waste rock was probably due to the dissolution of plagioclase, olivine and pyroxene.

In addition to solution pH, dissolution rates are dependent upon composition, solution ionic strength, temperature, and the presence of protective coatings or adsorbed inhibitors on particle surfaces. Dissolution rates are also controlled by the mineral chemistry such as the presence of cations that oxidize (e.g. Fe^{2+}). Sheet silicates such as micas, chlorite and clay minerals dissolve non-stoichiometrically by releasing octahedral cations (i.e. Al, Mg, Fe^{2+}, Ti, Cr, V) more readily than tetrahedral cations (Si, Al, Fe^{3+}) (Nagy 1995). Similarly, Brantley and Chen (1995) state that the dissolution of hornblende is non-stoichiometric with very fast early release of Ca. Additional factors to be considered include the presence of inhibitors and catalysts in solution. For instance the presence of aluminum in the solution may inhibit the dissolution of aluminosilicates. Similarly, organic ligands and increased dissolved salt concentrations may act as catalysts. Plants accelerate weathering in order to obtain nutrient elements from rocks (Berner 1995).

Different weathering reactions depend on variables such as soil solution concentrations of Al, Ca, Mg, Na, K, CO_2, organic acids and pH (Sverdrup and Warfvinge 1995).

3.5.2
Effect of Particle Size, Shape and Roughness on Dissolution

Because small particles have larger surface areas per unit mass, minerals in small particle size ranges will have more surface areas available for reaction and therefore dissolve more rapidly. For spherical and cubic particles, specific surface area S_s ($cm^2\,g^{-1}$) is defined as:

$$S_s = \frac{6}{\rho d},$$

where d is particle diameter or size (cm) and ρ is the mineral density ($g\,cm^{-3}$) (Anbeek 1994).

Particle shape is also important as the total surface area for a given sized particle varies by particle shape. For instance, platy grains have larger specific surface areas than cube-shaped, prismatic or rod-shaped grains. If particle shapes of minerals display a uniformity or the breakage habit of mineral grains is known, particle shape can be factored into the calculations. When it is assumed that particles have about equal length and breadth which define their size,[1] the thickness is greater than length for prismatic or rod-shaped particles, and thickness is shorter than length for platy mineral particles. The formula for specific surface areas of particles then becomes:

$$S_s = \frac{2(1+2\alpha)}{\rho \alpha d},$$

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[1] Sieve analysis by widely used Rotap shaker involves rotary and tapping motion, thereby maximizing the passage of particles through the nested sieves. In other words, particles with long aspect ratios – that is having thickness greater than screen openings – will pass through.

where a is the aspect ratio which is defined as the ratio of thickness to length. The aspect ratio will be equal to 1 for equigranular particles, less than 1 for platy particles and greater than 1 for prismatic particles. If the shape factor is taken into consideration in the calculations, prismatic particles will have less specific surface area whereas platy particles will have greater specific surface area in comparison with equivalent spherical or cubic particles.

Surface roughness is described as the ratio of the measured surface area based on gas sorption techniques such as BET to the equivalent geometric surface area based on particle size (White 1995). Incorporating the surface roughness factor, the specific surface area equation becomes:

$$S_s = \frac{2\lambda(1 + 2\alpha)}{\rho \alpha d} ,$$

where λ is the surface roughness factor. A surface roughness factor of 1 means that particles are smooth with surface areas equal to their geometric surface areas. White (1995) ascribed a mean surface roughness factor of 7 for a wide range of fresh mineral particles, 2.2 for crushed quartz grains, and 50–200 for primary silicate grains in sand size fraction of soil developed over weathered granitic bedrock. According to Blum and Stillings (1995), relatively unweathered feldspar particles have surface roughness factors in the 3 to 15 range. Surface areas of pyrrhotite samples with variable crystalline structures are 6 to 40 times greater than the theoretical geometric specific surface areas (Nicholson *et al.* 1997).

3.5.3
Assessment of Kinetics of Acid Production and Neutralization

An assessment of time frames involved in the dissolution of minerals can be made by the formulation of Lasaga (1984):

$$t = \frac{3.1536 \times 10^{-7} (d_0 - d_t)}{V_i R_i} ,$$

where d_0 is the initial particle diameter (cm), d_t is the diameter at time t, V_i is the molar volume of mineral i ($\mathrm{cm^3\,mol^{-1}}$), R_i is the dissolution rate of mineral i ($\mathrm{mol\,cm^{-2}\,s^{-1}}$) and t in years. Based on this formulation, a 5 mm anorthite grain will completely dissolve in a period ranging from about 3 months to 558 years, depending upon the choice of dissolution rates. In comparison, a similar sized calcite grain will dissolve within less than 15 h to 2.5 years.

Alternatively, oxidation of sulfides and dissolution of minerals can be modeled by using the shrinking core model. In this model, unreacted cores of particles shrink with time as the reaction front moves inward resulting from reactions with a liquid or gas. This model involves the following steps: (1) mass transport of the reactant through stagnant film around the original particle, (2) diffusion of the reactant through the product layer enveloping the unreacted core, (3) reaction of the reactant with the particle, (4) diffusion of products through product layer to the original particle surface, and (5) mass transport of products through stagnant film to bulk solution. Application of

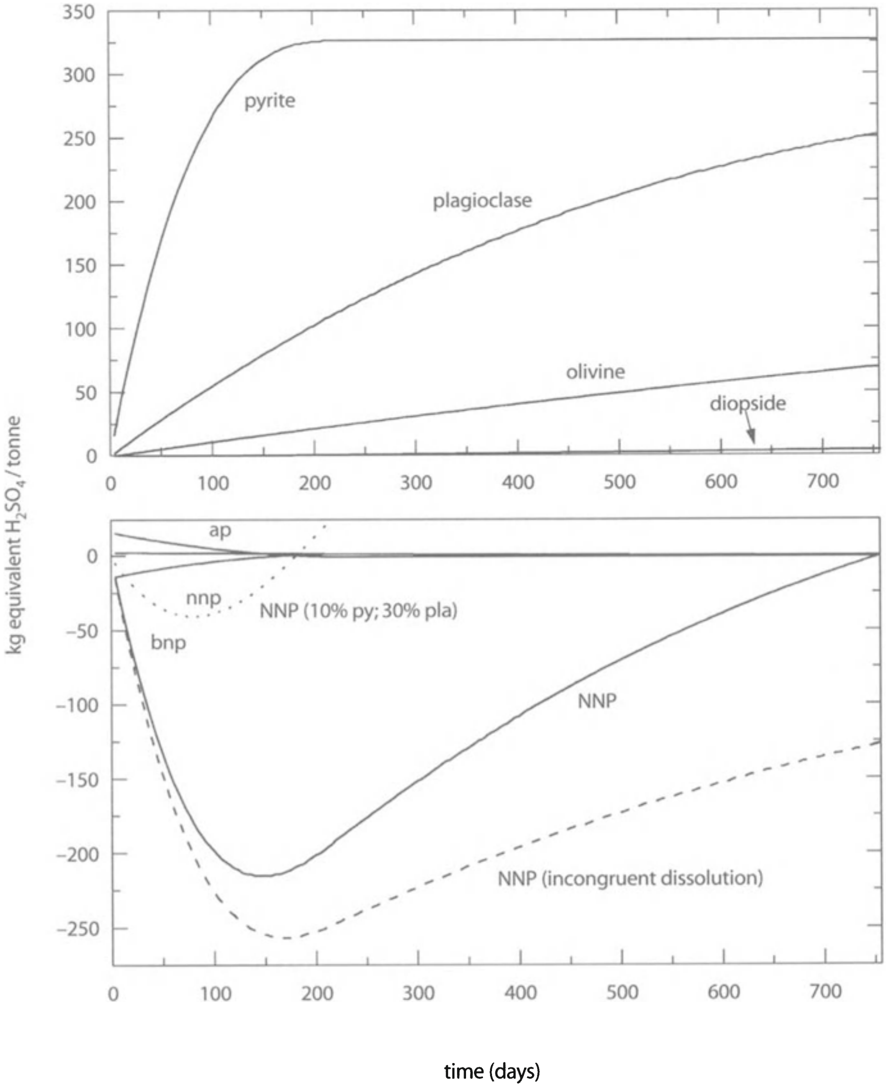

Fig. 3.4. *Above:* Variation in dissolved minerals expressed as cumulative acid and neutralization potentials (kg equivalent H_2SO_4 per tonne) with time for hypothetical tailings material consisting of 10% olivine, 20% diopside, 20% plagioclase, 30% chlorite and 20% pyrite with an average particle size of 50 µm. Surface roughness factor (λ) is assumed to be 1 and calculations performed at time increments of 4 days. *Below:* Variation of net neutralization potentials for dissolved minerals with time for the same material. *Solid NNP line* results where plagioclase completely dissolves. The case where plagioclase dissolution is incongruent is represented by *dashed line. Dotted NNP curve* is for 30% plagioclase and 10% pyrite. *Curves ap, bnp* and *nnp* are acid potential, bulk neutralization potential and net neutralization potential at each time increment, respectively

this model to reactions involving a gas phase is described by Levenspiel (1972) and to the assessment of pyrite oxidation rates by Nicholson *et al.* (1990) and Ritchie (1994b). It should be noted, however, that this model may be inadequate for long-term AMD prediction in waste rock piles as Otwinowski (1997) pointed out that the long-term oxidation rates are inversely proportional to the square of particle diameter because transport of oxygen in water becomes the rate limiting factor at later stages of oxidation.

Time involved in the reduction of a particle with the initial diameter of d_0 to d_t is expressed by the following formulas:
in the case where diffusion through the product layer is rate-determining

$$t_1 = \frac{2\rho_s d_0^2}{3sD_e C_R} \times \left[1 - 3 \times \left(\frac{d_t}{d_0}\right)^2 + 2 \times \left(\frac{d_t}{d_0}\right)^3\right];$$

in the case where surface chemical reaction is rate-determining

$$t_2 = \frac{\rho_s d_0}{s k_s C_R} \times \left(1 - \frac{d_t}{d_0}\right);$$

overall time in seconds is

$$t = t_1 + t_2$$

where ρ_s is the molar density of the reacting particle (mol cm^{-3}), s is the stoichiometric coefficient (moles of reacting particle/moles of reactant), D_e is the effective diffusion coefficient of the reactant in the product layer ($cm^2\ s^{-1}$), C_R is the concentration of the reactant (mol cm^{-3}), d_t is the diameter of remaining unreacted core (cm), d_0 is the original particle diameter (cm) and k_s is the reaction rate constant (cm s^{-1}).

An alternate approach to assess time frames of mineral dissolution is to employ an iterative approach similar to that of White *et al.* (1996). In this approach, masses of minerals dissolved at certain time intervals are calculated based on mineral surface area in each size fraction and mineral dissolution rates. The mass dissolved for mineral *i* during the period Δt is defined by:

$$M^j_{i,\Delta t} = M^j_i S^j_{S,i} R_i \Delta t\, \omega_i ,$$

where M^j_i is the total mineral mass (g) in the size fraction *j*, $S^j_{S,i}$ is the specific surface area for mineral *i* in size fraction *j* and w_i is the molecular weight (mol g^{-1}) of mineral *i*. Following subtraction of the mass dissolved, new nominal particle sizes and corresponding total surface areas are calculated. Calculations are performed iteratively until all the particles are completely dissolved.

In order to illustrate the magnitude of dissolution, oxidation and neutralization reactions with time, and to address the rate issues of the static NP and AP determinations, this iterative approach[2] was applied to a hypothetical tailings material containing various proportions of pyrite and silicates with moderate to high neutralization potentials.

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[2] Calculations were performed by a computer program (ARDMIN), written by the author to assess acid generation and neutralization potentials of samples and simulate mineral dissolution and oxidation reactions with time. The program calculates mineral fractions dissolved or oxidised in small time increments for various size fractions.

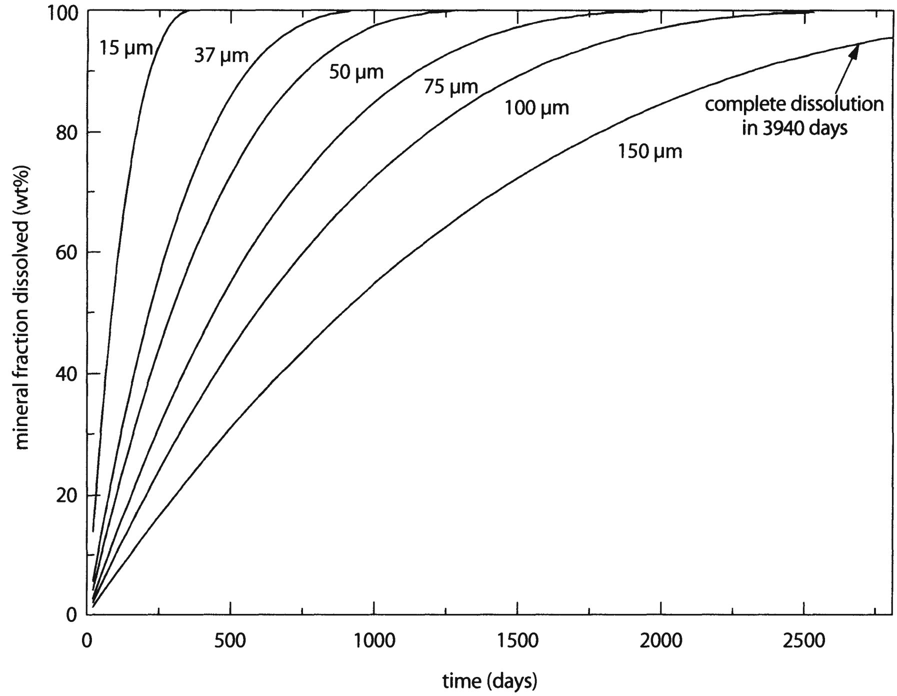

Fig. 3.5. Particle size effect on dissolution. Simulation carried out for plagioclase (An_{80}) at time intervals of 20 days and the surface roughness of 1. Plagioclase dissolution rate is $2.00 \cdot 10^{-13}$ mol $cm^{-2}\,s^{-1}$

Oxidation and dissolution reactions involving 20% pyrite, 10% olivine, 20% Ca-rich plagioclase (An_{80}), 20% diopside, and 30% chlorite with an average particle size of 50 µm were simulated at short time intervals. After removal of products from the particle surfaces following 4 days of reaction, new particle sizes were calculated for each mineral and calculations were performed iteratively. In essence, this approach simulates humidity cell tests where the reaction products are periodically removed from the particle surfaces. Cumulative net neutralization potential based on the amount of mineral dissolved, decreased rapidly due to oxidation of pyrite for about 21 weeks. From this point on, the cumulative net neutralization potential increased steadily to neutralize the acidity at 108 weeks (Fig. 3.4). This complete neutralization point corresponds to the dissolution of 89.5% plagioclase, 50.3% olivine, and 2.5% diopside. Only negligible amounts of chlorite dissolved during this period because of slow reaction rates; therefore, chlorite's contribution to the bulk neutralization during this period was insignificant. Complete oxidation of pyrite occurred after 30 weeks. If the tailings material did not contain olivine or diopside, complete neutralization was not possible. The ratio of plagioclase to pyrite (by weight) has to be greater than 1.2 to achieve neutralization within less than about 126 weeks. These results are based on the assumption that the minerals dissolve completely. If secondary minerals form during the dissolution, then neutralization potentials will be reduced to the extent that complete neutralization by plagioclase dissolution will not be possible for this material. In this case, by the time plagioclase is completely dissolved which corresponds to 186 weeks, net neutralization

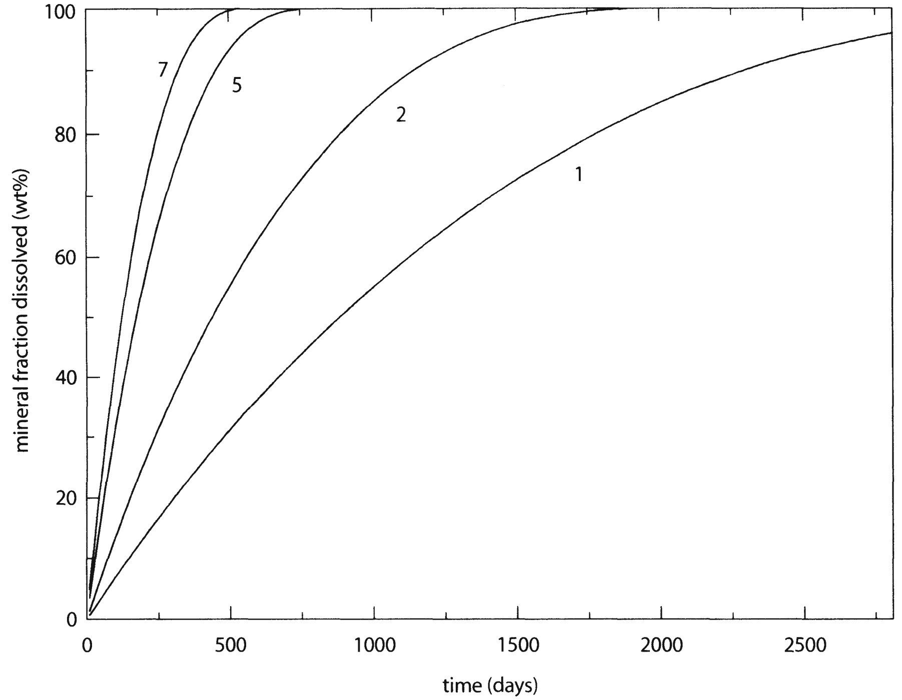

Fig. 3.6. Effect of surface roughness on dissolution. Numbers adjacent to *curves* are surface roughness factors (λ). Complete dissolution of particles with surface roughness of 1 occurs in 3940 days

potential will still be -75 kg equivalent H_2SO_4 per ton. Thus, such a tailings material containing approximately 20% plagioclase (An_{80}) and 10% olivine can not be considered with confidence to have the capacity to neutralize acidity resulting from the oxidation of 20% pyrite. A marginal neutralization capacity can be realized in no fewer than about 3 years where the plagioclase to pyrite ratio (by weight) is greater than 2.3.

To illustrate the magnitude of particle size effect on dissolution, distribution of Ca-rich plagioclase (An_{80}) in five particle size ranges (i.e. 15, 37, 50, 75, 100, and 150 µm) was simulated by ARDMIN. Results indicate that while complete plagioclase dissolution occur in 380 days for the 15-µm size fraction, it requires 3940 days for the 150-µm fraction to completely dissolve (Fig. 3.5). Contribution of smaller size fractions to the overall neutralization potential would be significant in short time frames. For instance, in 200 days, while only 13% (wt) of plagioclase in 150-µm size fraction dissolves, this is 86% in 15-µm size fraction. These figures are 47% for the 37-µm, 37% for the 50-µm, 26% for the 75-µm, and 20% for the 100-µm size fraction. The amount of plagioclase dissolving within such time frames are significant and illustrates the importance of mineral particle sizes in assessing the kinetics of neutralization potentials of silicate minerals. For instance, presence of plagioclase (An_{80}) in the 15-µm size fraction in abundances (by weight) that are about 1.6 times greater than pyrite could provide the waste material enough buffering capacity to neutralize the acidity in about 400 days. In comparison, plagioclase in the 50-µm size fraction with the same proportion will not be able to neutralize the acidity for at least 1390 days. In the case of the above tailings example, if the average particle size of plagioclase or olivine were 20 µm instead of 40 µm, neutralization of the acidity would have been achieved 64 weeks earlier.

As indicated earlier, estimates of particle roughness can significantly effect the dissolution time frames. As it is illustrated in Fig. 3.6, in the case of a surface roughness factor of 2, the overall time required to completely dissolve 150-µm plagioclase particles is 2090 days. A more realistic factor of 7 greatly reduces the overall time to 570 days. Since the surface roughness factor can vary up to several hundred (White *et al.* 1996), the use of surface roughness factors of 2 or 5 can be considered as conservative.

3.5.4
Uncertainties in Extrapolation of Laboratory Kinetics to Field Conditions

The value of predictive models based upon laboratory kinetics data may be limited when applied to large field scales. This is essentially a problem of scale reflecting the significance of uncertainties involved in the characterization of highly variable field conditions. Dissolution rates in laboratory are surface reaction limited. However, this may not be the case in field conditions, since dissolution of minerals may become transport limited. Transport rates and direction to and from reaction sites will depend on concentration gradients, gradient direction, diffusion coefficients, and diffusion paths. Although the data base is limited, Hochella and Banfield (1995) note that actual mineral dissolution rates are several orders of magnitude lower than laboratory rates. For instance, Drever and Clow (1995) quoted plagioclase dissolution rates of $2 \cdot 10^{-12}$ for reactor conditions, $2 \cdot 10^{-13}$ for column conditions, and $3 \cdot 10^{-14}$ for field conditions. This is essentially due to the differences in the mechanism of weathering. Possible causes of this discrepancy include: the assumption that the total surface areas are equally reactive which is not the case (White and Brantley 1995), aging of mineral surfaces, disappearance of high-energy sites on mineral surfaces with time, formation of protective secondary layers, presence of inhibitors in solution, saturation of solutions, and unrealistic estimates of mineral surfaces in contact with percolating water. Despite all these uncertainties, Sverdrup and Warfvinge (1995) indicate a good agreement between laboratory kinetics and observed field rates; therefore, field weathering rates can be calculated with very good accuracy. Especially at low pH conditions, laboratory rates may be closer to field rates since formation of secondary minerals is largely deterred (Hochella and Banfield 1995).

Otwinowski (1995, 1997) provides an approach to making long-term AMD predictions for waste rock piles based on laboratory and field data. The approach uses a scaling parameter that combines information about the waste rock characteristics and processes operating at different scales.

3.6
Concluding Remarks

The quality of the prediction of AMD potential of mine wastes and accompanying metal releases to the environment and the selection of an optimal remediation and the control technology for mine wastes require a good understanding of physical, chemical, and mineralogical characteristics of mine wastes.

Rates and mechanisms of dissolution reactions are important in terms of predicting the neutralization potential of minerals that are commonly present in waste rock and tailings. In addition to carbonates, some rock-forming minerals with potential to neutralize acid should be considered as effective in terms of providing additional long-term buffering capacity to mine wastes. Conventional acid base accounting tests have limitations and present difficulties in their interpretation. A new approach to determining the balance between acid producing and acid consuming components based on mineralogical composition of samples is proposed. This approach takes into consideration the presence of other sulfides and more than one type of neutralizing minerals.

Time frames of mineral dissolution can be assessed through the use of an iterative approach whereby the mass of mineral dissolved at certain time intervals are calculated based on mineral surface areas in each size fractions and mineral dissolution rates. Following subtraction of the mass dissolved, new nominal particle sizes and corresponding total surface areas are calculated. With this approach, an overall kinetic assessment of mine wastes in terms of their capacity to buffer the acidity resulting from the oxidation of sulfides, can be made based on mineralogical composition. This approach takes the static acid-base accounting tests to a step further by making possible their interpretations within a kinetic framework.

Particle size and shape are important factors in the assessment of AMD. Certain silicate minerals in fine size fractions may significantly contribute to the overall neutralization potential of waste material during early stages. As indicated in a preceding section, the amount of plagioclase dissolving can be significant and illustrates the importance of mineral particle sizes in assessing the kinetics of neutralization potentials of silicate minerals. Thus, further selective grinding of reactive silicate minerals may prove to be feasible technology to accelerate dissolution reactions and enhance the neutralization capacity.

Predictive models based upon laboratory kinetics data may be limited when applied to large field scales. Actual mineral dissolution rates are several orders of magnitude lower than laboratory rates; however, at low pH conditions, laboratory rates may be closer to field rates since formation of secondary minerals is largely deterred.

Fundamental basic data requirements for waste characterization include: mineralogical composition and characteristics, whole rock analysis including inorganic CO_2 concentration, particle size distribution, and static and kinetic tests. Mineralogical composition and characteristics include the types, abundances and textures of sulfide, carbonate, silicate, and hydroxide minerals. Particle size and surface roughness are important factors that control the rates of dissolution and oxidation reactions. Mineral modal abundances can be estimated by point counting, image analysis, quantitative XRD and from whole rock analyses by multiple regression and mass balance calculations.[3] Particle size, shape, surface areas, surface roughness, form and degree of crystallinity (e.g. disseminated, massive, framboidal, anhedral, euhedral) are determined by mineralogical techniques including image analysis and sieve tests. Liberation properties

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[3] MODAN, a computer program written by the author to estimate mineral abundances from whole rock chemical analyses and mineral compositions based on multiple regression and Gaussian reduction of mass balance calculations. The program is available from the author upon request through Email or on a 3.5-in. diskette (required with self-addressed envelope).

such as degree of liberation or encapsulation will determine the availability of particles to oxidation or dissolution reactions. Such information can be routinely analyzed by image analysis techniques. Once such basic information is obtained and assessed, more expensive laboratory and field test cells can be designed accordingly.

Acknowledgments

The author is grateful to Carl Weatherell for his efforts and help in providing copies of recent publications and to Henry Steger, Carl Weatherell, Genevieve Bechard, Philippe Dauphin (CANMET), Benoit Godin (Environment Canada) and Ron Nicholson (Beak Consultants and University of Waterloo) for reviewing the manuscript and for providing useful comments.

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Biomonitoring Environmental Contamination with Metallic and Methylmercury in Amazon Gold Mining Areas, Brazil

Olaf Malm · Jean R. D. Guimarães

4.1
Introduction

Mercury (Hg) has been used extensively in small-scale gold mining operations in the Amazon region of Brazil during the last 20 years (Pfeiffer and Lacerda 1988). Discharged directly into water bodies during different steps of the gold extraction and concentration, it can be oxidized and then methylated by physico-chemical and microbiological processes. In sediments and soils, mercury can remain in metallic form [Hg(o)] for a long time, specially under low redox potentials, a condition that is common in these environments. Mercury is released into the atmosphere during the burning of the gold amalgam as well as during the gold purification process. Release due to heating of the amalgam occurs mainly at the mining sites, since gold miners generally do not use retorts or other closed systems for the gold separation. On the other hand, important mercury emissions occur during the final gold purification in the gold dealers' shops, because the gold produced in the mining sites may still contain up to 10% mercury. Such emissions occur in urban environments and are significant sources of mercury exposure for the workers and for the population living nearby.

The two most important forms of mercury in the environment, from a toxicological point of view, are atmospheric [Hg(o)] vapor and methylmercury (CH_3Hg^+) in aquatic systems and both have a strong affinity for lipids, such as those in cell membranes. Besides its associated toxicological risks, emission of Hg(o) allows long-range mercury transport due to its high residence time in the atmosphere. Oxidation of Hg(o) to more reactive and water-soluble mercuric species increases wet and dry deposition of mercury, and in aquatic environments Hg^{2+} is partially converted to CH_3Hg^+, which is strongly accumulated and biomagnified along the foodchain. Methylmercury is among the most neurotoxic compounds and its concentrations in top aquatic predators are often a matter of concern for health authorities in areas where such organisms are the main protein source, as in most riverine villages in the Amazon.

Monitoring mercury levels in organisms such as fish and plants rather than (or in addition to) abiotic compartments is attractive because besides providing insights of the mercury load in the studied areas, these measurements give indications of the bioavailability of mercury and its potential risks to the biota, including man. Organisms such as lichens and bryophytes have been routinely used in large-scale surveys of atmospheric deposition of mercury and fish are the best indicator of mercury contamination in aquatic systems.

Evaluation of mercury contamination in the Amazon has been a challenge, due to its extension and poor infrastructure, not to mention the fact that gold mining activity is highly variable in space and time. Several research groups have been concerned with mercury contamination due to gold mining activities for the past 10 years, and most of them have worked with samples from the environment (soil, sediments, and fish) or humans (hair and urine). Analyses of mercury in air and water are complex tasks due to analytical difficulties. Nevertheless, air measurements have been carried out in mining areas, both in gold dealers' shops and their surroundings (Malm *et al.* 1991; Hacon *et al.* 1995) and extensive measurements of total mercury in water were recently made in the Tapajós basin (Roulet *et al.* 1996).

The purpose of this chapter is to summarize and discuss the results of research done during the past 10 years, with particular attention to two topics: (1) Biomonitoring of atmospheric mercury with a plant (atmospheric bromeliad); and (2) Assessment of mercury contamination in bodies of water using mercury concentrations in fish (Fig. 4.1).

4.1.1
Biomonitoring of Atmospheric Mercury

Traditional air samplers are often quite fragile instruments and usually allow only short sampling periods, which is a problem if the measurements must be done in remote areas and be representative of a wide range of atmospheric conditions. In contrast, plants may act as integrators of the concentrations of airborne pollutants over long

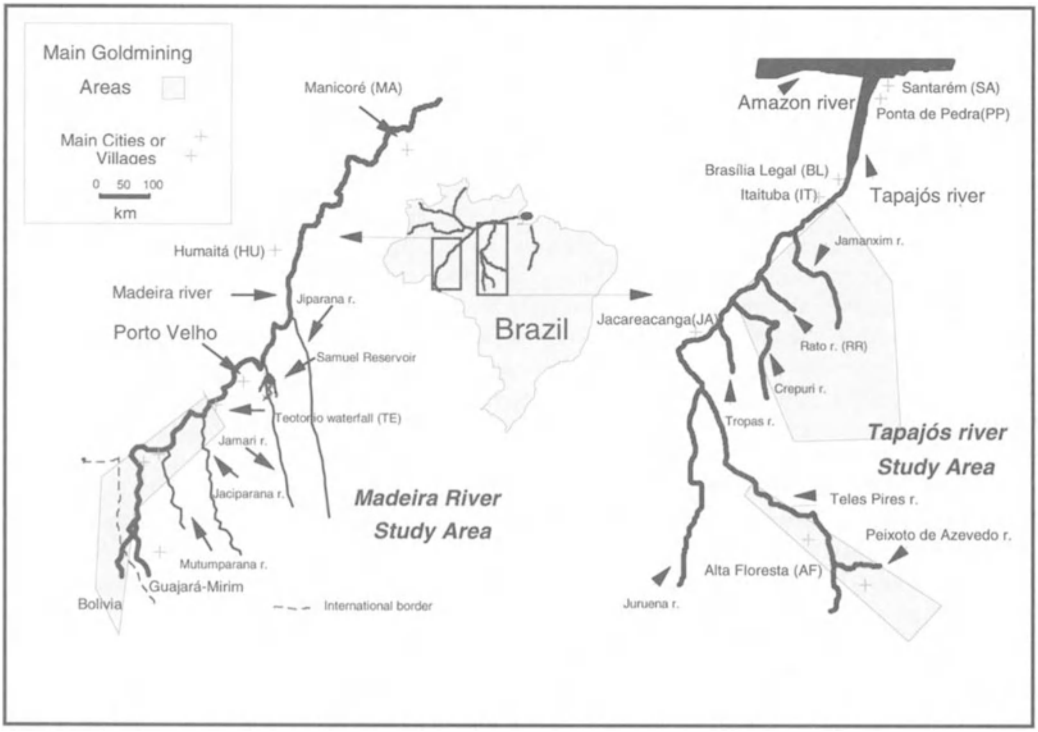

Fig. 4.1. Main gold mining areas in the Amazon region of Brazil

periods and their use as air pollution indicators has been widely reported in the literature. A classic example is the use of bryophytes to map atmospheric metal deposition in Denmark, Sweden, Finland, and Norway (Rühling *et al.* 1987; Steines 1993). However, species used for monitoring in temperate climates are not always suitable for use in tropical conditions, mainly because of their low resistance to desiccation.

The bromeliad *Tillandsia usneoides* (Linnaeus) has proved to be an efficient accumulator of atmospheric mercury in the vicinity of a chlor-alkali plant in Rio de Janeiro city (Calasans and Malm 1994). It was previously used in Brazil for the evaluation of fluoride in rainwater (Strehl and Arndt 1989) and is now being employed to monitor the emission of Hg(0) vapor during gold mining operations.

After acquiring experience in the use of *T. usneoides* to monitor an urban environment the authors applied this method to investigate the accumulation of mercury by this biomonitor in the vicinity of an isolated gold mining field in the jungle on the Tapajós River basin, as a function of time of exposure and distance from a fixed mercury source. In Alta Floresta, a town created in 1979 for farming development projects, which with the onset of the gold rush became an important regional gold trade center, we used this plant to investigate air mercury as a function of season, distance from the sources and height from the soil (Malm 1996a). Hacon *et al.* (1995) estimated that 0.5–2.5 tons of mercury are emitted annually to the atmosphere in Alta Floresta.

Biology of Tillandsia usneoides

Tillandsia usneoides belongs to the Bromeliaceae family and is a nonparasitic epiphyte that captures all its necessary nutrients and water directly from the atmosphere; therefore, it is classified as an atmospheric bromeliad (Benzing and Renfrow 1980). Concomitantly with nutrients, the plant accumulates pollutants such as heavy metals, including mercury. Its morphology and physiology make it ideal for preparation and handling of the transplanting systems. Its ramified morphology and small size provide a high ratio between surface area and mass, and consequently a high number of adsorption and/or absorption sites, which in turn results in a high efficiency for mercury uptake. *Tillandsia usneoides* is characterized by Crassulacean acid metabolism (CAM), opening the stomata mainly at night to minimize water loss; its tolerance to hydric stress makes it more appropriate than bryophytes or lichens for biomonitoring tropical environments. Additionally, it has a very slow growth rate (rank vegetative growth) which minimizes the problem of mercury dilution due to biomass increase.

4.1.2
Biomonitoring Aquatic Systems

Fish are the main source of methylmercury for humans in the Amazon region, as well as in other areas all over the world. The Amazon basin has more than 2000 species of fish, including about 200 of commercial importance. Fishing activities are higher in the dry season and the composition of the catch varies significantly with respect to the time of year (Goulding 1979; Malm *et al.* 1995a). Due to the high diversity and seasonal variations the comparison of mercury concentration in fish from different river basins or along time in a single sampling area becomes a complex task. Carnivorous (mainly piscivorous) fish from several gold mining areas have shown higher mercury concentrations than fish from lower trophic levels. The former must be monitored because they are good indicators of aquatic system contamination and pose the greatest threat to public health. Ingestion of contaminated fish may be critical when it occurs on a regular basis, because a very high proportion of methylmercury (>95%) is absorbed through the wall of the intestine, and also because this compound has a high degree of stability. Some human populations have fish as their main protein source, like in most riverine villages in the Amazon, where average daily fish intake may reach 200 g (Boischio *et al.* 1995).

Here, we will consider data reported by different authors for fish samples representing the Madeira, Tapajós, and Negro River basins, and two important hydroelectric reservoirs, all in the Amazon region of Brazil (Fig. 4.1).

Fish from sites considered to be in control areas without direct effects of mercury contamination usually have mercury levels lower than 0.2 $\mu g\ g^{-1}$ wet weight in rivers and 0.15 $\mu g\ g^{-1}$ in oceans. In contaminated areas mercury levels can reach 2 $\mu g\ g^{-1}$ or more and in carnivorous fish up to 20 $\mu g\ g^{-1}$, a level that is considered already lethal to fish (Meili 1991).

4.2
Materials and Methods

4.2.1
Atmospheric Biomonitoring

The biomonitoring systems consisted of two baskets with *T. usneoides* fixed under a transparent plastic cover. The systems were prepared in the laboratory in Rio de Janeiro and transplanted to the study areas. Each basket was made of a flexible plastic net with 0.5 cm pores and contained 5 g fresh weight of plants previously collected in a pristine area in Rio de Janeiro state, which has been used for this purpose for the last 5 years. Only the younger parts of the plants were selected for transplantation experiments. In the city of Alta Floresta, MT, an important gold commerce center, bioassays were performed at 200 sites at two different times of the year (15 and 45 days exposure) trying to identify the effects of prevailing winds and the influence of seasonal changes on atmospheric mercury.

One hundred systems were installed during the dry season (August–September 1995) when average daily temperatures range from 33 to 37 °C and during the rainy season (February–March 1996) when this range is 27–31 °C. Biomonitoring systems were suspended 2–20 m above the ground in open areas near and in gold shops, as well as in control areas. Relative occupational exposure was also evaluated by means of the systems installed inside the gold dealers' shops. A gold shop that had stopped operation was monitored 15 months after business interruption, when it was being partially used for another purpose.

In an isolated gold mining area in the jungle on the banks of the Rato River, a tributary of the Tapajós, 40 systems were installed in the dry season around a small custom vent hood designed by the miners where Au : Hg amalgams from nearby mining fields are regularly burnt. In both study areas one of the baskets was removed 15 days after installation and the second one 45 days after installation. The plants were transported to the laboratory in closed glass flasks.

4.2.2
Aquatic Biomonitoring

In the Madeira River basin, fish samples were collected from 1987 to 1994 along the Madeira River itself, from Guajará-Mirim to Manicoré, and some sites in certain tributaries (the Jamari, Candeias, Jaciparaná, and Mutumparaná Rivers), as well as the reservoir of the Samuel Hydroelectric Power Plant operated by Eletronorte, a subsidiary of the Brazilian federal electricity company. Fish were also collected from the Tapajós River from 1990 to 1996, and from the Negro River in 1993. Some of the presented data were reported by other research groups, over the past 4 years. Fish were caught with gill-nets or obtained from local fishermen who use a variety of fishing techniques. The fish were separated into three main classes based on feeding habits (Goulding 1979): Carnivorous (mainly piscivorous); microphagous (ingesting decomposing organic matter, algae, and ooze clay with some sediment); and omnivorous (eating mainly fruits and seeds). In some cases data were separated only in two categories, carnivorous and others.

4.2.3
Mercury Analysis in Plants and Fish

Mineralisation of the plants for total mercury analysis was done with a method adapted from Calasans (1994). The edible parts of fish (muscle) and some human hair samples were processed in accordance with the technique described by Agemian (1978). Determination of mercury was done by cold vapor atomic absorption spectrophotometry. Analytical quality control was done with routine analysis of reference material from NIES-Certified Reference Material No. 1, Pepperbush – Japan and Tuna Fish (sample 4) and Mussel (sample 5) from the European Commission Environment Institute (ECET) – Aquacon Sub-Project 2 "Mercury in the foodchain".

4.3
Results and Discussion

4.3.1
Atmospheric Biomonitoring

Concentrations of mercury in the plants placed inside the gold dealers' shops were remarkably high, reaching values of up to 26 μg g^{-1}, which is more than 300 times higher than background values (around 80 ng g^{-1}). A decrease in mercury concentrations was observed when the plants were moved away from the gold shops, in both dry and rainy seasons (Tables 4.1 and 4.2). Clear seasonal trends with higher values in the dry season were observed in the systems located outdoors and also inside the gold shops (Tables 4.1 and 4.2). Indoor systems generally presented an average value 3 times higher during the dry season than in the rainy season, probably due to the reduction of gold production in the wet period and to the higher temperatures in the dry season, which stimulate mercury vapor emission. On the other hand, because *T. usneoides* traps suspended particles besides absorbing mercury vapor, the lower dust concentration in the rainy season may also influence *T. usneoides* mercury levels.

In the dry season we observed an average value nearly 12 times higher inside the shops than in the surroundings (5–20 m) indicating the critical areas are inside the gold shops and in the immediate vicinity.

The average increase of mercury concentration in systems placed in the same sites in the rainy and in the dry season was 246% ($n = 32$) considering both 15 and 45 days' exposure (Table 4.3). Inside the gold shops this increase was 197% ($n = 7$) while for outdoor systems it was 136% ($n = 28$), not including the points at distance <5 m. This is probably reflecting the reduction of gold commercialization in the gold shops, but can also be related to the association of Hg(o) with particulate matter. Hacon *et al.* (1995), analyzing total mercury in different aerosol fractions in Alta Floresta, found an association between mercury and the two major aerosol sources, biomass burning and soil dust emissions, and suggested that high aerosol concentrations would favor the condensation of vapor phase mercury, a process that could be stimulated by the high humidity, ozone, and solar radiation. This aerosol–Hg association is probably more important during the dry season when the amount of dust in the air is also much higher, due both to soil resuspension and biomass burning, processes that are more intense in this season. Average daytime temperatures are also higher in the dry season, which

Table 4.1. Mercury concentration (ng g^{-1}) in bromeliads from urban areas and inside gold shops in Alta Floresta during the dry season (systems exposed 45 days)

Evaluated Area	No. of systems	Average	Min.–max.
Inside gold shops	12	4255	550 – 26 775
Immediate vicinity (≅5 m) from gold shops	02	12 185	1895 – 22 480
Sites between 5 and 20 m from gold shops	34	363	<dl – 2205
Sites between 200 and 500 m from gold shops	27	296	<dl – 895
Controls in Alta Floresta (>1 km from gold shops)	14	169	<dl – 420
Controls in Rio de Janeiro	03	200	<dl – 390

dl, Detection limit: 80 ng g^{-1}.

Table 4.2. Mercury concentration (ng g^{-1}) in bromeliads from urban areas and inside gold shops in Alta Floresta during the rainy season (systems exposed 45 days)

Evaluated Areas	No. of systems	Average	Min.–max.
Inside gold shops	10	1665	220 – 5255
Immediate vicinity (≅5 m) from gold shops	10	5245	2510 – 9450
Sites between 5 and 20 m from gold shops	57	450	105 – 1370
Control sites in Alta Floresta (>1 km from gold shops)	3	95	<dl – 120
Controls at Rio de Janeiro	2	<dl	<dl

dl, Detection limit: 80 ng g^{-1}.

stimulates mercury degassing from contaminated soils. Points very close to the goldshops showed much higher values during the dry period.

As a confirmation of the efficiency of atmospheric mercury uptake by plants, during the rainy season 10 of the 15 systems which presented the highest values after 15 days of exposure also had the highest concentrations after 45 days of exposure (67% correspondence) and ranked almost in the same order. In the dry season, this correspondence was higher than in the rainy season: 12 of the 15 systems which had the higher mercury concentrations after 45 days were the same. This suggests that for the places with the higher air mercury, a 15-day exposure was sufficient to allow a good integration. A similar study done in a chlor-alkali plant in Rio de Janeiro, where mercury concentrations were generally higher than found here, showed that 7-days' exposure was sufficient time for a good integration (Calasans and Malm 1994). Further away from the mercury sources, a 45 days' exposure will be more representative.

In the same locations we observed variations of mercury in plants according to height. During the rainy season, a relative increase of mercury in systems installed at higher locations and close to sources was observed, possibly reflecting the greater exposure of these systems to wind, which is more intense in that season and at these distances from soil, where the shielding effect of buildings is less pronounced. Considering the locations with systems exposed at at least 3–6 different heights, the highest concentrations were found at 2.5 and 4.5 m in the dry season and 3.5 and 5.5 m in the rainy season. This suggests that the transport of mercury is being controlled by particles, more abundant in the dry season and at lower heights.

The coefficient of variation between mercury concentrations in systems hung at the same locations but at different heights decreases with distance from the mercury sources. This may reflect the greater vertical mixing of mercury in the air at increasing distance from the mercury sources and also the decrease in the importance and/or variability of soil resuspension, as air turbulence is expected to be smaller in the less densely urbanized areas in the outskirts of the town. The points closer to the sources are also coincidentally closer to main streets with higher traffic. Strong winds causing resuspension of contaminated soil are more frequent during the dry season and the coefficient of variation of mercury in systems at different heights was higher by a factor of two during the dry season.

Systems hung on trees showed higher levels than the ones suspended on lamp-posts at the same distance from the sources. Even at points far away (around 350 m) from the sources high values of mercury were found in systems hung in a tall tree situated after a deforested area and downwind from the mercury sources. This suggests that the vegetation acts as a trap for suspended particles, creating a slightly more contaminated microsystem.

Biomonitoring systems installed inside the gold dealers' shops were quite useful to identify the shops with the worst ventilation conditions and/or higher reburning activ-

Table 4.3. Increase in atmospheric mercury (%), estimated with the biomonitor *T. usneoides*, between rainy and dry seasons in Alta Floresta

Location	Increase in mercury (%)
General (*n* = 32)	246
Inside gold shops (*n* = 7)	197
Outside gold shops (*n* = 28)	136

ity. Systems exposed inside an abandoned gold shop showed surprisingly high mercury values, which stresses the importance of establishing and enforcing regulations concerning the future uses of buildings previously used as gold shops.

In the second study area, an isolated gold mining village in the middle of the jungle called Piririma (Fig. 4.2) it was possible to locate and map the concentrations of mercury around a unique point source. Figure 4.2 shows the fast drop of the concentrations as one moves away from the source but also the probable influence of prevailing winds. It is worth noting the higher concentrations of mercury measured inside the houses close to the main burning place and even inside other residential units. This may reflect a residual contamination: amalgams used to be burned indoors before the construction of the rudimentary fume hood at some distance from the buildings.

As a complement to this work, experiments aimed at calibrating the biomonitor have been conducted for the last 3 years. *Tillandsia usneoides* samples are exposed in closed chambers to an air flow with known mercury concentrations, during periods

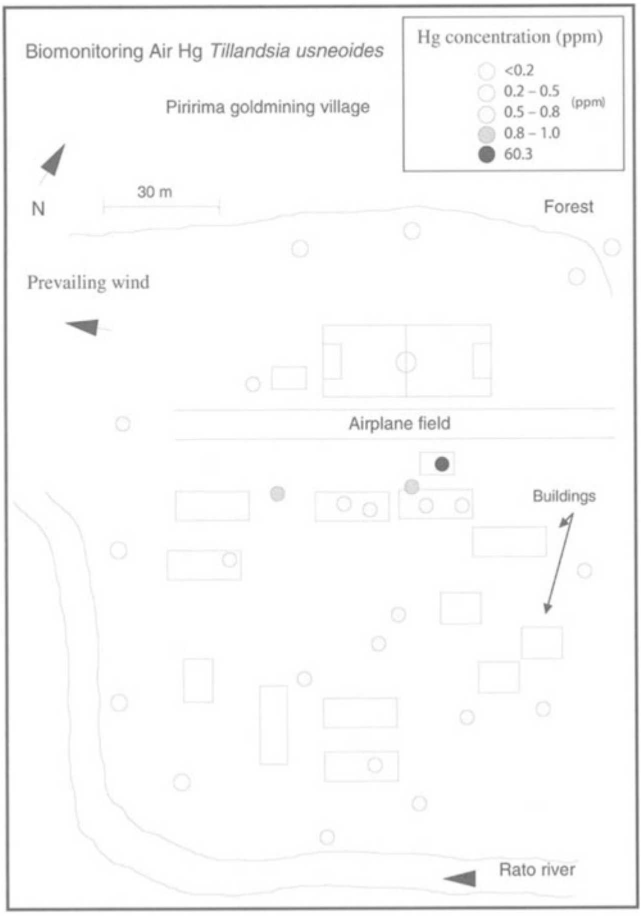

Fig. 4.2. Mapping of air mercury in a gold mining field with a biomonitor

comparable to the ones used in field experiments, and then analyzed for total mercury. Preliminary calibration curves obtained in these experiments were used to calculate the mercury concentration in the air of the gold shops which could be around 200 µg m^{-3}, which is in agreement with field measurements with mercury trapping solutions or gold amalgamators (Malm 1991; Hacon *et al.* 1995; Malm *et al.* 1995b) and much higher than the limit established by WHO (1990) for occupational exposure.

4.3.2

Biomonitoring Mercury in Water Systems: Mercury in Fish

The piscivorous and carnivorous group from the main Amazon River basins affected by gold mining usually presented average mercury values over the 0.5 µg g^{-1} maximum limit established by the Brazilian (Brazil 1975) or WHO (1990) guidelines. In some piscivorous/carnivorous fish species, a tendency for increasing mercury concentrations with increasing fish weight was observed. However, even for these groups of fish, a high variability was observed even when many fish of the same species are collected at the same time in the same net. This could be explained by the abundance of food through the long food chains in most Amazon River basins. A seasonal variability was observed for mercury in some piscivorous species, but more data are needed for several species in different river basins or reservoirs to verify if this is a general pattern or not. Table 4.4 summarizes mercury data on carnivorous and piscivorous fish from different areas in the Amazon region.

The data in Table 4.4 show a generally good agreement between the different investigations in the same areas, and the total number of samples is quite expressive. Mercury in fish from several areas presented average values close or over the 500 ng g^{-1} limit.

Table 4.4. Mercury concentrations (ng g^{-1}) in carnivorous or piscivorous fish from different Amazon areas

Sample origin	No. of species	No. of samples	Average	Range	Reference
Madeira River	34	370	850	165 – 3920	Malm *et al.* (1997)
Madeira River	22	154	665	60 – 3960	Gali (1997)
Madeira River	–	251	638	68 –11500	Barbosa *et al.* (1995)
Tapajós River	23	118	498	25 – 5960	Malm *et al.* (1997)
Tapajós River	12	212	499	46 – 2200	Uryu (1996)
Tapajós River	19	73	511	132 – 1354	Lebel *et al.* (1996)
Tapajós River	9	85	723	120 – 3580	Hacon *et al.* (1995)
Negro River	18	113	780	226 – 4231	Malm *et al.* (1995c)
Tucuruí Reservoir	8	121	1300	200 – 5900	Porvari (1995)
Balbina Reservoir	6	27	371	49 – 1103	Malm *et al.* (1996b)
Lakes in Amapá	3	74	481	26 – 1830	Guimarães *et al.* (1997)

A total of more than 400 individuals (from around 50 species) of fish from the Madeira River basin were analyzed at our laboratory. Despite the very high number of specimens, few representative time series from the same species and sampling locations are available. Figure 4.3 shows the distribution of mercury concentrations in fish ordered by ascending weight for fish of the same feeding habit. In the carnivorous group the weight interval shown can be considered as a logarithmic scale distribution since 70% of the fish investigated are in the range of 0.1–3 kg and the remaining ones reach weights up to 60 kg.

Fortunately, in the Madeira River basin carnivorous fish (with a quite high average mercury concentration of 850 ng g^{-1}) are not an important share (less than 20%) of the total fishery market while some other omnivorous and microphagous species such as Jatuarana (*Brycon* sp.), Curimatã (*Prochilodus nigricans*), Pacú (*Mylossoma spp.*), and Tambaqui (*Colossoma macropomum*) correspond to approximately 65% of the total fishery in weight (Goulding 1979). For a detailed risk assessment, it is important to consider this kind of information as well as fish intake by the local populations throughout the year and neurological evaluations should be done before issuing any advisory recommendations, because of the strong socio-economic impacts the latter may have. In the rainy season the food offer increases which would probably cause growth dilution of mercury concentrations. Junk (1985) and Val and Almeida-Val (1997) showed that alimentary habits of the fish as well as fat and protein contents changed significantly throughout the year.

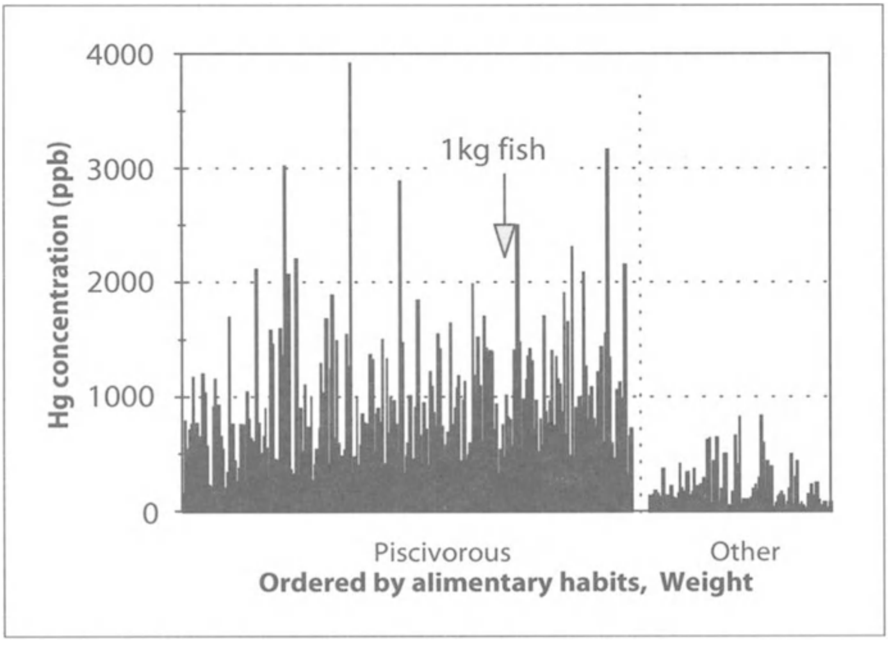

Fig. 4.3. Mercury concentrations in fish from the Madeira River Basin

No clear differences were observed for mercury in fish from different areas along the Madeira River, despite the fact that the sampling areas cover a far larger sector of the river than the one affected by gold mining activities. Mercury in fish is present essentially as methylmercury, and total mercury in fish is a function of the concentration and availability of methylmercury in water and in the fish diet, which are determined by a complex set of biotic and abiotic variables that are still poorly understood, specially in tropical environments. On the other hand, the issue of the relative importance of gold mining mercury emissions compared to other natural and/or anthropogenically intensified mercury sources such as biomass burning or soil erosion is a subject of debate. Longitudinal variations in fish mercury along the river will reflect the variation of the inorganic mercury sources, the factors affecting the methylation of the latter and last but not least the migratory pattern of sampled fish species.

In contrast to the Madeira River, fish collected throughout the Tapajós River basin presented a clear change in mercury concentration depending on sampling areas (Fig. 4.4), showing a decreasing pattern to downstream river basin. Much higher mercury values were observed in fish from the upper part of the Tapajós basin, at the Teles Pires River close to Alta Floresta city, but fish consumption is relatively low in this region.

The Tapajós River basin has been prospected for a longer time than Madeira (30 and 15 years respectively), but mercury concentrations in fish of the same species and weight range are higher in the latter basin. Elevated concentrations are found in the Tapajós River basin only in its upper reaches, while at Santarém, close to the confluence with

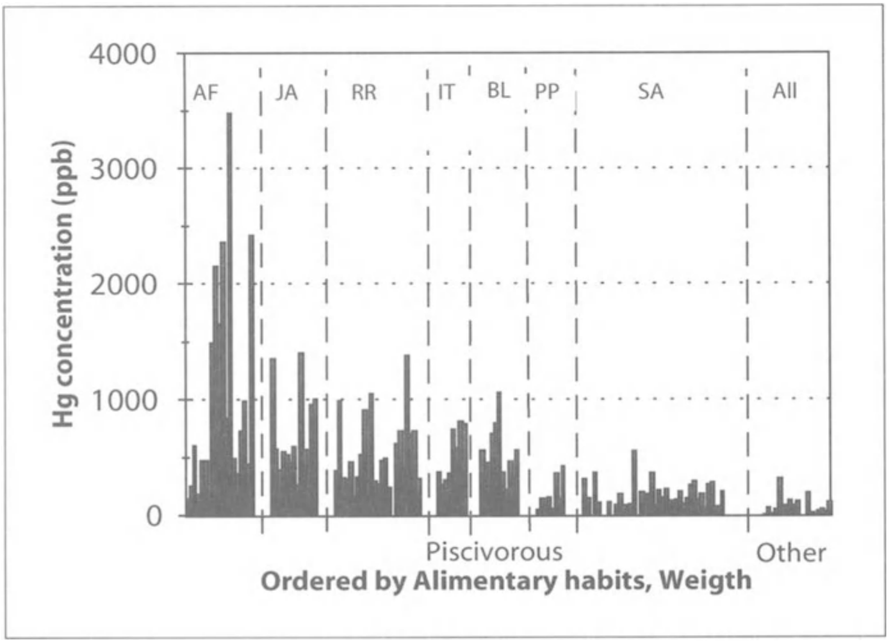

Fig. 4.4. Mercury concentrations in fish from the Tapajós River basin. Data presented from upstream to downstream in the following sites: Alta Floresta (*AF*), Jacareacanga (*JA*), Rato River (*RR*), Itaituba (IT), Brasilia Legal (*BL*), Ponta Pedras (*PP*) and Santarem (*SA*)

the Amazon River levels were close to background values (Malm 1995a). One possible explanation for this is that long distance transport of Hg(o) in the Tapajós River basin, with much lower flow rates and seasonal variation in water level, may be less efficient than in the Madeira River. Other factors to be taken into account are the contrasting biogeochemical characteristics of these two river basins. The Tapajós River is classified as a clear water system (Sioli 1985), with some features such as a lower pH and conductivity and higher humic content, which are expected to favor methylmercury production. The Madeira River is the better example of a white water river in the Amazon basin, with a high load of suspended solids, lower C, and higher pH and conductivity than the Tapajós River, but with an important mercury contribution from the leaching of the Andean cliffs and also from gold mining activities, in its Bolivian tributaries. It is interesting to note that the Negro River, with no significant history of gold mining activities presents quite high mercury values in fish samples (Forsberg *et al.* 1995).

4.4
Conclusions

Monitoring air mercury concentrations with the biomonitor *T. usneoides* proved to be a useful tool to trace variations of mercury levels in space and time in urban and semi-rural environments in the Amazon region. Biological monitoring using this atmospheric bromeliad presents some interesting advantages over determinations of mercury in air with traditional instruments, as it allows the assessment of hundreds of sites at the same time at a very low cost. Integrating exposure over long periods of time, this approach also solves, to a certain extent, a typical dilemma of atmospheric studies, that of either continuous sampling at usually very few locations or short sampling at a more representative number of stations. It also avoids the frequent operational problems endured by electronic equipment subject to high temperature, humidity, and dust levels besides inconstant electric energy supply.

Using *T. usneoides* we could identify critical areas in Alta Floresta with higher environmental and occupational mercury exposure risks and obtain evidence of variations of mercury in air as a function of season, height, and distance from the mercury sources. People overexposed should be monitored through mercury analysis in urine. As a recommendation concerning former gold shop offices, attention should be paid to the future uses of those stores, since the one studied herein showed high air mercury levels even after 15 months of interruption of activities. If the place is used as an office, with air conditioning and hence less ventilation, mercury in the air can reach even higher levels. Mercury impregnation of the walls, floor or ceiling is probably a long-term source of mercury source for successive future occupants of such buildings.

Indoor areas in cities or villages with poor air circulation are critical for Hg(o) human contamination, mainly through inhalation. Despite the small amount of mercury (around 5%), in the gold reburned in the gold shops, when compared to the total mercury used in the mining process, and the fact that most shops are equipped with some kind of fume hood, the occupational exposure frequently exceeds the standards and people in neighboring houses can also be contaminated because the hoods or other exhaust systems do not provide any removal of mercury from the air.

In comparison with the biomonitoring of mercury in air, the interpretation of data when using fish as biomonitors of mercury in water basins is far more challenging.

Evaluating water contamination through fish analysis allows the identification of critical areas for populations with a high fish intake and assessment of the risks involved, besides providing guidance for the efforts to understand the environmental conditions that can cause higher levels in fish. People overexposed must be monitored through mercury analysis of hair. Extensive work with fish from main river basins of the Amazon region is being carried out in orter to try to identify different patterns in areas with varying degrees of anthropogenic mercury impacts (Kehrig *et al.* 1998). However, the estimated mercury releases by gold mining activities in the different river basins do not correlate with the levels found in the biota, suggesting the influence of the many contrasting hydrological and biogeochemical features of the studied river basins and of the spatial and temporal variability of both natural and manmade mercury sources. Despite generally high mercury levels in fish, their variability in carnivorous or piscivorous fish discourages the establishment of restrictions of fish ingestion by the local population. A few top predator species could be blacklisted but this should be evaluated for each area and regularly revised. Attempts to change centuries-old alimentary habits of a human population may be justified in view of the early neurological effects documented in some riverine villages of the Amazon by Lebel *et al.* (1998), White *et al.* (1997), and Nielsen *et al.* (1997). However, the benefits of such measure have to be weighed against their potentially negative cultural and socio-economic impacts, whose quantification is not an easy task either.

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Chapter 5

Sodium Cyanide Hazards to Fish and Other Wildlife from Gold Mining Operations

Ronald Eisler · Donald R. Clark Jr. · Stanley N. Wiemeyer · Charles J. Henny

5.1
Introduction

Highly toxic sodium cyanide (NaCN) is used increasingly by the international mining community to extract gold and other precious metals through milling of high grade ores and heap leaching of low grade ores. Of the 98 million kg cyanide (CN) consumed in North America in 1989, about 80% was used in gold mining (Knudson 1990). In Canada, more than 90% of the mined gold is extracted from ores with the cyanidation process. This process consists of leaching gold from the ore as a gold–cyanide complex, and gold being recovered by precipitation (Simovic and Snodgrass 1985). Milling and heap leaching require cycling of millions of liters of alkaline water containing high concentrations of potentially toxic NaCN, free cyanide, and metal cyanide complexes that are frequently accessible to wildlife. Some milling operations result in tailings ponds of 150 ha and larger. Heap leach operations that spray or drip cyanide solution onto the flattened top of the ore heap require solution processing ponds of about 1 ha in surface area. Although not intentional or desired, puddles of various sizes may occur on the top of heaps where the highest concentrations of NaCN are found. Exposed solution recovery channels are usually constructed at the base of leach heaps. All of these cyanide-containing water bodies are hazardous to wildlife if not properly managed (Henny *et al.* 1994). In this account we emphasize hazards of cyanide from mining operations to fish and wildlife species and proposed mitigation to protect them.

5.2
Background

Heap leaching occurs when run-of-mine or crushed ore is stacked on an impermeable plastic pad on the ground surface, with spraying or dripping of a NaCN solution on the flattened top. Large leach heaps may include 272 000 tons of ore and tower 100 m or more (Alberswerth *et al.* 1989). In the milling of gold ores, a NaCN solution is percolated through the crushed ore to dissolve the gold particles (Ripley *et al.* 1996). In both leaching and milling processes, after the gold is chemically precipitated, the solution is adjusted for pH and cyanide concentration, and recycled to precipitate more gold. Eventually, the remaining solution must be treated to recycle the cyanide or to destroy it to prevent escape into the environment.

Alkaline chlorination of wastewaters is one of the most widely used methods of treating cyanide wastes. In this process, cyanogen chloride (CNCl) is formed, which at alkaline pH is hydrolyzed to the cyanate ion (CNO^-). If free chlorine is present, CNO^- can be further oxidized (Way 1981; Leduc *et al.* 1982; Simovic and Snodgrass 1985; Marrs and Ballantyne 1987). The use of sulfur dioxide in a high dissolved oxygen environment with a copper catalyst reportedly reduces total cyanide in high cyanide rinsewaters from metal plating shops to less than 1 mgl^{-1}; this process may have application in cyanide detoxification of tailings ponds (Robbins 1996). Other methods used in cyanide waste management include lagooning for natural degradation, evaporation, exposure to ultraviolet radiation, aldehyde treatment, ozonization, acidification–volatilization–reneutralization, ion exchange, activated carbon absorption, electrolytic decomposition, catalytic oxidation, and biological treatment with cyanide-metabolizing bacteria (Towill *et al.* 1978; USEPA 1980; Way 1981; Marrs and Ballantyne 1987; Smith and Mudder 1991). Additional cyanide detoxification treatments include the use of $FeSO_4$, $FeSO_4$ plus CO_2, H_2O_2, $Ca(OCl)_2$ (Henny *et al.* 1994), dilution with water, $FeSO_4$ plus H_2O_2, and $(NH_4)HSO_3$ (Wiemeyer, pers. comm.). In Canadian gold-mining operations, the primary treatment for cyanide removal is to retain mill wastewaters in impoundments for several days to months; removal occurs through volatilization, photodegradation, chemical oxidation, and, to a lesser extent, microbial oxidation. Microbial oxidation of cyanide is not significant at this time in mine tailing ponds, which typically have pH >10, a low number of microorganisms, low nutrient levels, large quiescent zones, and cyanide concentrations >10 mg l^{-1} (Simovic and Snodgrass 1985).

Cyanide seldom remains biologically available in soils because it is either complexed by trace metals, metabolized by various microorganisms, or lost through volatilization (Towill *et al.* 1978; Marrs and Ballantyne 1987). Cyanide ions are not strongly adsorbed or retained on soils, and leaching into the surrounding ground water will probably occur. Under aerobic conditions, cyanide salts in the soil are microbially degraded to nitrites or form complexes with trace metals. Under anaerobic conditions, cyanides denitrify to gaseous nitrogen compounds that enter the atmosphere. Mixed microbial communities capable of metabolizing cyanide and not previously exposed to cyanide are adversely affected at 0.3 mg HCN kg^{-1}; however, these communities can become acclimatized to cyanide and can then degrade wastes with higher cyanide concentrations (Towill *et al.* 1978). Acclimatized microbes in activated sewage sludge can often completely convert nitriles to ammonia, sometimes at concentrations as high as 60 mg total CN kg^{-1} (Towill *et al.* 1978).

In regard to cyanide use and toxicity in the recovery of precious metals, most authorities currently agree on nine points:

1. Metal mining operations consume most of the current cyanide production.
2. The greatest source of cyanide exposure to humans and range animals is cyanogenic food plants and forage crops–not mining operations.
3. Cyanide is ubiquitous in the environment, with gold-mining facilities only one of many sources of elevated concentrations.
4. Many chemical forms of cyanide are present in the environment, including free cyanide, metallocyanide complexes, and synthetic organocyanides, but only free cyanide (i.e., the sum of molecular hydrogen cyanide, HCN, and the cyanide anion, CN^-) is the primary toxic agent, regardless of origin.
5. Cyanides are readily absorbed through inhalation, ingestion, or skin contact, and are readily distributed throughout the body via blood. Cyanide is a potent and rapid-acting asphyxiant; it induces tissue anoxia through inactivation of cytochrome oxidase, causing cytotoxic hypoxia in the presence of normal hemoglobin oxygenation.

6. At sublethal doses, cyanide reacts with thiosulfate in the presence of rhodanese to produce the comparatively nontoxic thiocyanate, most of which is excreted in the urine. Rapid detoxification enables animals to ingest high sublethal doses of cyanide over extended periods without harm.
7. Cyanides are not mutagenic or carcinogenic.
8. Cyanide does not biomagnify in food webs or cycle extensively in ecosystems, probably because of its rapid breakdown.
9. Cyanide seldom persists in surface waters and soils owing to complexation or sedimentation, microbial metabolism, and loss from volatilization.

(Doudoroff 1976; Towill *et al.* 1978; Smith *et al.* 1979; Egekeze and Oehme 1980; USEPA 1980, 1989; Moore 1981; Vennesland *et al.* 1981; Leduc *et al.* 1982; Biehl 1984; Leduc 1984; Way 1984; Ballantyne and Marrs 1987; Evered and Harnett 1988; Eisler 1991; Smith and Mudder 1991; Hill and Henry 1996; Ripley *et al.* 1996).

5.3
Effects

5.3.1
Aquatic Ecosystems

Accidental spills of cyanide solutions into rivers and streams have produced massive kills of fish, amphibians, aquatic insects, and aquatic vegetation. Precious-metal waste-sources of poisonings have included storage reservoirs of concentrated solutions, overturned rail tank cars, and discharge of substances generating free HCN in the water from hydrolysis or decomposition (Leduc 1984; Alberswerth *et al.* 1989). Fish kills from accidental discharges of cyanide gold mining wastes are common (Holden and Marsden 1964; Leduc 1978; Towill *et al.* 1978; USEPA 1980; Albersworth *et al.* 1989; Ripley *et al.* 1996). In one case, mine effluents containing cyanide from a Canadian tailings pond released into a nearby creek killed more than 20 000 steelhead (*Oncorhynchus mykiss*; Leduc *et al.* 1982). In Colorado, overflows of 760 000 l NaCN-contaminated water from storage ponds into natural waterways killed all aquatic life along 28 km of the Alamosa River (Albersworth *et al.* 1989).

Data on the recovery of cyanide-poisoned aquatic ecosystems are scarce. In one case, a large amount of cyanide-containing slag entered a stream from the reservoir of a gold mine in Japan as a result of an earthquake (Yasuno *et al.* 1981). The slag covered the stream bed for about 10 km from the point of rupture, killing all stream biota; cyanide was detected in the water column for 3 days after the spill. Within 1 month flora was established on the silt covering the above-water stones, but there was little underwater growth. After 6–7 months, populations of fish, algae, and invertebrates had recovered, although species composition of algae was altered (Yasuno *et al.* 1981).

Cyanides do not persist in aquatic environments. In small, cold oligotrophic lakes treated with NaCN (1 mg l^{-1}), acute toxicity to aquatic organisms was negligible within 40 days. In warm shallow ponds, no toxicity was evident to aquatic organisms within 4 days after application of the same concentration of NaCN. In rivers and streams, cyanide toxicity fell rapidly on dilution (Leduc 1984). Cyanide was not detectable in water and sediments of Yellowknife Bay, Canada, between 1974 and 1976, despite the continuous input of liquid effluents containing cyanides from an operating gold mine. Nondetection was attributed to rapid oxidation (Moore 1981). Several factors contribute to the rapid disappearance of cyanide from water: bacteria and protozoans may degrade cyanide by converting it to carbon dioxide and ammonia; chlorination of water supplies can result in conversion to cyanate; an alkaline pH favors oxidation by chlorine; and an acidic pH favors volatilization of HCN into the atmosphere (USEPA 1980).

Cyanide concentrations in fish from streams poisoned with cyanide ranged between 10 and 100 µg total CN kg^{-1} whole body FW (Wiley 1984). Gill tissues of salmonids contain from 30 µg kg^{-1} fresh weight (FW) total CN to >7000 µg kg^{-1} FW under widely varying conditions of temperature, nominal water concentrations of free cyanide, and duration of exposure (Holden and Marsden 1964). Unpoisoned fish usually contained <1 µg total CN kg^{-1} FW in gills, although values up to 50 µg kg^{-1} FW occurred occasionally. Lowest cyanide concentrations in gill occurred at elevated (summer) water temperatures; at lower temperatures, survival was greater and residues were higher (Holden and Marsden 1964).

Fish are the most cyanide-sensitive group of aquatic organisms tested (Eisler 1991). Under conditions of continuous exposure, adverse effects on swimming and reproduction usually occurred between 5 and 7.2 µg free CN l^{-1} and on survival between 20 and

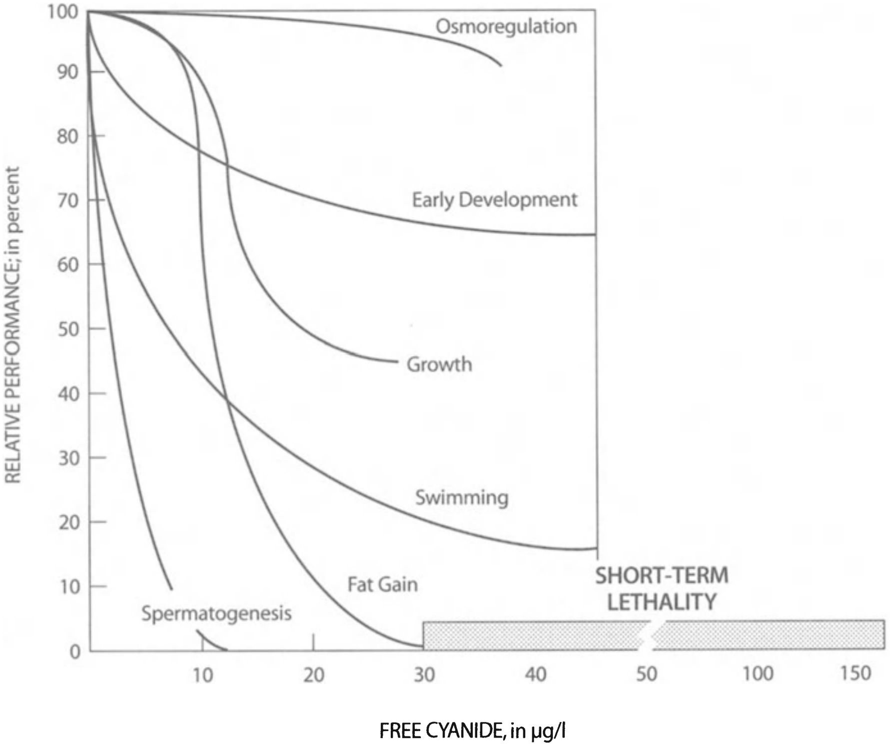

Fig. 5.1. Summary of lethal and sublethal effects of free cyanide on freshwater fish (modified from Leduc *et al.* 1982)

76 μg l^{-1} (Fig. 5.1). Other adverse effects on fish of chronic cyanide exposure included susceptibility to predation, disrupted respiration, osmoregulatory disturbances, and altered growth patterns. Free cyanide concentrations between 50 and 200 μg l^{-1} were fatal to the more-sensitive fish species over time, and concentrations >200 μg l^{-1} were rapidly lethal to most species of fish (USEPA 1980). Cyanide-induced pathology in fish included subcutaneous hemorrhaging, liver necrosis, and hepatic damage. Exposure of fish to 10 μg HCN l^{-1} for 9 days was sufficient to induce extensive necrosis in the liver, although gill tissue showed no damage. Intensification of liver histopathology was evident at dosages of 20 and 30 μg HCN l^{-1} and exposure periods up to 18 days (Leduc 1984). Cyanide has a strong, immediate, and long-lasting inhibitory effect on the swimming ability of fish (Leduc 1984). Free cyanide concentrations as low as 10 μg l^{-1} can rapidly and irreversibly impair the swimming ability of salmonids in well-aerated water (Doudoroff 1976). Osmoregulatory disturbances recorded at 10 μg HCN l^{-1} may affect migratory patterns, feeding, and predator avoidance (Leduc *et al.* 1982; Leduc 1984). Figure 5.1 shows that under laboratory conditions, fish experienced a significant reduction in relative performance (based on osmoregulation, growth, swimming, and spermatogenesis) at 10 μg HCN l^{-1} and survived indefinitely at 30 μg HCN l^{-1} (Leduc 1978, 1981; Leduc *et al. 1982*). Laboratory findings, however, may not be applicable to cyanide-stressed natural environments that are subject to wide fluctuations in physicochemical regimes. Cyanide, like many other chemicals, stimulates growth of fish during exposure to low sublethal levels. This phenomenon, known as hormesis, is little understood and warrants additional research (Leduc 1984).

Cyanide adversely affects fish reproduction by reducing both the number of eggs spawned and the viability of the eggs by delaying the process of secondary yolk deposition in the ovary (Lesniak and Ruby 1982; Ruby *et al.* 1986). Vitellogenin, a glycolipophosphoprotein present in plasma of fish during the process of yolk formation, is synthesized in the liver under stimulation of estrogen and subsequently sequestered in the ovary; it is essential for normal egg development. Exposure of naturally reproducing female rainbow trout (*Oncorhynchus mykiss*) to as little as 10 μg HCN l^{-1} for 12 days during the onset of the reproductive cycle produced a reduction in plasma vitellogenin levels and a reduction in ovary weight. The loss of vitellogenin in the plasma removed a major source of yolk (Ruby *et al. 1986*). Reproductive impairment in adult bluegills (*Lepomis macrochirus*) occurred following exposure to 5.2 μg CN l^{-1} for 289 days (USEPA 1980). Newly fertilized fish eggs were usually resistant to cyanide prior to blastula formation, but delayed effects occurred at 60 to 100 μg HCN l^{-1}, including birth defects and reduced survival of embryos and newly hatched larvae (Leduc *et al. 1982*). Concentrations as low as 10 μg HCN l^{-1} caused developmental abnormalities in embryos of Atlantic salmon (*Salmo salar*) after extended exposure (Leduc 1978). These abnormalities, which were absent in controls, included yolk sac dropsy and malformations of eyes, mouth, and vertebral column (Leduc 1984).

Among aquatic invertebrates, adverse nonlethal effects occurred between 18 and 43 μg l^{-1}, and lethal effects between 30 and 100 μg l^{-1} – although some deaths occurred between 3 and 7 μg l^{-1} for the amphipod *Gammarus pulex* (Eisler 1991). Aquatic plants are comparatively tolerant to cyanide; adverse effects occurred at >160 μg free CN l^{-1} (Eisler 1991). Adverse effects of cyanide on aquatic plants are unlikely at concentrations that cause acute effects to most species of freshwater and marine fishes and invertebrates (USEPA 1980).

Biocidal properties of cyanide in aquatic environments may be significantly modified by water pH, temperature, and oxygen content; life stage, condition, and species assayed; previous exposure to cyanides; presence of other chemicals; and initial dose tested. There is general agreement that cyanide is more toxic to freshwater fishes under conditions of low dissolved oxygen (Doudoroff 1976; Towill *et al.* 1978; Smith *et al.* 1979; USEPA 1980; Leduc 1984); that pH levels within the range 6.8 to 8.3 have little effect on cyanide toxicity but enhance toxicity at more acidic pH (Smith *et al.* 1979; USEPA 1980; Leduc *et al.* 1982; Leduc 1984); that juveniles and adults are the most sensitive life stages and embryos and sac fry the most resistant (Smith *et al.* 1978, 1979; USEPA 1980; Leduc 1984); and that substantial interspecies variability exists in sensitivity to free cyanide (Smith *et al.* 1979; USEPA 1980). Initial dose and water temperature both modify the biocidal properties of HCN to freshwater teleosts. At low lethal concentrations (i.e., near 10 µg HCN l^{-1}), cyanide is more toxic at lower temperatures; at high, rapidly lethal HCN concentrations, cyanide is more toxic at elevated temperatures (Kovacs and Leduc 1982a, b; Leduc *et al.* 1982; Leduc 1984). By contrast, aquatic invertebrates are most sensitive to HCN at elevated water temperatures, regardless of dose (Smith *et al.* 1979). Season and exercise modify the lethality of HCN to juvenile rainbow trout (McGeachy and Leduc 1988); higher tolerance to cyanide correlates with higher activity induced by exercise and higher temperatures, suggesting a faster detoxification rate or higher oxidative and anaerobic metabolism. Low levels of cyanide that are harmful when applied constantly may be harmless under seasonal or other variations that allow the organism to recover and detoxify (Leduc 1981).

Acclimatization by fish to sublethal levels of cyanide through continuous exposure was theorized to enhance their resistance to potentially lethal concentrations (Leduc 1981, 1984), but studies with Atlantic salmon and rainbow trout indicate otherwise. Prior acclimatization of Atlantic salmon smolts to cyanide increased their tolerance only slightly to potentially lethal concentrations (Alabaster *et al.* 1983). Juvenile rainbow trout previously exposed to sublethal concentrations showed a marked reduction in fat synthesis and swimming performance when challenged with higher cyanide doses; effects were most pronounced at low water temperatures (Kovacs and Leduc 1982a).

Knowledge of cyanide interactions with other chemicals is essential to the understanding of cyanide toxicokinetics and in evaluating risk to living resources. Additive or synergistic toxicity of free cyanide to aquatic fauna may occur in combination with ammonia (Smith *et al.* 1979; Leduc *et al.* 1982; Alabaster *et al.* 1983; Leduc 1984) or arsenic (Leduc 1984). However, conflicting reports on the toxicity of mixtures of HCN with zinc or chromium (Towill *et al.* 1978; Smith *et al.* 1979; Leduc *et al.* 1982; Leduc 1984) require clarification. Formation of the nickel–cyanide complex markedly reduced the toxicity of both cyanide and nickel at high concentrations in alkaline pH. At lower concentrations and acidic pH, nickel–cyanide solutions increased in toxicity by more than 1000 times, owing to dissociation of the metallocyanide complex to form hydrogen cyanide (Towill *et al.* 1978). Mixtures of cyanide and ammonia may have interfered with seaward migration of Atlantic salmon smolts under conditions of low dissolved oxygen (Alabaster *et al.* 1983). The 96-h toxicity of mixtures of sodium cyanide and nickel sulfate to fathead minnows (*Pimephales promelas*) was influenced by water alkalinity and pH. Toxicity, as measured by LC50 values, decreased with increasing alkalinity and pH from 0.42 mg CN l^{-1} at 5 mg $CaCO_3$ l^{-1} and pH 6.5, to 1.4 mg CN l^{-1} at 70 mg $CaCO_3$ l^{-1} and pH 7.5, to 730 mg CN l^{-1} at 192 mg $CaCO_3$ l^{-1} and pH 8.0 (Doudoroff 1956).

5.3.2
Birds

Many species of migratory birds – including waterfowl, shorebirds, passerines, and raptors – were found dead in the immediate vicinity of gold-mine heap-leach extraction facilities and tailings ponds, presumably as a result of drinking the cyanide-contaminated waters (Clark and Hothem 1991; Henny *et al.* 1994; Hill and Henry 1996). About 7000 dead birds – mostly waterfowl and songbirds – were recovered from cyanide-extraction, gold-mine leach ponds in the western United States between 1980 and 1989; no gross pathological changes related to cyanide were observed in these birds at necropsy (Allen 1990; Clark and Hothem 1991). No gross pathology was evident in cyanide-dosed captive birds (Wiemeyer *et al.* 1986), and this is similar to the findings of laboratory studies with cyanide and other animal orders that were tested and examined (Eisler 1991).

Free cyanide levels associated with high avian death rates have included 0.12 mg l^{-1} in air, 2.1–4.6 mg kg^{-1} body weight (BW) via acute oral exposure, and 1.3 mg kg^{-1} BW administered intravenously. In cyanide-tolerant species, such as the domestic chicken (*Gallus domesticus*), dietary levels of 135 mg total CN kg^{-1} ration resulted in growth reduction of chicks, but 103 mg total CN kg^{-1} ration had no measurable effect on these chicks (Eisler 1991; Hill and Henry 1996). First signs of cyanide toxicosis in sensitive birds appeared between 0.5 and 5 min after exposure, and included panting, eye blinking, salivation, and lethargy (Wiemeyer *et al.* 1986). In more tolerant species, signs of toxicosis began 10 min after exposure. At higher doses, breathing in all species tested became increasingly deep and labored, followed by gasping and shallow intermittent breathing. Death usually followed in 15 to 30 min, although birds alive at 60 min frequently recovered (Wiemeyer *et al.* 1986). The rapid recovery of some birds exposed to cyanide may be due to the rapid metabolism of cyanide to thiocyanate and its subsequent excretion. Species sensitivity to cyanide is not related to body size but seems to be associated with diet (Wiemeyer *et al.* 1986). Birds that feed predominantly on flesh, such as black vultures (*Coragyps atratus*), American kestrels (*Falco sparverius*), and eastern screech-owls (*Otus asio*), were more sensitive to NaCN than were species that feed mainly on plant material – with the possible exception of mallards (*Anas platyrhynchos*) – as judged by acute oral LD50 values (Table 5.1).

Some birds may not die immediately after drinking lethal cyanide solutions. Sodium cyanide rapidly forms free cyanide in the avian digestive tract (pH 1.3–6.5), whereas formation of free cyanide from metal cyanide complexes is comparatively slow (Huiatt *et al.* 1983). A high rate of cyanide absorption is critical to acute toxicity, and absorption may be retarded by the lower dissociation rates of metal-cyanide complexes (Henny *et al.* 1994). In Arizona, a single red-breasted merganser (*Mergus serrator*) was found dead 20 km from the nearest known source of cyanide, and its pectoral muscle tissue tested positive for cyanide (Clark and Hothem 1991). A proposed mechanism to account for this phenomenon involves weak-acid dissociable (WAD) cyanide compounds. Cyanide bound to certain metals, usually copper, is dissociable in weak acids such as stomach acids. Clark and Hothem (1991) suggested that drinking of lethal cyanide solutions by animals may not result in immediate death if the cyanide level is sufficiently low; these animals may die later when additional cyanide is liberated by stomach acid. More research is needed on WAD cyanide compounds and delayed mortality.

Table 5.1. Single dose toxicity of sodium cyanide (in mg NaCN kg^{-1} body weight) fatal to 50% of selected birds and mammals (listed from most sensitive to most tolerant)

Species	Oral LD50 (95% confidence limits)	Reference[a]
Mallard (*Anas platyrhynchos*)	2.7 (2.2–3.2)	1
Human (*Homo sapiens*)	3.0 estimated	2
American kestrel (*Falco sparverius*)	4.0 (3.0–5.3)	3
Coyote (*Canis latrans*)	4.1 (2.1–8.3)	4
Black vulture (*Coragyps atratus*)	4.8 (4.4–5.3)	3
Laboratory rat (*Rattus norvegicus*)	5.1–6.4	5, 6
Little brown bat (*Myotis lucifugus*)	8.4 (5.9–11.9)	7, 8
Eastern screech-owl (*Otus asio*)	8.6 (7.2–10.2)	3
House mouse (*Mus musculus*)	8.7 (8.2–9.3)	7, 8
Japanese quail (*Coturnix japonica*)	9.4 (7.7–11.4)	3
European starling (*Sturnus vulgaris*)	17 (14–22)	3
Domestic chicken (*Gallus domesticus*)	21 (12–36)	3
White-footed mouse (*Peromyscus leucopus*)	28 (18–43)	7, 8

[a] *1*, Henny *et al.* (1994); *2*, Way (1981); *3*, Wiemeyer *et al.* (1986); *4*, Sterner (1979); *5*, Ballantyne (1987); *6*, Egekeze and Oehme (1980); *7*, Clark *et al.* (1991); *8*, D.R. Clark Jr. (unpubl.).

Cyanide is a respiratory poison because of its affinity for the cytochrome oxidase complex of the mitochondrial respiratory chain (Keilin 1929; Nicholls *et al.* 1972). High dosages of cyanide are lethal through inhibition of cytochrome oxidase via cessation of mitochondrial respiration and depletion of ATP (Jones *et al.* 1984). Mallards given single oral doses of KCN (1.0 mg KCN kg^{-1} BW) at cyanide concentrations and amounts similar to those found at gold-mining tailings ponds (40 mg $CN^{-}l^{-1}$) – although it is NaCN that is used almost exclusively in mining – had elevated concentrations of creatine kinase in serum, suggesting tissue damage (Pritsos and Ma 1997). At 0.5 mg KCN kg^{-1} BW, mitochondrial function (an indicator of oxygen consumption) and ATP concentrations were significantly depressed in heart, liver, and brain (Ma and Pritsos 1997). Rhodanese and 3-mercaptopyruvate sulfurtransferase – two enzymes associated with cyanide detoxification – were induced in brain but not in liver or heart of KCN-dosed mallards. Although cyanide concentrations as high as 2.0 mg KCN kg^{-1} BW (at 80 mg $CN^{-}l^{-1}$) were not acutely toxic to mallards, the long term effects of such exposures were not determined and may have serious consequences for migratory birds exposed sublethally to cyanide at gold mine tailings ponds.

5.3.3
Mammals

Gold and silver mining are probably the most widespread sources of anthropogenic cyanides in critical wildlife habitat, such as deserts in the western United States (Hill and Henry 1996). Between 1980 and 1989, 519 mammals – mostly rodents (35%) and bats (34%) – were reported as found dead at cyanide-extraction, gold-mine mill tailings and heap leach ponds in California, Nevada, and Arizona (Clark and Hothem 1991). The list also included such species as coyote (*Canis latrans*), badger (*Taxidea taxus*), beaver (*Castor canadensis*), mule deer (*Odocoileus hemionus*), blacktail jackrabbit (*Lepus californicus*), and kit fox (*Vulpes macrotis*), as well as skunks, chipmunks, squirrels, and domestic dogs, cats, and cattle. Also found dead at these same ponds were 38 reptiles, 55 amphibians, and 6997 birds. At the time of this study (1980–1989) there were about 160 cyanide-extraction gold mines operating in California, Nevada, and Arizona and these mines were operating within the geographic ranges of 10 endangered, threatened, or otherwise protected species of mammals. Bats comprised 6 of the 10 listed species. Because bats were not identified to species, members of these 6 protected species could have been among the 174 reported dead bats (Clark and Hothem 1991). A population of Townsend's big-eared bats (*Plecotus townsendii*), one of the 10 protected species, may have been extirpated by cyanide at a nearby mine in California (Dr. P. Brown, pers. comm.). Badgers were another of the 10 protected species; 6 were counted among the 519 mammals reported dead. A vat-leach gold mine in South Carolina with a large tailings pond reported 271 dead vertebrates found in the immediate vicinity between December 1988 and the end of 1990; 86% were birds, 13% mammals (29 of 35 were bats), and the rest reptiles and amphibians (Clark 1991).

Signs of acute cyanide poisoning in livestock usually occur within 10 min and include initial excitability with muscle tremors, salivation, lacrimation, defecation, urination, and labored breathing, followed by muscular incoordination, gasping, and convulsions; death may occur quickly, depending on the dose administered (Towill *et al.* 1978; Cade and Rubira 1982). Acute oral LD50 values for representative species of mammals ranged between 4.1 and 28.0 mg HCN kg^{-1} BW and overlapped those of birds (Table 5.1). Despite the high lethality of large single exposures, repeated sublethal doses – especially in diets – are tolerated by many species for extended periods, perhaps indefinitely (Eisler 1991). Livestock found dead near a cyanide disposal site had been drinking surface water runoff that contained up to 365 mg HCN l^{-1} (USEPA 1980). Rats exposed for 30 days to 100 or 500 mg KCN l^{-1} drinking water had mitochondrial dysfunction, depressed ATP concentrations in liver and heart, and a depressed growth rate; little effect was observed at 50 mg KCN l^{-1} (Pritsos 1996). The adverse effect on growth is consistent with the biochemical indicators of energy depletion. However, the concentrations should be viewed with caution as cyanide may have volatilized from the water solutions prior to ingestion by the rats, due to presumed neutral pH.

Hydrogen cyanide in the liquid state can readily penetrate the skin (Homan 1987). Skin ulceration has been reported from splash contact with cyanides among workers in the electroplating and gold extraction industries, although effects in those instances were more likely due to the alkalinity of the aqueous solutions (Homan 1987). In one case, liquid HCN ran over the bare hand of a worker wearing a fresh air respirator; he collapsed into unconsciousness in 5 min, but ultimately recovered (USEPA 1980). No human cases of illness or death due to cyanide in water supplies are known (USEPA 1980). Accidental acute cyanide poisonings in humans are rare (Towill *et al.* 1978); however, a male accidentally splashed with molten NaCN died about 10 h later (Curry 1963).

5.4
Proposed Mitigation

Aquatic birds are naturally attracted to large open ponds, and efforts to deter or chemically repel them have been generally ineffective (Hill and Henry 1996). However, some chemical repellents when added to dump leachate pond water showed promise at reducing consumption of leachate water when tested on European starlings (*Sturnus vulgaris*), especially o-aminoacetophenone and 4-ketobenztriazine (Clark and Shah 1993). Exclusion from cyanide solutions or reductions of cyanide concentrations to nontoxic levels are the only certain methods of protecting vertebrate wildlife from mine water poisoning (Henny *et al.* 1994). Mortality of migratory birds from cyanide toxicosis may be curtailed at small ponds associated with leach heaps by screening birds from toxic solutions (Hallock 1990). Fencing and covering of small solution ponds with polypropylene netting have proved effective for excluding most birds, bats, and larger mammals, provided that the fencing and netting are properly maintained (Henny *et al.* 1994). A few mines in Nevada are now covering surfaces of small ponds with approximately 10.2 cm-diameter high density polyethylene balls (Wiemeyer, pers. observations); birds are no longer attracted to these ponds as water sources. Although initial costs of the balls are higher than installation of netting, there are no maintenance expenses for the balls, whereas netting needs continual maintenance. Gold mine operators in southern California and Nevada used plastic sheeting to cover the cyanide leach pond, resulting in a cessation of wildlife mortality. The comparatively high cost of this process was soon recouped through reduced evaporation of water and cyanide (Wiemeyer, pers. observations; Clark and Hothem, pers. observations).

Cyanide concentrations in the water column of mill tailings ponds were reduced at one Nevada site using naturally detoxified recycled tailings water (Henny *et al.* 1994). Lowering the cyanide concentrations in tailings ponds with hydrogen peroxide has been successful at a few mines in Nevada (Allen 1990), but this procedure is still preliminary (Clark and Hothem 1991). To reduce the potential for puddling on ore heaps, ores should be less compacted; this can be accomplished by reducing the clay content of the ores and stacking ores using conveyer belts rather than trucks (Henny *et al.* 1994). Puddling can also be reduced by careful monitoring of solution application rates and maintenance of solution distribution systems. Wildlife have been excluded from leaching solution on the heaps by substituting drip lines for sprinklers and covering the drip lines with a layer of gravel (Henny *et al.* 1994; Hill and Henry 1996). Some mines use small net panels over areas of puddling on heaps and in collection channels to exclude birds and mammals (Henny *et al.* 1994).

Water hyacinth (*Eichornia crassipes*) has been proposed as the basis of a cyanide removal technology. Hyacinths can survive for at least 72 h in a nutrient solution containing as much as 300 mg CN l^{-1} and can accumulate up to 6.7 g CN kg^{-1} DW plant material. On this basis, 1 ha of hyacinths has the potential to absorb 56.8 kg cyanide in 72 h, and this property may be useful in reducing the level of total cyanide in untreated wastewaters where concentrations generally exceed 200 mg CN l^{-1} (Low and Lee 1981). Large scale use of water hyacinths for this purpose has not yet been implemented, possibly due to disagreement over appropriate disposal mechanisms.

Free cyanide criteria currently proposed for the protection of natural resources include <3 $\mu g\ l^{-1}$ medium for aquatic life, and <100 mg kg^{-1} diet for birds and livestock

(Eisler 1991). For human health protection, free cyanide values are <10 $\mu g\ l^{-1}$ drinking water, <50 $mg\ kg^{-1}$ diet, and <5 $mg\ m^{-3}$ air (Eisler 1991). Additional research seems needed to establish legally enforceable standards and threshold limit values for potentially toxic cyanides in various forms, including HCN and inorganic cyanide. This includes research on low-level, long-term cyanide intoxication in birds and mammals by oral and inhalation routes in the vicinities of high cyanide concentrations, especially on the incidence of nasal lesions, thyroid dysfunction, and urinary thiocyanate concentrations (Towill *et al.* 1978; Egekeze and Oehme 1980). Research is also needed on threshold limits in water where birds and mammals may be exposed, including the role of CN-metal complexes, and on sublethal effects of free cyanide on vertebrate wildlife. In aquatic systems, research is needed on: (1) long-term effects of low concentrations of cyanide on growth, survival, metabolism, and behavior of a variety of aquatic organisms (Towill *et al.* 1978; Leduc *et al.* 1982; Eisler 1991); (2) adaptive resistance to cyanide and the influence, if any, of oxygen, pH, temperature and other environmental variables (Leduc 1981, 1984); and (3) usefulness of various biochemical indicators of cyanide poisoning, such as cytochrome oxidase inhibition (Gee 1987) and vitellogenin levels in fish plasma (Ruby *et al.* 1986).

Acknowledgments

We thank P.F.P. Henry and O.H. Pattee for scientific and technical review of the manuscript.

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Chapter 6

A Three-dimensional Finite Element Model to Predict Airflow and Pit Retention for an Open-Pit Mine

Ragula Bhaskar · Navin Tandon

6.1
Introduction

Surface mining operations (e.g., blasting, loading, hauling, and crushing) are sources of airborne particles. The estimation of concentrations of fugitive dust for an open-pit mining situation has traditionally been done using United States Environmental Protection Agency (EPA) models such as the Industrial Source Complex (ISC) model. Since the dust-producing operations at open-pit mines may occur at depths up to several hundred meters below grade, only a fraction of fugitive dust generated inside the pit escapes to the surface, where it may then be transported to mine boundaries. This tendency for particulate matter (PM) to remain inside the pit has been called "pit retention" (TRC 1985). The magnitude of pit retention is expressed as "escape fraction", defined as the mass fraction of emissions that escapes from the mine pit to the surface. The two mechanisms occurring simultaneously that contribute to the pit retention phenomenon are: the decoupling of the wind field in the pit from the wind field at the surface, and the deposition and settling of particulate on the mine pit surface and along the pit walls. A dispersion model ignoring the influence of pit retention will overpredict the downwind concentrations. The presence of a mine pit disturbs the airflow above and inside the pit, so that the plume of dust may not have the Gaussian distribution imposed by many dispersion models.

6.1.1
Approaches Used to Study Air Pollution Issues in Open-Pit Mines

There are three primary techniques identified in the literature that have been historically used to study problems related to the airflow and pit retention for surface mines: field experiments, wind tunnel modeling, and mathematical modeling. Each of these techniques has advantages and limitations. The level of uncertainty involved with these techniques is a function of assumptions utilized, number of independent parameters considered, and generalization and extrapolation of results.

Full-scale experiments are expensive and time-consuming, especially in complex terrain. Extensive measurements and analyses are required for wind, temperature, and concentration distribution to gain a sufficient understanding of the fundamental physics. Generalization from field data is difficult because of peculiarities of specific sites and meteorological conditions. Controlled variation of independent variables is generally not possible, and complicating factors are abundant. However, it is understood that field experiments can provide the "real-world" data to test the models. Although field studies in the vicinity of surface mines have undoubtedly been influenced by pit retention, very few studies have specifically addressed pit retention. There are studies reported in the literature in which the investigators detected discrepancies between the measured and modeled concentrations at surface mines, and attributed the discrepancies to pit retention (TRC 1985). In a field study conducted at four mines in the western United States, smoke generators at the bottom of the pits were used to release discrete 10-s puffs of diesel fuel smoke. An escape velocity, essentially the net upward velocity within each pit, was computed from the observed retention time of the tracers and the depth of each pit. This upward velocity, when compared to the downward settling and deposition velocity for different size particles, was the basis for the calculation of an escape fraction (TRC 1985).

Wind tunnel modeling comes under the general category of physical modeling or fluid modeling, and is an analog modeling technique for fluid-dynamic processes. In fluid modeling, a scale model of terrain, plant, buildings, and obstructions is used. Certain non-dimensional parameters must be duplicated in the wind tunnel model and the actual full-scale situation. Wind tunnel modeling has been typically employed to study plants in complex terrain or to determine the effect of building turbulence on dispersion from stacks. A detailed guide (Snyder 1981) has been published by the EPA to establish the procedures for fluid modeling. Fluid modeling has been found to be most effective in simulating neutral atmospheric conditions. The flow in a wind tunnel can be controlled and specific parameters can be independently adjusted. After the Clean Air Act Amendments of 1990 in the United States, the EPA's methods for modeling particulate matter less than 10µm in aerodynamic diameter (PM-10) for open-pit mines was reexamined (Perry *et al.* 1994; Thompson 1994). The associated wind tunnel study was performed at the EPA's Fluid Modeling Facility to investigate dispersion from surface coal mines (or similar sources) in support of the dispersion modeling activities. The study involved measurement of steady-state, tracer-gas (ethane) concentration fields downwind of model mines of various shapes, sizes, and orientations with low-momentum, point-source releases of a neutrally buoyant gas from various locations in the model pits. The mean flow in a mine model was observed as a large vortex, with the flow at the top of the mine in the direction of flow aloft and the flow along the mine floor moving upwind (against the direction of mean flow aloft). Considering the effect of the recirculation phenomenon, it was concluded that only the upwind edge of the mine model contributes to emissions.

Mathematical models encompass such concepts as empirical box and statistical models, semi-empirical Gaussian plume and trajectory models, and numerical multibox, grid and particle models. Mathematical models use mathematical techniques to represent the actual physical processes governing atmospheric flow dynamics and pollutant transport. Numerical models are very versatile. By making varying degrees of approximations and assumptions, numerical models can be tuned to each application. Advection by wind components, turbulent diffusion, chemical reactions, wet and dry deposition of pollutants, and other atmospheric processes can all be included in the numerical models. Several studies utilizing mathematical modeling methodologies were found in the literature (Lee 1977; Herwehe 1984; Perdikaris and Mayinger 1995). The major advantages in using numerical models include better control over input data specification, and the capability to provide useful information for meteorological and air pollution scenarios in a fast, reliable and inexpensive way compared with the observational approach.

6.1.2
Surface Mine Escape Fractions Models

Empirical equations are frequently used to estimate the escape fraction of dust from surface mines. Two of these equations, derived by Fabrick and by Winges, have been discussed in detail by TRC (1985). Fabrick derived a mine pit escape fraction equation that depends upon the width of the pit, the wind speed at the top of the pit, and the particle size distribution:

$$\in = 1 - V_d\left[\frac{C}{u}\left(\frac{1}{2} + \ln\frac{w}{4}\right)\right], \tag{1}$$

where $\in$ is the escape fraction, u is the wind speed (m s^{-1}), w is the pit width (m), V_d is the larger of deposition or settling velocity (m s^{-1}), and C is an empirical dimensionless constant with a value of 7.

Winges developed an equation to calculate the particulate escape fraction from surface mine pits. The escape fraction, $\in$, is given by:

$$\in = \frac{1}{1 + \left(\frac{V_d}{K_z}\right)H}, \tag{2}$$

where V_d is the larger of deposition or settling velocity (m s^{-1}), K_z is the vertical diffusivity ($m^2 s^{-1}$), and H is the pit depth (m).

The new ISC3 model (EPA 1995), with its algorithm for modeling impacts of particulate emissions from open-pit sources, considers the pit retention phenomenon and hence attempts to eliminate overprediction of PM-10 concentrations. The ISC3 open-pit source model can be used to simulate fugitive emissions from below-grade open pits. The ISC3 model allows the open-pit source to be characterized by a rectangular shape with an aspect ratio (length to width) of up to 10 to 1. The open-pit model accounts for partial retention of emissions within the pit by calculating an escape fraction for each particle size category. The escape fraction for each particle size category, $_{\epsilon\, i}$, is calculated as follows (EPA 1995):

$$\in_i = \frac{1}{1 + V_g/(\alpha U_r)}, \tag{3}$$

where V_g is the gravitational settling velocity (m s^{-1}), U_r is the approach wind speed at 10 m (m s^{-1}), and α is the proportionality constant whose value is set as 0.029. The variations in escape fractions across the particle sizes result in a modified distribution of mass escaping from the pit. Based on the fluid modeling performed at the EPA's Fluid Modeling Facility, pit emissions are assumed to have a tendency to escape from the upwind side of the pit. The open-pit algorithm simulates the escaping pit emissions by using an effective rectangular area source (a fraction of the entire pit opening) using the ISC3 area source algorithm. The shape, size, and location of the effective area source vary with the wind direction and the relative depth of the pit. It is assumed that because of the high level of turbulence in a mine, a pollutant is initially mixed prior to exiting the pit.

It can be concluded from the discussion above that empirical models have strong simplifying assumptions. The escape fraction equations generally consider few parameters, which may not be sufficient to characterize all the complexities of the pit retention phenomenon. If more site-specific and accurate results are desired, advanced mathematical tools, such as finite element models, can be used.

6.2
Overview of the Open-Pit Finite Element Model

This chapter primarily presents a three-dimensional finite element numerical modeling technique to simulate the airflow patterns, dust transport and diffusion, and to quantify the pit retention of dust for a large open-pit mine in the western United States. The Fluid Dynamics Analysis Package (FIDAP 7.5) was used in the study. Reynolds-averaged flow equations were solved to generate turbulent flow fields using the standard κ-ε (kappa-epsilon) turbulence model in combination with the near-wall approach. Subsequently, particle transport, diffusion, and pit retention were evaluated through the use of a Lagrangian stochastic model for a given flow simulation. Sensitivity studies were conducted in order to better understand the behavior of fugitive dust under various input conditions. The sensitivity to the following parameters was evaluated: wind speed, wind direction, atmospheric stability, source location and height, and particle size. The model showed the escape fraction to be a function of these meteorological and source parameters.

6.3
Theoretical Analysis

Numerical modeling has been used extensively to study air pollution problems. Herwehe (1984) developed a two-dimensional finite element model to simulate the transport, diffusion, and dry deposition of fugitive dust emitted from an idealized open-pit surface mine.

The detailed theoretical analysis and modeling technique utilized in this paper is included in Tandon (1998).

6.3.1
Atmospheric Turbulence Modeling

The various motions of the air in earth's atmosphere, from a slight breeze in the surface layer to a general atmospheric circulation of planetary scale, are turbulent. Atmospheric turbulence plays a fundamental role in the thermal and dynamic interaction between the atmosphere and the ground surface. Atmospheric turbulence also determines the spreading of pollutants in the air. Turbulent flows are represented in a majority of flow simulations by the ensemble-averaged conservation equations called Reynolds-averaged equations. The mean flow equations to simulate a turbulent isothermal flow with constant fluid properties may be represented as follows (Haroutunian and Engelman 1991, 1993):

Continuity:

$$\frac{\partial u_i}{\partial x_i} = 0\,; \tag{4}$$

Momentum:

$$\rho \frac{\partial u_i}{\partial t} + \rho u_j \frac{\partial u_i}{\partial x_j} = -\frac{\partial p}{\partial x_i} + \frac{\partial}{\partial x_j}\left[\mu\left(\frac{\partial u_i}{\partial x_j} + \frac{\partial u_j}{\partial x_i}\right) - \rho\overline{u_i' u_j'}\right]. \tag{5}$$

In the above equations, u_i are the components of the mean velocity vector in the Cartesian coordinate system x_i, t is the time coordinate, p is the mean fluid pressure, ρ is the fluid density, and μ is the molecular viscosity. This formulation allows the characteristics of the mean flow to be investigated without having to resolve all the intricate details of the turbulence field. A significant drawback of this approach, however, is that an unknown statistical correlation $\rho u_i' u_j'$ enters the flow equations as a result of the averaging process. This Reynolds stress tensor represents the mean turbulent flux of momentum in the three principal spatial directions. The prime notation used for the Reynolds stress tensor denotes a fluctuating variable. Mathematical turbulence modeling is utilized to approximate these stresses or fluxes in terms of mean flow characteristics. A large number of turbulence models have been explained by Rodi (1984).

The standard κ-ε turbulence model is a very useful model and was first proposed by Launder and Spalding (1974). By using the generalized Boussinesq eddy viscosity concept, Reynolds stresses can be expressed as (Haroutunian and Engelman 1993):

$$-\rho\overline{u_i' u_j'} \cong \mu_t\left(\frac{\partial u_i}{\partial x_j} + \frac{\partial u_j}{\partial x_i}\right) - \frac{2}{3}\rho\delta_{ij}\kappa\,, \tag{6}$$

where δ_{ij} is the Kronecker delta function, μ_t is the turbulent viscosity, and κ is the turbulent kinetic energy, which is expressed as:

$$\kappa = \frac{\overline{u'^2} + \overline{v'^2} + \overline{w'^2}}{2}\,, \tag{7}$$

where u', v', and w' are the velocity fluctuations in x, y, and z directions. The use of the Boussinesq approach shifts emphasis from modeling many unknown turbulent fluxes to a single unknown μ_t, which is expressed as:

$$\mu_t = c_\mu \rho \frac{\kappa^2}{\varepsilon}\,, \tag{8}$$

where $c_\mu = 0.09$ is an empirical model coefficient, and ε is the viscous dissipation rate of turbulent kinetic energy κ. The transport equations for κ and ε are represented by a two-equation model (Haroutunian and Engelman 1993).

The standard κ-ε model is not appropriate for modeling low turbulence near-wall regions adjacent to solid boundaries. In order to accurately resolve the velocity profiles in the near-wall finite elements, specialized shape functions are used. The functional form used for the velocity profile in the near-wall region is that developed by Reichardt (as explained by Haroutunian and Engelman 1991), which is as follows:

$$u^+ = f_R\left(y^+\right) = \frac{1}{k}\ln\left(1 + 0.4y^+\right) + 7.8\left[1 - \exp\left(-\frac{y^+}{11}\right) - \frac{y^+}{11}\exp\left(-0.33y^+\right)\right] \tag{9}$$

In this equation, k is the von Karman constant, u^+ and y^+ are the dimensionless velocity and distance, which are defined as:

$$u^+ = \frac{u}{u_*} \tag{10}$$

$$y^+ = \frac{\rho u_* y}{\mu}, \tag{11}$$

where u_* is the friction velocity.

In the near-wall region, the variation of turbulent viscosity (μ_t) is formulated by using van Driest's mixing length model (as explained by Haroutunian and Engelman 1991). Thus, μ_t is expressed as:

$$\mu_t = \rho l_m^2\left[\left(\frac{\partial u_i}{\partial x_j} + \frac{\partial u_j}{\partial x_i}\right)\frac{\partial u_i}{\partial x_j}\right]^{1/2}, \tag{12}$$

where l_m is the mixing length obtained from the van Driest's expression:

$$l_m = ky\left[1 - \exp\left(\frac{-y^+}{A}\right)\right]. \tag{13}$$

In the above equation, y is the normal distance from the wall, and A is an empirical constant that assumes a value of about 26 for smooth walls in the equilibrium near-wall layers. The dimensionless normal distance from the wall, y^+, is defined in terms of turbulent kinetic energy at the top of the element (κ_t), and is expressed as:

$$y^+ = \rho\left(c_\mu^{1/2}\kappa_t\right)^{1/2}\frac{y}{\mu}. \tag{14}$$

6.3.2
Particle Dispersion in Turbulent Flow

In order to predict particulate two-phase flows, the Lagrangian approach (as opposed to the Eulerian approach) has been utilized in this paper since it can handle two-phase flows consisting of polydispersed particle size distributions. The Lagrangian approach treats the fluid phase as a continuum and predicts the trajectories of particles in the fluid flow as a result of various forces acting on the particles. Elghobashi (1994) summarized that the interaction between particles and turbulence is usually specified as one-way coupling for dilute mixtures (as in the case of this chapter). Thus, while particle dispersion depends on the state of turbulence, particles have an insignificant effect on the flow.

In the Lagrangian approach, the motion of each particle of the dispersed phase is governed by an equation that balances the mass-acceleration of the particle with the forces acting on it. The particles are assumed to be spherical in this analysis. Considering that only drag and gravity forces are acting on the particle, the relevant governing equation for the motion of the particle is (adapted from FIDAP Manual 1993, 1995):

$$\frac{du_p}{dt} = \frac{(u_f - u_p)}{\tau} + \frac{(\rho_p - \rho_f)}{\rho_p} g , \tag{15}$$

where u_p is the particle velocity, u_f is the velocity of fluid, ρ_p is the particle density, ρ_f is the fluid density, and τ is the particle relaxation time.

Turbulence in the flow field transfers momentum to the particles and hence results in the spreading of particles. The fluid velocity variations in the turbulent field determine the extent of particle dispersion. The typical approach for the approximation of the velocity experienced by a particle is a "random walk" model, which assumes the carrier phase velocity to be a sum of local mean velocity and random fluctuations. A stochastic approach in conjunction with the κ-ε model (assuming isotropic turbulence) allows the evaluation of velocity fluctuation as follows:

$$u' = \lambda \left(\frac{2}{3} \kappa \right)^{1/2} , \tag{16}$$

where λ is a random number sampled from a normal distribution (between 1 and –1).

Due to the turbulent nature of the flow field, different particles introduced at a fixed point in the computational domain may have different trajectories. In this study, these particles were introduced and subsequently tracked with the passage of time. The ratio of the number of trajectories leaving the downwind mine boundary to the total number of trajectories introduced is used to represent the escape fraction $\in$. Pit retention is calculated as $(1-\in)$.

6.4
Model Development

The three-dimensional finite element open-pit model was developed using the Fluid Dynamics Analysis Package (FIDAP 7.5). The four model development steps are explained as follows.

1. Three-dimensional finite element mesh

In order to perform a computer simulation, it is essential to define the geometry of the flow domain and then generate the finite element mesh. A contour map of the area was used to select several points on the terrain at and around the open-pit mine. The lower boundary of the computational domain was specified by a surface containing the x, y, and z coordinates of these points. Subsequently, a mapped mesh consisting of 4-node linear elements was created on this surface. The upper boundary of the domain was specified as the height of the mixed layer determined from the rawindsonde (air sounding) data for this geographical area (Draxler and Heffter 1981). The two-dimensional mapped mesh on the surface was then projected in the z-direction to generate a three-dimensional finite element mesh composed of 8-node linear elements. The mesh grading was changed from fine to coarse in the upward direction. The mapped mesh generated on the ground surface is illustrated in Fig. 6.1. The six surfaces of the computational domain were specified as "open pit", "top", "north", "south", "east", and "west".

2. Boundary and initial conditions

Appropriate boundary conditions are needed to be specified for κ, ε, and the three components of the velocity vector at all the boundaries of the computational domain. The boundary faces can be classified as inlet planes, symmetry planes, outlet planes, and walls. The boundary conditions specified are dependent on the situation being analyzed. An example of a boundary condition is the specification of all components of the velocity vector as zero at the wall (ground surface).

In order to improve the convergence characteristics of the simulation runs, nonzero initial guess values are used for the variables.

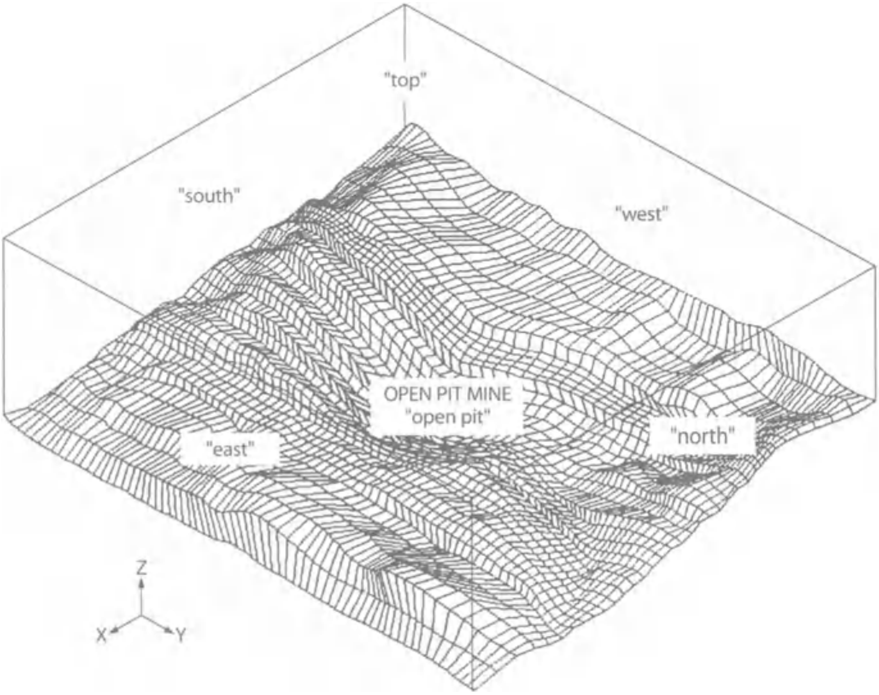


Fig. 6.1. Two-dimensional finite element mesh on the ground surface

3. Model definition data and execution

This analysis considered air to be incompressible and a Newtonian fluid. In addition, isothermal conditions were assumed. In order to minimize computational times, a "segregated solver" was used (FIDAP User Manuals 1993, 1995). After specifying the input information, the model was run on a Silicon Graphics Power Challenge XL super-computer. Each simulation run utilized about 70 to 80 h of processing time.

4. Particle characteristics and trajectories

As mentioned earlier, Lagrangian formulation with one-way coupling was used to describe the particle motion. Particle interaction for each of the six surfaces was specified as "escape", "rebound", or "trap". Particle trajectories were generated after the airflow was determined for a particular scenario. Since a stochastic model was used, 500 trajectories were introduced at each emission point and particles were tracked for each case. It was determined that 500 particles represent an optimal number of particles to obtain statistically significant results, while minimizing the computational costs. This provided an estimation of the pit retention for the simulation case.

6.5
Validation and Comparison

6.5.1
Numerical Tests and Validation

Numerical models are mathematical tools that use a set of numerical algorithms to describe the physical aspects of a problem. Therefore, it is essential to conduct numerical tests to validate a model to develop an understanding of its performance. The performance of a numerical model can be demonstrated by comparing the results with experimental/analytical results for some classical simple problems. If the model predicts results similar to those obtained with analytical or experimental studies, the model may be applied to more complex situations for which analytical/experimental results do not exist.

The details of numerical validation studies performed for this paper have been explained by Tandon (1998). Two validation studies were performed for the two components of the open-pit model, viz. the standard κ-ε turbulence model with the near-wall approach and the Lagrangian particle model.

The standard κ-ε turbulence model with the near-wall approach was used for a two-dimensional, steady, incompressible flow over a backward-facing step. The numerical results obtained in terms of the length of the recirculation region were compared to the available experimental data (Kim 1978, as cited in the FIDAP Examples Manual). This comparison demonstrated good performance of the technique utilized in this paper. A finite element model involving the transport of a small particle in two-dimensional laminar flow was used to validate the Lagrangian particle formulation. In this case, the settling velocity of the particle was determined and compared to the settling velocity obtained by utilizing the Stokes terminal velocity formulation. A close match between these values demonstrated satisfactory validation of the particle formulation utilized in this paper.

6.5.2
Idealized Versus Actual Geometries for Open-Pit Mines

The EPA's ISC3 model is frequently used to support the agency's regulatory air modeling programs in the United States. As was explained in Section 6.1.2, the ISC3's open-pit algorithm is based on the fluid modeling performed at the EPA's Fluid Modeling Facility. This wind tunnel modeling determined the presence of a recirculatory airflow inside a pit. Therefore, the ISC3 model implicitly assumes this recirculatory profile, and the pit emissions are consequently assumed to have a tendency to escape from the upwind side of the pit.

The application of finite element modeling for the actual pit in the present study indicated absence of such a recirculatory profile. For comparison purposes, a two-dimensional investigative analysis was performed by developing finite element flow models for idealized rectangular and trapezoidal cross-section configurations (similar to EPA's wind tunnel studies) and the actual pit geometry (Tandon 1998; Tandon and Bhaskar 1998). In all the cases, the evaluation was conducted for neutral atmospheric conditions and a wind speed of about 10 km h^{-1}. The results demonstrated that idealization of pit geometry (trapezoidal and rectangular cross-sections) induces a flow separation on the upwind edge of the pit, thereby causing a recirculatory wind profile inside the mine. These numerical modeling results mimicked EPA's wind tunnel results for idealized configurations. However, the recirculatory profile was absent for the actual pit configuration, which indicated that the presence (or absence) of a recirculatory profile is a function of pit geometry. The qualitative match between finite element and wind tunnel modeling results for idealized pit configurations demonstrates that the finite element modeling technique can be expected to predict realistic airflow patterns for the actual pit geometry.

6.6
Sensitivity Studies and Results

Due to the high degree of variability associated with meteorological parameters and emission sources in a typical mine, sensitivity studies were performed on the numerical model to determine the relative impact of each variable (Tandon 1998). Model sensitivity to the following parameters was evaluated: wind speed, wind direction, atmospheric stability, source location and height, and particle size. Atmospheric stability is a classification scheme to describe the atmospheric dispersion, that takes into account both the effects of mechanical turbulence and those due to buoyancy. There are six stability classes: A, B, and C represent unstable conditions, D is neutral, E and F are stable conditions.

The simulations in the study were planned so as to expedite performance of the sensitivity studies. The simulation cases used in the study are shown in Table 6.1. The input conditions for these simulation cases were either observational values for the site or values computed using meteorological principles mentioned in the literature (Panofsky and Dutton 1984; Zannetti 1990). The background information related to the specification of different input parameters in order to perform sensitivity studies is included in Tandon (1998).

The airflow patterns, dust dispersion, and pit retention were found to be functions of the source and meteorological parameters. For instance, the escape fraction was de-

Table 6.1. Simulation cases

Case no.	Wind speed (miles/h)	Wind direction (from)	Atmospheric stability[a]
1	6	North	D
2	6	North	A
3	6	North	F
4	2	North	D
5	10	North	D
6	30	North	D
7	6	South	D
8	6	West	D
9	6	East	D

[a] *A*, very unstable; *D*, neutral; *F*, stable.

termined to have a positive correlation with wind speed and a negative correlation with particle size. It was determined that the unstable atmospheric conditions induce the maximum spread of the dust plume in the horizontal and vertical directions, due to higher magnitudes of wind fluctuations and eddy sizes as compared to the neutral and stable atmospheric conditions. It was determined that as the depth of the emission source inside the pit increases, the escape fraction decreases. The escape fraction was determined to have a positive correlation with the height of the emission source above the ground. The site-specific conditions played an important role in determining the airflow patterns and the fate of dust particles.

Figure 6.2, illustrating the particle trajectories obtained for Case 1 (listed in Table 6.1), is presented as an example. In this particular simulation, 500 particles (each with an aerodynamic diameter of 10 µm) were released at a pit bottom ground-level emission location. Since 59 particle trajectories escaped the pit boundaries, the escape fraction was determined to be 11.8%.

Simulation Case 1 was also used to study the sensitivity of the escape fraction to aerodynamic particle size (with unit density) for a pit bottom ground-level emission point. As shown in Fig. 6.3, as aerodynamic particle size increases, the escape fraction decreases. This is due to the higher values of terminal settling velocities (and hence more gravitational settling) for larger particles.

Generally speaking, the escape fractions for the open-pit model in the study were determined to be significantly lower compared to other previously established approaches. The comparison between the finite element modeling results and the existing empirical escape fraction equations (i.e., Fabrick, Winges, and ISC3 model algorithm) is presented in Fig. 6.4. Due to the simplified nature of these empirical equations, certain assumptions were made in order to perform the comparative analysis. For instance, the pit width (w) was specified as 8150 ft (2485 m) in the Fabrick's equation. The pit depth (H) was specified as 2750 ft (838 m), and the vertical diffusivity (K_z) was specified as 6.74 $m^2 s^{-1}$ (value calculated for neutral stability, 10 km h^{-1} wind speed) in the Winges model. The values of deposition velocities (V_d) for different particle sizes

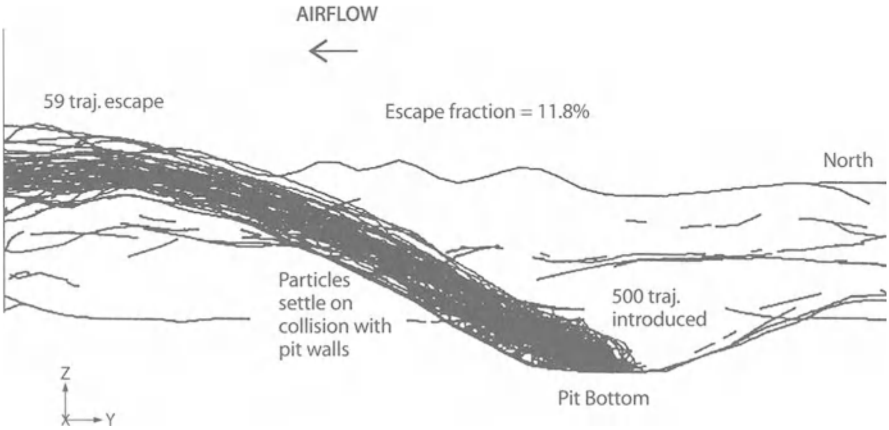


Fig. 6.2. Particle trajectories for Case 1 (viewed from *x* direction)

Fig. 6.3. Sensitivity of escape fraction to aerodynamic particle size

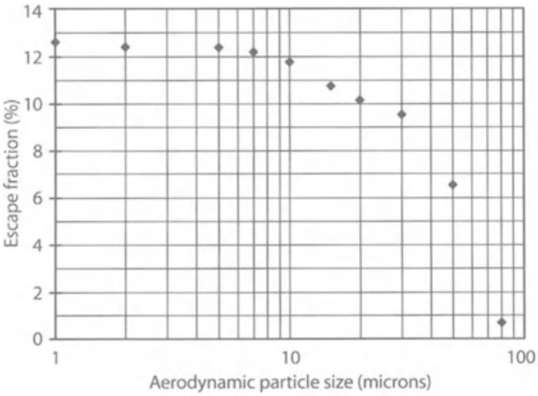
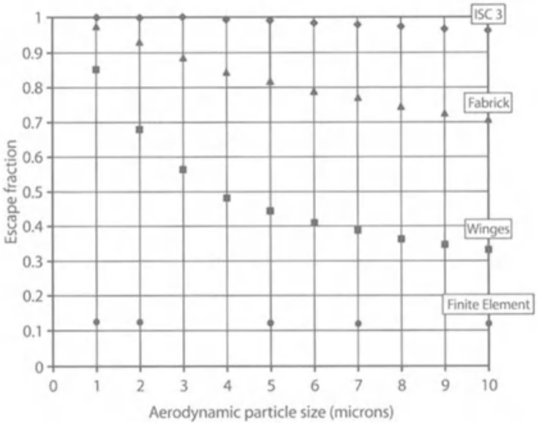


Fig. 6.4. Comparison with empirical escape fraction models



were computed using Fig. 10.4 of Hanna *et al.* (1982), using a roughness height (z_0) of 10 cm and a particle density of 1 g cm^{-3}. As illustrated in Fig. 6.4, the finite element model predicts escape fraction values much lower than the empirical escape fraction equations.

The open-pit algorithm of EPA's ISC3 model is based on wind tunnel studies and implicitly assumes the presence of recirculatory airflow in a pit. Numerical modeling using the finite element technique presented in this paper demonstrated that the idealization of pit geometry (as in the wind tunnel studies) may induce recirculatory airflow in a pit. However, air recirculation may not universally occur with actual pit geometry (Tandon 1998; Tandon and Bhaskar 1998).

6.7
Concluding Remarks

The technique presented in this paper can provide useful insights regarding the airflow patterns, dust dispersion, and pit retention phenomena for the open-pit mine as a function of varying parameters. This methodology is introduced as one of the alternate approaches that may be used to evaluate similar kinds of problems. The model is capable of simulating non-Gaussian dispersion, and can be expected to provide results that are more representative of actual open-pit mine conditions. The approach presented in this chapter addresses various independent source and meteorological parameters. Thus, the analysis presented is a more comprehensive technique as opposed to specifying a few independent parameters in empirical models (such as the EPA's ISC3 model escape fraction equation). However, the analysis of three-dimensional turbulent two-phase flows using the finite element method can be computationally very expensive.

Although the open-pit model is believed to be quite useful in its present form, there is substantial scope for improvement and future work. The technique presented has the potential to be used to derive generalized relations incorporating several independent parameters, which may be more useful from a practical standpoint. A few recommendations for improving model performance are: use of a finer mesh; better parameterization of flow regimes and turbulence parameters in complex terrain; model validation and testing against observational data; use of an anisotropic turbulence model; and, better representation of the interaction between the dust plume and the ground.

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Chapter 7

Mine Health and Safety: Industry's March Towards Continuous Improvement – The United States Experience

R. Larry Grayson

7.1
Introduction

For the period 1980–1989, mining still ranked as the number one United States industry with respect to the average annual rate of traumatic occupational fatalities (Anonymous 1993). Its rate (per 100 000 workers) was 31.91 versus 25.61 for the construction industry, 23.30 for the transportation/communications/public utilities industries, and 18.33 for the agriculture/forestry/fishing industries. The next highest figure was three times lower at 6.73. The largest causes of death were machines (7.39 fatalities per 100 000 workers), motor vehicles (5.27), struck by falling objects (4.33), explosions (2.46), electrocutions (2.27), and falls of persons (1.89). Mining's death rates by machines, struck by falling objects, and explosions were the highest among all industries, while mining ranked second in death rates by motor vehicles (11.44 in transportation/communications/public utilities), electrocutions (3.99 in construction), and falls of persons (6.56 in construction).

A unique trend toward high death rates for younger workers exists in mining for the period of study. No other industry had such a trend. The rates were 72.9 for the 16–19 age group, 50.3 for the 20–24 group, 32.1 for the 25–34 group, 26.0 for 35–44, 26.8 for 45–54, 26.5 for 55–64, and (the usual exception) 60.8 for the 65+ group. Nonetheless, recent evidence indicates that mining's overall performance may now be surpassing the performances of construction and agriculture/forestry/fishing (Anonymous 1992a). The total accident incident rates in 1990 were 8.3, 14.2, and 11.6, respectively, for mining, construction, and agriculture/forestry/fishing.

Impacts of mining on the work environment of miners can be severe unless proper, well-designed and timely measures are implemented to mitigate them. Often the timing of counteractions require intervention following mining activity, i.e., the implementation of remedial measures. Both underground and surface mining generate potentially hazardous work conditions and situations which may impact the health and safety of miners. This chapter will specifically outline the past history of fatalities, injuries, and illnesses (mostly based on the United States experience but with an international comparison of fatality rates), the potentially hazardous conditions and situations that develop, and the state-of-the-art measures used to control the impacts on workers. The long-term drive to mitigate and remediate the impacts of hazards on miners has now led to systematic efforts by the mining companies to improve the welfare of the workers continuously, generally invoking a total loss control perspective. Conventional wisdom and evidence indicates that the most efficient mines are indeed the safest ones (Ary 1989; Terezopoulos 1995).

7.2
Health and Safety Impacts of Mining Activities

Mining operations can broadly be classified according to whether they use underground or surface mining methods. Table 7.1 gives the number of operations and miners employed in the United States' mining sectors in 1995 (Reich and McAteer 1997a–e). The number of mines in the aggregates industry dominates (3398 in stone; 6021 in sand and gravel), followed by coal (2356), and nonmetal (627). Employment is dominated by the coal industry (85 652), followed by aggregates (30 968 in stone; 32 970 in sand and gravel). These figures do not include processing facilities, independent shops, and contractors.

In general, fewer miners are employed at surface mines for comparative production, and the incidence of accidents/injuries is lower. Longwall mining, a method of underground mining used primarily for producing coal, has an overall injury incident rate comparable with surface mining, but the underground longwall mine still has a much higher incident rate because of the hazards miners encounter when performing support operations, including room-and-pillar mining to set up longwall panels.

Underground hardrock mining has a much lower overall injury incident rate than underground coal mining, which will be evidenced by statistics presented in a later section. There are physical conditions and work situations which account for the differences, and these differences will be delineated next. In doing so, the hazardous conditions and situations which impact the work environment will be presented separately for underground and surface mining. A typical table of contents from a series of international conferences on mine safety (Green 1985) includes the following topics: rock and gas outbursts, strata control, transport, accidents and rescue, gas drainage and ventilation, monitoring, fires, explosions, ignition protection, explosives and blasting, and miscellaneous topics. The hazards or situations presented next generally encompass the hazards that occur within these standard topical areas.

7.2.1
Underground Mining

Hardrock mining generally employs the unit operations of drilling, blasting, excavating and loading, and hauling, with a few support activities also being required, as needed. The support activities may involve roof/ground support, ventilation, delivering of supplies and materials, maintenance, drainage of water, and advancement of development workings and support systems. In coal mining, the unit operations of drilling, blasting, excavating and loading are combined into a single operation by using continuous mining machines or longwall mining systems (shearer- or plough-based, although ploughs are rapidly disappearing). However, each type of mining generates similar potentially hazardous conditions or work situations, or both. The factors contributing to hazardous conditions in underground mining are outlined next separately.

7.2.1.1
Dusts, Gases, and Fumes

As mining progresses, dusts and possibly gases are generated from the orebody and the host rock. Gases and fumes are generated by the equipment used. Distinguishing fea-

Table 7.1. Number of mines and employees at US mines in 1995

Mining sector	Mines U/G	Surface	Miners U/G	Surface
Coal	1081	1275	51777	33875
Metal	118	176	7560	17509
Nonmetal	57	570	3904	5832
Stone	102	3296	1940	29028
Sand and Gravel[a]	–	6021	–	32970

U/G; underground

[a] Includes 824 dredge operations employing 4669 miners.

tures between underground hardrock and coal mining are the presence of methane gas in coal mines and the methods of extraction (and equipment) which generate differential amounts of dust and requirements for ground control.

Dusts generated by mining activity include crystalline silica (mostly alpha quartz). Crystalline silica was identified as a probable carcinogen to humans by the International Agency for Cancer Research, which has recently voted to change the classification to "carcinogenic to humans" (Carroll 1997). As such, special concern has been generated among mining health researchers, enforcement agencies, miners representatives, and mining companies. Silicosis can be a deadly illness, sometimes causing death in very short periods of time. A classical case of uncontrolled exposures to crystalline silica is evidenced by the tragedy at Hawks Nest (or Gauley Bridge) in West Virginia, United States, in the early 1930s, where 764 workers died of acute silicosis during excavation of a tunnel (Burtan 1984). The matter of control of free silica dust will be discussed in greater detail later under mitigation and remediation (Sect. 7.4).

The uncontrolled exposure of miners to coal mine dust also causes significant disability and mortality, via coal workers' pneumoconiosis (and progressive massive fibrosis in more advanced stages). In the United States, approximately 1.2 billion $ is currently paid annually to "black lung" recipients. ("black lung" is a US political term which encompasses all coal-related disabling diseases of the lungs.) Extensive worldwide research has been intensively undertaken since World War II by virtually every major mining country to control dust exposures and disease incident rates. State-of-the-art control methods will be presented later.

Most other dusts are considered nuisance dusts, i.e., long and substantial exposure could lead to some health impact, but generally the impact does not result in disability at significant rates under normal working conditions. The amount of worker exposure to nuisance dusts is also generally well controlled.

Of course, methane in coal mines has long been a major cause of mine disasters, which are destructive events causing four or more fatalities (according to the current US definition of a disaster). The extent of the impact of methane explosions will be highlighted during the presentation of historical fatality statistics (Sect. 7.3). The methods of controlling methane in the mine atmosphere will be discussed in some detail later, including methane drainage.

Other dangerous gases that may be encountered in mining operations include carbon monoxide (poisonous and explosive), carbon dioxide, hydrogen (explosive), hydrogen sulfide (poisonous), oxides of nitrogen (poisonous), sulfur dioxide (poisonous), and other natural gases (explosive). The control of methane and diesel emissions will be the primary focus in the later section concerning mitigation and remediation, but dilution by ventilation is the primary method of control for these gases.

With the proliferation of diesel equipment in mining, emissions are a concern. Some of the previously mentioned gases (e.g., carbon monoxide, oxides of nitrogen, and sulfur dioxide) are generated by diesel equipment, but diesel particulate matter (DPM) is also generated, which is classified as a carcinogen primarily because of the polycyclic aromatic hydrocarbons it contains (Anon. 1988; Lowndes and Moloney 1996). The control of diesel emissions will be discussed in the later section on mitigation and remediation.

7.2.1.2
Roof, Floor, and Ribs

As an orebody is extracted, the natural ground condition surrounding it is disturbed. The amount of pressure developed on the walls of excavations depends on depth, the amount of water and gas trapped in pore spaces of the remaining rock, and tectonics and other geologic processes, which can develop high horizontal stresses. Methods for stabilizing the roof, floor, and ribs around excavations will be outlined in Section 7.4.

7.2.1.3
Noise

Noise is generated in mining operations primarily by equipment. Drills, excavators and loaders, haulers, crushers, ventilation fans, and other equipment can generate noise pressure levels well in excess of 90 decibels (A scale), which is the level at which hazardous exposure to workers begins by the US definition. In a National Institute for Occupational Safety and Health Study (Anonymous 1981), findings showed that 30% of miners age 50, who worked approximately 30 years in the coal mining industry lost 40 dB in their hearing, while 50% of them lost 25 dB. Currently in the United States a new "action" level of 85 dB is proposed, which will require training of affected miners placed on noise protection and annual audiometric hearing tests (MSHA 1996; Burns 1997). There are many ways to control noise exposures, and these will be presented in the section on mitigation and remediation.

7.2.1.4
Heat

The major sources of heat in mines are the conversion of potential energy to thermal energy as air descends shafts, machinery, and geothermal heat (Pickering and Tuck 1997). There are numerous other sources of heat, including explosives, oxidation, falling rock, mine lighting, etc. The efficiency of miners in performing work can diminish substantially as ambient effective temperatures exceed 28 °C. Miners can also cramp, faint, and suffer exhaustion when working at elevated temperatures for prolonged periods. Ultimately when a miner works an extended period without body-temperature reduction, a heat stoke may result (body core temperature rises beyond 41 °C). Methods for controlling the ambient work environment temperature and worker exposure to heat will be addressed in the section on mitigation and remediation.

7.2.1.5
Workplace Activities

Workers are often exposed to hazardous combinations of conditions and situations in the workplace, e.g., working around equipment, especially near deteriorated physical conditions, working on scaffolding, performing tasks in confined space, etc. Many routine tasks require repetitive motion, which places repeated stress on important parts of the miners' body leading to chronic motion-related diseases such as carpal tunnel syndrome. Handling materials of various configurations also places stress on key body components, especially the back. Muscle sprains and strains are a particular problem in getting required supplies and materials to and around the workplace. Methods for mitigating the impacts on miners will be discussed later. Also, statistics concerning general occupational illness data will be presented later to highlight the problems here.

7.2.1.6
Work Shift Design

Many progressive mining companies have designed work shifts to ensure a quality life for their employees. It is common to see scheduling for four 10-h shifts in the US, followed by 3 days off. There are many other variations of shift scheduling which attempt to arrange favorable combinations of on-duty and off-duty times for workers and meet the companies' demands for production.

Many companies do not manage favorable shift scheduling for their miners. Instead production demand and cost govern the scheduling of workers, and the weekly working hours per worker are often quite high. In such situations, it is common to observe miners working 10 or more hours per day, 6 days per week. Many of the miners, usually voluntarily, work the seventh day as well. In unfavorable scheduling situations, fatigue and inattention can become problems, along with a general resentment of forced work hours.

7.2.1.7
Emergency Preparedness and Response

Mine disasters (involving five or more deaths by past definition) persisted, especially in underground coal mines, until approximately 1975 in the United States. Table 7.2 summarizes the number of US mining disasters over time, which have steadily declined to a very low number since 1976 (less than one per year). Strict enforcement of federal mining regulations began after the Coal Mine Health and Safety Act of 1969, and after the Mine Safety and Health Amendments Act of 1977 for other mining sectors. Although internal vigilance and enforcement are the key components in preventing mine disasters, emergency preparedness practised in its broadest perspective is also very effective in preventing disasters and in ensuring a swift and effective response when a disaster does occur.

Table 7.2. Number of documented mine disasters in the United States (Anonymous 1997)

Period	Coal mines	M/NM mines	Total
Through 1875	19	4	23
1876–1900	101	17	118
1901–1925	305	51	356
1926–1950	147	23	170
1951–1975	35	9	44
1976–present	13	1	14

M/NM, metal/nonmetal.

7.2.2
Surface Mining

Hardrock and coal surface mining methods and unit operations are similar. Generally drilling, blasting, excavating and loading, and hauling, with a few additional support activities are required. The support activities generally involve maintenance of slope stability, maintenance of haul roads, delivery of supplies and materials (e.g., drill bits, blasting caps, and powder), equipment maintenance, drainage of water, and advancement of operations and support systems. Although many of the potentially hazardous conditions are similar to the ones encountered in underground mining, the specific generation of the hazards and the situations in which the miners encounter them are dissimilar. Thus, the factors contributing to potentially hazardous conditions in surface mining are again outlined separately.

7.2.2.1
Dusts, Gases, and Fumes

Silicosis is the major concern in surface mining, whether hardrock or coal. In the United States in 1992, a silicosis alert was initiated by the National Institute for Occupational Safety and Health (NIOSH) for drill operators (Anonymous 1992b). In 1996 the Mine Safety and Health Administration (MSHA) joined NIOSH for a general alert on silicosis because of an unusually high incidence of silicosis in Pennsylvania among drill operators at surface coal mines (Anonymous 1996b). The effort involved an intensive information campaign, training program, and call for voluntary chest X-rays.

Coal workers' pneumoconiosis is also a concern at surface coal mines and preparation plants, but not to the extent as in underground mines for high-risk occupations. Dust control measures for surface mines (primarily for dust generated from drilling, hauling, and crushing) are different from the ones implemented in underground mines, and these will be covered in the section on mitigation and remediation (Sect. 7.4).

7.2.2.2
Slope Stability

The primary ground control concern in surface mines is slope stability, and the handling of unstable slopes when they occur. Generally it is not a problem because of good mine design and operational practices, but when wet conditions occur in a mine, once-stable slopes can become unstable. The methods for designing and maintaining stable slopes will be covered briefly in the section on mitigation and remediation.

7.2.2.3
Noise

The sources of exposure of miners to noise at surface mines are similar to those at underground mines, i.e., equipment. Of course, the equipment used is quite different. The proposed new regulations will apply to both surface and underground mines. The methods of noise control will be discussed later.

7.2.2.4
Heat

The source of heat stress at surface mines is primarily ambient air temperature and humidity, especially in arid and semi-tropical climates. Also, areas with little air movement can develop in pits, which intensifies the effect of the heat. Methods for control of the exposure of miners to heat will be discussed in the section on mitigation and remediation.

7.2.2.5
Workplace Activities

Surface miners may also be exposed to hazardous combinations of conditions and situations in the workplace, e.g., working around equipment, especially near highwalls, performing maintenance tasks in confined space, etc. Repetitive motion tasks and materials handling also place stress on important parts of the miners' bodies leading to chronic motion-related diseases and sprains and strains. Methods for mitigating the impact on miners will be discussed later.

Comments on work shift scheduling for underground workers apply to surface miners as well. No further comments are needed here. Emergency preparedness and effective response to emergencies, although rare for surface mines, are also considerations for surface mines.

7.3
Historical Fatality, Injury, and Illness Experiences

As shown previously in Table 7.2, mine disasters occurred frequently in the United States during the period 1901–1925. At this time, the industrial revolution was in full bloom. Table 7.3 shows the number of fatalities that occurred in coal mining and other sectors by 5-year periods beginning in 1911, just following the worst period for US fatalities (1905–1910) which galvanized Congressional action in 1910, via the Organic Act, to create the US Bureau of Mines. The Organic Act emanated in reaction to "Bloody December" in 1907 when four major coal mine disasters claimed 692 miners' lives, but even more, because of a privately funded study which showed that a single Pennsylvania county logged 528 fatal mine-related accidents in a 1-year period. Five thousand other

Table 7.3. Mine fatality experience in the United States (expressed as 5-year average since 1911)

Period	Coal fatals per year	Other fatals per year	Total fatals per year	Workers per year (thousands)	Fatals per 100 miners
1911–1915	2507	813	3320	1216	0.273
1916–1920	2419	764	3183	1341	0.237
1921–1925	2215	482	2697	1058	0.26
1926–1930	2235	462	2697	996	0.27
1931–1935	1241	181	1422	699	0.20
1936–1940	1265	269	1534	762	0.20
1941–1945	1311	270	1581	676	0.23
1946–1950	870	185	1055	661	0.16
1951–1955	522	168	690	532	0.13
1956–1960	380	167	547	480	0.11
1961–1965	274	175	449	424	0.11
1966–1970	246	180	426	390	0.11
1971–1975	151	171	322	396	0.08
1976–1980	131	122	253	521	0.05
1981–1985	107	70	177	439	0.04
1986–1990	68	54	122	357	0.03
1991–1995	51	48	98	339	0.03

mine workers in the county were maimed or totally disabled during the same year (Traweek 1994).

Also shown in Table 7.3, by 5-year period, are the average total number of workers employed in mining and the average number of fatalities per 100 miners (an incident rate). Until approximately 1971, the fatality experience was dominated by coal. Since 1991, the fatality experience of coal and other sectors is comparable. The incident rate has generally declined steadily over the entire period since 1911, but most rapidly following World War II (1946–1950) and following the passage of the Mine Safety and Health Amendments Act of 1977 (1976–1980). The great progress was made by a combination of factors, which will be discussed in greater detail later.

Table 7.4 gives details for 5-year periods beginning with 1931–1935 on the average annual number of mines, average hours worked per year, average number of fatalities per year, and the average fatal incident rate (IR) for the coal industry. Comparing Tables 7.3 and 7.4 shows that the fatal IR was generally higher for coal than for other sectors, including for the period 1991–1995. Generally this occurs because of the underground coal mine experience. However, there has been a convergence of both general sectors toward the 0.03 level, with a nearly equal annual number of fatalities recently.

Table 7.5 gives even greater detail for comparisons between surface and underground mines for the coal and metal/nonmetal sectors since 1978. It is clear from Table 7.5 that

Table 7.4. Coal industry mines, employment, and fatality statistics in the United States (expressed as 5-year average since 1931)

Period	Number of mines per year	Worker hours per year (millions)	Fatals per year	Fatal IR
1931–1935	5875	810	1241	0.31
1936–1940	6821	827	1265	0.31
1941–1945	6969	1003	1311	0.26
1946–1950	9254	816	870	0.21
1951–1955	9513	522	522	0.20
1956–1960	10727	432	380	0.18
1961–1965	9762	252	274	0.22
1966–1970	7245	243	246	0.20
1971–1975	4625	293	151	0.10
1976–1980	6648	418	131	0.06
1981–1985	6067	385	107	0.06
1986–1990	4688	314	68	0.04
1991–1995	3474	260	51	0.04

IR, incident rate.

underground coal and metal/nonmetal mines have comparable fatality rates since 1978, as do surface coal and metal/nonmetal mines. As a matter of fact, in order to reduce fatality rates beyond a current plateau (0.03–0.04 level), new focuses are being employed today to target poorer-performing mine sizes and prioritize interventions for specific jobs with poor fatality and serious-injury rates (Tisdale 1993; Peters 1994; Grayson 1997).

As mentioned by Traweek (1994) in his addendum, there were many processes occurring which had an impact on improving mine safety: "...decreases in mine employment; new technologies; improved equipment; enlightened management; employee representative (sic) and employee participation; changing social mores...". He states further: "Within this context, legislation and regulatory activities play important, but only contributory roles. Legislation and regulation have never been and can never be the prime movers in mine safety. That role must fall to the operator and to the employee".

This representation is an accurate assessment of the many factors which had led to the great improvement in mine safety and health. Today, the factors have converged to demand continuous improvement emphasis by companies toward the goals of zero fatalities and zero injuries. Many significant awards are given each year in the United States to companies which accomplish these goals.

Figure 7.1 presents a comparison of the US underground coal mine fatality rate to other major coal-producing countries in the world between 1971 and 1987, as determined by Wagner (1989). It is apparent that Germany and the United Kingdom consistently

Table 7.5. Comparison of fatality incident rates for US underground and surface mines by coal and metal/nonmetal sectors

Year	Fatal IR U/G coal	Fatal IR U/G M/NM	Fatal IR Surface coal	Fatal IR Surface M/NM
1978	0.07	0.12	0.03	0.06
1979	0.09	0.09	0.02	0.06
1980	0.08	0.07	0.03	0.05
1981	0.11	0.09	0.03	0.04
1982	0.08	0.10	0.03	0.03
1983	0.06	0.08	0.02	0.05
1984	0.11	0.11	0.03	0.05
1985	0.06	0.12	0.02	0.03
1986	0.07	0.07	0.04	0.04
1987	0.05	0.07	0.03	0.06
1988	0.04	0.06	0.03	0.03
1989	0.06	0.06	0.03	0.03
1990	0.06	0.07	0.02	0.04
1991	0.06	0.06	0.02	0.04
1992	0.06	0.05	0.03	0.04
1993	0.05	0.13	0.04	0.02
1994	0.04	0.06	0.04	0.02
1995	0.05	0.05	0.04	0.04

had better fatality incident rates, which is primarily attributable to the extensive amount of longwall mining done in each country. Although Australia and the United States had similar mixes of continuous mining and longwall mining, South Africa had a predominance of continuous mining. The South African underground coal mine fatality incident rate was understandably consistently higher than the other countries, while Australia and the United States had comparable fatality experiences.

The United States nonfatal days lost (NFDL) injury experiences in coal and metal/nonmetal mines since 1978 are compared in Table 7.6 for underground and surface mining methods. There has been no marked improvement in NFDL IR performances for either sector in either underground or surface mines since 1982. An apparent significant change occurred in underground coal mines between 1980 and 1986, but significant under-reporting of accidents occurred during this period, especially in small mines (GAO 1987). Also noted by the US General Accounting Office (GAO), an effect of changes in the definition of reportable accidents can be seen in the increase of NFDL IRs from 1986 to 1987 in each sector and mine type.

Table 7.7 gives a more detailed breakdown of the rates of fatalities, injuries, and accidents, along with the severity measure (the number of lost and restricted work days

Table 7.6. Comparison of nonfatal days lost (NFDL) incident rates for US underground and surface mines by coal and metal/nonmetal sectors. *IR*, incident rate

Year	NFDL IR U/G coal	NFDL IR U/G M/NM	NFDL IR Surface coal	NFDL IR Surface M/NM
1978	10.41	9.74	3.52	4.12
1979	11.88	10.67	3.52	4.26
1980	12.13	9.39	3.48	3.62
1981	10.88	8.58	3.43	3.17
1982	9.94	5.80	3.27	2.51
1983	8.25	5.96	2.71	2.80
1984	8.28	5.26	2.66	2.80
1985	7.40	4.86	2.72	2.74
1986	8.05	5.35	3.21	3.19
1987	12.19	5.70	3.39	3.88
1988	12.93	6.73	3.51	4.33
1989	12.39	6.75	3.54	4.65
1990	12.15	6.31	3.57	4.60
1991	12.28	5.87	3.34	4.32
1992	11.61	4.80	3.17	4.28
1993	10.18	5.37	2.61	3.71
1994	11.08	5.69	3.12	3.72
1995	9.90	4.64	2.52	3.12

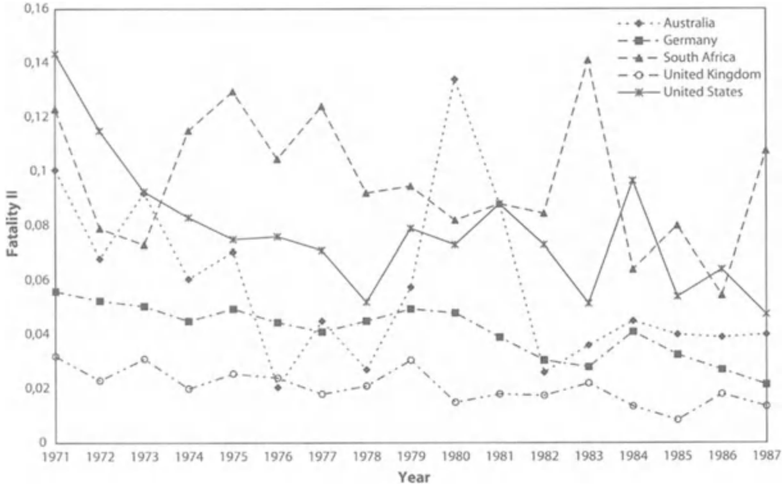

Fig. 7.1. International comparison of coal mine fatality rates (normalized for 100 employees)

Table 7.7. Fatality, nonfatal days lost (NFDL) injury, and total accident incident rates and severity measures for underground and surface mines by sector (data for the United States in 1995)

Sector	Fatalities	Fatal IR	NFDL	NFDL IR	Total IR	Overall SM
Coal underground	25	0.05	5426	10.57	13.50	772
Coal surface	10	0.03	942	2.75	4.06	288
Metal underground	3	0.04	422	5.79	9.75	498
Metal surface	5	0.03	469	2.52	4.23	290
Nonmetal underground	3	0.08	146	3.83	6.34	712
Nonmetal surface	3	0.05	137	2.41	4.19	433
Stone underground	0	0	75	3.71	5.53	122
Stone surface	11	0.04	1201	4.15	6.86	345
Sand/gravel surface	4	0.02	881	3.56	5.62	269
Sand/gravel dredge	2	0.04	182	3.77	6.10	335

SM, severity measures.

multiplied by 200 000 and divided by the number of employee-hours worked) by underground and surface mines within each industry sector for 1995 (Reich and McAteer 1997a–e). It is clearly shown in Table 7.7 why underground coal still gets the greatest attention from enforcement agencies (25 fatalities and 5426 lost-time accidents with an NFDL IR over 10), even though the fatal IR is comparable with the other sectors. It is also apparent why surface stone operations have received recent enforcement agency attention (11 fatalities were the second highest total for a sector in 1995). Underground coal leads in the other measures of injury rate and severity measure as well, while metallic and nonmetallic underground mines are relatively high in severity measure (and fatal IR for nonmetallic underground mines). By focusing on data such as this and job-related rates, MSHA has been successively targeting sectors, mines, and jobs for national and regional enforcement attention and informational/public press efforts.

Table 7.8 breaks down the US reported occupational illnesses for the mining industry by sector for 1995 (Reich and McAteer 1997a–e). These data are collected and summarized annually. The data shown are for underground and surface mines for operators only (i.e., not contractors). It is clear that the underground coal sector again presents the greatest need for improvement based on magnitude, followed by the metallic sector. Based on the number of employees in each sector, an incident rate for occupational illnesses can be calculated, giving a better glimpse at areas of need. The occupational incident rates per 100 workers are 0.53, 0.55, 0.47, 0.30, and 0.10 for coal, metallic, nonmetallic, stone, and sand and gravel, respectively. Based on these numbers, the metallic sector has the highest incident rate (primarily because of repeated trauma disorders), followed by coal (primarily because of both dust diseases of the lungs and repeated trauma disorders). The underground coal industry has the far greater problem in dust diseases of the lungs with two-thirds of the coal cases.

Table 7.8. Occupational illnesses in mining by sector (data from the United States in 1995)

Type\sector	Coal	Metal	Nonmetal	Stone	Sand/gravel
Skin diseases	3	4	1	12	3
Dust diseases – lungs	207	8	5	9	2
Respiratory – toxic agents	8	1	2	1	0
Poisoning	3	1	0	3	0
Disorders – nontoxic physical agents	2	12	3	15	5
Disorders – repeated trauma	214	109	32	49	22
All others	21	2	3	5	1
Total	458	137	46	94	33

Table 7.9. Cases of coal workers' pneumoconiosis and silicosis mortality for three periods in the United States

Period	Silicosis No. of cases	Cases per year	Pneumoconiosis No. of cases	Cases per year
1968–1978	9431	857.4	26 123	2374.8
1979–1990	4313	359.4	29 344	2445.3
1991–1992	569	284.5	3704	1852.0

Silicosis is a lung disease which spans a number of industries, not just mining. Table 7.9 presents National Institute for Occupational Safety and Health (NIOSH) data on the number of US deaths attributable to silicosis for three time periods (Anonymous 1996a). It is clear that the rate of incidence has been decreasing; however, a substantial rate persists. Coal mining claims the largest proportion at 9.5%, followed by metallic mining at 8.6%, and nonmetallic mining and quarrying at 4.9%. The single largest job category found among death certificates was "mining machine operator" at 16%, followed by "laborers, except construction" at 11.7% (the next highest proportion was 4.5%). A breakdown of cases by state showed that Pennsylvania led the nation with 2483 in the first period, 755 in the second period, and 106 in the last period. Ohio was a distant second in each period with 1036 cases, 456, and 59.

As mentioned earlier, coal workers' pneumoconiosis (CWP) costs the United States $1.2 billion per year in disability payments (currently, it has been as high as $2 billion in the past). Table 7.9 gives a breakdown of CWP mortality by time period, as was done for silicosis. Pennsylvania, an old coal mining state with both anthracite and bituminous coal deposits, had the vast majority of cases with 20 965 in the first period, 16 994 in the second, and 1797 in the third. This represents 80.3%, 57.9%, and 48.5% of the total for each period. West Virginia is a distant second in case numbers with 1305 in period one, 3339 in period two, and 591 in the third period. The coal mining industry accounted for 76.1% of the total cases (the next highest percent was 3.8), and mining machine operators was found on 70.4% of the death certificates (the next highest percent was 3.3). It is clear that much more work needs to be done in eliminating CWP; the rate has not declined nearly as quickly as expected since the implementation of the US Federal Coal Mine Health and Safety Act of 1969.

7.4
Methods of Mitigation and Remediation

There are three primary methods for achieving compliance with regulations, as well as mitigating or remediating the impacts of conditions and situations on miners in the workplace: education, engineering, and enforcement. Commitment by management is required for having much impact, regardless of the type of intervention. Once management manifests its commitment, then continuous improvement of all aspects of mine performance is the goal, reinforced by frequently changing initiatives aimed at specific objectives concerning successive targeted areas. Specific methods for mitigating and/or remediating the various conditions/situations addressed earlier will be presented next.

7.4.1
Dusts, Gases, and Fumes

The primary method of controlling dusts, gases, and fumes in underground mines as well as mitigating and remediating their impact in the workplace is dilution through effective ventilation. Mining regulations generally specify permissible exposure limits for specific dusts, gases, and fumes and, often, a minimum quantity of fresh air that is required at various locations in a mine. Usually the ventilation scheme for a mine is carefully planned, often requiring enforcement agency approval, engineered, and implemented. Thorough monitoring of mine ventilation is also required by law, usually on a shiftly or daily basis, to ensure that the proper quantity and quality of air is moving in the planned directions throughout the mine.

The second line of defense for maintaining a healthy and safe work environment is enclosures and water sprays for dust-generating equipment, incorporation of surfactants in spray water, ventilation control devices, and other dust-allaying methods or work procedures (e.g., sprinkling roadways). Most mining handbooks include various effective methods for controlling and mitigating the effect of dusts. Haney *et al.* (1988) outline common methods for controlling dust in US underground coal mines, while Mohamed *et al.* (1996) give a general overview of effective methods. These methods are for the most part effective for either coal dust or crystalline silica, as well as other entrained dusts. The methods are also generally applicable to surface mines, although they must be adapted to specific situations and equipment in a unique way. For example, adaptation of these general methods to control crystalline silica dust in surface mines was included in the NIOSH Alert of August 1992 (Anonymous 1992b).

Control of diesel equipment emissions generally requires monitoring of tailpipe gases, the use of effective (preferably regenerative) on-board scrubbers, and specified quantities of air. An excellent review of these techniques is given by Lowndes and Moloney (1996). Of course, a good maintenance program is required to maintain emissions levels of equipment over time, especially with changing engine duty cycles.

The last line of defense is the use of respirators or other breathing apparatus (e.g., self-contained breathing apparatus, air-sweeping helmets, etc.) and administrative controls. In the United States, all other engineering measures must be taken first. When the primary and secondary measures cannot ensure a healthy work environment, then an operation is found in non-compliance with the regulations and the miners must wear protection, or administrative controls may be exercised. Even when a working place is in compliance, miners must be provided respiratory protection on request. Many companies request that their workers wear protection as good working practice.

7.4.2
Roof, Floor, Ribs, and Slopes

To ensure the safety of workers in mines, effective design of underground mine entries, pillars, roof, floor and ribs must be incorporated using rock mechanics/ground control analysis methods. Similarly, stable slopes must be designed for surface mines in consideration of geology, time, and anticipated conditions. Mining handbooks and rock mechanics textbooks contain extensive coverage of design principles, techniques, and formulas for a wide variety of conditions. The international literature is replete with technical papers on these topics. Recurring conferences and symposia keep experts on top of the latest developments.

The modern trend in analysis is to examine sites carefully for existing problem phenomena, instrument the site to attain rock properties and magnitudes of deformations, and then perform a detailed engineering analysis using finite element, boundary element, finite difference, or hybrid modeling. Brady and Brown (1993) provide a comprehensive textbook on general rock mechanics while Peng's (1978) textbook deals more specifically with coal mining.

Often times following analysis, remedial measures of various types are needed to control actively-changing conditions. For example, in underground coal mines, installation of cribs (of various stiffness), trusses, rib or floor bolts, cable bolts, and different types of roof bolts may be used to supplement the initial method of support. In hardrock mines, supports may be added when none were used initially.

7.4.3
Heat

Well-designed ventilation and air conditioning are the primary methods for controlling the impact of heat on workers. McPherson (1993) provides an excellent textbook on engineering aspects. Secondly, cool drinking water should be made available for workers in mines where elevated temperatures occur. Light clothing should be worn in such environments, with "ice" jackets being used in onerous climates. As mentioned by Pickering and Tuck (1997), "the workforce which is required to work in hot environments should be trained to recognize the symptoms of heat strain and to adopt sensible work habits…". In practice, workers should also be acclimatized over time, be in good physical condition, and intersperse periods of rest and refreshment with work periods.

7.4.4
Workplace Activities

Workplace interventions in the area of ergonomics have helped drive continuous improvement in safety performances throughout the mining industry. Companies have been implementing systematic programs to eliminate sprains, strains, and other musculoskeletal injuries (Grayson *et al.* 1998). Back injuries have been given special attention by the US government (Gallagher *et al.* 1990) as well as companies. Special attention has also been directed at specific problems, such as worker-machine interfaces during roof bolting (Anonymous 1994a,b). Other problem areas such as handling of supplies and materials, operating machinery in confined space, vibration, lack of visibility, using the proper tools for specific tasks, and using proper protective devices for safety in performing work are addressed systematically as well. The National Institute for Occupational Safety and Health has published a useful general industry primer for systematically addressing musculoskeletal disorders in the workplace (Cohen *et al.* 1997).

7.4.5
Work Shift Design

As mentioned earlier, many companies design shift schedules to avoid long-term burden to their workers. One example is evidenced by Mingo Logan Coal Company in West Virginia which studied work scheduling in a number of other industries before deciding on a schedule that would both motivate their workforce and achieve company production demands (Ladd 1990).

7.4.6
Emergency Preparedness and Response

Although the frequency of mining disasters has diminished substantially, as discussed earlier, the conditions and potential for disasters exist nonetheless. Mine fires persist in US mines even though they are generally controlled or extinguished promptly. Thus, companies maintain a readiness to respond to emergencies through mine rescue teams, required by regulation, fire brigades, emergency medical technicians, first aid training for all employees, and doctors on-call. Recurring training for first aid (annual) and mine rescue teams (monthly or bimonthly) is required by regulation. Emergency medical technicians must be certified by the state in which a mine is located, and recertification is required for currency (generally every 3 years).

In addition, many underground mines have installed a mine-wide monitoring system which can detect incipient fires (generally with CO sensors), including spontaneous combustion in remote or isolated gob (goaf) areas. Further, a reliable fire-fighting system is installed, maintained, and checked periodically for proper functioning and capability.

7.5
Conclusions

The mining industry has made great strides in ensuring that its workers can work an entire career without incurring death, disability, or serious injury. A comparison of statistics over the past 80 years evidences the progress. This is an international success. Workplaces and work activities have been engineered to attain these successes, and today continuous improvement is sought systematically to enhance the quality of the miners' lives. Education, enforcement and engineering are combined to address changing issues and problems, but the commitment of management best guarantees good results. At long last, the health and safety performances of mines has begun to surpass the performances of other important industries, such as construction, agriculture, and forestry. It is our hope, as a mining community, that deaths, disabilities, and serious injuries will be systematically eradicated over the coming decades by even better methods for mitigation and remediation of mining conditions and situations that have caused the problems in the past.

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Section B

Treatment Methods for Mine Effluents and Rehabilitation of Mine Tailings and Overburden Materials

Chapter 8

Treatment of Coal Mine Drainage with Constructed Wetlands

William W. Hellier

8.1
Why Constructed Wetlands Are Used to Treat Coal Mine Drainage

In most parts of the world where coal is mined, water pollution from the resulting drainage is a matter of serious consideration. For example, coal mine drainage is the greatest cause of pollution to the streams of Pennsylvania (USA). Coal mine drainage currently pollutes over 3900 km of Pennsylvania's 87 000 km of streams (Arway 1996). Pennsylvania Department of Environmental Protection (DEP 1980) records estimate the amount of pollution from coal mine drainage at $2.27 \cdot 10^7$ tons per year of acidity as $CaCO_3$ equivalent.

Coal mine drainage often is acidic, because of the oxidation of pyrite (FeS_2). Pyrite is present in the coal itself, in the rock immediately underlying the coal, and in the rock, or overburden, overlying the coal. When the coal and rock are exposed to atmospheric O_2 and H_2O, the familiar sequence of pyrite oxidation reactions occurs:

$$2\ FeS_2(s) + 7\ O_2 + 2\ H_2O \text{ fi } Fe^{2+} + 4\ SO_4^{2-} + 4\ H^+ \qquad (1)$$

$$4\ Fe^{2+} + O_2 + 4\ H^+ \text{ fi } 4\ Fe^{3+} + 2\ H_2O \qquad (2)$$

$$Fe^{3+} + 3\ H_2O \text{ fi } Fe(OH)_3(s) + 3\ H^+ \qquad (3)$$

$$FeS_2(s) + 14\ Fe^{3+} + 8\ H_2O \text{ fi } 15\ Fe^{2+} + 2\ SO_4^{2-} + 16\ H^+. \qquad (4)$$

Concentrations of chemical species (mg l^{-1}) will henceforth be represented by brackets; e.g., [Fe]. The water resulting from contact with the oxidizing pyrite has low pH, and elevated [Fe]. Because the acidic conditions favor dissolution of Al, Mn, and other metals from the minerals present in the rock, coal mine drainage often has elevated [Al] and [Mn] as well as [Fe]. Other metals such as Zn, Cu, Co, and Ni are sometimes present in detectable concentrations. Coal overburden often contains $CaCO_3$ or $CaMg(CO_3)_2$, either of which dissolves and imparts alkalinity in the form of HCO_3^- to the coal mine drainage. This can result in alkaline or partially neutralized mine drainage which may still contain excessive concentrations of Fe, Al, and Mn. The pH of mine drainage follows a bimodal distribution, with peaks at pH 3.2 and 6.2 (Hellier 1994).

Operators of coal mines are required by US federal regulations and by Pennsylvania regulations promulgated in accordance with federal regulations to conform to discharge

Table 8.1. Best available technology (BAT) discharge limitations (mgl^{-1})

Pollutant parameter	Discharge limitations 30-Day average	 Daily maximum
Iron	3.5	7.0
Manganese	2.0	4.0
Suspended solids	35	70
pH	$6.0 \leq pH \leq 9.0$ at all times	
Alkalinity	Alkalinity > acidity at all times	

limitations based on best available technology (BAT) during mining (Table 8.1). More stringent limitations may be imposed on discharges to streams designated as needing special protection, and discharge limitations are applied to other pollutants, such as Al, when the regulatory authority finds that such limitations are necessary to protect the quality of the receiving stream.

The coal miner is required by law to post a bond which cannot be released until mining has been completed, the mined area has been reclaimed and revegetated, and all pollution has been abated. The criteria for release of the bond are given in the Rules and Regulations of the Pennsylvania Department of Environmental Protection (DEP 1997). In hundreds of cases, the coal miner has found that when the site has been reclaimed, post-mining groundwater seeps that do not conform to the effluent limitations shown in Table 8.1 remain. If a coal mine is abandoned and the bond forfeited, the coal miner loses his privilege to mine coal anywhere in the United States. In hundreds of cases, the only option to operators of coal mines who wish to continue in business has been conventional, or chemical, treatment of the post-mining groundwater discharges. Conventional treatment is an effective means of pollution control accepted by the US Environmental Protection Agency (EPA) and by DEP. The polluted drainage is mixed under oxidizing conditions with an alkaline chemical such as NaOH, $Ca(OH)_2$, or Na_2CO_3, thereby neutralizing the acidity and promoting the formation of solid oxides and hydroxides which are removed from the water by settling in a series of basins.

From an environmental point of view, the citizens are faced with pollution from coal operations where the miner has forfeited the bonds and abandoned the mine site. Either the coal miner's bonding company or the state of Pennsylvania is left with the problem of abating pollution from the abandoned site. Additionally, a large majority of the mine sites discharging polluted water were abandoned before modern laws and regulations were passed. No current miner has a legal or financial responsibility to abate pollution emanating from such sites.

From an economic point of view for mine operators currently in business, a continued need for conventional treatment can make continuing in business unprofitable. The cost of conventional treatment is often high. Money spent on conventional treatment is unavailable for other business purposes. Finally, bonding companies often are reluctant to supply bonds to operators of coal mines who are experiencing ongoing treatment liability. Because of the environmental and economic considerations, both the coal mining industry and the various government regulatory agencies have been seeking innovative means to abate water pollution from coal mine discharges without the need for conventional treatment. One growing innovative abatement technique involves the construction of artificial wetlands, in which natural processes are used to abate the polluted water.

8.2
Natural Treatment Processes Occurring in Constructed Wetlands

8.2.1
Neutralization of Acidity with Bicarbonate

Acidity, the capacity of a water to neutralize strong bases, is usually expressed as $CaCO_3$ equivalent (mg l^{-1}). Acidity is the sum of proton acidity, commonly expressed as H^+ and metal cations (Fe^{2+}, Fe^{3+}, Mn^{2+}, Al^{3+}) which have the potential to undergo hydrolysis and thereby liberate H^+. Acidity, if not neutralized, can depress pH below 6.0, thereby violating effluent limits, causing the water to be injurious to aquatic life, causing damage to structures such as bridges, and increasing the cost of treating drinking water. Therefore, a primary objective in designing a constructed wetland is the removal of acidity.

In terms of $CaCO_3$ equivalent, the following simple expression is derived from Mendeleev's periodic table:

$$\begin{aligned}\text{Acidity (mg l}^{-1}\text{ as CaCO}_3) = 49.7 \cdot 10^{(3-\text{pH})} + 1.8\,[Fe^{2+}] + 2.7\,[Fe^{3+}] \\ + 5.6\,[Al^{3+}] + 1.8\,[Mn^{2+}] + \pounds,\end{aligned} \qquad (5)$$

where £ represents the contributions of other cations, such as Zn^{2+}, Cu^{2+}, or Ni^{2+}, which may be present. In a wetland, the neutralization is performed almost entirely by HCO_3^-; e.g.:

$$H^+ + HCO_3^- \text{«}\ H_2O + CO_2 \qquad (6)$$

$$Fe^{3+} + 3\ HCO_3^- \text{fi}\ FeOOH + H_2O + 3\ CO_2 \qquad (7)$$

$$Al^{3+} + 3\ HCO_3^- \text{fi}\ Al(OH)_3 + 3\ CO_2 \qquad (8)$$

$$2\ Mn^{2+} + O_2 + 4\ HCO_3^- \text{fi}\ 2\ MnO_2 + 2\ H_2O + 4\ CO_2. \qquad (9)$$

In alkaline or partially neutralized discharges, HCO_3^- is initially present due to previous dissolution of $CaMg(CO_3)_2$ or $CaCO_3$; e.g.:

$$H_2O + CO_2 \text{«}\ H_2CO_3^- \qquad (10)$$

$$CaCO_3 + H_2CO_3 \text{«}\ Ca^{2+} + 2\ HCO_3^- \qquad (11)$$

$$CaCO_3 + H^+ \text{«}\ Ca^{2+} + HCO_3^-. \qquad (12)$$

If sufficient bicarbonate alkalinity is not present, it must be imparted to the discharge either by dissolution of limestone or by generation through biological processes. Limestone dissolution occurs in a pretreatment structure known as an anoxic limestone drain or in the wetland substrate. Biological bicarbonate generation also occurs when organic substrate materials are oxidized and sulfate is reduced; e.g.:

$$SO_4^{2-} + 2\ CH_3CHOHCOOH \rightarrow 2\ CH_3COOH + H_2S + 2\ HCO_3^- . \tag{13}$$

8.2.2
Oxidation of Metal Cations

Iron and manganese initially are present in mine drainage in the II oxidation states: ferrous ion, Fe^{2+}, from the oxidation of pyrite, and manganous ion, Mn^{2+} from dissolution of clays, Mn^{2+} in solid solution in pyrite, or carbonate materials such as $MnCO_3$. An important process of metal removal in constructed wetlands is the oxidation of Fe^{2+} and Mn^{2+}. Under conditions where dissolved oxygen is present in the water, the following reactions occur:

$$4\ Fe^{2+} + O_2 + 4\ H^+ \rightarrow 4\ Fe^{3+} + 2\ H_2O \tag{14}$$

$$2\ Mn^{2+} + O_2 + 4\ H^+ \rightarrow 2\ Mn^{4+} + 2\ H_2O. \tag{15}$$

For further discussion of the removal of Fe^{2+} and Mn^{2+}, the reader is referred to the US Bureau of Mines (1988). The rates of both these reactions increase as dissolved O_2 concentration increases and as hydroxyl ion concentration increases, or as pH increases. Under acidic and near neutral conditions, bacterial oxidation of Fe^{2+} is an important mechanism; however, under alkaline conditions, the dominant mechanism for the oxidation of Fe^{2+} is an abiotic process, which has been modeled as first order in $[Fe^{2+}]$ and $[O_2]$ and second order in $[OH^-]$, proceeding quite rapidly in solution at circumneutral pH. The oxidation of Mn^{2+} is autocatalytic, accelerated by the presence of MnO_2 at surfaces. The half-life of the oxidation reaction was found to be 37 min at pH 8.71, but 785 min at pH 8.08 (Morgan *et al.* 1985). The oxidation is catalyzed by γ-FeOOH. However, when Fe^{2+} is present, it tends to reduce Mn^{4+}.

8.2.3
Hydrolysis of Metal Cations

Metals are removed from solution by hydrolysis, which proceeds spontaneously and rapidly at circumneutral pH:

$$Fe^{3+} + 3\ H_2O \rightarrow FeOOH + H_2O + 3\ H^+ \tag{16}$$

$$Al^{3+} + 3\ H_2O \rightarrow Al(OH)_3 + 3\ H^+ \tag{17}$$

$$Mn^{4+} + 2\ H_2O \rightarrow MnO_2 + 4\ H^+. \tag{18}$$

The solubility of the resulting oxides decreases with increasing pH, and they are precipitated on the wetland substrate and at surfaces. The acidity liberated by the release of H^+ during hydrolysis must be neutralized by the bicarbonate neutralization reactions given above.

8.2.4
Reduction of Metal Cations

Within an organic substrate, the dissolved $[O_2]$ becomes depleted, and conditions change from oxidizing to reducing. In wetlands where the flow is primarily horizontal, this occurs within 1 cm of the surface, except where oxygenation from plant roots is occurring. In wetlands with a significant vertical flow component, the oxidizing zone may extend lower. Sulfate reducing bacteria function best under conditions of $pH > 5.5$, and produce H_2S by the reaction discussed earlier. Some of this gas escapes to the atmosphere taking with it acidity, while some reacts with reduced metals:

$$Fe^{2+} + H_2S \rightarrow FeS + 2\,H^+ \tag{19}$$

$$Mn^{2+} + H_2S \rightarrow MnS + 2\,H^+ . \tag{20}$$

The rates of these reactions are much slower than the rates of oxidizing reactions. Some production of carbonate may occur in wetland substrates:

$$Fe^{2+} + HCO_3^- \rightarrow FeCO_3 + H^+ \tag{21}$$

$$Mn^{2+} + HCO_3^- \rightarrow MnCO_3 + H^+ . \tag{22}$$

The reactions of Fe and Mn with H_2S and with HCO_3^- liberates H^+; it is therefore important to maintain excess alkalinity in the substrate by means of dissolution of HCO_3^- from $CaCO_3$ or by means of biological generation or some combination thereof. Hydrolysis reactions are independent of $[O_2]$ and occur within organic substrates; they are responsible for the retention of Al observed in many wetlands.

8.2.5
Uptake of Metals by Plants

It has been shown that emergent plants such as *Typha latifolia* accumulate Fe and Mn in their tissues, and that algae accumulate metals such as Mn. However, the accumulation accounts for less than 1% of the metal loading removed by wetlands constructed to treat coal mine drainage (Sencindiver and Bhumbla 1988). Unless the plants are harvested, they cannot themselves be the dominant means of metal or acidity removal. Their main functions are to cause the flow on the wetland surface to be more uniform, thereby making the settling of metal hydroxides and oxides more efficient, to provide reaction surfaces for metal removal, and to enhance the appearance of the constructed wetland system. As they die and decay, the plants also can act as a carbon source for sulfate reducing organisms.

8.2.6
Transport Processes in Constructed Wetlands

The processes transporting momentum, heat, and mass are at work to varying degrees of importance in constructed wetlands. The equation of mass conservation (continuity equation) and momentum conservation (Navier–Stokes equation) applied to a wetland modeled as a continuous medium describe the flow in the wetland. A commonly used simplifying model for subsurface flow is Darcy's Law, which for one-dimensional flow is:

$$v = -k\, dh/dx, \tag{23}$$

where v is the velocity, k the hydraulic conductivity, h the hydraulic head, and x the distance. Flow patterns are important when considering the degree of contact of polluted water with the wetland substrate and its influence on rates of pollutant removal. Heat transfer in a wetland, as affected by ambient air and influent water temperature, and to some extent by the heats of chemical reactions, has not received much theoretical consideration, although it is known that reaction rates are slowed at lower temperatures. There is considerable evidence that wetlands are not as efficient during winter months in temperate climates. Mass transfer occurs by bulk mass transfer, and diffusion, of course with chemical reaction.

Most models attempt to portray the wetland as some combination of a batch reactor, plug flow reactor, and well-mixed reactor. The chemical rate processes are usually modeled on the basis of a quasi zero order or first order chemical reaction. When temperature data is unavailable wetlands are usually modeled as isothermal systems.

8.3
Design Considerations

8.3.1
Alkalinity or Acidity of Mine Discharge

The Pennsylvania Department of Environmental Protection performed an analysis to determine the types of treatment which, in its best professional judgment, would represent the best available technology for the treatment of post-mining discharges (Hellier *et al.* 1994). The Department's Best Professional Judgment Analysis divided mine drainage into five subcategories based on the net acidity (acidity minus alkalinity) or alkalinity (alkalinity minus acidity) of the untreated mined drainage:

1. Very acid: net acidity > 300 mg l^{-1} as $CaCO_3$
2. Moderately acid: 10 £ net acidit £ 300 mg l^{-1} as $CaCO_3$
3. Weakly acid: 0 £ net acidity < 100 mg l^{-1} as $CaCO_3$
4. Weakly alkaline: net alkalinity < 80 mg l^{-1} as $CaCO_3$
5. Strongly alkaline: Net alkalinity ‡ 80 mg l^{-1} as $CaCO_3$

The Best Professional Judgment Analysis found that conventional treatment is usually the best available technology for discharges of subcategory 1. For moderately acid to strongly alkaline discharges, wetlands as a best available technology should be designed on the basis of chemistry and loading, using criteria similar to those presented

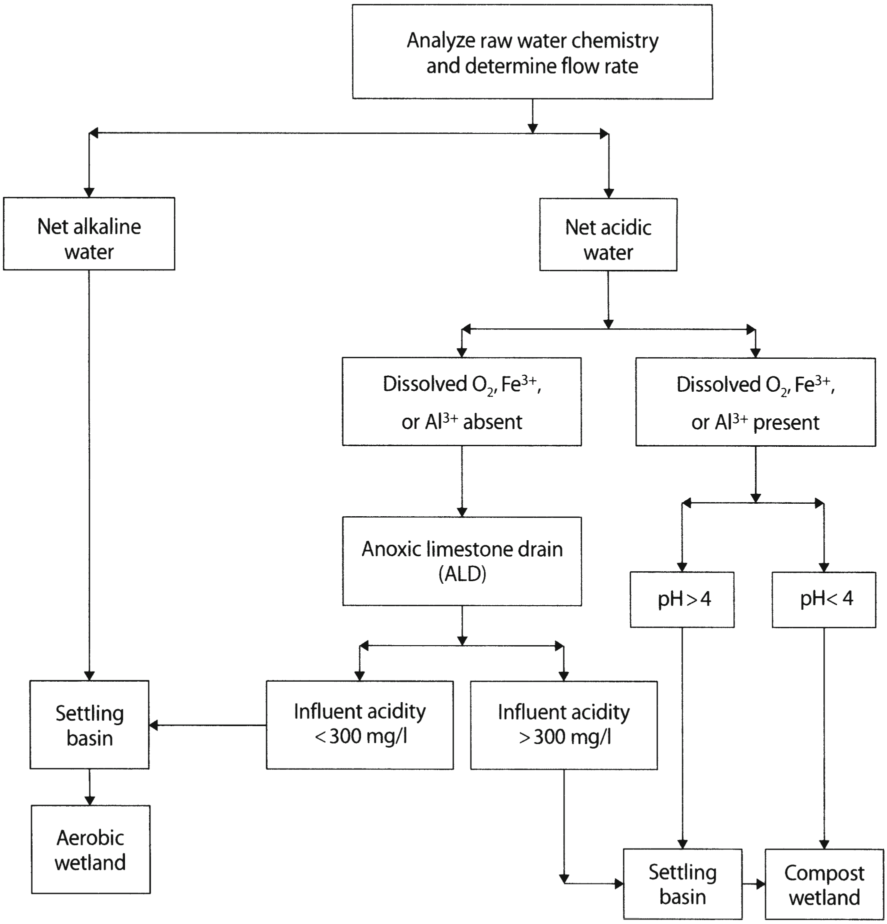

Fig. 8.1. Decision making chart

by Hedin et al (1994), an adaptation of whose decision making chart (their Fig. 12) is presented here for the reviewer's or designer's use (Fig. 8.1). Combinations of these passive systems, such as sequential alkalinity producing systems (SAPS), can be designed and reviewed on a site-specific basis (Kepler and McCleary 1994). Because of continuing developments in the design of sequential systems, some regulatory staff have adopted modifications to Fig. 8.1 to facilitate regulatory technical review. An example of such a modification, due to J. Corey Cram (pers. comm.) of Pennsylvania DEP Hawk Run District Mining Office, is presented as Fig. 8.2.

8.3.2
Removal of Metals from Alkaline Discharge

If the mine discharge is alkaline, but has elevated [Al], [Fe], and [Mn], an aerobic constructed wetland or settling basin is the usual choice for the removal of metals. An

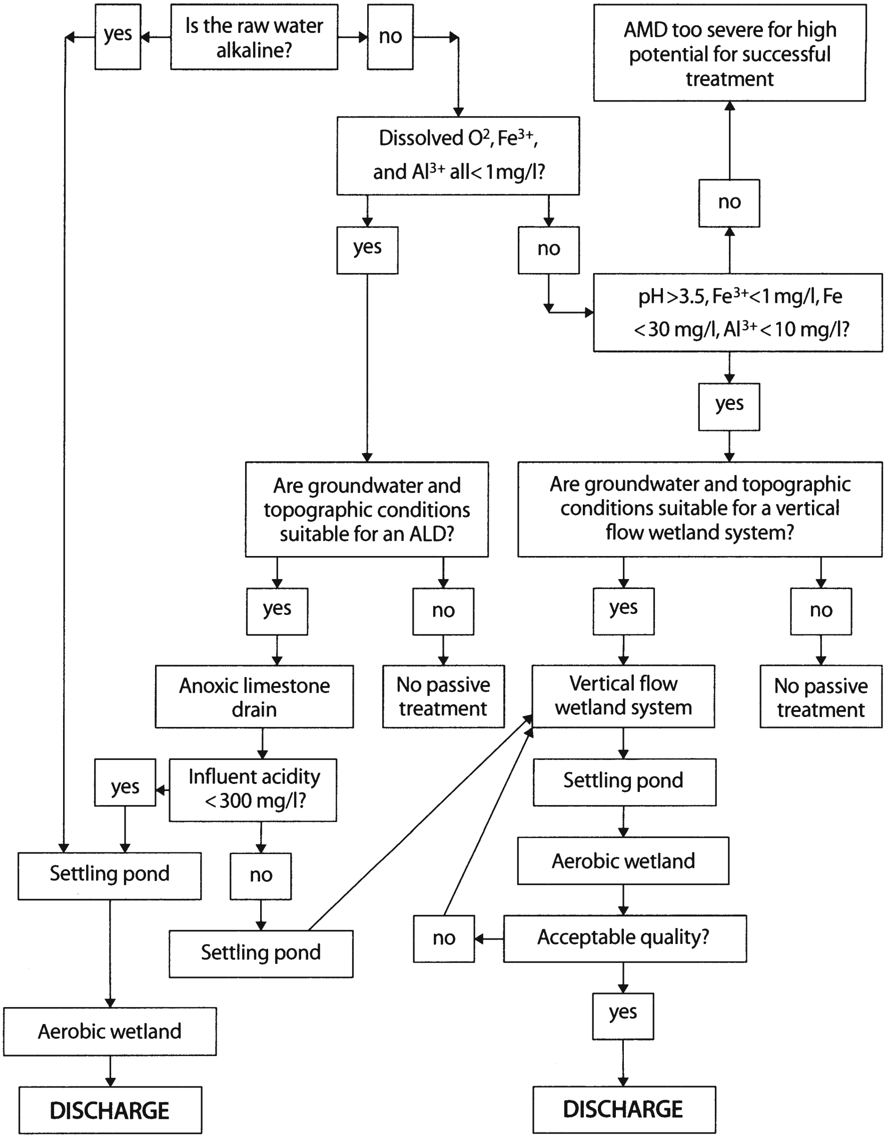

Fig. 8.2. Refinement to decision making chart (Fig. 8.1). *AMD* acid mine drainage; *D.O.* dissolved oxygen; *g.w* groundwater; developed by J. Corey Cram (1997)

aerobic wetland is one which has been designed and built to take advantage of oxidation processes under alkaline conditions to remove metal cations, whereas an anaerobic wetland is one designed and built to take advantage of reducing conditions to generate alkalinity and retain metal cations as sulfides and carbonates. Aerobic wetlands and settling basins rely on the oxidation of Fe^{2+} (and to a lesser degree Mn^{2+}), hydrolysis of Fe^{3+} and Al^{3+}, and settling and retention of oxidation and hydrolysis products such as FeOOH, MnO_2, and $Al(OH)_3$. These processes are only effective in water which has a net alkalinity, which may be initially present in the mine drainage or be imparted to acidic water by means of conventional treatment or by the passive systems on the right hand part of the flow chart. During a given period, solid oxides and hydroxides of iron, aluminum, and other cations will accumulate in aerobic wetlands or settling basins. The wetland design will provide either for periodic cleaning or for added volume reserved for solids retention, or for a combination of both. Aerobic wetlands must be designed to produce an effluent with a residual alkalinity greater than effluent acidity. It is desirable for the net effluent alkalinity to be greater than 20 mg l^{-1}.

8.3.3
Imparting Alkalinity to Acid Discharge

If the mine drainage is acidic or weakly alkaline; i.e., falls into subcategories 1 through 4, the designer can consider using one of several methods to impart alkalinity to the mine drainage before it is directed into an aerobic constructed wetland. Possibilities include aerobic limestone channels, anoxic limestone drains, horizontal flow anaerobic wetlands, and vertical flow anaerobic wetlands.

8.3.4
Aerobic Limestone Channels

An aerobic limestone channel imparts alkalinity to the mine drainage by passing it over and through a limestone ($CaCO_3$) channel under aerated conditions. The dissolved O_2 oxidizes the metal cations while the limestone dissolves to impart bicarbonate alkalinity to the drainage, thereby neutralizing acidity. The limestone particles in these channels often become coated, or armored, with FeOOH, retarding the dissolution of $CaCO_3$. Furthermore, precipitation of FeOOH and $Al(OH)_3$ can physically fill the channel unless sufficient velocity is provided to keep the channel flushed. Both of these considerations tend to limit the use of aerobic limestone channels for imparting alkalinity. Mine drainage which is weakly acidic with only slightly elevated [Fe] and [Al] is probably most amenable to this type of pretreatment.

8.3.5
Anoxic Limestone Drains (ALD)

The problem of armoring and plugging, encountered in many aerobic limestone channels, may be avoided by constructing anoxic limestone drains. Anoxic limestone drains are most effective when the dissolved $[O_2]$, $[Fe^{3+}]$, and $[Al^{3+}]$ in the influent each approaches zero. They also impart alkalinity at greater rates when dissolved $[CO_2]$ is high. They should therefore be designed to exclude oxygen and retain carbon dioxide. Groundwater exchange and invasion by roots of woody plants should be kept to a minimum. An artificial liner may be considered, and the drain should be covered by at least 1 m of compacted earth. The limestone should be kept inundated, and deflectors or dams might be necessary to achieve inundation. The drain should not be expected to produce more than about 275 to 300 mg l^{-1} of alkalinity. It should be made from the purest limestone, $CaCO_3$, economically available; $CaMg(CO_3)_2$, or dolomite, is not as effective. The detention time should be at least 14 h (Hedin *et al.* 1994).

While some calculations have been suggested to design the size of the anoxic limestone drain on the amount of limestone necessary to achieve the design effluent alkalinity during the expected life span, consideration must be given to interference from Fe^{3+} and Al^{3+} precipitation, physical collapse, plugging, or a greater limestone removal rate at the influent (more acidic) end. The quantity of limestone to be used (tons) will substantially determine the design life of the ALD. The design will also consider flow velocities, detention times, and maintenance practices to be followed if, due to unexpected oxidizing conditions, the limestone becomes ineffective due to armoring with FeOOH or other material, or due to plugging with $Al(OH)_3$ or other material. The effluent of the anoxic limestone drain is directed into an aerobic wetland or settling basin to remove the metals.

8.3.6
Horizontal Flow Wetlands

Anaerobic, or compost, wetlands can often be used where Fe^{3+}, Al^{3+}, or O_2 are present in the influent. The compost wetlands function by reacting the mine drainage with alkalinity in the form of HCO_3^- generated from dissolution of limestone present in the substrate, and from the reduction of Fe^{3+} and SO_4^{2-} The iron and sulfate reducing bacteria produce bicarbonate as part of their life processes, so long as favorable pH levels are present. Beginning about 1 cm below the surface, substrates tend to become anoxic, or reducing. Some wetlands rely on diffusion of CO_3^{2-} from the substrate into the aerobic surface water to impart alkalinity to water there. Others are horizontal flow systems which have a permeable compost designed to maximize contact between the anaerobic substrate and the polluted water.

8.3.7
Vertical Flow Wetlands

Still others are designed as vertical flow reactors, many with limestone or other underdrains (Kepler and McCleary 1994). It is theorized that $Al(OH)_3$, FeS, and $FeCO_3$ are some of the entities retained in the substrate matrix, and studies are continuing. There are few data on the long-term performance of wetlands wherein mixing of the polluted water and the substrate was part of the design; and questions remain unanswered about how long the substrate will continue to function and whether the flow rate through the organic matter can be maintained. The amount of carbon required and the expected substrate life based on sulfate reduction can be used as predictive tools, but long term effects of the decrease of substrate permeability and availability of reaction surfaces due to accumulation of precipitated metal compounds are not known. For this reason, the designer is urged to be conservative in designing anaerobic wetlands, possibly considering alternating systems in parallel.

Another suggested design approach is to base the design life of the anaerobic wetland on the amount of limestone included in the substrate, using the same calculations as one would when designing an anoxic limestone drain. Typical diagrams of an anoxic limestone drain, an aerobic wetland, and a horizontal and vertical flow anaerobic wetland are shown in Fig. 8.3.

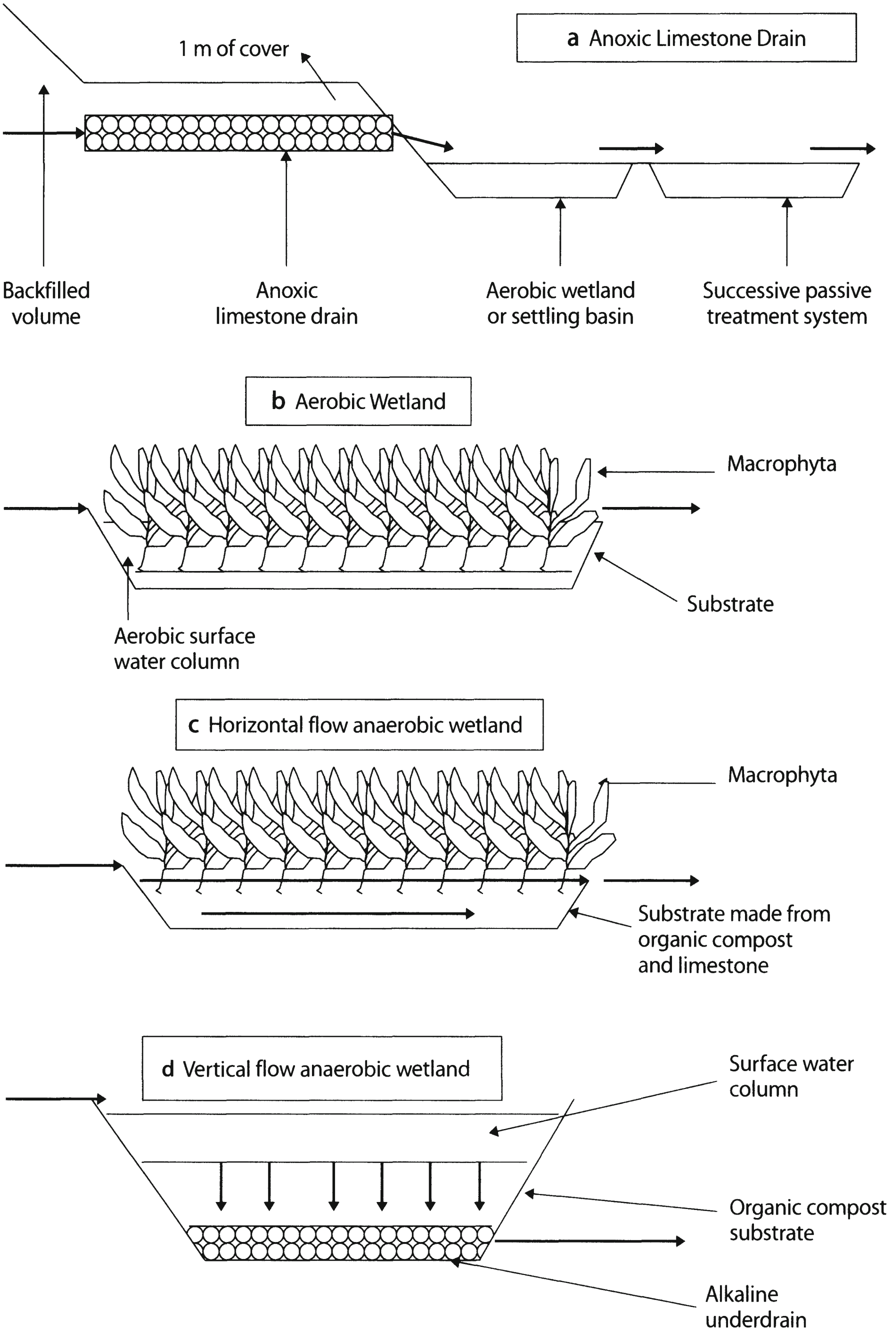

Fig. 8.3. **a** Anoxic limestone drain. **b** Aerobic wetland. **c** Horizontal flow anaerobic wetland. **d** Vertical flow anaerobic wetland

8.3.8
Substrate Design

The substrate is usually composed of an animal manure waste, such as aged horse manure or spent mushroom compost. Pulverized limestone or limestone chips are sometimes mixed with the organic substrate to add alkalinity. Without the presence of sufficient alkalinity, the bicarbonate generating organisms may become overwhelmed by the acidity from the mine drainage and cease to produce bicarbonate. Mixtures of about 50% each by volume of organic substrate and limestone are commonly used.

Vertical flow anaerobic wetlands are commonly designed with a limestone underdrain often supplemented with piping to convey the treated water to the next stage of the treatment process.

8.3.9
Sequential Passive Treatment Systems

Because of the varying effects and limitations of the different types of passive systems, the practice of designing several types of passive systems in series is becoming increasingly common. One might, for example, find an anoxic limestone drain for imparting alkalinity, followed by an aerobic wetland for oxidizing Fe^{2+}, followed by an aerobic wetland for removing Fe^{3+} and Al^{3+} by hydrolysis and settling, followed in turn by an aerobic limestone bed to remove Mn^{2+} by oxidation. A series of constructed wetlands, particularly one including vertical flow anaerobic wetlands, is often called a SAPS (Successive Alkalinity Producing System; Kepler and McCleary 1994).

Anoxic limestone drains and vertical flow anaerobic wetlands can be sized on the basis of the amount of limestone or other alkaline material necessary to impart the design alkalinity requirement for a specified period of time. For example, the amount of limestone required to impart 300 mg l^{-1} of alkalinity to a discharge flowing at 100 m^3 day^{-1} for 25 years, assuming 90% of the limestone to be composed of $CaCO_3$ and the process efficiency to be 75% would be:

$$\frac{300\,\text{mg}\,\text{l}^{-1} \times 100\,\text{m}^3\,\text{day}^{-1} \times 365\,\text{days year}^{-1} \times 25\,\text{years}}{0.90 \times 0.75} = 406\ \text{tons of limestone}.$$

This would apply, for example, to a polluted discharge with 260 mg l^{-1} of acidity for which a residual alkalinity of 40 mg l^{-1} is required. Careful consideration should be given to the hydraulic gradients through the anaerobic part of the SAPS.

8.3.10
Inlet and Outlet Structures

The inlet structure should be designed so as to permit the convenient collection of water quality samples, and should be provided with a means to measure influent flow (pipe outlet for bucket and stopwatch measurements, weir, flume, etc.). The outlet structure should also be provided with a means of measuring flow and gathering samples, and should have a deflector or some other outlet device to protect the outfall from solids carry-over into the effluent. The outlet structure must be protected from structural damage from freezing, and from damage due to animals such as beavers.

8.3.11
Plants

Plants serve the primary function of promoting uniformity of flow across the surface of the wetland. Secondary functions of plants are to provide colonization sites for microorganisms which oxidize Fe^{2+}, and to provide some measure of metals uptake. They also serve to stabilize and retain the substrate and to make the wetlands aesthetically pleasing. Plants should be selected on the basis of their tolerance for the mine drainage to be treated, as well as their local availability. Species diversity should be encouraged.

The dominant macrophyte in constructed wetlands in Pennsylvania is *Typha latifolia,* with some colonization by *Typha angustifolia.* Other macrophytes used in wetlands include but are not limited to *Juncus effusus* and *Phragmites communis.* The designer has much flexibility is this aspect of wetlands design.

8.3.12
Stability

The sides should be designed with an inward slope of between 2:1 (horizontal:vertical) and 3:1 (18 to 27 degree angle measured from horizontal). This maximizes stability, makes it easier for equipment that is used in building or maintaining the wetlands to operate, and to some degree discourages (but is not guaranteed to eliminate) mosquitoes. The bottom should have a level to slightly upward slope in the direction from influent to effluent. If *Typha latifolia* are to be planted, surface water column depths of more than 50 cm are usually detrimental. However, wetlands are commonly designed with deeper zones to allow storage of precipitated metal oxides and hydroxides.

Wetlands must be designed so that their physical integrity is maintained. Surface water runoff must be excluded from the wetlands to minimize erosion damage, as well as to maximize detention time. A system of ditches or raised berms designed in a manner similar to the design of diversion and collection ditches is the usual means to accomplish this. The wetland must be provided with a freeboard of not less than 1 m above the free water surface. The freeboard serves a twofold purpose: (1) it prevents overtopping of the wetland during storm events, and (2) provides long-term detention of the accumulated solids. The engineer should calculate how much wetland capacity will be occupied by settled solids at the end of its design life, and is strongly encouraged to design the freeboard accordingly.

8.3.13
Liners

Exchange of the treated water with the groundwater system must be prevented, using either a clay liner or an artificial liner where necessary. The designer should be satisfied that the liner material will not itself contribute pollution to the water being treated. The principal reason for trying to eliminate groundwater exchange is to protect the

Table 8.2. Recommended treatment rates for sizing wetlands

	AML criteria (g m^{-2} day^{-1}) Influent alkaline	Influent acid	Compliance (g m^{-2} day^{-1}) Influent alkaline	Influent acid
Iron	20	nap	10	nap
Manganese	1.0	nap	0.5	nap
Acidity	nap	7	nap	3.5

AML, abandoned mine lands; *nap*, Not applicable.

groundwater from pollution, but it is also important to eliminate unmeasured effects of groundwater exchange when determining the effectiveness of the wetland.

8.3.14
Animal Damage

The sides of the wetland must be protected from damage from such animals as muskrats; where muskrats present a problem, they can be discouraged by incorporating a structure such as chain link fence into the sides of the wetlands.

8.3.15
Sizing of Wetlands

The sizing of vertical flow anaerobic wetlands and anoxic limestone drains has been discussed above. Aerobic wetlands and horizontal flow anaerobic wetlands may best be designed using the criteria given by Hedin *et al.* (1994) (Table 8.2). In Table 8.2, the abandoned mine lands (AML) criteria column gives the design sizing of wetlands constructed to remove acidity, Fe, and Mn from discharges when reclaiming AML. The compliance criteria column gives guidelines for design area necessary for constructed wetlands to achieve compliance with the criteria given in Table 8.1.

8.4
Monitoring and Regulatory Considerations: Case Studies

Passive treatment systems installed to treat coal mine drainage should receive sufficient monitoring to characterize their performance. Monitoring data are necessary to aid in further development design criteria for passive systems. Furthermore, monitoring provides the coal miner an opportunity to take corrective measures in time to avoid the discharge of water which does not conform to effluent criteria. At a minimum, the flow rate, pH, alkalinity, acidity, [Fe], and [Mn] should be determined for the influent and effluent. In some cases, [SO_4^{2-}], [Al], [Ca], or other metals concentrations may provide useful information. Furthermore, the operator may wish to measure [Fe^{2+}] as well as total [Fe]. For sequential systems, it would prove beneficial to obtain the same information for the effluent of each subsystem within the wetlands system. The consultant and operator are recommended to obtain advice and direction from their appropriate

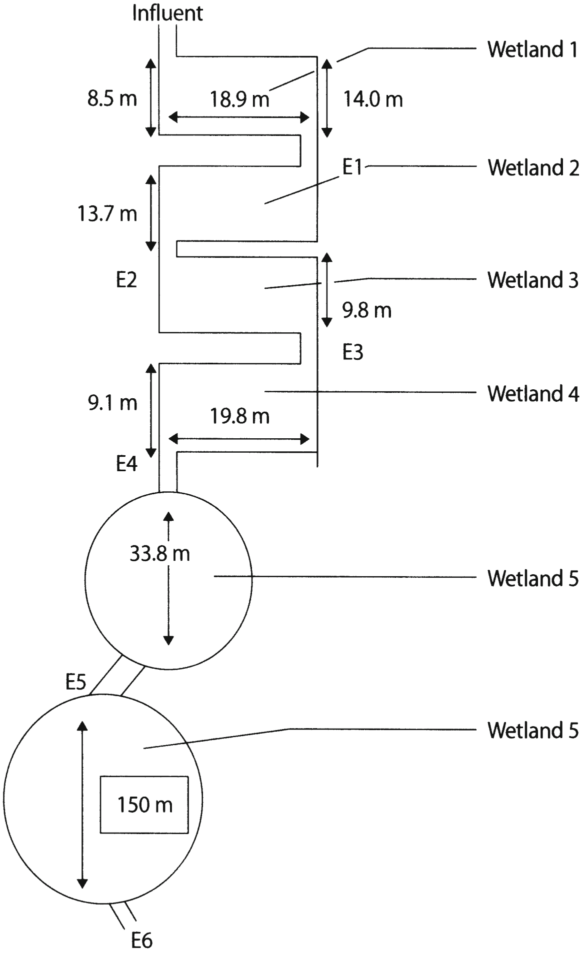

Fig. 8.4. Case Study 1: wetlands in Clearfield County, Pennsylvania

regulatory authorities before constructing and operating a wetlands system to treat coal mine drainage.

8.4.1
Selected Case Studies

Case Study 1 (Fig. 8.4) consists of six sequential aerobic wetlands in Brady Township, Clearfield County, Pennsylvania, which were built in 1985 to remove Fe^{3+} and Mn^{2+} from an alkaline mine drainage. The illustrative data given in Table 8.3 were obtained by the Department in 1990. Flows were measured in some cases by bucket and stopwatch, in other cases by estimation. The mine drainage is directed into wetland 1, which has an average water depth of 38 cm and an organic substrate depth of 76 cm. This wetland's effluent, E1, becomes the influent to wetland 2, where the respective water and substrate depths are 18 cm and 37 cm. The effluent, E2, is then directed into wetland 3 with respective water and substrate depths of 15 cm and 57 cm. That wetland's effluent, E3, is

Table 8.3. Performance of Case Study 1 wetlands in Brady Township, Clearfield County, Pennsylvania

		Influent				Effluent				Removal	
WL (area)	Date	Flow (m^3 day^{-1})	Net Alk (mg l^{-1})	[Fe] (mg l^{-1})	[Mn] (mg l^{-1})	Flow (m^3 day^{-1})	Net Alk (mg l^{-1})	[Fe] (mg l^{-1})	[Mn] (mg l^{-1})	Fe (g day^{-1} m^{-2})	Mn (g d^{-1} m^{-2})
1	5/24	109	62	31.5	34.3	65	20	12.5	32.1	11.73	7.35
223m^2	6/22	164	42	29.0	30.0	164	16	8.9	28.4	14.71	1.17
	10/10	216	48	31.6	32.2	216	29	12.4	35.3	18.60	-3.00
	11/21	259	8	22.6	25.3	259	-44	8.2	26.1	16.80	-0.93
	12/12	203	70	27.7	26.3	203	56	15.9	26.4	10.74	-0.09
	Mean	190	46	28.5	29.6	181	15	11.6	29.7	14.86	1.13
2	5/24	65	20	12.5	32.1	82	12	3.0	30.4	2.14	-1.46
266m^2	6/22	164	16	8.9	28.4	164	14	1.6	28.9	4.49	-0.31
	10/10	216	29	12.4	35.3	194	19	3.1	33.6	7.80	4.11
	11/21	259	-44	8.2	26.1	259	-26	1.7	23.1	6.31	2.92
	12/12	203	56	15.9	26.4	229	12	6.6	23.0	6.54	0.35
	Mean	181	15	11.6	29.7	186	3	3.2	27.8	5.66	0.84
3	5/24	82	12	3.0	30.4	123	14	0.4	30.1	0.98	-6.05
199m^2	6/22	164	14	1.6	28.9	164	16	0.3	28.3	1.09	0.49
	10/10	194	19	3.1	33.6	194	14	0.4	29.4	2.66	4.10
	11/21	259	-26	1.7	23.1	220	-28	0.5	21.1	1.65	6.73
	12/12	229	12	6.6	23.0	212	28	9.7	21.5	-2.73	3.59
	Mean	186	6	3.2	27.8	182	9	2.3	26.1	0.92	2.01
4	5/24	123	14	0.4	30.1	43	10	0.4	30.7	0.20	13.28
178m^2	6/22	164	16	0.3	28.3	164	18	0.3	28.6	0.00	-0.28
	10/10	194	14	0.4	29.4	194	-12	0.3	30.0	0.07	-0.66
	11/21	220	-28	0.5	21.1	220	22	0.5	21.2	-0.01	-0.12
	12/12	212	28	9.7	21.5	212	28	1.9	20.9	9.29	0.71
	Mean	182	9	2.3	26.1	167	13	0.7	26.3	1.68	2.14
5	10/10	194	-12	0.3	30.0	194	-20	0.0	18.4	0.12	4.41
511m^2	11/21	220	22	0.5	21.2	220	18	0.0	16.9	0.21	1.85
	12/12	212	28	1.9	20.9	212	22	0.6	18.7	0.53	0.91
	Mean	209	13	0.9	24.0	209	7	0.2	18.0	0.28	2.47
6	11/21	220	18	0.0	16.9	220	9	0.7	9.3	-0.41	4.32
390m^2	12/12	212	22	0.6	18.7	212	8	0.0	8.6	0.35	5.50
	Mean	216	20	0.3	17.8	216	9	0.4	8.9	-0.02	4.92

WL, wetland.

received by wetland 4, which has a maximum 60 cm of substrate but which is provided with deep and shallow zones, causing the water depth to vary from 30 cm to 185 cm. The effluent E4 is directed into wetland 5, which has no organic substrate and has >100 cm of water depth. The effluent E5 is then directed into wetland 6, with a 4 cm water depth and a naturally developed organic substrate ranging from 30 to 72 cm deep.

Wetlands 1, 2, and 3 have about 70% surface coverage by *Typha latifolia* with some *Leersia arizoides*. Wetland 4 has *Typha* only along its edges, while wetland 5 has for its first 17 m mostly *Leersia*, then 9.1 m length of deep water, then 9.1 m length of *Typha*. Wetland 6 has about 95% cover of *Leersia*.

The Fe removal rate in the first two wetlands compares favorably with the removal rates suggested in Table 8.2. In the remaining wetlands, the mass of Fe removed per day per meter squared of wetland (g day^{-1} m^{-2}) is less because there is less Fe to be removed. The wetland treatment system is an example of one which was given little or no maintenance after having been built, and demonstrates that wetlands can successfully be built to remove Fe^{3+} from an alkaline discharge. The removal rate of Mn^{2+} is higher than that suggested in Table 8.2, and increases through the wetland system as [Fe] decreases. We believe that *Leersia* in wetlands remove the Mn by providing reaction surfaces upon which MnO_2 is precipitated under local alkaline, oxidizing conditions; al-

Table 8.4. Performance of Case Study 2 wetlands in Eldred Township, Jefferson County, Pennsylvania

WL	Area (m^2)	Influent Flow ($m^3\,day^{-1}$)	Net Alk ($mg\,l^{-1}$)	[Fe] ($mg\,l^{-1}$)	[Mn] ($mg\,l^{-1}$)	Effluent Flow ($m^3\,day^{-1}$)	Net Alk ($mg\,l^{-1}$)	[Fe] ($mg\,l^{-1}$)	[Mn] ($mg\,l^{-1}$)	Removal Fe ($g\,day^{-1}.m^{-2}$)	Mn ($g\,day^{-1}.m^{-2}$)
1	162	91	−37	34.5	11.2	69	−38	3.6	12.4	17.79	0.99
2	145	69	−38	3.6	12.4	69	−28	1.4	12.2	1.05	0.10
3	248	69	−28	1.4	12.2	9	−16	1.1	9.7	0.35	3.06
4	120	9	−16	1.1	9.7	9	−3	0.1	7.6	0.07	0.15
5	117	9	−3	0.1	7.6	76	−1	0.2	6.0	−0.12	−3.36
6	125	76	−1	0.2	6.0	76	4	0.1	4.4	0.06	0.98
7	129	76	4	0.1	4.4	76	168	0.2	1.3	−0.06	1.84
8	131	76	168	0.1	1.3	76	28	0.3	0.6	−0.12	0.41
9	13	76	28	0.1	0.6	76	90	0.30	0.4	−0.59	1.18

WL, wetland.

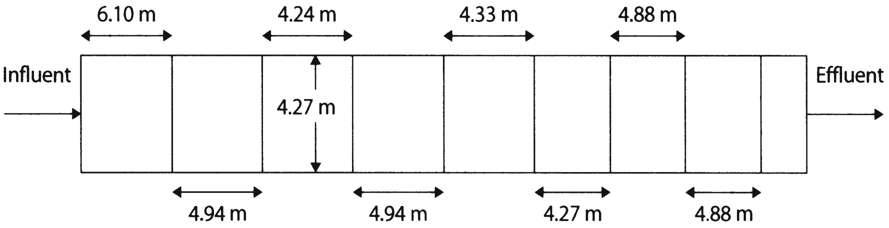

Fig. 8.5. Case Study 2: wetlands in Eldred Township, Pennsylvania

though some Mn is taken up by the plants, this cannot account for the observed removal rate of about 50%.

The Case Study 2 wetland system (Fig. 8.5) in Eldred Township, Jefferson County, Pennsylvania, consists of nine wetlands in series. The post-mining discharge, which has net acidity, is piped into wetland 1, and the effluent is from wetland 9. The wetlands are separated by limestone boulders, around and through which the effluent from a wetland cell flows into the succeeding wetland. The surface water depth averages 10 cm. The wetland substrate, which is composed of limestone chips and spent mushroom compost, averages 30 cm depth. Iron removal efficiency in wetland 1 is 90%, and 97% in the combined wetlands 1 and 2. Alkalinity is progressively imparted through the first eight cells. This illustrates a horizontal flow wetland system wherein Fe^{3+} is removed by hydrolysis, and alkalinity is imparted to the surface water column from the substrate and from the limestone barriers. The progressive removal of Mn^{2+} may be attributed to oxidation and precipitation on the surfaces of the limestone boulders between wetlands.

The dominant macrophyte in wetlands 1 through 4 is *Typha latifolia*, but more diversity occurs in wetlands 5 through 9, where *Leersia* and *Solidago* spp. predominate. The surface coverage by macrophytes is essentially 100%. Average performance during a 3-month study period is shown in Table 8.4.

8.5
Summary

Constructed wetlands have become a best available technology for the treatment of coal mine drainage. Wetlands offer the operators of coal mines the hole of considerable financial savings and the possible release of performance bonds. They also offer a means for abating pollution as abandoned mine lands for which no current coal mine operator has liability are reclaimed under State and Federal programs. Designers of passive systems such as wetlands must consider in their designs the several mechanisms by which pollution is removed from mine drainage. It is of prime importance in any treatment system design to produce a final discharge whose alkalinity exceeds its acidity. Several mechanisms and suggested designs for producing alkalinity have been discussed above.

Acidity in mine drainage is present not only as H^+ but also as metal cations, such as Al^{3+}, Fe^{2+}, Fe^{3+}, and Mn^{2+}, which can be removed in wetlands systems. Design strategies have been suggested to take advantage of oxidation to remove Fe^{2+} and Mn^{2+} and of hydrolysis and settling to remove Fe^{3+} and Al^{3+}. Removal of cations as sulfides has also been discussed. Consideration has been given to flow configurations, substrate design, inlet and outlet structures, and the role of plants.

The author conducted a study which resulted in a report to the EPA, which contains a more detailed search of the literature on constructed wetlands treatment of coal mine drainage (Hellier 1996). There are also several other publications which the reader may find useful; for example, EPA (1988), US Department of Agriculture and EPA (1995), Reddy and Smith (1987), Hammer (1989), Hammer (1992), Cooper and Findlater (1990), and Moshiri (1993). As for any developing technology, innovative ideas are continually being advanced and the performance of passive treatment systems improved. The reader is encouraged to share suggested improvements with those who are attempting to abate pollution from mine drainage.

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Chapter 9

Underwater Placement of Mine Tailings: Case Examples and Principles

Derek V. Ellis and James D. Robertson

9.1

Introduction

9.1.1

The Canadian Mine Environment Neutral Drainage (MEND) Program

Over the past 10 years, the Canadian government/industry (Mine Environment Neutral Drainage) MEND program has undertaken a detailed study of mine drainage with the objective of developing mitigating procedures (Sly 1996). An important conclusion is that placement of tailings underwater can prevent or reduce acid rock drainage (ARD). This technical conclusion has been supported by an independent review (Rawson 1992).

There have been several follow-on developments by the MEND program. A *Design Guide for Disposal of Reactive Tailing in Constructed Impoundments* is being developed, based on the field studies that have been undertaken of mine sites where tailings were placed underwater in lakes. Currently two flooded impoundments have been built in Canada, but monitoring results are not yet in the public domain.

Simultaneously with the MEND developments, there has been increasing concern with tailings placement in the sea; from ARD, other forms of contamination, and other environmental impacts. This has resulted in several reviews of which the latest (including deep ocean tailings placement) is by Ellis and Poling (1995). Where an ore body has potential to develop ARD, the contamination risk from waste rock and tailings is greatest when they are placed with exposure to rain, air and its ample oxygen, and acid-generating soil bacteria (Perry 1995), e.g. in rivers and at the edges of lakes and the sea. The risk can be reduced if oxygen is limited, i.e. underwater and especially where pH changes can be neutralized, e.g. deep in the sea.

9.1.2

Range of Environmental Impacts and Biodiversity Recovery from Tailings Placement

Tailings wherever they are placed have other environmental impacts than acid generation and contamination (Table 9.1). Essentially they can smother land, lakebed or seabed; they can change biodiversity or temporarily eliminate it; and they can take productive habitat away from human use. Controlled underwater placement potentially can mitigate some or all of these other impacts.

Where tailings are placed on land, in rivers and on shorelines the smothering effect may impact areas populated by people with traditional rights. In rivers and lake shallows the smothering effect may be on productive and intensely used ecosystems; par-

Table 9.1. Environmental risks from mine tailings

Chemical contamination	- From acid rock drainage and other geochemical processes - Damages ecosystem and resource use downstream from site - Recovery to an undegraded ecosystem likely to be very slow, and measured in decades or more
Habitat smothering	- From deposition of tailings, smothering living organisms and their habitats - Does not contaminate the ecosystem - Occurs where deposition is at a rate greater than organisms can avoid or grow through the deposits - Recovery on land is measured in decades for tree growth - Recovery underwater may be quicker, e.g. 1–5 years
Catastrophic system collapse	- From earthquake or torrential rains undermining structural stability - May cause sudden and massive loss of life and resources. - Recovery complex
Landform changes	- Changes habitat - Recovery on land measured in decades for tree growth - Recovery underwater may be quicker, 1–5 years
Water turbidity and siltation	- Reduces river and lake use by changing river channels and flood plains, biological productivity and fishery resources - Recovery to previous water clarity may be measured in days; but ecosystem consequences may only recover over periods of years
Socio-economic changes	- Encompasses changes in resource use and consequent changes to people - Recovery complex

Recovery from physical/chemical/biological processes in the first five categories above is taken to be return of the mine's surrounding ecosystem to a sustainable successional state sufficiently advanced that biodiversity will progress over time to a climax (equilibrium) biological community unless interrupted by other impacts, or natural setbacks.

Table 9.2. Time periods to recovery for benthic biodiversity. Information accumulated from tailings deposits and dredging sites (CCORE 1996)

Fine-grained deposits:	
muds/silts/clays; may contain some rocks and boulders	1 year
Medium-grained deposits:	
sands; can contain some silts/clays and/or gravels	1–3 years
Coarse-grained deposits:	
gravels; can contain some finer fraction and some rocks and boulders	5 years
gravels with many rocks and boulder	>5 years

Recovery is taken to be growth of a definable community of species which represents an early successional stage leading towards an eventual high-biodiversity, but probably highly variable, climax (equilibrium) community.

ticularly for example subsistence, artisanal or commercial fisheries. In contrast, tailings placed deep in lakes and the sea may smother only bed deposits below the productive surface zones, and with relatively low fisheries production or potential (Ellis and Ellis 1994).

There is a further impact risk from placing tailings in an on-land impoundment wherever earthquake, landslide, intense rainstorm or snow-melt with flooding may occur. The potential risk is the catastrophic collapse of the tailings impoundment. Such collapses have occurred sufficiently frequently in recent years even with modern dam technology (UNEP 1997) that the risk should not be ignored. The consequences of impoundment collapse are magnified if the downstream region is heavily populated, or supports valuable biodiversity or fishery resources. The west coasts of Alaska, the USA, and Canada are examples. The risks of earthquakes, storm rainfall, and snow-pack meltwater flooding, are all high. The rivers support the spawning grounds of the Pacific salmon, the most important commercial fishery species on the west coast of North America. Placement deep in the sea eliminates these on-land placement risks.

9.1.2.1
Biodiversity Recovery

Another factor to be considered in tailings placement is the potential for recovery of the impacted ecosystems. Recovery can be either natural (recolonization) or engineered (reclamation). Mine reclamation on-land is now formalized by required codes of practice in many countries (e.g. AMIC, ND). However, very little has been known until recent years about the recovery of biodiversity and resources following tailings placement underwater. Our knowledge of seabed biodiversity recovery has increased rapidly over the past ten years (e.g. Ellis and Hoover 1990a; CCORE 1996). We now know that seabed recovery (ecological succession) after defaunation events may proceed to the point of sustainable moderate biodiversity within 1–5 years depending on the particle size of the seabed deposits (Table 9.2). However, little detailed information is yet available for lakes.

Knowledge about recovery of tailings-smothered habitat allows predictions of the time during which such habitat will be unproductive, and to what productive use it can recover. The concept of ecological succession to a sustainable biodiversity can be useful here. A 1-to 2-year recovery of seabed biodiversity from low or absent to moderate, with consistent appearance of particular species is a form of ecological succession. An even higher biodiversity after several more years confirms both the succession and sustainability. The new knowledge about biodiversity on tailings from proven marine placement systems means that predictions of time-to-recovery to a sustainable succession can be made at new sites in at least general terms.

9.1.3
Reviewing and Updating Tailings Placement Regulations

Many regulations controlling tailings placement were developed as precautionary controls 20–30 years ago due to the obvious environmental impacts of the past, and lack of information on how these impacts could be controlled cost-effectively. It is noticeable that early regulations are now being reviewed and changes implemented in some areas.

In Canada, an Environmental Effects Monitoring Program is now ongoing to assess the adequacy of effluent regulations under the Federal Fisheries Act. This is a longstanding piece of legislation from which regulations controlling mining and other wastes have been developed. An independent review of submarine tailings placement was recently completed (Golder 1996).

In the USA, the Federal Register (EPA 1996) stated that the Environmental Protection Agency was recommending to exempt dewatered tailings at the proposed AJ Mine, Alaska, from the earlier Effluent Limitation Guidelines as "...extreme topographic and climatic conditions at the A-J project site render it impractical to treat and dispose of tailings in a tailings impoundment."

The latest international agreement on placing wastes in the sea, the 1996 Protocol for the 1972 London Dumping Convention, includes the following statements:

- Article 7, Internal waters: "... each Contracting Party shall at its discretion either apply the provisions of this Protocol or adopt other effective permitting and regulatory measures to control the deliberate disposal of wastes or other matter in marine internal waters ..."
- Article 4, Dumping of wastes or other matter: "1. Contracting Parties shall prohibit the dumping of any wastes or other matter with the exception of those listed in Annex 1."
- Annex 1, Wastes or other matter that may be considered for dumping: "The following wastes or other matter are those that may be considered for dumping ... 5. inert, inorganic geological material ..."
- Article 8, Exceptions: "1. The provisions of article ... 4.1 ... shall not apply ... in any case which constitutes a danger to human life ... if dumping appears to be the only way of averting the threat and if there is every probability that the damage consequent upon such dumping will be less than would otherwise occur."

There is an unusual feature in these Articles. ARD-generating tailings, after placement in the sea, can be the "inert, inorganic geological materials" of Annex 1. This potential, in association with Article 8 (noting where other options may cause more damage), effectively means that tailings placement in the sea under some circumstances is in compliance with the Convention.

In summary, controls on tailings placement underwater are being reviewed by regulatory authorities. This reflects the growing experience with new methods of tailings management around the world.

9.1.4
Organization of this Chapter

This series of case examples of tailings placement underwater, and the development of such systems, is separated into two sections: lake and marine. Cases in the marine section are ordered to follow the development of effective regulations. At some of the cases, the system developed has not been brought into operation, but the approaches used in the development are informative as mitigation procedures for use at other sites.

Only summaries of each case history can be given here. Mining and environmental details are available from the references quoted for each case history. Two other cases

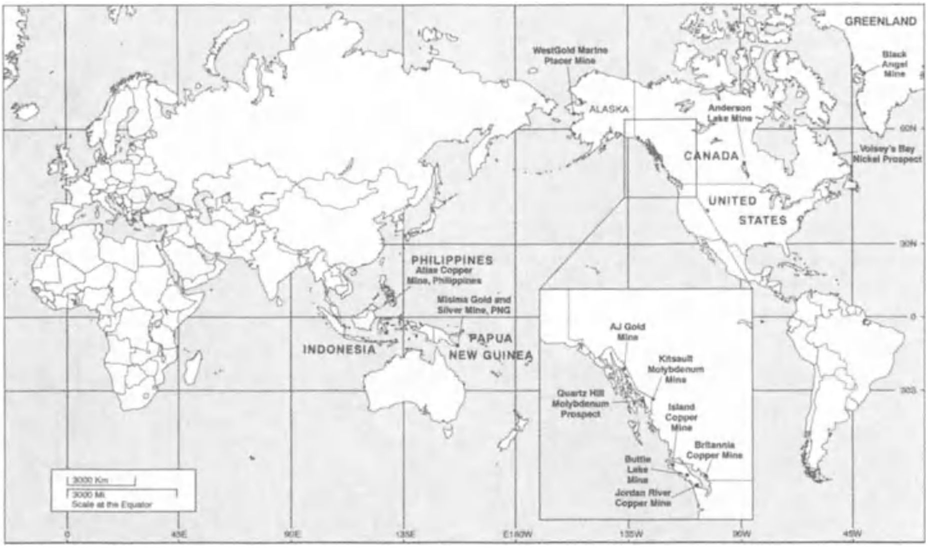

Fig. 9.1. Location of case examples reviewed or referenced

with good documentation on tailings placement are the Black Angel Lead-Zinc Mine in Greenland (Poling and Ellis 1995) and the WestGold Marine Placer Mine in Alaska (Garnett and Ellis 1995). Acid rock drainage was not an issue in these cases, although both presented contamination issues from other causes.

9.2
Case Histories (Fig. 9.1)

9.2.1
Lake Case Histories

Under the MEND program four lakes were examined (Sly 1996). These studies applied sophisticated sediment coring methods and dialysis arrays ("peepers") and analysis to extremely low detection levels to determine the metal profiles in the tailing sediments. The work demonstrated the natural geochemical processes which act to stabilize most sulfide minerals in the tailings and limit diffusion of specific metals into the porewater and overlying lake waters.

The current design for the application of this methodology has been limited to man-made impoundments as the main hydraulic and chemical variables can be actively quantified and controlled. Additional control strategies can be added for specific metals if needed, e.g. antimony, arsenic, molybdenum. The application of this technology to fresh water lakes is a more technically complex and politically sensitive issue which requires additional site-specific work. Many factors such as biological productivity, land use, regulatory limitations and risk analysis must be assessed. However, the generic geochemical processes which make a water cover an effective ARD control system have been successfully proven and documented through these studies.

9.2.1.1
Anderson Lake (Canada)

Anderson Lake in the province of Manitoba (Fig. 9.1), is a meso-eutrophic lake approximately 6 km long. It was operated by Hudson Bay Mining and Smelting Co., and actively received tailing from mines processing copper, lead and zinc ores, from 1979 until closure in 1995 (Rescan 1996). The tailings were deposited in relatively shallow water as the mean depth was 2.1 m. Studies which were conducted during the operating period demonstrated that metals were retained in the tailings through natural geochemical processes; and the efflux to the porewater and overlying lake water was very limited. As the lake was ice-covered for significant parts of the year, anoxic conditions developed to create a reducing environment which further assisted the geochemical stability of the tailing deposit. These studies indicated that for some metals there was a net influx of metals into the tailings from the overlying water column.

The availability of natural organic materials in this lake provided additional conditions to create a reducing environment which would enhance the development of metal sulfides and reduce metal efflux. Although organic material is not essential for the geochemical processes to work, this observation yielded an additional generic variable for achieving stable systems for mine closure.

9.2.1.2
Buttle Lake (Canada)

Tailings were deposited from a lead, zinc, and copper mine into Buttle Lake which is a large oligotrophic lake (35 km) located in mid Vancouver Island, British Columbia (Pedersen *et al.* 1994; Rescan 1995). The lake site functioned from 1966 to 1984 when the disposal system was changed to on-land, hence the case provides a closed underwater placement site. The site has a well established ARD problem associated with the underground mine, the waste rock and tailings. The original regulatory regime allowed the tailings to be deposited into the lake as a surface discharge when operations commenced in 1966. The tailings were located at a maximum depth of 87 m. As the lake did not freeze in the winter, high oxygen levels were present at depth throughout the year. However, the research determined that the sediments were anoxic below depths of 2 or 3 cm. At the time of the tailings deposition into the lake its ARD potential was not fully appreciated. Public pressures on the regulatory agencies resulted in the operating permit being revised to an on-land tailings system in 1984. The studies of the lake tailings deposits were conducted from 1984 through to 1995 (Pedersen *et al.* 1994).

The revised operating permits adopted an on-land subaerial tailings system which created beaches of tailings and required a treatment plant for metal removal. This system aggravated the tailings' ARD reactivity in allowing access to air and water in the wet west coast climate. Consequently the new system was an excellent comparison to the historic underwater tailings that was studied. A flooded tailings pond would have mimicked the lake system and prevented the ARD problem.

The measurement of limited metal effluxes into the porewater also demonstrated minimal metal efflux into the overlying lake water. The results also illustrated that the thin veneer of natural sediments (4 mm year^{-1}), which had deposited over 10 years of inactive use, further prevented any efflux of metals to the overlying water body.

9.2.1.3
Voisey's Bay Nickel Prospect (Canada)

Voisey's Bay is under development in Labrador in Canada's easternmost province of Newfoundland (Fig. 9.1). As a coastal site, submarine placement was a logical prospect for tailings management, but the alternative of subaqueous lake disposal is the preferred option at this site. There are several moderate sized, and not very productive, subarctic lakes near the ore-body (Cole *et al.* 1997). Initial laboratory tests have suggested that long-term reactivity of the tailings will be reduced under subaqueous lake conditions, although there appears to be an even greater reduction under seawater cover (Mugo *et al.* 1997).

There are two significant environmental features at this site in Canada. It has had a role in generating a review of national tailings placement regulations (Golder 1996). If Voisey's Bay uses lake placement for its sulfide deposit wastes, it will be the first such lake disposal system to come under the on-going regulatory scrutiny of a country with a history of requiring assessments and public reporting of those assessments.

9.2.2
Marine Case Histories

9.2.2.1
Britannia Copper Mine (Canada)

The Britannia Copper (Fig. 9.2) Mine was an old mine by the time provincial (British Columbia) pollution control regulations were developed in 1969. It operated continuously from 1899 to 1974, and during that time it generated a serious ARD problem which still remains in the area (McCandless 1995). It is now the site of considerable research on the marine geochemistry of ARD (Drysdale and Pedersen 1992).

The minesite is located alongside a 300 m-deep fjord basin separated by a 60 m sill from the outer fjord. Contaminated acid tailings water flowed down a creek 5 km to the sea, and polluted the surface brackish water. Intertidal organisms have been contaminated widely, and eliminated at the creek mouth and nearby shoreline. During the last years of operations the tailing was discharged at the shoreline. This fine material deposited in the nearshore area with precipitated ARD from the creek while coarse material deposited at depth.

By 1988, a quantitative assessment of ecosystem recovery in the area showed high biodiversity on both nearshore and deep tailings (Ellis and Hoover 1990b). The biodiversity was as high as on the reference stations. However, the shoreline which had been subjected to contaminating leachate remained impoverished (Dunn *et al.* 1992). The results showed that the tailings successional biodiversity was different from the biodiversity in a nearby reference area without tailings. This is to be expected. Mud bottom biodiversity is dependent on settlement of drifting larvae, and varies enormously with the relative success of brood stocks from place to place, and from time to time. The variability is also due in part to the first colonizers determining, to some extent, those which can settle later (Matthews *et al.* 1996). Any mud bottom ecosystem carries biological indicators of its past; and the organisms living on tailings capped by subsequent river silt deposits will reflect the species from which the deposits colonized.

Mud bottom successional stages and climax (equilibrium) communities can vary enormously between places and from year to year.

The significance of this case history for submarine tailings placement is that the ARD-generating tailings which deposited deep in the sea were colonized by abundant sea-life. In contrast, where the ARD-generating tailings, and acid waste water, dispersed alongshore, the resultant marine contamination was not neutralized, and has persisted for many years. The geochemical studies conducted on the deep-water tailings illustrated virtually no metal efflux from the tailings material (Drysdale and Pedersen 1992). The metal loadings were covered by natural sediments from the Squamish River system. Also, although there was no seabed area specifically targeted to receive the tailings, they were effectively contained by the fjord topography of an enclosed basin.

9.2.2.2
Atlas Copper Mine (The Philippines)

The operators of Atlas Copper Mine on Cebu Island were the first to design a submarine tailings disposal system from which the impacts have had some documentation. The system engineered was a simple pipeline, extending 500 m by bridge offshore out over 30 m depth (Ellis *et al.* 1995a). At the end, the pipe was shaped vertically down into the water, discharging tailings at 10 m water depth to disperse with the tides and cur-

Fig. 9.2. The mill site at Britannia Copper Mine 20 years after closure in 1974. The mill, which was built on the hillside, is shown *left*. The stream which received tailings and effluent is discharging to the sea showing how the mine's effluent dispersed alongshore

rents in unpredicted ways. The concept that a specific area could be targeted to receive and contain the tailings was not a vision of the times.

The system was developed in 1971 as the previously river-discharged tailings were making the estuary un-navigable for marine traffic including the mine's concentrate loading vessels. Tailings are no longer discharged as the mine closed in 1988.

The Philippines government is currently reviewing the concept of submarine tailings disposal in the context of its new Mining Act. The Atlas site potentially can provide good documentation for the regulatory authorities on the manner, extent and rate at which a tropical seabed covered by tailings has recovered its biodiversity and supported fisheries.

9.2.2.3
Jordan River Copper Mine (Canada)

The Jordan River Mine (Fig. 9.3), which re-opened during 1972–1974 under the British Columbia provincial Pollution Control Act 1969, is an example of early regulations that did not manage to produce a successful submarine tailings placement system (Ellis *et al.* 1995a). The site was a high energy wave-swept coast: one of Canada's few surfing beaches was only one kilometer away to the west. A pipeline was permitted to discharge at 20 m depth about 1 km offshore. The pipeline broke repeatedly on this high energy shore, so that tailings were dispersed alongshore, to the point that a metal placer deposit formed at a natural deposition site some 2–6 km to the east (Kilby 1991). The discharge permit was issued on the basis that the tailings would deposit in a 30 m nearshore trough, but the pipeline system designed and built did not attain this target.

Fig. 9.3. At Jordan River Mine, the tailings line broke repeatedly and discharged tailings along shore. The oceanographic conditions and the pipeline design causing this are now known and can be avoided elsewhere

Some metal contamination of intertidal organisms that developed, at the site and downcurrent, was maintained for at least 5 years. However, the 30–50 m of shoreline with smothered habitat being exposed to wave action returned to normal beach morphology within 6 months of closure. The intertidal biodiversity was restored (Ellis and Popham 1983).

9.2.2.4

Island Copper Mine (Canada)

For 25 years, Island Copper Mine (Fig. 9.4) discharged its potentially acid-generating tailings to a clearly defined target basin of 100-to 200-m depth within a fjord enclosed by a 17 m sill (Ellis *et al.* 1995b). The bulk of the tailings remained within the basin, and the seabed recovered much of its biodiversity within 1–2 years wherever the tailings deposits stabilized (Ellis and Hoover 1990a). Five particular benthic species regularly

Fig. 9.4. At the Island Copper Mine, tailings were discharged for 25 years deep into adjacent fjord, where they deposited in the targeted area, with little surface turbidity. The pit is shown here, surrounded by waste rock under reclamation. The waste rock placed on the fjord side of the pit was to allow extension of mining right to the edge of the fjord. Subsequently, shallow embayments were cut into the dump edge to act as nursery areas for young salmon on their seaward migration from neighboring spawning streams. (From Island Copper Mine)

appeared within 1–2 years. They subsequently maintained themselves, and can be considered bioindicators of a sustainable ecological succession.

Some tailings were resuspended and upwelled away from the basin, mostly depositing at rates about 1 cm year^{-1} or less. At this slow deposition rate the benthos was so little affected that the biodiversity could not be distinguished statistically from that at unaffected sites. Less than 0.3% of tailings escaped from the inner fjord, and without biological consequences. A commercial crab fishery has been maintained throughout the life of the mine over and around the tailings deposits; and salmon continue to migrate to spawning streams and a hatchery in the fjords.

Acid generation has not occurred in the tailings, even on the thickest tailings beds near the outfall. Bioaccumulation studies have clearly identified trace metal elevations only as copper present at twice natural levels on the concentrate-loading dock. The copper is presumably derived at the dock from fugitive concentrate dust during vessel loading, and not from the tailings.

The 0.3% tailings escaping from the inner fjord were resuspended and upwelled from the targeted area, by periodic tidal jets scouring the bottom of the 150 m deep pit, and then deflected by underwater cliffs back to the surface. The mine's environmental consultants during the development phase 1969–1971 did not have sufficient information to recognize this phenomenon, therefore, they were not able to predict the tailings escape. Developments in oceanographic research over the past 20 years, plus Island Copper's experience of an extensive environmental monitoring program, now ensure that appropriately designed data gathering can be implemented to properly target a tailings receiving area. An environmental impact assessment can predict where tailings re-suspension and upwelling may occur, at what intensities, and whether it will have an environmental impact.

The target area was achieved at Island Copper Mine by means of an outfall design which is now a standard for submarine tailings placement (Fig. 9.5). Where the tailings pipeline reached the sea edge, it discharged to a tank which functioned for de-aeration of the tailings slurry and to mix the slurry with seawater. A further pipeline descended below the tank to 50 m depth. Both de-aeration and seawater mixing increased the density of the slurry, and made it more coherent, with reduced dispersion of particles from the density current as it flowed over the surface of the seabed to the targeted deposition area. Seismic surveys showed that the tailings slurry normally flowed as a density current (Hay 1982).

The Island Copper Mine case history is significant in that all stages from development, through operations to final closure have been under regulation. All stages have been well assessed and well documented. However, in addition, because of the extensive database the mine has had opportunity to request and obtain modified permits for changes in mine procedures and throughput based on the assessment results. Island Copper shows that well-monitored systems can document environmental impact, can introduce changes for further mitigation if needed, and can justify changes in the regulatory controls.

9.2.2.5

Kitsault Molybdenum Mine (Canada)

The Kitsault Molybdenum Mine re-opened under the new federal mining effluent regulations in 1981 (Pedersen *et al.* 1995). It generated considerable social controversy as the environmental success of the Island Copper Mine (opening 1971) had not yet been clearly documented. Kitsault used the Island Copper outfall prototype with de-aeration/seawater-mix chamber, also discharged at 50 m depth.

Tailings flowed as a slurry density current to depth and remained within the fjord basin where targeted. Newly developed sensitive oceanographic instrumentation detected slightly raised turbidity levels at the discharge depth of 50 m. These were taken to be eddies of tailings separating at the discharge point. Hence they appeared to contravene the new regulations even though there was some uncertainty at the time whether the turbidity field was a natural river plume or not.

The mine closed in 1982 after 18 months of operation, but it has been subjected to ecological recovery studies by the federal government (Burd 1997). Essentially, the deep smothered benthos recovered to a moderate biodiversity within 1–2 years (as at the Island Copper Mine wherever tailings stabilized), but the biodiversity was different from the reference stations; similar to the Britannia Mine (Ellis and Hoover 1990b). The latest government and university studies (Macdonald and O'Brien 1996; Odhiambo *et al.* 1996) at the site have confirmed that tailings did remain within the targeted area of the fjord basin. In addition, even if the raised water-column turbidity at 50 m depth was due to tailings rather than a natural river plume they did not disperse and settle beyond the targeted area.

The importance of this case history is that it confirms (1) the 1-to 2-year rate of recovery to a moderate sustainable successional biodiversity like the one at the Island Copper mine; and (2) the species present will not be identical to those at nearby refer-

Fig. 9.5. The tailings outfall at Island Copper Mine. De-aeration/seawater-mix tank is shown in *foreground*. The tailings input line to the tank is visible on the shoreline to *left* at point of descent underwater. Not shown is the tank output line which descends to 50-m depth. The original outfall, with input by open flow, is shown *right*, and remained operational as a standby

ence stations similar to the findings at the Britannia mine. Also, this case shows that the development of new regulations and new assessment technologies requires cautious and extended use. Preliminary observations e.g. the turbidity detected at Kitsault by new instruments after the discharge started, may be misleading.

9.2.2.6
Quartz Hill Molybdenum Prospect, Alaska (USA)

Quartz Hill was developed in the USA under the well-known National Environmental Protection Act 1972, which appears to prohibit submarine tailings placement. The mine never opened, but the extent and intensity of environmental impact assessment and prediction are unparalleled. Of the 100 million US$ spent on the development, about 30 million US$ were spent on environmental studies (Hesse and Ellis 1995).

The environmental consultants developed sophisticated near-field and far-field mathematical models to predict underwater flow of the tailings slurry in two fjords with potential as deposition targets. The models were based on data from Island Copper Mine. These are prototype numerical models which have been used for similar predictions at other sites, including Misima where subsequent assessments confirmed their validity.

The mine progressed towards approval by state and federal agencies, and eventual permitting, in spite of repeated challenges from opponents. The US Environmental Protection Agency (EPA) designated a preferred target area, located in the remotes of the two fjords. In 1991, the developers withdrew their application and sold the property. It remains undeveloped.

9.2.2.7
Misima Gold and Silver Mine (Papua New Guinea)

The Misima minesite in Papua New Guinea (Fig. 9.6) was re-developed in 1988 on a larger scale than the previous underground workings (Jones and Ellis 1995). The novel deep placement system developed has achieved its environmental targets, including deposition of the tailings in a 1000-to 1500 m-deep basin nearshore. It is the prototype adaptation of the Island Copper system to tropical seas with coral reefs near the surface. A further novelty is that the outfall has been located in the mine's vessel docking area.

The system at Misima is also a significant development for tailings placement on an exposed coast. The depth of discharge from the outfall pipe (112 m) is below the surface mixing layers, as well as below the euphotic zone of biological productivity. This ensures that the tailings remain in deep water from which they cannot be upwelled under normal oceanographic conditions.

Assessment of biodiversity recovery at Misima with the very deep tailings deposition is challenging, since large oceanographic vessels are required for sampling with heavy equipment (Ellis and Ellis 1994). As a result, at Misima, the biodiversity recovery assessments of the deep tailings beds were started with bacterial and mieofaunal sampling using corers. The results show that an apparently normal micro-organism biodiversity is present (NSR 1997), implying that larger organisms, the benthos and their predators (fish and crustacean shellfish) are also present.

The widespread existence of micro-organisms on tailings suggests that either they colonize rapidly or that they can sustain themselves under slow deposition of the tailings (as at Island Copper Mine). There is an implicit suggestion here that after discharge below a surface mixing layer, it could be desirable to disperse tailings (provided they are inert in the sea) over as wide an area as possible. The intent would be to reduce the rate of deposition to a level that would be tolerable by many of the organisms. To some extent this occurs at Misima, where thin plumes of tailings fines separate out at deep pycnoclines, dispersing down current and settling at very slow rates (NSR 1997).

In summary, Misima is a mine with a submarine tailings placement system for a tropical, open coast with coral reefs. It has introduced several mitigative innovations in submarine tailings placement and can be used as an informative example for other mines in the tropics, and on open, wave-exposed coasts.

9.2.2.8
Re-opening the AJ Gold Mine, Alaska (USA)

The AJ Mine was under re-development in 1996, when the US EPA published in the Federal Register (EPA 1996) a note of a proposed regulation change which would allow placement of tailings deep in the sea at that site (see Sect. 9.1.3). The mine was previously denied a permit for placement of its tailings on-land due to risk of collapse in this

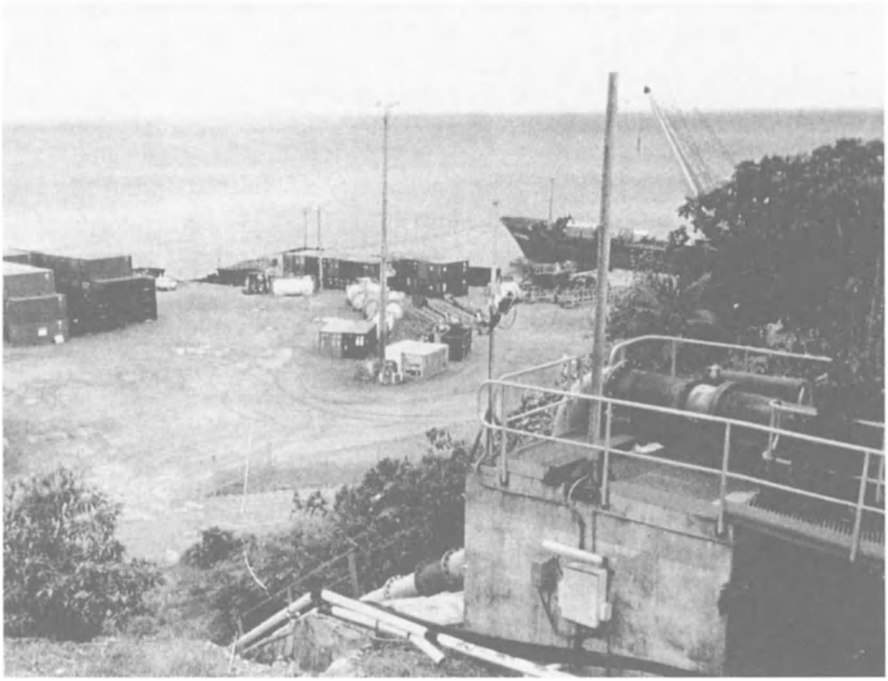

Fig. 9.6. At the Misima Gold and Silver Mine (Papua New Guinea), the tailings line is embedded in the docking area, with the de-aeration/seawater-mix tank at the seaward edge (between the two stacks of containers)

earthquake zone and inability to achieve US EPA effluent standards due to the very wet climate. The only options available were submarine placement or no mine. The re-development plan was extended to consider submarine tailings placement, but the proposed redevelopment has now been abandoned.

There was considerable socio-economic controversy about the re-opening (e.g. Torok and Lindsay 1996) even though old tailings dumps alongshore appear to have a normal intertidal biodiversity about 50 years after the last closure (D.V. Ellis, pers. observation). There are heavily fished salmon runs through the inlet where the tailings would be discharged, and many commercial fishermen were concerned that the stocks would be impacted.

9.3
Screening Criteria for Submarine Tailings Placement (STP)

Information from the above case histories, and others (Poling and Ellis 1995) have been used to develop a set of screening criteria for STP. A two-step set was first developed (Ellis *et al.* 1994), and this is now expanded to a three-step system (Table 9.3). The steps represent increasing data gathering and expense, hence increasing complexity in decision making.

The intent of the screening system is that Step 1 can be an inexpensive desktop evaluation seeking evidence of a "fatal flaw" for STP. Step 2 places the environmental risks from STP at the site into context with the risks from other tailings placement options there. Step 2 can be repeated whenever new information becomes available. Step 3 requires detailed environmental data gathering from which the definitive decision to adopt or reject STP can be made. A set of guidelines for implementing the screening criteria has been developed (Ellis and Connolly 1996). These have also been presented in workshops to industry and regulatory agencies (D.V. Ellis, pers. comment).

9.4
Conclusions

There is now a sufficient number of well documented cases of tailings placement underwater to establish that this option for tailings management at some sites can elimi-

Table 9.3. Screening criteria for determining applicability of submarine tailings placement at coastal and island mines (extended from Ellis *et al.* 1994 by the authors)

Step 1	– Preliminary physical assessment – Simple desktop evaluation of maps and charts can show whether land and sea distances and topography are suitable for a pipeline between mill and offshore waters
Step 2	– Preliminary risk assessment – Evaluation of comparative on-land and submarine environmental risks can be based on physical parameters such as earthquake and typhoon risk, and socio-economic importance to subsistence, artisanal and other commercial users
Step 3	– Definitive risk assessment – Intensive environmental and socio-economic data gathering and evaluations are needed over a relevant time period such as 1 or more years to provide predictions of impacts, and to determine potential mitigative changes in mine-mill conceptual design

nate or reduce the environmental risks from other options. In cases where the benefits have not been reached, the causes have been shown at some sites, and can be avoided elsewhere.

The key conclusions from the accumulated case histories are:

- The Canadian MEND program has established experimentally and in principle that underwater placement of tailings reduces the risk of ARD.
- Two Canadian lakes, Anderson and Buttle Lakes, have provided operating-site data confirming the MEND program's conclusion of reduced metal efflux from tailings placed underwater rather than on-land.
- Controlled underwater lake placement is currently scheduled for the Voisey's Bay nickel development in Canada.
- At Island Copper Mine, ARD in the mine tailings and subsequent contamination of the surrounding ecosystem, has been prevented by placing the tailings at 100–150 m in a fjord. The case history provides substantial operating-site support for the MEND program conclusion.
- At Island Copper Mine, bioaccumulation of trace metals has been limited to mussels on the concentrate loading dock; and is not related to the tailings.
- At Island Copper Mine, the tailings after deposition in the sea meet the 1996 requirement of the London Dumping Convention for marine discharge: they are "inert, inorganic geological materials".
- Biodiversity following deposition of inert tailings on the seabed can recover to sustainable levels of ecological succession within 1–2 years after the tailings stabilize through colonization by a range of opportunist species. This has been shown at both Island Copper Mine and Kitsault Molybdenum Mine.
- Where there is recovery to a sustainable biodiversity, the actual species present may differ considerably from nearby unaffected areas, which in turn will differ from each other. This is clearly shown at Island Copper Mine, Kitsault Molybdenum Mine and Britannia Copper Mine.
- At Island Copper Mine, Kitsault Molybdenum Mine and the Misima Gold and Silver Mine the outfall location and design placed all, or almost all, the tailings in the targeted deposition areas. At Britannia Copper Mine the tailings were fortuitously constrained to an inner fjord basin.
- At Island Copper Mine there have been low levels of tailings deposition outside the targeted area, and of tailings resuspension to the water column. These have had no detectable effect on biodiversity of the seabed, nor the biological productivity of the water column.
- At Island Copper Mine a crab fishery in the fjord receiving the tailings was sustained during the mine's 25-year existence, and stocks of Pacific salmon were maintained.
- New regulations and new instrumentation for monitoring environmental conditions need careful and continuing appraisal. At the Kitsault Mine some new instrumentation produced results which appeared to contravene a new set of regulations even though subsequent assessments showed that the STP system achieved its placement target.
- Misima Gold and Silver Mine in Papua New Guinea has placed its tailings at 1000-to 1500 m-depth on a wave exposed coast, without environmental impact on nearby

coral reefs and local fisheries. It has achieved this by discharging below a 100 m-deep surface mixing zone, so that tailings cannot be upwelled to surface under the prevailing oceanographic conditions.

- At Misima Gold and Silver Mine the targeted deposition area is so deep (1000–1500 m) that biodiversity assessment has been limited to bacteria and mieofauna. The micro-organisms on the tailings cannot be distinguished from that off the tailings.
- Results of the deep tailings placement at Misima Gold and Silver Mine suggest that after the descending tailings density current has flowed through the base of the surface mixing layer, dispersal of fines at deeper pycnoclines has some environmental advantage. Dispersal and subsequent slow settling will reduce deposition rates to levels which may be tolerable by the resident fauna. This of course is dependent on the tailings being inert. Rapid deposition in these cases means covering the benthos at rates too rapidly for resident organisms to burrow through or avoid.
- There are sufficient cases of shoreline and shallow-water tailings placement in the sea demonstrating environmental degradation by contamination as well as by smothering, that the expected risks at any coastal or island site from this option must be very carefully appraised. In principle the option of shoreline or shallow disposal to the sea, is not recommended.
- At three sites, where submarine disposal was considered but not implemented, Quartz Hill and the AJ Mine in Alaska, USA, and the Voisey's Bay nickel development in Canada, the process of developing a marine disposal system is informative for other sites. In particular the tailings near-field and far-field mathematical dispersal models developed at Quartz Hill have a general utility. The three cases illustrate current requirements for marine tailings placement in North America (the USA and Canada).
- A three-step set of screening criteria has been developed for appraising STP. It allows progressive screening based on increasing accumulation of information.

9.5
Recommendations

- At any proposed mine-site, a range of tailings management options should be considered, and the risks for each option identified. Risk analysis methodologies should be used to give estimates of the probability that the risks will happen over both the short and long term, e.g. collapse of tailings impoundments during earthquakes, major storms and flood events, fishery contamination and losses.
- A decision whether to use the underwater tailings placement option or not at a mine-site should be based on detailed scientific investigations of the physics, chemistry and biology of the receiving area, whether to a flooded on-land impoundment, to a lake or to the sea.
- Socio-economic effects and perceptions should be predicted to help understand the various risks to, and potential impacts on, local inhabitants and on the nation as a whole.
- Regulations controlling tailings management should be reviewed and updated at intervals to recognize and incorporate new relevant information obtained by operating experience at mine sites around the world.

Acknowledgments

We thank the many people who comprise our two networks of colleagues in the mining industry, government agencies and universities. In writing this chapter we have been able to bring much information together from many sources. We thank you all.

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Chapter 10

Reduction of Mercury Emissions from Gold Mining Activities and Remedial Procedures for Polluted Sites

Marcello M. Veiga · John A. Meech

10.1
Introduction

The modern gold rush in developing countries began in the 1980s. It has involved millions of people who have become artisanal miners in order to escape complete social marginalization. In Latin America today, over one million people are involved directly with artisanal gold mining, producing between 115 and 190 tons of gold (Au) and, as a consequence, emitting over 200 tons of mercury (Hg) per year to the environment (Veiga 1997). Including Africa and Asia, the number of artisanal gold miners may be as high as 6 million. Most developing countries face enormous social and environmental problems derived from poor mining practices when considered together with the lack of economic alternatives.

Amalgamation is a preferred method used by artisanal miners to extract fine gold. For centuries, it has been known that mercury is an avid collector for gold forming an amalgam-alloy paste that can be easily separated from other solid material in an ore. There are many methods and devices used to contact mercury with an ore (Lins *et al.* 1994), each of which have their own specific efficiency, effectiveness and impact on the environment. Mercury is relatively inexpensive. Although it is sold in Brazilian "garimpos" (artisanal mining sites) at five times the international price, mercury has a cost equivalent to 0.012 g of gold per ton of ore processed (Veiga and Fernandes 1990).

A variety of mining and amalgamation methods are used by artisanal miners. Together with the fate of contaminated tailings and Au–Hg separation procedures, these methods define the extent of mercury losses from a specific site. If concentrates are amalgamated, the main emission source derives from burning amalgam in open pans. This operation produces a gold doré that contains 2 to 5% residual mercury. When the doré is melted at gold shops located in urban areas, further release of mercury vapors takes place (Veiga and Meech 1995a).

When gravity concentrates are amalgamated, the mineral portion is separated from amalgam by panning, forming a tailing that is usually dumped into a stream to create a "hot spot". Panning is done in water-boxes, in pools excavated in the ground, or at creek margins. Excess mercury is removed by squeezing it through a piece of fabric. The excess is generally recycled but some is lost to the tailing.

Amalgam usually contains about 60% gold and so must be retorted or burned in an open pan. When retorts are not used, atmospheric emissions represent as much as 50% of the total mercury introduced into the process. However, when amalgamation is conducted properly and retorts are used, losses are very low – as little as 0.05% (Farid *et al.* 1991). This recycling practice suggests one of the first points of attack to reduce the burden on the environment.

Mercury has been detected in the blood of miners, gold dealers and citizens living near emission sources at levels known to produce symptoms of mercurialism (Malm *et al.* 1990; GEDEBAM 1992). Mercury is rapidly taken up by aquatic species if it becomes oxidized and then is transformed into methylmercury (CH_3Hg^+) – the most poisonous form of mercury for aquatic biota and man. The high organic acid content of Amazonian sediments and waters favors oxidation of metallic mercury dumped by miners or precipitated from the atmosphere (Tromans *et al.* 1996). Such soluble Hg-complexes are easily transformed into methylmercury. Riparian communities in which fish is the staple diet are showing extremely high levels of mercury in blood (GEDEBAM 1992) and in hair (Martinelli *et al.* 1988; Pfeiffer *et al.* 1991; Barbosa *et al.* 1995; Malm *et al.* 1995; Boischio and Henshel 1996).
The main questions related to artisanal mining raised by many experts are:

- Can developing countries accommodate artisanal mining in a controlled and organized fashion?
- How can an appropriate legal framework be provided?
- Can this activity help contribute to reducing poverty by imposing order on the present chaos?
- What techno-economic support is needed to develop processes with less environmental impact?

Mercury pollution is one of the most serious environmental problems, but it is just the tip of the iceberg of the environmental, social and economic problems associated with artisanal gold mining activities. This chapter is intended to provide technical background for decision-makers coping with high levels of mercury emissions. Several possible solutions are examined and the most practical and feasible measures are highlighted. Some of the procedures described are being considered for adoption by international organizations such as the United Nations Industrial Development Organization (UNIDO).

10.2
Attempts to Control Mercury Use

Protests by certain international environmental groups have caused some governments to attempt to enforce laws on artisanal miners. In most cases however, only "lip service" has been paid to the problem since it is difficult to provide the social services and economic stimulation needed to replace the positive benefits (money and food) generated by this informal economy. For developing countries, artisanal mining represents an embarrassment in direct contrast to the desires of the dominant society to pursue modernities and efficiencies. The activity is labor-intensive accommodating unskilled workers and is relatively immune to the impoverishing ravages of economic cycles (Brooks 1993).

However, following a decade of these "rushes", the true effect on the global environment is now clear. Groups wishing to protect the environment are trying to impede the progress of artisanal mining. Deprived of strong support from national and regional governments, artisanal mining has become a marginal activity. Yet, despite or perhaps, because of this negative position in society, informal miners still predominate and the destruction of the environment continues.

In the Amazon region, concern for the environment has become the focus of harsh criticism of mining activities. For many people, Amazon is regarded as "the lungs of the world", an untouchable sanctuary of nature. Serious ecologists believe that living organisms have their own rights and so humanity must adjust its behavior to accommodate these interests (Merchant 1992). However, it is important to understand that among the five million species that inhabit the Amazon, more than 25 million people live in poorly organized communities. They mainly use inadequate farming methods and eke out their survival, without technical assistance, by being employed on several major agricultural projects. Clearly, these people have rights as well.

A centralization of economic and political power into the hands of a small elite has occurred in Amazonia. Because of land-ownership inequities, informal mining has been an important escape valve for the predictable outcome of an agrarian crisis (Hecht and Cockburn 1990). It must be appreciated that the conflict over artisanal mining is not between humanity and nature, but rather, between humans at the bottom of society's hierarchy and those at higher levels.

Often, the media plays a significant role in creating fantasies about mining and mercury pollution that causes significant misunderstanding by members of society. In the Disney cartoon Pocahontas, North American artisanal miners (colonizers) are depicted as villains who destroyed trees, lands and native cultures. On January 16, 1997, the North American TV series "Tarzan" presented an episode in which an artisanal miner using a small sluice box killed more than 100 animals, including fish, panthers, monkeys and birds by discharging mercury into a river. Of course, Tarzan stopped this ecocide and saved the rain forest from further peril.

In 1995 a Venezuelan newspaper published stories claiming metallic mercury was dripping out of water taps in remote jungle villages. In 1987 a Brazilian union leader of artisanal miners drank a glass of metallic mercury in front of TV cameras to show that this liquid metal is inoffensive. Mercury was an effective weapon to use against the miners but the attackers could not handle spectacles such as this one because of their poor understanding about the technical issues of mercury poisoning.

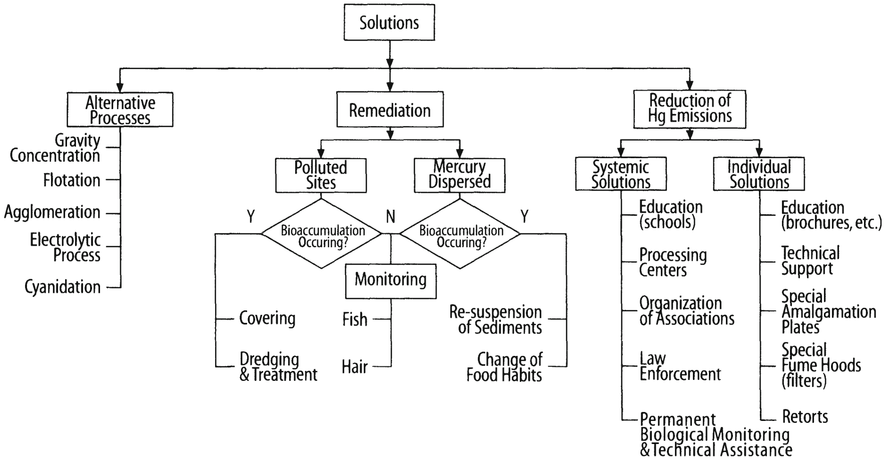


Fig. 10.1. Solutions for mercury pollution from artisanal gold mining

In all press reports, it is clear that the truth about mercury is a "taboo" that cannot be discussed by ordinary people; its understanding is reserved only for a privileged elite of "specialists". The press frequently looks for sensationalism instead of providing reliable advice. Fantasies, panic and the advancement of political interests result from the misinformation provided by the press. Affected communities (miners and fish consumers) are ignored. Future generations will inherit a legacy shaped by the continuing discharge of mercury and the failure to address the issue directly. Education is a prerequisite for a long-term solution to mercury emissions – a process in which useful information is provided to miners, riparian communities, health-care workers, mines inspectors, etc. Solutions must be provided rapidly to avoid an epidemic. The success of implementation requires acceptance by the offending members of these communities.

10.3
Proposed Solutions

As mercury pollution has many implications, a multi-disciplinary approach is needed to address the problem. Legal measures to control informal mining and mercury emissions can help, but to date have not worked on their own. Mercury is used illegally in almost all Latin American countries. According to Priester (1995), the situation requires solutions that combine protection of the environment and workers with an evaluation of the impact of the solution on the economics of the mining venture. An illegal miner will not pay out "a single dollar unless two are returned".

Solutions have received little attention from researchers and governments. Studies are the preferred process. Invariably, the "blind" solution has been "Stop the use of mercury"; but this noble goal requires an integrated approach. Three main actions are delineated in Fig. 10.1: (1) search for alternative processes to the use of amalgamation; (2) implement remedial procedures for polluted sites; and (3) apply measures to reduce mercury emissions.

10.3.1
Alternative Processes

Amalgamation is applied in "garimpos" for two purposes: to recover fine gold from an ore and to extract the gold to a very-high grade product which can be stored in one's pocket. The first application is the one which must be stopped. This will avoid mercury emissions directly to the aquatic environment with tailings. To stop the second application is extremely unlikely – garimpeiros will always want to hold their gold in their hands until they can have faith in the security of the community which they inhabit. So, our goal should be to encourage use of amalgamation only for gravity concentrates and to see that it is conducted in a sensible and controlled fashion.

Amalgamation of the total ore is attractive since the gravity process is easily adaptable to amalgamation. Mercury can be added directly to the primary extraction operations without requiring a second processing stage. To eliminate mercury use with the total ore, we must find alternative methods to recover fine gold. In the initial stages of a "rush", fine gold is rarely an issue – the attraction of a site generally is because of the presence of "easy-to-recover" coarse gold nuggets. But as the ore becomes depleted, the miners will turn to amalgamation to maintain their gold production by recovering fine gold. Alternatively, some may begin reprocessing old tailing or waste dumps using mercury for fine gold pick-up.

Attempts to introduce new gravity concentration equipment, such as shaking tables, spirals, automatic panners, etc., to eliminate amalgamation, have not been very successful. Mercury use has declined, but never eliminated. The principle of these methods is based on using gravity to clean an initial (rougher) concentrate to obtain a rich final concentrate for smelting. In one processing center in Venezuela, Gemini tables process concentrates obtained from sluice boxes located on-board barges. Instead of amalgamation, the tables work as cleaners to produce a concentrate that is smelted and sold to the Bank. A gold-rich middling product is produced which must be amalgamated to extract very fine gold (Veiga 1996), but this material is very low in weight relative to the ore and so a significant decrease in mercury consumption results. When combined with the fact that amalgamation is performed in a centralized facility, a very significant drop in emissions takes place.

Centrifuges, such as the Falcon or Knelson Concentrators, have potential as primary gravity concentrators for fine gold as well as for cleaning ordinary gravity concentrates. A new model of the Falcon Super Bowl uses a fluidized bed spinning-bowl that can process up to 60 tons of solids per hour, applying a centrifugal force of 200 G. Concentrates can reach grades above 20 000 g Au ton^{-1} in two stages (rougher and cleaner) which can be directly smelted, potentially avoiding an amalgamation step. Knelson Concentrators has developed a continuous high-G force unit as well.

Similar centrifuges have been manufactured in Brazil (MacKnelson) and are being used by many artisanal miners. Despite being rough copies of the Knelson Concentrators, they do provide improvement in gold recovery and a reduction in mercury use.

Froth flotation has been tried in a few South American artisanal operations to concentrate fine gold. But even with concentrate grades of Au as high as 3000 g ton^{-1}, the product still requires amalgamation or cyanidation – it is not easy to directly smelt such concentrates. Lins *et al.* (1994) have studied flotation of gold contained in gravity concentrates. Using xanthate collectors, concentrates were upgraded from 13 g Au ton^{-1} to 3000 g Au ton^{-1} at 82% recovery. Flotation of coarse gold (>0.43 mm) is ineffective so a two-stage process is necessary. Flotation cannot compete economically with amalgamation as used by artisanal miners.

Experiments with *coal-oil agglomeration* show some promise. Agglomerates of coal and oil (5 mm) are formed and contacted with a gravity concentrate pulp. Recoveries of 90% are possible. Envi-Tech Inc., Edmonton, Canada, has developed a novel agglomeration process using a proprietary adsorbent. Following 5 to 10 min of intense agitation, the gold-loaded adsorbent is separated by froth flotation achieving 70% recovery. The adsorbent contained 350 g Au ton^{-1} (Envi-Tech 1997).

In another selective adsorption technique attempted by Lins *et al.* (1994), *melted paraffin* was used to collect gold in an acidic media. Despite encouraging results, none of these methods provide a simple, cheap alternative to amalgamation for an unskilled artisanal miner. They may also have other environmental problems as significant as mercury emissions.

A *salt-electrolytic process* to leach gold has been developed by the Center of Mineral Technology (CETEM) in Rio de Janeiro, Brazil, and tested in a pilot plant in the Tapajós region of Brazil (Souza 1991; Sobral and Santos 1995). This process has the potential to replace amalgamation of gravity concentrates. Material with as little Au as 1 $\mu g\ g^{-1}$ is mixed with a sodium chloride solution (1.0 M) which is transformed by electrolysis into a mixture of sodium hypochlorite-chlorate. More than 95% of the gold dissolves within 4 hours and is collected on a graphite cathode. The solution is recycled, minimizing effluent discharge and plastic tanks are used, reducing investment and replacement costs. The main drawback is the need for trained personnel to control operating variables (pH, current density, etc.).

Cyanidation is a process beginning to appear in artisanal mining operations in Andean countries and, to a lesser extent, in Brazil. Cyanidation of ores and flotation concentrates has been adopted widely by numerous organized mining companies. The use of amalgamation has essentially been banned by the establishment of effluent discharge standards that are difficult, if not impossible, to meet when amalgamation is used. Cyanidation has replaced amalgamation as the preferred recovery method in ores that contain extremely fine gold or gold in solid solution. As coarse gold requires long retention time for cyanide leaching, it is generally removed by a gravity process prior to submitting the material to flotation and/or cyanidation.

Despite high gold recoveries, cyanidation techniques require much more skill and investment than simple amalgamation. In such cases, artisanal miners need technical support. A small cyanidation plant can be set up for use by a small mining community, but this is not a general solution for all cases of artisanal mining. Another important issue is the occupational risk for operators. Although part of the residual cyanide is naturally degraded by sunlight and heat, total cyanide destruction requires complex methods, such as the INCO SO_2/air process. Although the environmental impacts of cyanide are usually lower than those of mercury, the consequences of occupational exposure can be rapid and very dramatic. Cyanide does not bioaccumulate like mercury, rather it may kill-off food sources in the food-web (Chap. 5, this Vol.).

The possibility of replacing amalgamation with other processes is remote, but must be pursued. For an artisanal miner, mercury is an easy and efficient way to extract fine gold. When amalgamation is applied to gravity concentrates, more than 90% of the gold is recovered. Alternative processes must be investigated, but no extraordinary breakthrough should be expected.

The only possibility to eliminate wide-spread use of amalgamation is to promote the creation of *Processing Centers* where miners can take their gravity concentrates for processing by specialists using leaching or amalgamation methods in a controlled environment. This approach is viable but will hardly be adopted by all miners dispersed throughout the jungles. Implementation demands a highly organized program combining efforts from governments and miners.

10.3.2
Site Remediation Requirements and Monitoring Programs

The political and technical decision to apply remedial procedures to a polluted site is usually based on an assessment of environmental risk by investigating a site to establish the level of present and future pollution. Most countries have mercury guidelines for fish, water and air but not for sediments. The Environmental Agency of Japan has established 25 $\mu g\ g^{-1}$ as a provisional minimum standard to remove Hg-contaminated sediments (Buffa 1976) while in British Columbia, Canada, the Ministry of Environment uses the standards shown in Table 10.1. These are isolated cases of soil guidelines. Therefore, determination of the mercury level in a sediment is insufficient to support a decision for remediation. So, the critical question is: how polluted must a sediment be before remediation is recommended?

Sediments are witnesses to a contamination process that takes place over many years. In metallic form, mercury is not easily transported from a sediment into the aquatic environment and hence, into the food chain. Its solubility is extremely low and its high specific gravity can keep it in place deep within sediment voids. Unfortunately, metallic mercury may be transformed into its most toxic form, methylmercury (CH_3Hg^+). This species is highly soluble (unlike metallic mercury) and readily assimilated by organisms. Between 70 and 95% of the mercury in fish is in the form of CH_3Hg^+ (Huckabee *et al.* 1979). Methylation is usually a bacteria-mediated process occurring primarily in the top 1–2 cm of sediments. In studies done on Canadian reservoirs, Kelly *et al.* (1994) concluded that the level of total mercury in a sediment is a poor predictor of CH_3Hg^+, since certain environments enhance its production irrespective of the total mercury content.

Some microorganisms actually promote demethylation, i.e. transforming CH_3Hg^+ back into metallic mercury. In a sediment, strains of methylating bacteria live with demethylating microbes and so the ratio of these strains controls the rate of CH_3Hg^+ production (Ramlal *et al.* 1986; Guimarães *et al.* 1995). Since production of CH_3Hg^+ in sediments is a reversible process, the actual rate is often governed by the how fast CH_3Hg^+ is removed into overlying waters (Parks *et al.* 1984).

Metallic mercury emitted by a mining operation must be oxidized to be methylated by bacteria. Oxidation is feasible in the presence of organic acids but requires the availability of dissolved oxygen (Tromans *et al.* 1996). After oxidation, methylmercury is readily produced by either biotic or abiotic processes. Organisms accumulate CH_3Hg^+ so fast that the actual concentration of this compound in water is very low (D'Itri 1990). In fact, mercury often goes undetected in water by standard analytical methods and so, sediment and fish samples are the only accurate alternatives available.

Sediment core analyses have been used to investigate mercury accumulation over time (Lacerda *et al.* 1991). Since mercury has high affinity for organic matter and high mobility in the soil profile, lithogenic factors can mask the data, showing higher mercury concentrations with organic horizons. Comparison of mercury levels in mining

Table 10.1. Guidelines for mercury levels ($\mu g\,g^{-1}$) in soils and sediments, British Columbia, Canada (Ministry of Environment 1989)

Mercury level	Remarks
< 0.1	Usually found in organic soils as a natural background
0.1 – 2	Soil is considered slightly contaminated, but remediation is not required
2 – 10	Sediment is contaminated and requires remediation to levels $<2\ \mu g\,g^{-1}$ if the land is used for residential and recreation purposes. Remediation is not required if the land is used exclusively for commercial or industrial activities
> 10	Significant contamination. For soils containing Hg exceeding this level, all uses of land are restricted pending application of appropriate remedial measures to reduce Hg concentrations to $<10\ \mu g\,g^{-1}$

sites with sediments from other regions of the world can be misleading due to the regional geological and mineralogical differences (Lacerda and Salomons 1997).

To establish the main mercury-bearing components in a sediment, sequential extraction is useful. This technique shows how strongly mercury is bound to suspended particles, giving indirect information about mercury mobility and availability to the water column (Ferreira and Veiga 1995). Unfortunately, the level of pollution is not established although the degree of contamination of the environment from a particular soil may be indicated.

Quite often, monitoring programs are used to understand mercury behavior in sediments. Despite the academic merit of such work, little information is obtained that can assist remedial procedures. A mercury monitoring program of stream sediments is useful to: (1) establish the contamination source; (2) establish liability; and (3) identify highly polluted sites ("hot spots") that need cleaning-up. In the case of active sites, the first two points are already known. As metallic mercury has low mobility in aquatic systems, sediment sampling must focus on the location and size of the "hot spots". Monitoring programs to verify the amount of mercury in a sediment are extremely costly and unlikely to derive reliable results. Emissions can be estimated indirectly by the ratio of gold produced to mercury lost. In regions where concentrates are amalgamated and retorts are not used, this ratio is typically 1. When mercury is spread on the ground, as in hydraulic mining activities, the ratio can be as high as 3.

In summary, remedial procedures are necessary when mercury bioaccumulation is indicated by a monitoring program. However, the only indicator of bioaccumulation is biota, in particular fish, specifically carnivorous fish, preferably those with low mobility, such as black piranha. The risk of bioaccumulation can also be predicted by examining natural variables (sediment type, mercury levels in sediments, conductivity, Eh, pH, water content, etc.) Such data can indicate the presence of environmental conditions that may lead to bioaccumulation. Depending on the nature of emissions, mercury can be dispersed in the sediments or concentrated in hot spots. The different procedures to remediate such sites are delineated in Fig. 10.1.

10.3.3
Highly Polluted Sites ("Hot Spots")

Whenever amalgamation takes place, in a water-box or a river bed, tailings are discharged into the stream creating "hot spots" where the mercury concentration can be hundreds of $\mu g\ g^{-1}$. If mercury droplets are visible in a sediment after panning, the sediment generally contains more than 3 $\mu g\ g^{-1}$. Whether "hot spots" should be dredged or covered is a decision based on an evaluation of bioaccumulation, costs involved in the dredging operation and spoil treatment options.

Covering procedures aim to prevent release of mercury from sediments to the water. The extent of pollution and the hazard potential establish the material to be used as covering. Each case needs an evaluation that includes laboratory experiments. Either inert or reactive (adsorbent) material can be used but the system must be enclosed (Veiga and Meech 1995b). In a river, this technique would be unsatisfactory since erosion during high flows will re-expose the contaminated river bottom.

Covering "hot spots" is a technique in which oxidation of metallic mercury is controlled by adsorption, thus hindering the action of methylating agents. Remedial actions focus on reducing oxygen access to the water-sediment interface and/or adsorbing the oxidized mercury species. This method was used in Minamata Bay to control methylmercury production in sediments. In that case, the mercury discharged was already oxidized in the form of $HgCl_2$.

A series of covering procedures for polluted sediments is suggested for testing. Some of the materials that can be mixed with sand or silt to cover "hot spots" are as follows:

- Laterite crusts: iron oxides adsorb oxidized mercury
- Pyrite (FeS_2): reacts and precipitates oxidized mercury as a sulfide
- Fibers: adsorb oxidized mercury released from sediments
- Rubber scrap (e.g. old tires): adsorb oxidized mercury
- Scrap iron (cementation process): precipitate Hg(II) soluble compounds

The adsorption capacity of lateritic materials such as hydrous ferric oxides (HFO) and manganese oxides is well-known and tropical countries are rich in laterite crusts as weathered-products of iron-yielding minerals. These crusts can be used to cover "hot spots". To our knowledge, no testwork has yet been done with these materials. Testwork is warranted.

Experiments using pyrite to overlay Hg-rich sediments has been performed in laboratory and small-scale field tests in Sweden (Jernelöv and Lann 1973). Formation of HgS occurred – a compound with very low solubility. An area of 1 ha in Lake Gårlången, Sweden, polluted with mercury from municipal wastewater was covered with a 3 cm-layer of mine tailings (sulfide-rich). Methylmercury released from the sediment and fish accumulations were reduced five fold in 3 weeks. Over the years, analyses of mercury in fish from the whole lake indicate a significant decrease in bioaccumulation levels. Pyrite is a common component of tailing in some gold mining activities which work primary quartz vein ores. Such materials should be tested.

A large list of cheap organic fibers such as wool, chicken feather, hair, onion skin, nylon, and sugar cane bagasse, have been studied as mercury adsorbents (Feick *et al.* 1972; Asai *et al.* 1986). The mechanism of adsorption is not yet understood but sulfur-bearing sites are believed to play an important role since desorption is extremely difficult. Laboratory studies have also demonstrated that waste wool and wool/polyester blend fibers remove 90 to 95% of soluble mercury compounds (organic and inorganic) within 24 h of contact (Tratnyek 1972). Due to obvious aesthetic and environmental impact issues, these fibers cannot be applied to large polluted areas. However, since artisanal miners are major consumers of carpet for use in gold sluice concentrators, discarded carpets could be a potential source of material to cover "hot spots". Field tests should be tried.

Over a wide pH range (2 to 13) ground-up automotive tires can remove as much as 99% of mercury from water in 15 min. The excellent adsorptivity of rubber for mercury compounds is due to high contents of sulfur and carbon black (Beszedits 1979). The technique should be tested on a small scale study in some highly polluted site in mining areas. Rubber scrap has been tested in Nicaragua by the EHP-USAID team, but results have not been reported (Edwards and Ault 1996).

Laboratory studies have shown that iron scrap rapidly and efficiently removes Hg(II) and CH_3Hg^+ from water by converting these soluble forms into elemental mercury. When Hg(II) is cemented (reduced and precipitated) with iron, hydrous ferric oxide is formed, which is also an effective precipitator for mercury species. Discarded shredded automobile bodies are suggested as an iron source. Reduction of mercury compounds to metallic mercury (Hg^o) occurs either under aerobic or anaerobic conditions. Methylmercury reduction is more efficient at acidic pH, while the removal of Hg(II) is better at neutral or slightly alkaline pH (EPA 1972). This technique has good potential to be tested in highly polluted sites. Iron scrap dumped into "hot spots" or mixed with sand may contribute to an efficient covering of sediments to control mercury entry into the water column.

Dredging procedures remove the source of contamination from the water system. They are expensive measures and are recommended only when the gold content in the spoil can return part of the costs or when mercury bioaccumulation cannot be controlled by covering. Usually, "hot spots" have high gold content. A separate processing plant is commissioned to extract gold and mercury from dredged material. Dredging can be a definite measure for highly polluted spots where mercury pollution is highly concentrated, but there are three issues regarding environmental impact: (1) dispersal of mercury into streams during the dredging operation; (2) treatment and disposal of the contaminated spoil; and (3) covering of the dredged site.

Aquarium experiments with simulated mechanical dredging indicate that the amount of mercury dispersed in the water column is of the order of 2 to 10% of that removed. The majority of the dispersed mercury is associated (adsorbed) with suspended particles. An effective method to prevent redistribution of mercury is to use a suction dredge in place of a mechanical one but this leads to a high percentage of water in the reclaimed spoil. So, settling ponds and aluminum sulfate are necessary to clarify water before returning to water courses (Feick *et al.* 1972).

In Venezuela, miners while processing alluvial gold from sediments in the Caroni River between 1991 and 1994, extracted about 0.5 g Hg kg^{-1} of concentrate produced by sluice boxes placed on-board their barges (Bermudez *et al.* 1995). Considering that a Caroni River miner averages 40 kg of concentrate per 1400 tons of ore processed in sluice boxes, this gives 20 g Hg per 40-h shift, or about 0.02 g Hg ton^{-1} of riverbed sediment. Assuming over 80 dredges are working in the lower Caroni area, processing about 2 million tons of ore per year, over 40 kg Hg is recovered from the river annually. Since mercury is no longer used on board these dredges, these miners are actually contributing to the removal of mercury from the river.

Whenever possible, a sealing treatment is recommended for a dredged spot, such as covering with inert or adsorbent material to guarantee that the remaining mercury is immobilized at the site. Treatment of dredge spoil is also essential.

10.3.3.1
Treatment of Dredge Spoil Material

When placed in a landfill, a polluted sediment can lose mercury to the water (when oxidation occurs) or to the atmosphere (evaporation). Mercury losses can be prevented by covering the dredged sediments disposed in a landfill with appropriate material. Mercury removal from the dredged material is also possible. Retorting is the only process which provides complete mercury extraction, but this is clearly impractical. However, there may be interest in recovering the gold content from these spoils and hence the associated mercury as well.

Experiments with 80 tons of polluted sediments (6.5 $\mu g\,g^{-1}$ of Hg) dredged from "hot spots" were performed in Poconé, Brazil (Farid *et al.* 1991). Centrifuges recovered 70 to 80% of mercury reaching a grade of 1% Hg (10 000 $\mu g\,g^{-1}$) in the concentrate. Higher extraction is difficult since fine dispersed droplets of metallic Hg and mercury bound to hydrous ferric oxides are not recovered. Gold associated with the spoil was also concentrated in the centrifuges with a recovery of around 60%. Gravity methods can reduce mercury levels significantly in the spoil but since the tailings are still highly contaminated, disposal must follow safe procedures, such as covering with adsorbents.

A new technology has recently been developed in Brazil to remove mercury from contaminated sediments. Manufactured by a Brazilian company, Rio-Sul, special silver-based plates can be used to amalgamate gold avoiding mercury emissions. The plate is made of a special alloy significantly different from the older copper-plate amalgamation process used by artisanal miners. Between 8 and 16 plates are placed in a sluice box so that a cascade effect is obtained when a pulp concentrate is processed. When gold specks or fine particles (as fine as 37 µm) contact the plates, gold is captured and firmly fixed to the plate surface. No mercury loss from the plates has been observed when amalgamating gravity concentrates. When the plates are loaded, the amalgam is removed by scraping the surface with a piece of plastic.

These plates have been tested successfully to remove mercury from contaminated tailings (Veiga *et al.* 1995b). Amalgamation tailings from 35 dredges operating in the Piranga River, state of Minas Gerais, Brazil, were treated with the Rio-Sul plates. About 145 kg of mercury and 880 g of gold were extracted from 91 tons of tailing treated in 13 days. Approximately 90% of the mercury was recovered in two operating cycles and the tailing was then disposed safely into lined pools.

Hydrometallurgical processes to treat dredged material are also being studied. The electrolytic method developed by CETEM, that was previously described as an alternative method to recover gold, can also be applied to Hg-contaminated sediments. The process was tested in two pilot plants in the Poconé and Tapajós regions, Brazil (Souza 1991; Sobral and Santos 1995). In Tapajós, the mercury extraction was above 99% and the final solids contained 60 to 300 μg Hg kg^{-1} (R. Santos, CETEM, Brazil, pers. comm.).

Flotation to remove mercury from dredged sediments has also been studied using synthetic samples. Recovery was as high as 95% (Lima and Araújo 1994), but with actual contaminated material, extraction fell to 30 to 68% (Lins *et al.* 1994). The presence of superfine mercury droplets and mercury adsorbed onto other minerals, such as HFO, explain these low recoveries.

10.3.4
Mercury Dispersed on Sediments

Mercury dispersion over large areas takes place from emissions to the atmosphere from amalgam burning in open pans, deforestation by burning, and from the flooding of land. Dispersed mercury pollution is more difficult to treat but several methods have been attempted.

Procedures to minimize mercury bioaccumulation have been applied in Canada and Sweden where fish from natural and man-made reservoirs have shown increased mercury levels over time (Rudd and Turner 1984; Lindqvist *et al.* 1991). Mercury sources in these countries are industrial emissions from coal combustion, pulp and paper or chlor-alkali effluents. Sometimes the source is unknown. Amelioration procedures such as selenium, liming or intensive fishing are applicable for enclosed environments such as lakes and reservoirs but no short term results have been reported. These techniques could be tested in Hg-polluted mining areas, but cost is a major impediment.

Suspension of sediments is a method studied in some Canadian impoundments (Parks *et al.* 1984). Clayey ferruginous sediments have an enormous capacity to absorb heavy metals in solution, inhibiting mercury uptake by fish (CETEM 1989). Some sediments are ineffective adsorbents and so field trials are necessary to confirm the technical feasibility of this remedial activity. Application of this method is fraught with difficulty since increased water turbidity can have negative impact on aquatic life. As well, mercury adsorbed onto suspended particles may be released in the presence of salts or organic acids in the water column. This procedure is unlikely to be successful in the Amazon.

Changing the diet of riparian people is one practical option which can produce immediate results in regions where mercury is dispersed and bioaccumulation exists. A massive educational campaign is needed and it was done in many developed countries. In James Bay, Canada, most fish from La Grande hydroelectric dam are contaminated with mercury. A brochure was issued for First-Nations people to describe mercury sources, biotransformation, mercury in the human body, dietary recommendations, etc. The booklet also includes a list of recipes in which fish is diluted with vegetables to reduce the amount of methylmercury ingested per meal. This is a difficult measure for many communities in which fish is the main source of food. However, projects are starting in the Brazilian Amazon to introduce concepts of sustainability based on fish- and chicken-farming. This is definitely an interesting option since it also encourages economic activities other than mining.

Avoiding consumption of carnivorous fish is a useful measure for an individual who wishes to avoid mercury contamination. Unfortunately, for most riparians, carnivorous fish are preferred to other species. It should be noted that one can also be contaminated with mercury from non-carnivorous fish. For example, if a person consumes 200 g of fish daily containing 0.2 $\mu g\ g^{-1}$ of Hg (well below the 0.5 $\mu g\ g^{-1}$ limit for human consumption of the World Health Organization) approximately 40 µg of methylmercury is ingested. This is well above the Allowable Daily Intake (ADI) for both Canada and the WHO. The Canadian ADI guideline, exclusively for diets based on fish, is 13 µg per person (CWQG 1987) while WHO has adopted an ADI level of 30 µg CH_3Hg^+ as safe.

The increase in methylmercury levels in aquatic biota in artificial impoundments has become a significant problem around the world which has forced many fish-eating people to change their diet. The phenomenon is associated mainly with the amount and quality of organic matter in flooded sediments which affect microbial activity and increase methylation rates (Stokes and Wren 1987; Lucotte *et al.* 1995). The effect was recently identified in the Guri hydroelectric reservoir, Venezuela where about 42% of 219 fish samples showed mercury levels above 0.5 $\mu g\ g^{-1}$ (Veiga 1996). About 90% of the most popular carnivorous fish in the region (*Raphiodon vulpinus*) showed an average mercury level of 2.7 $\mu g\ g^{-1}$ (0.17 to 8.25 $\mu g\ g^{-1}$; 31 samples). The government of Bolivar state, UNIDO and the La Salle Foundation have distributed leaflets to the affected communities showing how much fish can be consumed safely each week. Children and pregnant women are the main target of the campaign and an ADI of 15 µg of mercury was adopted for them.

10.3.5
Reduction of Mercury Emissions

Mercury is a very cheap reagent for extracting gold (296 US$ per oz, New York, December 1997). If more control was exerted on mercury sales, the price would increase (even with smuggling) and the miners would probably stop using mercury in riffled sluices to amalgamate the whole ore and stop spreading mercury on the ground. Some miners still believe that mercury moves through the ground to trap all the gold it comes into contact with. This is incorrect, inefficient and environmentally unacceptable. Miners should be encouraged to use amalgamation for treating concentrates only. We must convince miners to adopt an initial gravity operation without using mercury. Amalgamation is efficient for coarse (>75 µm) and for free or partially liberated gold particles (Wenqian and Poling 1983). Process efficiency depends on the capability of contacting mercury with gold particles. Amalgamation can be properly conducted with negligible levels of emission and occupational exposure if correct instructions are followed and a controlled centralized facility is used.

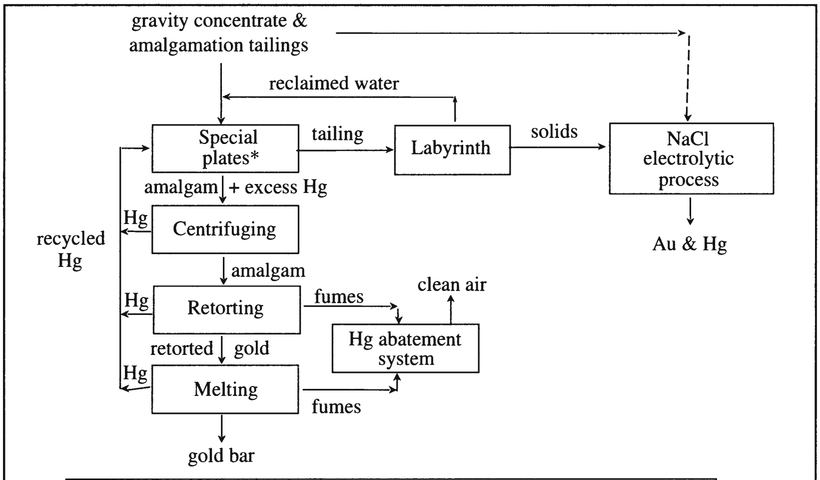
Reducing mercury emissions is a feasible and practical way to address the problem. Figure 10.1 depicts two approaches: systemic solutions and individual solutions. Systemic solutions depend on institutions, agencies and private companies for implementation while individual solutions require each individual miner to adopt suggested measures. Education is a pre-requisite for long-term reduction of emissions regardless of which approach is emphasized. Educational measures consist of a set of recommendations for miners to adopt safe practices for the environment and themselves. These measures may reach the informal mining community by an indirect route – through skilled people in frequent contact, such as priests, equipment suppliers, social assistants, mine inspectors, health-care workers, etc.). Miners must be convinced they are being affected by mercury vapor and that irreversible health problems are being generated for themselves, their neighbors, friends and families. A few brochures explaining the dangers are being distributed by international institutions and by miner associations. A coordinated effort is needed to avoid the gap between artisanal miners and different sectors of society due to inaccurate information.

10.3.5.1
Systemic Solutions

One durable systemic measure is to introduce environmental issues into formal education process in the schools. A miner's children can play a positive role in changing the attitudes of his or her parents with respect to poor amalgamation practices. A very creative solution has been implemented in Venezuela: Amalgamation Centers. This solution can be reproduced in other developing countries. Miners take their gravity concentrates to these Centers to be safely processed by technical operators. In the government-owned Amalgamation Centers in Venezuela, the service is free. In private Centers, miners pay 0.7 US$ per kg of concentrate to be amalgamated. Based on the model of the Carhuachi Center, a remarkable Amalgamation Center in Caroni River, UNIDO and a Venezuelan Non-Governmental Organization (PARECA) have designed a new Processing Center called UNECA – UNit of gold Extraction and Controlled Amalgamation (Veiga 1997). Gold is extracted by amalgamation using special plates or by leaching using the salt-electrolysis process. The electrolysis process eliminates amalgamation while the amalgam plates reduces mercury use to a negligible level. Special retorts and melting furnaces are used under fume hoods with iodine-impregnated charcoal filters (Fig. 10.2).

The UNECA-type Processing Center is suitable for installation in a mining village or any central location in a gold-mining region to facilitate transportation of gravity concentrates. Gold recovery is actually improved and exposure of operators to mercury is insignificant. For a miner who takes his concentrate to a Processing Center, there is the added benefit of reducing costs in his own processing plant. These Centers also play an important role in bringing information about mercurialism caused by mercury vapor or contaminated fish ingestion. Miners can be given brochures and additional instructions while they wait for the processing of their concentrates. The Centers can provide advice for miners on how to improve their production. They can become a meeting place for purposes of education and organization.

The concept of the UNECA Processing Center is similar to the Mining Center set up in 1989 in Zimbabwe (Shamva Mining Center established by IT[1]. This facility provides access for about 200 miners to a central mill. Although successful, this method is not quite as flexible as the Venezuelan Processing Centers. In the UNECA Centers, miners can continue their mining and processing activities on their own without interference. Only concentrates are processed in a central place. The Centers can be commissioned quickly with full support of environmental agencies and Mining Departments. Small-Miner Associations can be encouraged to open their own Processing Centers.



NOTE: * amalgamation can be replaced with NaCl electrolytic process or other leaching process

Fig. 10.2. Steps in a UNECA Processing Center

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[1] Intermediate Technology (IT) is a UK-based international development agency founded in the 1960s to enable poor people in three continents to develop and use skills and technologies which give them more control over their lives and to contribute to the sustainable development of their communities.

Another important systemic measure involves organization of artisanal mining activities. The creation of Small Miner Associations or Unions is an important step in organizing mining activities. They can become an effective channel for introducing clean techniques.

Law enforcement is an important element in controlling mercury sales but there are other inspection requirements as well. Miners must discontinue mercury use in their sluices and the spreading of mercury on the ground in hydraulic monitoring operations. The aim must be to compel miners to consider only amalgamating concentrates. This reduces emissions and gives an opportunity to introduce new techniques in the future, such as the Processing Centers.

Policing must also be applied to those gold dealers who melt gold and release mercury into the urban atmosphere. The use of small scrubbers or filters must be enforced. These dealers are usually rich companies which can easily afford to introduce safety equipment. Legal control of miners burning mercury in open pans is more difficult since this operation is done rapidly and miners often move about from area to area. Education and the creation of Processing Centers is the only sensible way to end these deadly practices.

Permanent biological monitoring and technical assistance should also be highlighted. Monitoring programs have been the focus of environmental agencies and researchers for some time now. They help to establish the bioaccumulation levels in a region if fish and human hair samples are analyzed. Such programs are usually expensive since chemical laboratories and specialized personnel are needed to accurately determine the mercury levels. Only rarely are the data and results communicated back to the population being studied.

A computer program (HgEx) has been developed to help transfer knowledge and allow rapid diagnosis of bioaccumulation risk even when biota samples are unavailable (Veiga and Meech 1992; Veiga and Meech 1995a). Non-technical users such as health-care workers, environmental and mining inspectors, miners, biologists, and local people can have a picture of the intoxication level of individuals subjected to mercury exposure or those who consume fish as their main diet. The software is complementary to permanent monitoring programs, since it can help determine the need for such studies and can provide an indication of the magnitude of the problem at its initial stages.

Technical assistance comprises measures to bring practical solutions to miners to improve their productivity and reduce mercury emissions. In Latin America this approach has been adopted by NGOs and some government agencies. It must become a permanent policy of Mining Departments which need trained people to deal with artisanal miners in a trustworthy manner. Miners are usually suspicious of government representatives, even when they bring profitable solutions. Engineers, technicians, or consultants give suggestions but rarely follow the outcome of their ideas, creating distrust amongst the miners. In 1991, Ivo Lubrina, president of the Amazonian Union of "Garimpeiros", declared in an interview: "Thanks to radio and TV, garimpeiros are concerned now about mercury, but they do not know exactly why. As there is no direction from government or technical people, everything continues as before. I would say that the transfer of news is happening like a rotten onion: it is going from one hand to another" (Barbosa 1992).

Most developing countries have Mining and Environmental Agencies, usually with low budgets and unmotivated personnel. Corruption among such people is often rampant. This further increases the distance between technical solutions and artisanal miners. Most independent miners are very proud of their simple technologies even though they know improvements are possible with some help. An intelligent approach must be devised to train government personnel to guarantee they provide continuing and reliable technical assistance.

10.3.5.2
Individual Solutions

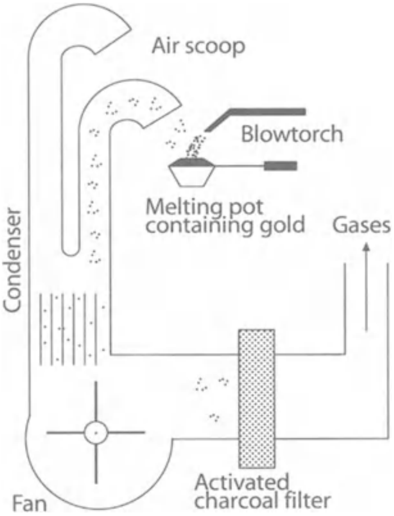
These ideas are directed at the individual artisanal miner to reduce his/her emissions of mercury. As mentioned above, amalgamation of the whole ore is the main source of mercury emission. This is ineffective and mercury losses can be as high as 50% of the mercury originally introduced in the amalgamation process. Amalgamation barrels are efficient methods to provide a controlled and smooth amalgamation of gravity concentrates. However amalgamation tailing which still contains residual mercury must be treated. The special Rio-Sul plates, as described above to extract mercury from contaminated sediments, are also an effective way to extract gold from gravity concentrates with negligible mercury losses. As many miners still employ copper amalgamation plates to process the whole ore, if the special-plates are misused, then the impact of this new technique will be negative. Technical assistance is fundamental.

Retorts can be used with recoveries above 95%, allowing the mercury to be recycled and resulting in substantial reduction in air pollution and occupational exposure (Braga *et al.* 1995). There are many types of retorts. Some are made with stainless steel while others use inexpensive cast iron. A new retort was developed by the German company ThermEx made of special glasses with stainless steel gaskets and clamps. Miners can see the distillation process and retorted gold keeps a shining color reaching better price in some gold shops. The cost of this retort is around 1 oz of gold and it is being popular in artisanal mining operations in Africa.

Mercury losses during retorting depend on the type of connections or clamps used. A homemade retort built with standard plumbing water pipes can be easily assembled to reduce mercury emissions. The distillation chamber of the RHYP retort is made by fitting an end plug in which the amalgam is placed. The size of the retort can vary from 5 to 20 cm in diameter. An iron tube is connected to the elbow bend by a thread, bushing, or welding. The tube should be at least 50 cm long and should curve downwards to permit good condensation of gaseous mercury without using coolant. For better performance, the retort can be immersed in a charcoal bed in order to heat it as a whole and avoid mercury leakage (Veiga *et al.* 1995a). A hole in the ground or an iron bucket with charcoal can be used. When the fire is lit, after 1 hour, all of the mercury should be retorted off. Use of this retort is easily understood and accepted by miners because it is inexpensive and all materials are familiar and accessible.

Methods for mercury abatement from gold shops are available and can be easily implemented. In 1989, a Brazilian company developed a special condensing fume-hood. The prototype had a series of condensing plates coupled with activated charcoal filters impregnated with iodine (Fig. 10.3). This equipment reduces mercury emission drastically. More than 99.9% of mercury is retained by this special fume-hood (Veiga and Fernandes 1990). Less than 40 $\mu g\ m^{-3}$ of mercury was detected in the interior of the shop during a gold melting operation compared with other measurements as high as 300 $\mu g\ m^{-3}$ in unprotected shops (Malm *et al.* 1990). Similar equipment is used at the

Fig. 10.3. Special fume-hood with condenser and filter



Amalgamation Center of Carhuachi in Venezuela. This simple solution must be applied to all gold dealers in Latin America, which will result in a significant reduction of emissions in urban centers.

10.4
Conclusions

Artisanal gold mining in developing countries is a temporary activity that will persist up to the exhaustion of easily extractable gold. Artisanal activities on primary ore deposits are inefficient and expensive. In all regions where miners insist on applying rudimentary methods to extract and process primary ores, the results are not encouraging. The activity has created significant social and economic problems in municipalities where mining is the main economic activity, due to the migratory nature of miners. It is clear that gold panners who conduct very-small scale mining for their subsistence will exist for many decades, as long as new discoveries of alluvial gold ore are made in remote regions.

Once mechanized operations are introduced into an artisanal mine, the activities tend to become organized. At this point technical and legal support to meet mining and environmental regulations should be made available. Governments must find a way to transfer legal titles to those who have actually discovered the majority of gold deposits in developing countries. This is an important way to stop migration of these miners and to reduce the consequent damage to the environment.

Mercury pollution has been used by different sectors of the society as a useful villain and the affected communities have been ignored. The numbers of academic research and monitoring programs are far higher than the numbers of solutions suggested or effectively implemented in the field. Education is a pre-requisite for long-term solutions to mercury emissions. Creation of Processing Centers to amalgamate or leach gravity concentrates is perhaps the most concrete systemic solution to reduce emissions or eliminate mercury use. Miners can be convinced to bring their gravity concentrates to these Centers as can be seen by the examples in Venezuela. Amalgamation can be conducted safely and without mercury emission. Leaching processes, such as the CETEM salt-electrolysis process is a practical and environmentally friendly way to recover gold from concentrates. Other important measures to reduce mercury emissions are:

- Stop amalgamation of the total ore.
- Use special plates to amalgamate gold from gravity concentrates.
- Use retorts to separate mercury from gold.
- Use filters in shops melting gold.
- Introduce education about mercury use in schools.
- Create Miner Associations as a form of union organization.
- Enact and enforce Mining Laws.
- Set up permanent biological monitoring programs and provide technical assistance.

Different approaches are necessary to remediate polluted sites depending on whether pollution is dominated by atmospheric emissions or solid waste disposal. Metallic mercury is relatively stable in aquatic environments but its oxidation in the presence of organic material is a key step favoring biomethylation.

Procedures to treat "hot spots" should attempt to create conditions to bind any Hg(II) formed, hindering its availability to microorganisms. Covering with adsorbent materials is the most favorable method to irreversibly fix mercury. Rubber, carpet, pyrite, and iron scrap are materials easily available to gold mining sites that could be used to treat tailing. Laterite material is also available in tropical countries and can be an effective adsorbent.

Unfortunately, in regions where mercury is dispersed and bioaccumulation is evident, the only practical way to achieve immediate results is a massive educational campaign to change the diet of riparian people. Care and concern for the local population must come first in any plan attempted.

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Chapter 11

Wastewater Renovation with Mine-Derived Fill Materials

Autumn-Lynn Harrison · Raymond B. Reneau, Jr. · Charles Hagedorn

11.1 Introduction

The use of fill materials from mining activities for wastewater renovation is prohibited in virtually all countries that have regulatory structures governing the collection and treatment of residential and municipal wastewaters. Such prohibitions are in place because adequate evaluations are difficult to perform on mine-derived fill materials due to variability in properties that could affect permeability such as texture, density, and lack of structure (Converse and Tyler 1985). Groundwater pollution becomes a potential problem when wastewater is applied to mined materials such as spoils since they tend to be unconsolidated, very porous, chemically variable, and have a low percentage of soil-size particles and a high percentage of rock fragments (McCormick and Borden 1973). Rock fragments may be considered a dilutant in mine spoil as there is essentially no wastewater flow through the fragments, and very little renovation of wastewater occurs as it moves between the fragments (Ross *et al.* 1987). A high percentage of rock fragments in mine spoil may substantially decrease the renovating ability of that spoil and could lead to rapid movement of inadequately treated wastewater to the underlying water table (Peterson *et al.* 1994).

Mine spoil derived from surface mining consists of blasted rock, pebbles, and cobbles embedded in a matrix of finer rock fragments and soil-sized materials (Howard 1979). The chemical and physical composition of these materials is highly variable. Frequently, between 60–80% of mine spoil particles are greater than 2 mm in size. Topsoil derived from these surface mined areas usually contains a mixture of all the materials found above bedrock, including blasted rock. Regulations in the United States classify mine spoil as a fill material because the overburden is usually removed to obtain some valuable deposit such as coal, and then replaced (US Congress 1977).

11.1.1 Wastewater Application on Mine-Derived Fill Materials

Very limited research has been conducted on the use of mine-derived fill materials for renovation of wastewater. Converse *et al.* (1978) reported that the state of Pennsylvania, the USA, currently permits mound systems on fill material provided that the fill material has had at least four years to settle and the site meets specified morphological criteria. This study also mentioned that disturbed sites such as minespoils, landfills, slag, and bottom or fly ash piles pose special problems and should be evaluated closely to determine whether or not effluent will move through these materials and what type of purification occurs within these materials. Hunt (1971) irrigated spoil material with treated municipal sewage effluent and sludge. He concluded that the spoil was 98% effective at removing P in the effluent, although when combined with sludge applications the NO_3^--N concentration exceeded the US Public Health Service standard for potable water.

Kardos *et al.* (1979) studied sewage effluent as an ameliorative agent applied to bituminous strip mine spoil and anthracite refuse. They concluded that: (1) the pH was higher in the leachate that received the highest application; (2) the application rates significantly increased the concentration of NO_3^--N and NH_4^+-N in the leachate; and (3) the concentration of NO_3^--N was increased over time while NH_4^+-N peaked and then leveled off. McCormick and Borden (1973) studied the percolate from mine spoils treated with sewage effluent and sludge and concluded that spoil material appears to be a satisfactory renovation medium for sewage effluent, although applications of large amounts of sewage sludge and effluent simultaneously may temporarily cause groundwater polluting conditions. Brandes *et al.* (1975) examined three fill materials: limestone, a sand/limestone mixture, and a sand/red mud mixture. The red mud is a byproduct of aluminum extraction from bauxite and is available in great quantity at aluminum mines. It contains compounds of Al (22–37%), Fe (24–26%), Ca (2–4%), Na, and Si. Brandes *et al.* (1975) concluded that of the three materials, red mud, when used in a mixture with medium-sized sand in proper proportions, appeared to be the most effective at removing P, BOD, suspended solids, and fecal coliforms from domestic sewage.

Brooks *et al.* (1979) evaluated the effects of iron ore overburden material on the renovation of secondary sewage effluent. Weekly application of effluent, usually over a 24- to 72- h period, was applied at a rate of 6 cm $week^{-1}$ in eight applications over a 9-week period. The results indicated that 98, 89, and 99% of NO_3^- plus NO_2^--N, total Kjeldahl-N, and total P were removed, respectively. A US Environmental Protection Agency (US EPA 1975) study investigated the performance of conventional gravity flow waste treatment systems in fill material composed of sand, silt, and shell that had been dredged from coastal canals. The results indicated that leachates were transported rapidly to the canal waters. These rapid transmissions were evident in high nutrient levels in ground and canal waters and violations of bacteriological standards.

While there are few published results regarding the use of fill materials from mining for municipal wastewater treatment, there are essentially no published results regarding placement of onsite wastewater treatment and disposal systems (OSWTDS) in mine spoil (Peterson 1991). As a result, the small quantity of available literature fails to address most of the important performance related questions regarding the use of fill materials to renovate residential and municipal wastewaters. More extensive research will be needed to determine the performance of OSWTDS, particularly alternative technologies, and municipal wastewaters that have received different levels of treatment, when placed in disturbed soil or mine spoil.

11.1.2

Mound Systems and Fill Material

In the United States, mound systems that use fill materials are permitted by some states as an alternative OSWTDS for treating residential wastewater where natural soils have limiting features such as shallow depths to a water table, bedrock, hardpans, and claypans, and do not allow adequate treatment depth below trench bottoms to fulfill requirements of a conventional OSWTDS (Converse *et al.* 1978; Hoover *et al.* 1981). Engle and

Hermanson (1982) examined fill/mound systems to determine the relation, if any, between system failure and quality of fill and installation procedures. They reported that soils considered good for filter beds are generally coarse-textured, single grained, contain little organic matter, and have structural units that are weak and easily crushed. Engle and Hermanson (1982) also reported that properties of soil fills that determine how they will settle and compact in a filter bed are soil moisture content, organic matter content, soil structure (size and strength of structural units of sand, silt, and clay particles), particle size distribution (percentages of the various sand, silt, and clay particles), and particle shape and roughness. Effects of moisture on compaction could be controlled by being certain that the fill material at the time of placement is dry enough to minimize compaction.

However, additional properties are inherent in fill materials and must be carefully considered when considering fill materials for wastewater renovation. A fill with low organic matter, poor structure, and a high percentage of clay will tend to compact more readily than soils high in organic matter with well developed structure and low clay content (Engle and Hermanson 1982). Compaction is a major concern when working with spoil materials as sources of fill due to increased bulk density which may lead to inadequate adsorption and purification of applied wastewater. Converse *et al.* (1978) reported that rockiness also may present problems for siting of mound systems as wastewater renovation is probably minimal in rocky materials, and therefore greater detailed examination of such materials must occur before use with mound systems.

There is considerable variety regarding the fill material used within mounds. Hoover *et al.* (1981) reported that the state of Pennsylvania (US) requires by law that sandy fill materials contain 5–15% clay. Willman *et al.* (1981) investigated the effect of clay content and sand mineralogy on septic tank effluent (STE) renovation within mounds. They concluded that sand type influences the renovation of wastewater and is more important than the amount of clay. Converse *et al.* (1978) reported that the standard fill material in Wisconsin (US) is a medium textured sand, and that work had been done in West Virginia (US) to evaluate bottom ash as a fill material. They also reported that sandy loams, soil mixtures, strip mine spoil and slags have been used and tested as a source of fill for mounds. However, these results have not been published.

11.1.3
The Appalachian Mining Region

The development of southwest Virginia's coal mining areas is severely limited by a lack of building sites and waste disposal facilities (Howard 1979). Much of the land area in the coal mining region of southwest Virginia consists of steep slopes with shallow soils. The shallow soils effectively limit the volume of soil available for waste disposal purposes, and the steep slopes increase the possibility of seepage and surface pounding of effluent (Commonwealth of Virginia 1982). New technologies to reclaim mined lands have the capacity to produce large level expanses of land suitable for building sites by using the mine spoil to reshape the land contours. However, the problem of waste disposal remains. These areas are usually too remote for construction of centralized sewers to be economically feasible.

The suitability of wastewater disposal systems in fill materials is uncertain because of the dramatic effects that surface mining activities have on soil physical properties

(Brown *et al.* 1978). The variability, large percentage of coarse fragments, and unconsolidated nature inherent to fill material could potentially lead to inadequate purification of wastewater and eventual ground water contamination. Therefore, current Virginia Department of Health regulations forbid placement of OSWTDS in any fill material, including mine spoil (Commonwealth of Virginia 1982). However, there is the potential for placing selected fill material in designated disposal and treatment sites as reclamation of mined lands proceeds. The implementation of the Surface Mining Control and Reclamation Act requires the placement of topsoil above mine spoil in mined land reclamation areas (US Congress 1977). Our research on fill materials (mine soil and mine spoil) involves evaluating alternative OSWTDS technologies for their applicability to treating wastewater in fill materials with respect to both removal and renovation of biological and chemical contaminants and hydraulic performance. Howard (1979) characterized the geology of the Wise Formation coal mining area where our research is conducted as consisting of a heterogeneous group of strata characterized by abrupt lithologic and geochemical facies changes involving calcareous and ferruginous sandstones and siltstones, shales, mudstones, conglomerates and coals.

Howard (1979) also reported on the source of P in mine spoils of the Wise Formation and concluded that the bulk of the P in the rocks (and therefore the mine spoil) was associated with a phosphate or carbonate mineral, and was most likely associated with calcite and particularly dolomite. He also stated that P may leach from fresh mine spoil, but eventually reacts with oxides to precipitate as insoluble Fe and Al phosphates.

11.2
Pollution Concerns

11.2.1
Biological Contaminants

Residential and municipal wastewaters contain microorganisms that are considered to be carriers of disease. The four main groups of organisms, listed in order of increasing size are: enteric viruses, bacteria, protozoans, and helminths. Size has an important influence on the fate of disease agents as the larger organisms are more readily entrapped or retained by most soils as wastewater percolates through the soil. Sobsey (1982) summarized the diseases caused by microbial agents in domestic wastewaters. Enteric viruses cause such diseases as meningitis and hepatitis, while bacteria may cause typhoid fever and gastroenteritis, among others. Protozoans can cause amoebic dysentery and giardiasis, and helminths (worms) are known to cause a number of chronic diseases that include anemia and gastroenteritis. Ziebell *et al.* (1975) reported that humans are subject to a number of enteric diseases, during which time pathogens are shed in the feces in tremendous numbers. And, although most patients cease to shed after recovery, some continue as carriers, even though they are superficially well.

Viruses and bacteria are probably the more important microbial disease agents associated with OSWTDS because their smaller size makes them less likely to be removed than protozoans and heminths (Sobsey 1982). Fecal coliform counts are thought to be reduced by dilution, soil filtration, sedimentation, adsorption, and dieoff as the bacteria move through the natural soil system (Sobsey 1982; Stewart and Reneau 1982).

Peterson and Ward (1988) reported that the percentage of fines (silt- and clay-sized particles) are a major factor leading to bacterial retention, whether by straining (clogging) or adsorption, and that transport of most pathogenic organisms could be limited by a soil profile of sufficient depth with adequate amounts of clay, silts, fine sands, and organic matter. Macropore flow through saturated sandy or strongly structured soils may result in greater travel distances by pathogenic organisms (Reneau *et al.* 1989). Adsorption to soil particles is enhanced with bacteria and viruses due to the charged nature of these organisms (Tyler *et al.* 1978). Because of the small size of virus particles, some researchers consider viral adsorption to soil constituents to be the primary mechanism of immobilization (Green and Cliver 1975; Gross and Mitchell 1985).

Although adsorption tends to be the most important mechanism for enterovirus retention, biochemical degradation should not be discounted as an important means to remove viruses from wastewater, and to inactivate the virus (Gross and Mitchell 1985). Virus adsorption depends on pH, flow rate, the presence of cations, soluble organics and clay as well as the soil composition. Virus adsorption is greater with a low pH, high cation exchange capacity, and in soils with a high surface area (Sobsey 1982). Lower level of virus adsorption occurs with an increase in flow rate and concentrations of soluble organic compounds, since soluble organics interfere with virus adsorption. Because effluent moves through coarser, gravely soils more rapidly than finer clayey soils, virus adsorption is decreased in the coarser soils. This phenomenon of adsorption may be followed by desorption if these variables are altered (Wellings 1980).

Saturated soil conditions tend to increase pathogen travel distance from OSWTDS (Converse *et al.* 1975; Stewart and Reneau 1988). These conditions can sometimes be overcome by the use of mounds or other alternative OSWTDS (Stewart and Reneau 1982). Computer simulation studies have shown that enteric coliform bacteria have the potential to travel great distances in nutrient-poor, coarse-grained soils. Peterson and Ward (1988) noted that in coarse-grained soils fecal coliform bacteria will eventually be transported beyond the often accepted 1.2 m of suitable soil depth, they therefore recommended modification of existing regulations to account for bacterial transport in coarse-grained soils.

Tyler *et al.* (1978) discussed the difficulty of detecting bacteria and viruses in soil and waste disposal systems. Because of this difficulty, indicator and tracer organisms are often used, with the thought that where the indicators are found the pathogens of public health significance may also be found. Fecal streptococci and fecal coliform bacteria are used as indicators of fecal pollution because they are: (1) present in high numbers in feces of man and warm-blooded animals, (2) able to survive outside the body, (3) detectable quantitatively, and (4) found in high numbers relative to pathogens. Reneau *et al.* (1989) also discussed the need for improved detection methods for bacterial pathogens and suitable indicators for viral and protozoal pathogens.

11.2.2
Nitrogen and Phosphorus

The nutrients N and P are of concern when evaluating OSWTDS effectiveness (Tyler *et al.* 1978; Reneau *et al.* 1989). Excessive amounts of NO_3^- may be toxic to infants and young animals, and both NO_3^- and P have been shown to encourage eutrophication of surface waters, although P does not appear to pose a hazard to human health.

The N contained in STE is primarily in the NH_4^+ form (Wall and Webber 1970). When this ammonified effluent reaches an aerobic soil environment, it is subjected to nitrification that converts the NH_4^+ to soluble NO_3^- with limited NO_2^- remaining in the soil under adequate operating conditions (Wall and Webber 1970). Nitrates are very mobile and move readily through the renovating media. Brown *et al.* (1978) suggested that the best way to deal with NO_3^- leachate is either by limiting the number of OSWTDS per unit area, removing the N from the effluent before it is applied or possibly by using vegetation to help remove it after application. Denitrification (reduction of NO_3^- to N_2O and N_2) is also a potential mechanism for NO_3^- loss from these systems. Magdoff and Brown (1975) reported denitrification rates of 32% in a sand or sand loam till.

A well-drained soil has the capacity to fix and immobilize large quantities of P from sewage effluents. Most field studies have indicated that P contamination is limited to shallow groundwaters adjacent to OSWTDS and that P adsorption continues at reduced rates, under saturated conditions (Reneau *et al.* 1989). Magdoff and Keeney (1975) reported P retention was initially a result of adsorption and in the later stages probably a result of precipitation as calcium phosphate. Reneau *et al.* (1989) reported that precipitation depends on pH, concentrations of P, Fe, Al, and Ca, competing anions, and reaction time. Slightly soluble P compounds of Fe and Al form in acid soils, while in calcareous soils compounds containing Ca and P form. Sawhney and Hill (1975) observed that soil used for OSWTDS has the potential to regenerate P adsorption sites with time.

11.3
Fill Material Column Study

In our preliminary study two fill materials were evaluated in columns, mine soil-fill (a mixture of Jefferson, fine-loamy, siliceous, mesic Typic Hapludult and Muskingom, fine-loamy, mixed, mesic Typic Dystrochrept soils) and mine spoil-fill (blasted rock material associated with the Taggart Marker and Low Splint Bench coal seams of the Upper Middle Wise Formation) (Roberts *et al.* 1988; Peterson *et al.* 1994, 1997). Mesic soils are those formed under the moderate temperature ranges commonly found in temperate regions. Septic tank effluent (STE) and sand filter effluent (SFE) were applied to mine soil-fill columns at four loading rates (0, 5.4, 10.8, and 21.6 $l\ m^{-2}\ day^{-1}$) and mine spoil-fill columns at one loading rate (21.6 $l\ m^{-2}\ day^{-1}$) for 20 weeks.

11.3.1
Removal of Nitrogen, Phosphorus, and Coliform

The NO_3^- concentration in mine soil-fill column leachates increased with increased SFE application at the two highest application rates, changed very little with increased STE application rates, but remained below the US regulatory limit (10 $mg\ l^{-1}$) in all cases. There was a much higher concentration of NO_3^- in the SFE than in the STE due to nitrification. Nitrate-N in the mine spoil-fill column leachate was higher than 10 $mg\ l^{-1}$ (Peterson *et al.* 1997).

Ammonium-N concentration decreased when influent was passed through the mine soil-fill columns, a result of NH_4^+ being nitrified. However, NH_4^+ in the mine soil-fill column leachate did increase with STE application at the highest application rate. The relationship between NO_3^- and NH_4^+ in both types of fill column leachates indicates that a larger percentage of the fill volume may be anaerobic where STE is applied.

There was a general trend of higher N losses (denitrification) occurring in columns receiving STE, possibly due to a greater availability of carbonaceous material in STE. Nitrogen losses via denitrification were estimated to range from 15.0 to 36.8%. The concentration of P in STE was reduced between 74.4 and 99.1% after passing through the soil-fill columns, but by only 47.4% after passing through the spoil-fill columns. The high levels of P in the spoil-fill column effluent was probably due to the high levels of P in the mine spoil-fill. We concluded that the mine soil-fill was effective at renovating wastewater in regards to P, while the mine spoil-fill was much less effective because of the large quantities of P and alkaline pH present in the minespoil. However, data in the literature indicate that mine spoil upon oxidation may become a net fixer of P.

The sand filter was an effective pretreatment mechanism for reducing (up to 97.7%) fecal coliforms present in STE (Table 11.1). When column influent types were compared (STE vs SFE) fewer fecal coliforms were present in column effluents where SFE was applied. This was observed for both the mine soil-fill and mine spoil-fill columns and was not unexpected since the average fecal coliform count in SFE was much lower than that present in STE (630 vs 27400 organisms ml^{-1}, respectively). The reduction in fecal coliform numbers after passage through columns (average of both mine soil and mine spoil) was 99.8% and 97.9%, respectively, where SFE and STE were applied to the columns. The results also showed that, at high rates of application of SFE and STE, there was no difference between the effectiveness of the mine soil-fill and mine spoil-fill with respect to fecal coliform removal (Peterson *et al.* 1994).

Table 11.1. Average fecal coliform densities (determined from median value of log normalized distribution of measurements) as influenced by influent type, application rate, and fill type

	Influent type			
	Sand filter effluent (SFE)		**Septic tank effluent (STE)**	
Application rate ($l\ m^{-2}\ day^{-1}$)	**Organisms ($m\ l^{-1}$)**	**Reduction (%)**	**Organisms ($m\ l^{-1}$)**	**Reduction (%)**
	Column influent			
	630	97.7	27400	–
	Column effluent			
Soil-fill				
5.4	1.2	99.9	13	99.9
10.8	25	99.9	24	99.9
21.6	43	99.8	585	97.9
Spoil-fill				
21.6	70	99.7	1650	94

Reduction percentages are based on organisms (m/l) in STE influent. Soil-fill SFEs are the soil-fill columns that received SFE ($l\ m^{-2}\ day^{-1}$); soil-fill STEs are the soil-fill columns that received STE ($l\ m^{-2}\ day^{-1}$); Spoil-fill SFEs are the spoil-fill columns that received SFE ($l\ m^{-2}\ day^{-1}$); Spoil-fill STEs are the spoil-fill columns that received STE ($l\ m^{-2}\ day^{-1}$).

11.3.2
Conclusions

The column study indicated that, based on N and P renovation, OSWTDS may be an appropriate option in selected soil and spoil fills with limited risk for environmental degradation. The results also demonstrated that large reductions in biological contaminants can be achieved with 60 cm of unsaturated soil-fill at lower dose rates (10.8 l m^{-2} day^{-1}) or when pretreatment such as sand filtration is employed. Mine spoil-fill was effective at renovating wastewater regarding inorganic N, but the waste may require additional treatment to remove P beyond that provided by the septic tank before application. Also, lower loading rates may be required for spoil-fill installations. It is unknown how a conventional OSWTDS would perform in mine spoil, although the removal of chemical and biological contaminants would probably not be as efficient as that of an alternative OSWTDS that was simulated in this study (influent was dosed at regular intervals and uniformly to the soil volume).

11.4
Mine Soil-Fill Field Study

Based on encouraging results from the column study, a field study is being conducted on a reclaimed coal strip mine in Wise County, Virginia. The site is typical of reclaimed strip mines where the landform could physically support housing and other forms of residential or commercial development. STE for the field portion of the project was collected in a 3785-l pump chamber installed between the septic tank and the distribution box at a private residence. A high head pump was used to pump the STE to the field site, which is located at an elevation of 11.9 m above and 67.4 m upslope of the private residence. The alternative systems employed at the site were low pressure distribution (LPD), vegetated subsurface bed (VSB) constructed wetland, and slow rate spray irrigation.

11.4.1
Low Pressure Distribution (LPD) Systems

The LPD systems were placed above the reclaimed mine land surface in a mound constructed from mine soil-fill using design parameters from the column study to determine adequate depth for renovation of wastewater. The configuration of this mound was based on the US EPA onsite manual (US EPA 1980). The mine soil-fill was estimated to have an infiltration rate of 48 l m^{-2} day^{-1} and the reclaimed mine land an infiltration rate of 10 l m^{-2} day^{-1}. The mound height was based on a mine soil-fill depth of 0.76 m and an absorption bed depth of 22.9 cm. The cap at the center of the bed received an additional 0.3 m of soil-fill. The central core of the mound was 1.2 m and the ends and sides of the mound extend 3.7 m beyond the central core. The total length of the mound was 32.9 m and the width was 8.5 m.

Six experimental LPD systems were placed in the mound. Each system consisted of an absorption bed 0.6 m wide, 2.7 m long, and 22.9 cm deep. The absorption bed was constructed with 2.5- to 3.8-cm-diameter gravel and was dosed through a distribution system that consisted of a single perforated line. The separation distance between experimental systems that received the same rate of STE application was 1.5 m, while for systems that received different rates of STE it was 3.0 m.

Two rates of STE application, 9.2 and 19.3 l m^{-2} day^{-1} were selected based on Virginia Health Department regulations and the column study. Each application rate was replicated three times. Results from the column study indicated that acceptable wastewater renovation should occur in 0.76 m of mine soil-fill material at the 9.2 l m^{-2} application rate. Since renovation should be more effective in the field because of the absence of horizontal boundaries present in the columns, both the 9.2 and 19.3 l m^{-2} day^{-1} rates were employed. Each LPD system was dosed twice per day. For the 9.2 and 19.3 m^{-2} day^{-1} rates, each dose was 15.7 and 7.4 l, respectively, for a total application of 31.4 and 14.8 l day^{-1}. Application of STE to the systems was initiated in October 1991.

11.4.2
Vegetated Subsurface Bed (VSB) Constructed Wetland and Spray Irrigation

The VSB constructed wetland was designed in combination with the slow rate spray irrigation system. The VSB was used as an economical method for additional treatment of STE prior to spray irrigation. Effluent that discharges from the VSB passes through a water level control box, a tablet chlorinator and into a 113-l contact chamber. The chlorinated VSB effluent then spills into a pump chamber when fresh VSB effluent is added to the contact chamber. The chlorinated VSB effluent was spray-applied to tall fescue (*Festuca arundinacea* Schreb.) through a stationary riser. The spray irrigation system was designed to apply 7.2 l min^{-1}. VSB effluent was sprayed for 15 min twice each night, at 1 and 3 a.m. This system applies 1.25 cm of chlorinated VSB effluent to the soil each week. The sprinkler system consists of an impact type sprinkler placed 45 cm above the soil surface. The sprinkler is made of stainless steel (designed for effluent application) and has a frost proof design for winter application. The sprinkler had a 0.28 cm nozzle and delivered 7.2 l at 30 PSI to a circular area 122 cm in diameter with a low trajectory (7 degrees) and large drop size to minimize aerosol drift.

The VSB was designed for a 5.5-day effluent retention time. Data from the US EPA design manual (US EPA 1988) indicated that this retention time should decrease N and not result in excess stagnation of effluent. For a 5.5-day retention time it was necessary to design a VSB that would be capable of holding a total volume of 1363 l. The gravel used as the liquid reservoir was estimated to have a total pore volume of 40%. To meet the necessary volume requirements, the VSB was constructed with a liquid reservoir that was 0.45 m deep, 0.45 m wide, and 14.2 m long. The VSB was lined with a butyl rubber liner that was greater than 30 mm in thickness. The first 1.2 m of the VSB was filled with rock 7.5–17.5 cm in diameter and the remaining 13.0 m of the trench was filled to a depth of 0.45 m with 2.5- to 3.7-cm-diameter (drainfield sized aggregate) gravel. The VSB was then covered with 15 cm of pea gravel (0.3- to 0.7-cm-diameter). Emergent vegetation was established in the pea gravel, consisting of cattail (*Typha latifolia*), woolgrass (*Scirpus cyperinus*), iris (*Iris pseudacorus* and *Iris verisicolor*), horsetail (*Equisetum hyemale*), duck potato (*Sagittaria latifolia*), and lizards' tail (*Saururus cernuus*). A total of 312 of STE was applied to this VSB each day. This was accomplished by filling and discharging the 106-l dosing reservoir twice daily.

11.4.3
Results from LPD Mine Soil-Fill System

Selected chemical and biological parameters for the LPD systems have been monitored for 5 years to date (Tables 11.2–11.4; Reneau *et al.* 1995). The NH_4^+-N present in STE was effectively nitrified (conversion to NO_3^-) as indicated by the data collected from January of 1992 to the present (Table 11.2). In fact, the concentration of NH_4^+-N in the solution percolating through the mine soil-fill in the mound was generally lower than the concentration of NH_4^+-N present in the control wells. Thus these systems had adequate aerobic soil volume for NH_4^+ to be nitrified and for NH_4^+ concentrations to decrease to background levels after travel through 0.76 m of mine soil-fill.

The mine soil-fill was very efficient at removing PO_4-P from the wastewater. Greater than 99% of the applied P was removed by the mine soil-fill at both application rates (Table 11.2). It should also be noted that the PO_4-P present in the leachate never exceeded the concentrations present in the control wells. This was in contrast to the higher PO_4-P concentrations measured in the column study at the highest STE application rate. The pH values were lower where higher applications of wastewater were applied (Table 11.2). This was attributed to acidity produced from the increased nitrification where larger quantities of NH_4^+-N were applied to the system.

Table 11.2. Means for chemical constituents in shallow groundwater beneath and downslope of low pressure distribution systems, background controls, and septic tank effluent

	Application rate							
	Low (9.2 l m^{-2} day^{-1})				High (19.3 l m^{-2} day^{-1})			
Depth (cm)	NH_4-N (mg l^{-1})	Cl (mg l^{-1})	PO_4-P (mg l^{-1})	pH	NH_4-N (mg l^{-1})	Cl (mg l^{-1})	PO_4-P (mg l^{-1})	pH
0.0 m Downslope								
0	0.90a[a]	62a	0.06ab	7.14	0.94ab	61.1a	0.08ab	6.33
60	0.40a	43b	0.10ab	6.79	0.88ab	60.1a	0.09ab	6.27
1.5 m Downslope								
0	0.90a	43b	0.07b	7.02	0.64ab	33.4bc	0.08ab	6.74
60	0.41a	32c	0.17a	7.35	0.38ab	34bc	0.08ab	6.89
Control								
60	1.1a	16.2d	0.11ab	6.28	1.1a	16.2d	0.11ab	6.28
Septic tank effluent								
–	40.7	82.3	4.91	7.41	40.7	82.3	4.91	7.41

0-cm Depth: surface reclaimed mine land (effluent has passed through 0.76 cm of fill), 60 cm depth: located 60 cm below surface of reclaimed mine land; 0.0 m downslope: sampling wells located at downslope edge of low pressure distribution system; 1.5 m downslope: sampling wells located 1.5 m downslope of low pressure distribution system; control: wells located upslope and outside zone of influence of domestic wastewater disposal.

[a] Numbers followed by different letters for each parameter within columns are significantly different ($P < 0.05$) as determined by Duncan's Multiple Range Test (SAS).

The biological quality of the leachate percolating through the mine soil-fill was very good and samples from wells usually contained few, if any, fecal coliforms (Table 11.3). In most instances, the coliform counts were attributable to only one or two positive counts from the 60 samples collected. The effectiveness of the mine soil-fill to biologically renovate wastewater was demonstrated by the fact that coliform counts were generally lower in samples collected below the mound than samples collected from the control wells. Thus, 0.76 m of mine soil-fill was effective in reducing fecal coliforms to background levels. This implies that complete biological renovation of the wastewater had been achieved.

The quality of the leachate with respect to NO_3^--N has changed markedly since the study was initiated. At the beginning of the study (01/92–09/92), the NO_3^--N concentration in the mine soil leachate was high, as expected (Table 11.4). However, beginning in 10/92 the concentrations of NO_3^--N present in the leachate started to decrease. Using chloride (Cl^-) as a conservative ion, we can estimate the quantity of NO_3 that has disappeared due to biological processes (Stewart and Reneau 1988). If we use the sample collected at the reclaimed mine surface at the low rate of application, we would estimate that 11% of the NO_3 was lost via denitrification in the sampling period from 01/92 through 04/92. The loss from this same portion of the mound at the 10/93–06/94 sampling period was estimated at 87%. This increase indicates that, as these systems mature, zones that enhance denitrification also appear to be developing. Thus the potential for contamination of ground and surface waters in these mine soil fill materials may be much less than initially estimated. The NO_3^--N concentrations during the cooler times of the year were somewhat higher, perhaps indicating decreased denitrification.

Table 11.3. Fecal coliform counts in shallow groundwater beneath and downslope of low pressure distribution systems, background controls, and septic tank effluent

	Application rate **Fecal coliforms (organisms/100 ml)**	
Depth (cm)	**Low (9.2 l m^{-2} day^{-1})**	**High (19.3 l m^{-2} day^{-1})**
0.0 m Downslope		
0	0	0
60	0.8	36.3
1.5 m Downslope		
0	0	0.1
60	0	1.1
Control		
60	5.7	5.7
Septic tank effluent		
–	1.27×10^5	1.27×10^5

0 cm Depth: surface of reclaimed mine land (effluent has passed through 0.76 cm of fill); 60 cm depth: located 60 cm below surface of reclaimed mine land; 0.0 m downslope: sampling wells located at downslope edge of low pressure distribution system; 1.5 m downslope: sampling wells located 1.5 m downslope of low pressure distribution system; control: wells located upslope and outside zone of influence of residential wastewater treatment.

During the months when evapotranspiration exceeded precipitation, very low concentrations of NO_3^--N were present in the wells. While it is not currently possible to predict if all systems placed in mine soil-fill will behave in a similar manner, these data indicate that a layer of finer soil-fill material might be considered when reclaiming mine lands for residential and commercial development. This layer of finer soil would help develop an anaerobic layer that could enhance denitrification.

11.4.4
Results from VSB Wetland and Spray Irrigation

Data for the VSB effluent is the mean of data collected over the past 60 months (Table 11.5). Fecal coliforms decreased with increased detention time and coliform reductions ranged from 89 to 96% from 1.8 to 5.5 days. For NH_4-N, PO_4-P, and Cl the concentrations decreased at the 1.8- and 3.6-day detention times and increased at the 5.5-day detention time. This is attributed to the fact that samples for the 1.8- and 3.6-day detention times are collected from the center of the wastewater column while the 5.5-day

Table 11.4. Means for nitrate in shallow groundwater beneath and downslope of low pressure distribution systems and background control wells (data expressed in mg l^{-1})

Depth	**Date**					
(cm)	**01/92–04/92**	**05/92–09/92**	**10/92–02/93**	**03/93–09/93**	**10/93–06/94**	**07/94–06/95**
Low application rate (9.2 l m^{-2} day^{-1})						
0.0 m Downslope						
0	28.0ab[a]	23.5abc	2.83b	4.67a	2.02a	0.25a
60	24.9ab	32.1a	3.81ab	2.98ab	1.45a	0.14a
1.5 m Downslope						
0	18.lb	29.5a	0.42b	0.28b	0.68a	0.19a
60	17.7b	11.9abcd	1.05b	0.18b	0.48a	0.25a
High application rate (19.3 l m^{-2} day^{-1})						
0.0 m Downslope						
0	37.0a	16.7abcd	0.29b	1.55b	0.97a	0.27a
60	35.2a	26.6ab	8.9a	2.78ab	2.69a	0.43a
1.5 m Downslope						
0	15.5b	6.0 cd	0.17b	0.17b	0.31a	0.14a
60	17.5b	6.6cd	0.20b	0.00c	0.92a	0.16a
Control						
60	1.6	0.37	0.06	0.12	0.14	0.11

0-cm Depth: surface of reclaimed mine land (effluent has passed through 0.76 cm of fill); 60 cm depth: located 60 cm below surface of reclaimed mine land; 0.0 m downslope: sampling wells located at downslope edge of low pressure distribution system; 1.5 m downslope: sampling wells located 1.5 m downslope of low pressure distribution system; control: wells located upslope and outside zone of influence of domestic wastewater disposal.

[a] Numbers followed by different letters for each parameter within columns are significantly different ($P < 0.05$) as determined by Duncan's Multiple Range Test (SAS).

detention samples are collected at the outflow from the VSB and are removed from the bottom of the water column. Additional research with other VSB systems has shown that wastewater in VSBs is highly stratified and that samples collected from the surface of the VSB will yield samples of a better water quality than those reported in Table 11.5 (Huang *et al.* 1994). During the next year the outfall from the VSB will be changed to the top of the water column. Thus water of improved quality should emerge from the VSB in the future. There was a 43% reduction in BOD_5 at the 3.6-day retention time and a 38% reduction in BOD_5 reduction in the discharge from the VSB. Much higher reduction in BOD_5 should occur in the discharge from the VSB during the next sampling year.

The spray irrigation system adequately decreased the biological and chemical contaminants in the wastewater after passage through 60 cm of minespoil (Table 11.6). The NH_4^+-N and fecal coliforms in samples collected from the spray irrigation area were lower than concentrations present in the control wells. In fact, the fecal coliform counts

Table 11.5. Effect of retention time in the vegetated subsurface bed (VSB constructed wetland) on selected biological and chemical contaminants

		Septic tank effluent	Parameter		
Fecal coliforms	Cells/100 ml	$1.27 \cdot 10^5$	10060	5500	3200
BOD_5	(mg l^{-1})	123	100	80	86
NH_4-N	(mg l^{-1})	40.7	39.9	40.4	45.3
NO_3-N	(mg l^{-1})	0.01	0.05	0.05	0.11
PO_4-P	(mg l^{-1})	4.91	4.19	4.85	5.01
Cl	(mg l^{-1})	82	71	70	77
pH		7.41	7.99	7.96	7.93

Table 11.6. Means for selected chemical constituents in shallow groundwater (6 cm) beneath the spray irrigation system

	Spray field	Control wells	VSB effluent
NH_4-N (mg l^{-1})	0.3	1.3	45.3
NO_3-N (mg l^{-1})	1.32	0.56	0.11
PO_4-P (mg l^{-1})	0.24	0.08	5.01
Cl (mg l^{-1})	27.7	17.9	77
Fecal coliforms (organisms/100 ml)	<1[a]	17	3200
pH	7.85	6.43	7.93

[a] Samples were collected 60 cm beneath the reclaimed mine surface. Positive samples were only present for two of the 20 samples collected. These values were both <10.

may actually be lower than the 1 count per 100 ml reported. Two positive samples were collected from the spray irrigation site during the study period. Both of these samples had fecal coliform counts of <10 counts per 100 ml. A maximum value of 10 counts per 100 ml was assigned to these samples. Samples collected from shallow groundwater beneath the spray irrigation system indicated an acceptable reduction in P and N (average 95% decrease). The low quantities of NO_3^--N present in the waters beneath the spray field were also acceptable as our design calculations allowed for 10 mg l^{-1} of NO_3^--N. Spray irrigation appears to be an acceptable means for treating and disposal of residential wastewater on existing reclaimed mine lands.

11.4.5
Conclusions from Field Study

Based on 5 years of sampling, the mine soil-fill material employed was an excellent wastewater renovation medium. In the LPD systems, biological and chemical renovation of wastewater was effective at two different application rates. Only a few positive samples for fecal coliforms were collected over the study period. The average fecal coliform counts in effluent that had passed through 0.76 m of mine soil-fill were lower than the average counts present in the control wells. Ammonium and PO_4^{3-}-P, both potential contaminants, were also reduced to background level after passage through the mine soil-fill. The most surprising observation was the decrease in NO_3^--N with time. There was approximately an 87% decrease in N in the mine soil-fill system during the winter months after the start-up phase of the study. This is an important observation since NO_3^--N is one of the most mobile contaminants present in soils where domestic wastewater is applied for treatment purposes.

The spray irrigation system was also extremely effective. The biological and chemical contaminants were generally reduced to background levels after travel through 60 cm of the reclaimed mine surface. The constructed wetland associated with the spray irrigation system was not as effective as other wetlands that we have monitored, probably due to placement of the sampling wells at the bottom of the wetland bed.

The data collected over the past 5 years indicate that mine soil-fill can be used to effectively renovate domestic wastewater. Wastewater can be applied at application rates as high as 19.3 l m^{-2} day^{-1} on at least 0.76 m of mine soil-fill. Low pressure distribution or drip irrigation systems should be used to ensure uniform effluent distribution to the mine soil-fill. Spray irrigation was also shown to be an effective mechanism for applying residential wastewater to existing reclaimed mine lands. When spray irrigation is employed, additional treatment of septic tank effluent is necessary prior to application. Regulatory agencies are not likely to permit spray application of untreated wastewater and such a practice is not appropriate for mine-derived fill in any case. This additional treatment could be achieved with constructed wetlands, as in our study, with sand filters, or with small extended aeration systems.

While our results are not applicable to all types of fill materials, we have demonstrated that there are some application technologies that can be used to renovate residential wastewaters in mine-derived fill materials, and with such technologies it appears feasible to allow closely regulated development to proceed on carefully evaluated reclaimed mined lands.

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Chapter 12

Environmental Effects of the Deposition and Re-use of Colliery Spoils

Krystyna M. Skarzynska · Piotr Michalski

12.1
Origin and Nature of Colliery Spoils

Coal resources were formed in zones of vast and waterlogged morphological depressions with very rich vegetation, and in regions of subsidence resulting from mountain formation. In periods of relative stability organic matter accumulated, whereas in times of intensive tectonic movements that caused subsidence, coal deposits were buried by material eroded from the surrounding uplands. Hence, sediments which overlay coal beds generally consist of an alternating sequence of thin layers or lenses of clayey shale, mudstone and sandstone, as well as occasional fine-grained conglomerates. In regions where the subsidence of sea basins reached considerable depth, layers of limestones can appear (Fig. 12.1). Mudstones usually contain mixtures of coal in pelitic form or thin laminates and interbeddings characteristic of coal shales. Larger fragments of carbonized plant detritus appear in sandstones and conglomerates because alluviation of these sediments resulted in the washing out of clay-sized fractions. The filling of depressions up to the level of the water table resulted in first the slow and then the fast development of vegetation. Following the next tectonic event, another cycle of coal deposition occurred. Hence, the organic matter content is highest in the vicinity of the coal layers and decreases considerably away from these layers.

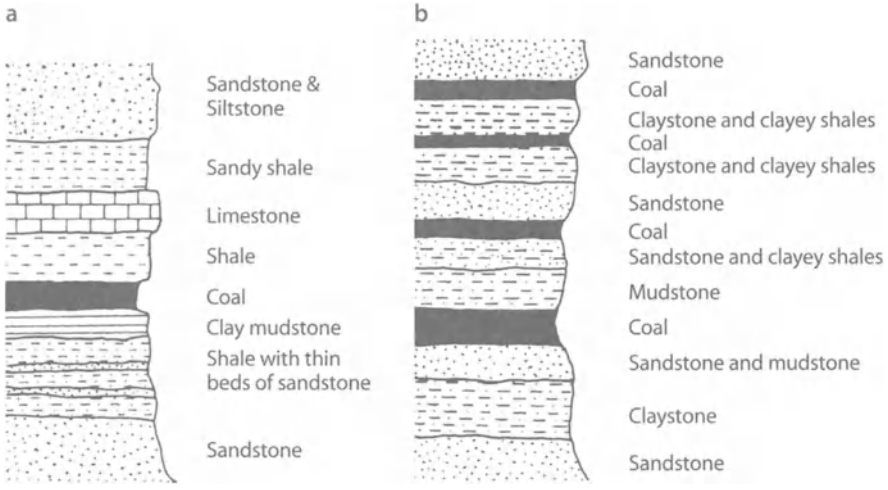

Fig. 12.1. Profiles of rocks accompanying coal deposits: **a** the UK Rainbow 1987; **b** Upper Silesia coal field, Poland (Skarzynska 1995a)

Colliery spoil is the waste from the extraction of coal by mining. It consists of a mixture of rocks that accompany the coal seams and is separated from pure coal in washery plants and extracted from shaft driving and preparatory works. Colliery spoil can be classified into two main groups on the basis of origin: (1) Mining wastes extracted mainly from routine preparatory procedures and exploitation of coal seams. These materials are usually coarse-grained wastes ranging in size up to 500 mm in diameter (2) Washery wastes obtained during the coal separation process. These materials can be subdivided into:

a Coarse-grained washery wastes, which vary from 10 to 250 mm in diameter, and are produced from suspension plants
b Fine-grained washery wastes, which range in size from 0.5 to 30 mm in diameter, and are produced from sedimentation processes
c Slurries or tailings resulting from flotation procedures and having sizes finer than 1 mm in diameter

The range of the above sizes depends upon the technology of the specific coal production operation and is, therefore, probably not the same in every coal producing country. However, corresponding values should not be very different because extrac-

Table 12.1. Estimated accumulation of colliery spoils

Country	Million tons
China	1200
France	200
UK	2000
Japan	600
Poland[a]	700
South Africa	1000
USA	3000

[a] Upper Silesia coal field.

Table 12.2. Petrographic composition (%) of colliery spoils from selected countries

Rock	Czech Republic	Germany	Spain	France	Poland[c]
Claystones	10 – 15	47 – 66	7 – 10	–	40 – 98
Mudstones	40 – 75	8 – 36[a]	5 – 40	–	2 – 40
Coal shales	–	2 – 14	5 – 30	50 – 80[b]	2 – 25
Sandstones	5 – 15	0 – 4	10 – 80	20 – 40	0 – 33
Coal	15 – 20	5 – 15	4 – 30	0 – 20	3 – 18

[a] Clayey-sandy shales + sandy-clayey shales.
[b] Shales and coal shales.
[c] Upper Silesia coal field.

tion procedures are usually very similar in countries with similar levels of mechanization. The total amount of colliery spoils depends not only on the total coal production but also on the geological conditions prevailing at a particular mine and the mode of extraction. The ratio between the amount of wastes generated to coal production in various countries is different. For example, in France, the ratio is such that 2 tons of colliery spoil is brought to the surface in order to extract 1 ton of saleable coal (Berthe 1984). In the United Kingdom and in Poland, the ratio is about 0.6–0.7 tons of wastes for the recovery of 1 ton of coal. In Germany, however, the amount of wastes is 0.86 tons for 1 ton of coal (Toussaint 1984). As a result of coal mining activity, huge quantities of colliery spoils have been accumulated in coal producing countries as shown in Table 12.1.

12.2
Composition

12.2.1
Petrographic and Mineralogical Composition

The petrographic composition of colliery spoil depends on the kind of rocks accompanying coal seams and consequently on the particular coal basins and fields where coal is extracted. There are some basic rock types that constitute colliery spoils, but their proportions in wastes from different mines vary considerably. The basic rocks forming almost all colliery spoils are: claystones, mudstones, coal shales, and sandstones with variable admixture of pure coal. The mineralogical and chemical composition as well as the physical and mechanical properties of these rocks are site-specific and, therefore, the variability in their proportions results in changes of nearly all the properties of the colliery spoils. The variation in the petrographic composition of colliery spoils in some European countries is presented in Table 12.2.

The mineralogical composition of colliery wastes is very important when considering their potential utilization and influence on the environment. The properties of this material are greatly influenced by the presence of clay minerals since they are often the major component of it. Typically, colliery spoils contain about 50–70% clay minerals, 20–30% quartz, and 10–20% other minerals and carbonaceous matter (Table 12.3). Clay minerals are essentially hydrous aluminum silicates having a layered sheet structure. The mixed layer illite-montmorillonite is the most expansive of the clay minerals, especially when sodium is the inter-layer cation (Taylor and Spears 1970).

The other minerals are chlorite, pyrite, siderite, and ankerite. Gypsum and jarosite may also be present, but usually they originate in tips as a result of pyrite oxidation of carbonate rocks. The presence of pyrite is important since it indicates the progress of weathering and whether the material is still chemically active. Such factors as the strength of the rock particles, climatic conditions, and the age of heaps considerably influence chemical weathering of wastes, i.e. the stage of decomposition of minerals, and the final transformation which leads to the development of clayey soils.

The presence of carbonaceous matter is a characteristic feature of colliery spoils. The organic carbon content does not usually exceed 2.5% by weight. The inorganic carbon content, however, is in the range 18–25% and influences the properties of the waste as well as its long-lasting chemical and thermic behavior.

12.2.2
Chemical Composition

The chemical composition of colliery spoils is of great importance when we consider potential utilization of colliery spoils or their possible environmental impacts associated with leaching of the soluble components. The variation in chemical composition of colliery spoils from different countries is shown in Table 12.4 (Gutt and Nixon 1979; Skarzynska 1995a). Silica (SiO_2) is the most abundant component and accounts for between 19 and 67% of colliery waste. Aluminum oxide (Al_2O_3) is the second largest component (8–37%), followed by ferric oxide (Fe_2O_3) (2–31%). The most important from the environmental point of view is the content of soluble salts as chlorides and sulfates.

Table 12.3. Mineralogical composition (%) of colliery spoils from selected countries

Minerals	Belgium	Czech Republic	Germany	Spain	UK	CIS	Poland
Illite	80	10 – 45	41 – 66	20 – 60	10 – 31	5 – 30	28 – 82
Kaolinite	12	20 – 45	4 – 25	3 – 30	10 – 40	1 – 60	9 – 65
Mixed layer illite-montmorillonite	0	0 – 0.5	–	8	6 – 18	6 – 40	0 – 5
Chlorite	5	0 – 15	1 – 3	0 – 7	2 – 7	–	0 – 10
Quartz	8	5 – 10	13 – 27	5 – 57	15 – 25	–	3 – 37
Pyrites	0.5	0 – 25	0.5 – 5	–	2 – 10	0.2 – 8	0 – 8
Carbonaceous matter	10	0 – 25	5 – 10	4 – 30	5 – 25	8 – 40	15 – 30

CIS, Commonwealth of Independent States (the former USSR).

Table 12.4. Chemical composition (%) of colliery spoils from several countries

Chemical components	Czech Republic	Spain	UK	USA	Germany	France	Poland
SiO_2	42 –52	38 –67	19 –67	43 –61	47 –60	58	19 –67
Al_2O_3	15 –24	8 –30	15 –27	14 –37	22 –27	26	15 –27
Fe_2O_3	3.5 – 6.7	2.4 – 9.7	3 –10	2 –31	5 –11	9	3 –10
K_2O	1.8 – 3.4	1.8 – 4.3	1 – 5	–	4.5	1	1 – 5
CaO	0.4 – 2.8	0.4 – 5.0	0.2 – 4	0.1 – 10	–	0.5	0.2 – 0.4
MgO	1.0 – 1.6	0.5 – 1.8	0.5 – 3.5	0 – 3	2 – 7	1.5	0.5 – 3.5
Na_2O	0.5 – 0.8	0.3 – 1.1	0.4 – 1.0	1 – 5.5		3.5	0.4 – 1
TiO_2	0.6 – 0.8	1.1 – 1.2	0.5 – 1.0	0.8 – 2.2	0.8 – 1.1	–	0.5 – 1
P_2O_5	0.21 – 0.25	0.2	0.5 – 0.25	–	–	–	0.5 – 0.25
SO_3	0.2 – 1.0	0.6 – 1.0	0.2 – 7.5	0 – 2	0.75 – 8.5	0.5	0.2 – 7.5

12.2.2.1
Chloride Content

Although it is recognized that the determination of chloride content is important, little attention has been paid to its concentration in colliery spoils. In the United Kingdom, however, routine tests are performed to ascertain the chloride level of colliery spoils to be used in civil engineering structures. Results obtained show that chlorides rarely exceed 0.1%, a level which satisfies British Standards for engineering soils (Rainbow 1987). Chloride levels of colliery spoils from Polish and Czech Republic coal fields indicate that significantly lower levels prevail, in the range 0.01–0.06%. If colliery spoils are stored in an uncompacted state, percolating rainwater can dissolve the chlorides and increase salinity to values higher than found in seawater, which could potentially damage both underground and surface water supplies (Talyor and Cripps 1984; Twardowska *et al.* 1990). In addition, when wastes are used as a civil engineering material it is necessary to determine how the chlorides may affect steel and concrete incorporated in the structures. High levels of chlorides could be disadvantageous. Reports indicate that sodium chloride and calcium chloride negatively affect both steel and concrete (Skarzynska 1995a).

12.2.2.2
Sulfate Content

The determination of sulfate content in colliery spoils is very important in determining their usability in the production of concrete and cement. According to the national standards of the majority of European countries, the content of water soluble sulfates should not exceed 0.2% SO_3; if wastes are to be stabilized with cement, the content of acid soluble sulfates cannot be higher than 1% SO_3.

Sulfates occurring mainly in claystones, mudrocks, mudstones and marls are in the form of calcium, magnesium or sodium salts. Sulfur is therefore often present in colliery spoils, usually in one or more of the following forms: ferrous sulfate ($FeSO_4$); magnesium sulfate ($MgSO_4$); calcium sulfate ($CaSO_4$); ferrous sulfide or pyrite (FeS_2). Pyrite (FeS_2) is undoubtedly the main form of sulfur occurring in colliery spoils. When exposed to atmospheric factors it is oxidized to ferrous sulfate ($FeSO_4$) and sulfuric acid (H_2SO_4) and this chemical reaction is strongly exothermic. Further oxidation produces ferric sulfate which in the presence of alkaline compounds eventually produces calcium, magnesium, sodium or potassium sulfate.

12.2.2.3
pH Value

Colliery spoils at the time of extractions are normally neutral or slightly alkaline, but as weathering proceeds the spoil may become increasingly acidic. This is due to the continuous oxidation of pyritic rocks lying on the surface of a spoil heap. The extent of oxidation of pyritic materials within the body of a tip is dependent upon the rate at which air can penetrate into the tip and this is often controlled by the particle size distribution of the material, the degree of compaction, and the degree of saturation. The products of pyritic oxidation are extremely acidic and water soluble, but they may

Table 12.5. pH values, moisture content and optimum moisture content of coal mining wastes from different countries

Country	pH (H_2O)	Moisture content (% dry weight)	Optimum moisture content (% dry weight)
Belgium	7 - 8	–	–
Czech Republic	7.2 - 7.8	–	–
France	9 - 11	8 - 9	–
Germany	5 - 7	5 - 8	9 - 19
Poland	4 - 7.5	4 - 18	6 - 19
Spain	–	2 - 14	7 - 11
UK	4 - 7.5	2 - 28	6 - 25
USA	–	4 - 10	7 - 9

be neutralized by reaction with other alkaline minerals. The effect of these neutralizing reactions produces a predominantly neutral or slightly alkaline material and this pH seldom falls below a value of 5 (Table 12.5).

12.3
General Geotechnical Characteristics

12.3.1
Introduction

The geotechnical properties of colliery spoils vary from source to source and often within a spoil heap. In general, coal mining wastes are comparable to boulder clays or clay-gravel-mix soils. The basic difference is in the formation process. The natural process of weathering and disintegration is replaced by mechanical action. Hence, because of the lack of natural selection, fragments of soft, easily disintegrating rocks are mixed with hard ones. Moreover, the exposure time of this material to atmospheric agents is extremely short when compared with the exposure time of natural soils. Another difference is in the amount of easily ignitable coal. Moreover, rapid relief of stress on rocks mined from a considerable depth underground will result in lamination and fissures in the rocks, thereby accelerating their future breakdown (Skarzynska 1995a).

It is well established that colliery wastes are a material in a transitional phase tending toward residual state. They are, therefore, susceptible to mechanical disintegration and physical and chemical weathering. These processes proceed in various stages, depending both on petrographic composition and on prior history. The residual product is an inert soil. Because of the variation in grain size, and addition of coal particles as well as different degrees of physical and mechanical weathering, properties of coal mining waste do not fully correspond to those of conventional soils. From a geotechnical point of view, natural soils are usually classified and characterized by two groups of properties, i.e. physical and mechanical. There are numerous properties within each of these groups. However, in civil engineering practice, only some of them are determined on the basis of the particular use of the soil. Generally, standard geotechnical parameters are used to describe the properties of coal mining wastes, but some modifications are needed, due to the specific character of this material. In this section, the basic geotechnical properties influencing the usability of colliery spoils for civil engineering purposes are described.

12.3.2
Physical and Mechanical Properties

Grain size distribution is a basic property that influences many other important geotechnical parameters. Grain size distribution is a direct result of the materials' petrographic composition, the utilized mining and preparation procedure and the types and magnitude of physical and chemical weathering. Initially, coal mining wastes can be classified as coarse-grained material where cobble and gravel fractions predominate and a shortage of the finer fractions occurs. The effect of mechanical factors and duration of weathering can significantly modify the original grain size distribution. It is clear from Table 12.6 that the disintegration of colliery wastes from the Upper Silesia coal field (Poland) increases with time resulting in a decrease of coarse fraction and an increase of finer fractions. The envelopes indicating particle size distribution of colliery spoils from different countries are illustrated in Fig. 12.2 (Skarzynska 1995a).

Table 12.6. Grain size distribution (%) of wastes from the Upper Silesia coal field (Poland)

Kind of wastes	Cobbles	Gravel	Sand	Silt and clay
Fresh wrought	37 - 80	15 - 53	2 - 6	2 - 4
From new tips	34 - 38	43 - 48	10 - 15	3 - 8
From old tips	4 - 18	46 - 59	20 - 21	2 - 30

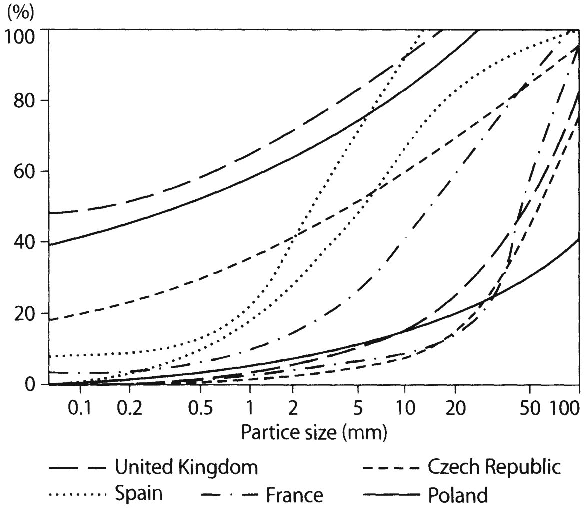

Fig. 12.2. Particle size distribution of colliery spoils from different countries

12.3.2.1
Moisture Content and Optimum Moisture Content

Since the absorption properties of individual rock particles are usually low, the moisture content of colliery spoils is mainly a function of grain size distribution and atmospheric conditions. The optimum moisture content for the maximum compaction of colliery spoils depends on grain size distribution. In fresh wrought wastes where coarse fractions predominate the moisture content is usually low, about 4–7%, but it may temporarily be slightly higher after prolonged periods of precipitation (Table 12.5; Skarzynska 1995a).

12.3.2.2
Unit Weight

The unit weight of colliery spoils is a function of the specific gravity and porosity of the rocks constituting spoil heaps. The moisture content of the wastes also has an influence on the value of unit weight. For example, the unit weight of Polish colliery spoils when loosely tipped ranges from 13 kN m^{-3} in freshly tipped material to 19 kN m^{-3} in old heaps (Table 12.7; Skarzynska 1995a).

12.3.2.3
Permeability

The permeability coefficient of colliery spoil depends on its grain size distribution and compaction. For fresh wrought material taken directly from the washery plants or from new tips, the coefficient is usually in the order of 10^{-5}–10^{-4} m s^{-1}. Higher compaction can decrease the coefficient of permeability by up to two orders of magnitude. Permeabilities of colliery wastes from some countries are listed in Table 12.7 (Skarzynska 1995a).

Table 12.7. Unit weight, permeability coefficients, and shear strength of colliery spoils from various countries

Country	Unit weight (kN m^{-3})	k_{10} (m s^{-1})	Angle of internal friction (degree)	Cohesion (kPa)
Belgium	20	–	–	–
CIS	6 - 17	–	–	–
Czech Republic	14 - 19	10^{-7} - 10^{-3}	28 - 40	0 - 10
France	–	–	20 - 45	–
Germany	15 - 21	10^{-8} - 10^{-3}	37 - 51	14 - 52
Poland	13 - 19	10^{-8} - 10^{-3}	26 - 50	2 - 80
Spain	–	10^{-8} - 10^{-6}	33	10
UK	8 - 21	10^{-11} - 10^{-4}	21 - 50	0 - 96
USA	–	10^{-7} - 10^{-5}	28 - 40	0 - 45

CIS, Commonwealth of Independent State (the former USSR).

12.3.2.4
Swelling

The swelling of colliery wastes depends on the petrographic composition, grain size distribution and on the intensity of physical and chemical weathering. The process of swelling depends mainly on: (1) inhibition of water by surface absorption on the edges and on the surface of clay mineral crystals as well as on sorption of carbonaceous matter in micropores. This kind of swelling is accompanied by parallel texture as well as the presence of montmorillonite and mixed-packed minerals of illite/smectite type and a considerable amount of organic matter and the expansion of rock material resulting from mining at considerable depth; (2) alteration of the clay minerals, that is, transition of illite to "swelling" minerals of the montmorillonite type; and (3) the presence of sulfides (e.g. pyrites) which are oxidized to jarosities and gypsum in developing incrustations along the surface of interbedding cracks.

The main cause of swelling is the absorption of water by the filling material and the transition of the sulfides during weathering. Swelling and soaking are directly related and fast soaking wastes with their low weathering resistance weather more quickly and have a high swelling capacity. Colliery spoils are susceptible to water inhibition so swelling should be taken into consideration when material is to be used in engineering structures.

12.3.2.5
Shear Strength

Shear strength, commonly determined in shear box apparatus, is defined by the angle of internal friction and cohesion. Both of these parameters depend on grain size distribution, degree of compaction and moisture content. For example, in the Polish colliery spoils the angle of internal friction ranges from 26–40 degrees in uncompacted materials to 29–50 degrees in compacted material, whereas cohesion ranges from 2–22 to 2–80 kPa, respectively. The shear strength parameters of coal mining wastes from various countries are given in Table 12.7.

12.3.2.6
Compressibility

The compressibility of colliery spoils can be characterized by the modulus of compressibility and the modulus of deformation. These are basic geotechnical parameters that determine the settlement of colliery wastes. These parameters depend on the compaction and moisture content as well as on the petrographic composition and grain size distribution of the material. Unfortunately, the compressibility of colliery spoils has not been seriously investigated because of technical difficulties. There are only a few data sets from laboratory and in situ tests. In general, the deformation and compressibility moduli of colliery spoils are relatively low and close to the moduli of cohesive natural soils of low plasticity. For example, at the loading of 50 kPa the values of original modulus of compressibility are 6–8 MPa, but with the increase of loading to 300 kPa the modulus values and variation increase to 17–75 MPa. This is probably due to the different materials and their compaction, because an increase of moisture content decreases friction among the rock fragments and thereby lowers the modulus value. The extent of weathering may also influence the values of moduli. Long-term compressibility tests showed that 50–70% of settlement occurs within the first 10 days and the remaining 30–50% within 70–340 days after loading. Furthermore, it has been observed that the moduli of compressibility are higher for freshly dumped than for weathered material deposited in old dumps. In summary, it can be concluded that colliery spoils, in general, can be used under certain conditions for engineering structures as a substitute for conventional soils.

12.4
Application of Colliery Spoils

12.4.1
Deposition

If not utilized immediately after mining, colliery spoils have to be dumped. Currently, huge quantities of coal mining wastes have to be deposited in numerous tips covering large areas. The deposition of colliery spoils creates both technical and ecological problems particularly in densely urbanized and industrialized areas. Procedures of spoil dumping have developed over recent decades. Initially, the stockpiles were formed in conical shapes of considerable height. Then in Poland, large piles with a flat upper surface at a height of 20 m were erected and the stepping of slopes was introduced when the total height was greater. More recently, and particularly in the UK, tips have been landscaped to mimic the shape of natural hills. They are blended into the surrounding area using landscaping criteria, thereby creating various possibilities for future use, including forestry, agriculture, allotments and leisure (recreation and sport). There are also other methods of depositing colliery spoils such as disposing of wastes from several mines in abandoned sand and gravel quarries (Poland) or in the sea (the UK). Recently, some countries (Germany, Poland) have engaged in the joint tipping of mine and domestic wastes (Fig. 12.3). Another method of depositing colliery spoils is to pack them back into the mines from which they come, using them as a fill. Currently, however, only small quantities of wastes are deposited in this way, primarily in India. Regardless of the method used, depositing colliery waste stockpiles affects the natural environment and some prevention measures should be undertaken.

12.4.2
Use in Civil Engineering Structures

A primary use of untreated colliery spoils is as bulk material for various types of earthworks. In fact, this is what these spoils are primarily used for, and this use can consume large quantities of waste materials. The use of mining wastes as a construction material for embankments of roadways, railways, rivers and dams has increased in the last 30 years. The use of this material in the construction of road embankments is well established especially in Germany, France and the UK, and recently it has been used for these embankments in Spain and Poland. Colliery spoils have been not only incorporated into embankments as the sole fill material but also used for sub bases and bases in combination with cement, or used to stabilize embankments by means of a variety of binding agents such as lime, bitumen or tar. Embankments designed for railways are fundamentally similar to those used for roads and two examples of railway embankment construction using colliery spoils are known in the UK.

The most common use of coal mining wastes is their application in hydraulic engineering. The material has been used extensively for construction of flood prevention structures, small dams, and dikes of industrial sedimentation ponds. Particularly in areas affected by mining subsidence, where outflow of surface waters is disturbed, large quantities of wastes have been used to construct river embankments to prevent flooding of adjacent areas (Poland, the UK). Colliery spoils have been also successfully used for construction of impoundment dams and dikes for hydraulically deposited pulverized fuel ashes from power stations. Ash lagoons have been constructed with the use of colliery spoils in the USA, UK and Poland.

Recently, colliery spoils have been more frequently used in the reclamation of derelict and contaminated land. Excavations caused by the removal of sand and gravel as well as land subsidence resulting from underground coal mining are filled with colliery spoils and then covered with layers of impermeable or organic soils, depending on the restoration aim. The important point in designing a landfill is the consideration of its future restoration. For instance, if the site is to be used as an industrial or residential development, high compaction is necessary to avoid excessive and unequal settlement. There are also several possibilities for the usage of colliery spoils in harbor engineering, seashore protection quays and caisson islands built offshore (Chuter 1980). Some small quantities of colliery spoils are also used for brick making and the production of lightweight aggregates (the UK, Poland), but these will not be considered as the proportion of the material used is very small.

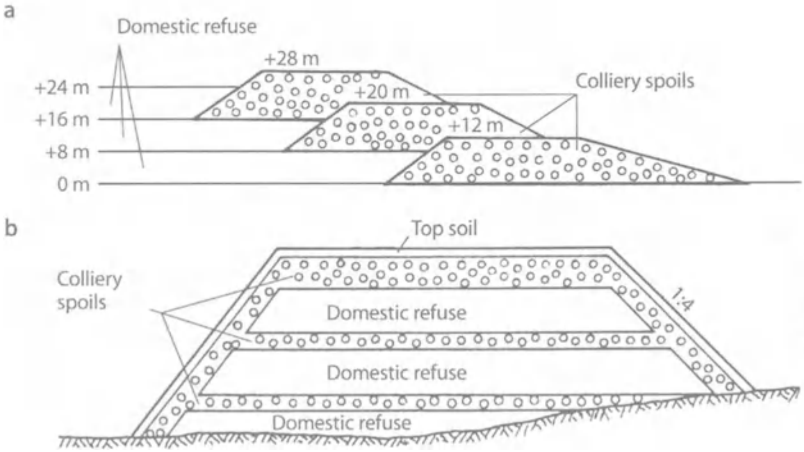

Fig. 12.3. Joint tipping of mine and domestic wastes: **a** Germany; **b** Poland

12.5
Environmental Impacts of Colliery Spoil in Dumps and Structures: Strategies for Mitigation

12.5.1
Risk of Spontaneous Combustion

The main components of colliery spoils are noncombustible. However, combustible coal rocks containing a large amount of carbonaceous substances, coal, carbonaceous interlayers and pyrite are also present. Moreover, other combustible materials, such as wood, rags, etc., can be found on dumping grounds. The main combustible materials in stockpiles are coal (15–20%) and pyrite (1%). Spoils originating directly from mining or sorting plants undergo grain segregation when loosely tipped on the dumpingground. This process is conducive to spontaneous heating and self-ignition of the material due to the easy access of air into the interior of the dumping ground. Shape and height of the dump also significantly affect spontaneous heating and self-ignition (Szafer 1985).

Two types of fire can occur in minestone dumping grounds: exogenic or endogenic. Exogenic fires result from an external source of heat, e.g. lightning strikes igniting the dump surface, poor protection from welding operations, dumping of hot cinders, etc. These fires usually can be more easily prevented by avoiding the initial sources. Endogenic fires result from the self-heating of carbonaceous matter or of pyrite, and in the final stage of the process, spontaneous ignition occurs. The expansion of smolder pockets can extend to a large part of, or even the entire, dumping ground. Endogenic fires occur when three main criteria are fulfilled: (1) easy access of air into the interior of the dumping ground, i.e. a sufficient amount of oxygen to enable burning; (2) the presence of a sufficient amount of material prone to oxidation; and (3) possibility of excessive heat accumulation in the dumping ground without proper dissipation, and thereby resulting in a gradual increase of temperature.

The self-ignition process on a dumping ground is probably due to the chemical reaction of pyrite, water and oxygen. Hydrogen sulfide is released and the characteristic "rotten egg" odor can be noticed on tips. Heat is produced as the temperature continues to rise. The hydrogen sulfide will burn if sufficient oxygen is present, producing sulfur dioxide. This is a gas which may also be formed directly from pyrite at higher temperatures. With a further increase in temperature, the coal and other carbonaceous material in the tip begin to burn. Consequently, this produces a rapid rise in temperature of the dumping ground, even up to 120 °C (Rainbow 1987). At the same time the oxygen content of the material is rapidly reduced and carbon monoxide is produced. Subsequently, this gas can react with other materials, producing free sulfur as well as other sulfur oxides. If the amount of the carbonaceous material is quite small, the rate of burning is reduced and the temperature decreases. The degree of temperature achieved, or sustained, depends on the rate of heat dissipation into the surrounding ground or air. Once the temperature rises above 50–70 °C, the rate of reaction increases rapidly and spontaneous ignition is likely to occur. Spontaneous combustion is dangerous to the surrounding environment because emanating gases, including carbon monoxide, sulfur dioxide and hydrogen sulfide, are highly toxic to humans, fauna and flora and the burning material can menace the surrounding urban and industrial areas, particularly when incorporated into engineering structures. Therefore, there is strong necessity to prevent the spontaneous heating and combustion of colliery stockpiles and structures.

An essential element of endogenic fire prevention is early detection of increased temperature within the materials. This may be monitored by temperature measurements at inspection sites, air composition analysis within the tip and, more often, by making temperature measurements at the surface of the dump with infrared thermometers. In particular, the prevention of spontaneous ignition can be carried out by using proper dumping technology, that is, dumps should be properly constructed and not just heaped (Cook 1990). Depending on the requirements of the amount of material dumped and on the future development, the shape, height and slope of the dump should be designed and constructed to meet strict density specifications. Up to 1960, all dumping grounds were formed from loose material. The effect of planned construction of the tips using motorized equipment to compact the material has almost completely eliminated the potential combustion of minestone in dumping grounds.

In Poland, two main practices are employed for the prevention of spontaneous combustion of colliery spoil. The first involves the mixing of the material containing potentially combustible elements with other materials which provide sealing material. These industrial spoils include power station ash, clay or soil from excavations. These materials should amount to more than 20–30% of the total volume of dumped spoils. Another option is interbedding the spoil tips with the above-mentioned materials in order to prevent any leakage of gas. The thickness of the colliery spoil interbedding layer should be more than 0.5 m and the stockpile should be covered by a layer of different material at least 1 m in thickness (Kawalec 1974; Szafer 1985). The second method is based on building spoil tips in layers, modifying the procedure as to whether they are above ground or beneath the ground level (Szafer 1985). The spoil tip is constructed in layers with layer thickness equal to the vertical reach of the compacting plant, e.g. 1–2 m. Compaction is accomplished by five passes of a vibratory roller having a mass of 7–10 tons and exerting a force of 20 tons. The angle of the spoil tip slope should be 15–25 degrees. The thickness of the compaction layers of waste fill will depend inversely on the content of combustible materials. Combustion of colliery dumping grounds, both above and beneath the ground level, is reduced or prevented by cutting off oxygen inflow to the deepest parts of the tips and to fire "nests". In Poland for fire prevention in spoil tips 10 m high, the upper surface as well as the slopes are compacted by vibratory rollers to a depth not less than 1 m and the inclination of slopes is reduced to 1 : 3.5. Whenever the height of the spoil tips exceeds 10 m and slope reduction is not possible, fire prevention is accomplished by building an adjacent band embankment which is compacted in layers and the upper layer of the spoil tip is compacted to 1 m. The adjacent embankment can also be built from colliery spoil.

Another method for the prevention of spoil tip fires is the application of different suspensions. In China and South Korea, an innovative method of extinguishing colliery spoil fires is the spraying of dumps with a layer of lime sludge. This sludge neutralizes SO_2 gas emitted from the burning dumps and eventually the fire is extinguished. Better results, however, have been obtained by mixing the lime sludge with sodium hydroxide (Grobbelaar 1988).

In the Commonwealth of Independent States, suspensions of lime are widely used to prevent spontaneous combustion as well as extinguish spoil tip fires (Zborshchik *et al.* 1986). Moreover, research has shown that it is possible to use lime suspension effectively to eliminate burning spoil tips by the cooling of the liquid which fills the interstices of the spoil mass. The calcium hydroxide reacts with the gases and dissolved substances formed as a result of the redox reaction. This method has been used in a number of collieries in the Donetsk coal basin. Extinguishing the fires in tips using a 5% lime suspension is achieved with vacuum pumps, injection pumps and shallow injectors. This involves treating the tip surface with lime suspension and, at the same time, creating a vacuum in the interstitial area. The gases and vapors are separately drawn off into a storage vessel. This technique has been further modified to rapidly eliminate the combustion sites within the tip, while the extinguishing proceeds. The lime suspension is injected into the tip by means of one of several injectors approximately 6 m long. The centrifugal pump draws the suspension through the intake pipe from the mixer and feeds it into the injectors along the injection line to the apex of the tip. Injection pressure of the suspension ranges from 0.008 to 0.015 MPa and is monitored by a pressure gauge. The intake pipe of the tip vacuuming system is placed over the extinguishing zone. The maximum effect of gas and vapor removal from the tip is within a vertical distance of less than 3–3.5 m between the extinguishing and vacuuming zone. The vapor emission process can last several days after the suspension has been injected. This is due to the gradual penetration of the suspension by gravity into the combustion locations within the tip. Some of the fluid injected is not converted to vapor and, as a result, fills up the filtration volume of the nonburning spoil tip. Also, some of the vapor formed by the high temperature condenses en route. Therefore, the amount of fluid injected into the tips differs from that of the vapor emitted. Control of the extinguishing process is recommended in order to prevent pollution of the atmosphere.

In Poland, extinguishing of burning tips is done by hole injection, using a seal pulp (Szafer 1985) which consists of a mixture of smokebox dust and bentonite with water in the ratio 10 : 1. The amount of water determines the density of the mixture and the density should be 14.7–15.7 kN m^{-3}. The pulp is forced under 5–6 · 10^5 atm. of pressure into the interior of the tips by means of steel tubes having perforations in the lower section. The tubes are placed in the holes, which are set in rows, not less than 5 m apart.

12.5.2
Influence on Surface and Groundwaters

Unburned colliery spoils contain water soluble chemicals, which can be dangerous to the environment (Herzig *et al.* 1986; Szczepańska 1987; Twardowska *et al.* 1988, 1990). Due to water filtration through the dumps or engineered structures constructed from colliery spoils the leaching of sulfates, chlorides and some heavy metals (Mn, Zn and Fe) occurs, resulting in contamination of underground and/or surface waters. Products of leaching are produced where surface precipitation or groundwater flows through the dumps.

The general topography of the ground surface determines the size and range of vertical motion of water in weathering material and influences chemical leaching. Erosion of steep slopes can be significant, causing the redistribution of weathering products and the uncovering of the underlying strata. Time is a significant element in forming and changing secondary weathering products. Freshly dumped wastes are always alkaline in nature, but with the lapse of time the alkaline minerals are leached and an acid medium may be formed because of pyrite oxidation. The change of conditions from alkaline to acidic is a side-effect of the production of the weathering products.

A detailed investigation of the influence of colliery wastes stored in dumping grounds and earth structures built from this material on surface and groundwaters has been conducted in Poland. Samples of water taken from rivers and wells situated close to stockpiles, and from piezometers installed in profiles perpendicular to the river embankments made of colliery spoils were chemically analyzed. The samples were taken in winter when the water level was low, during the spring thaw, and during summer and autumn.

According to Twardowska (1980), the concentration of sulfate in water extracted from a fresh, slightly compacted colliery spoil heap was 3000–4500 mg l^{-1}. In addition, it was observed that the sulfate content slowly decreased with time. The chloride concentration in mine water ranged from 287 to 3925 mg l^{-1}. Although the water filtrating through the embankment made from coal mining wastes and the water in the rivers had high amounts of chlorides, the level of chlorides in groundwater was low. In contrast, Mn concentration was relatively high (0.3–0.6 mg l^{-1}) in the groundwater.

Finally, it should be pointed out that for a dam under construction on a site away (150–200 m) from a colliery spoil damage zone, the inorganic anion content of water, in general, is at acceptable levels. Depending on well location, both chloride and sulfate concentrations are on a whole within class I water purity. Samples taken from the excavation site within the bowl differ in chloride and sulfate content depending on their proximity to the colliery stockpile. In general, the concentrations of the heavy metals were within acceptable limits. Unlike the chloride content, the heavy metal content of the filtration water through the embankments or in the rivers was relatively low. All water samples, regardless of location, were essentially neutral or only slightly alkaline as indicated by pH measurements.

Evaluation of these extensive tests made at different locations indicates that it is highly important to take necessary precautions for the protection of surface and groundwater when such coal mining wastes are being incorporated into engineering structures. The primary consideration is proper compaction and natural development. Depending on the size and type of the construction and local conditions, the use of nonpermeable screens and drainage is recommended for protection.

Along with the consideration of groundwater, observations have been made on the intrinsic vegetation in the dumps and earth structures built from colliery spoils. It has been noted that the growth reflects, to a great degree, the chemical changes taking place in the surface of the fill. It has been shown that advantageous changes will occur in structures made from colliery spoils when infiltration and erosion are limited, moisture content is decreased by evaporation, and outflow of chemical components is decreased (Skarzynska and Setmajer 1985).

Generally, it can be stated that, without supplementation, growth begins first at the bottom of the embankment where the fill is better irrigated. Growth is least on steep slopes as well as on higher sections or narrower surfaces of embankments. Moreover, it is recommended that certain species of grass, such as white goosefoot (*Chemopodium album*) should be included to produce a covering in fills and structures made from waste materials. It has been recognized that a good development of grasses from the family Papillionaceae and trees such as acacia (*Robinia pseudoacacia),* birch scrubs (*Betula pendula*) and American poplars (*Pepulus canadensis*) should be employed. The application of grass mixtures, which enrich the subsoil with nitrogen, and the planting of deciduous trees, which enrich the atmosphere with oxygen, have been recognized as the most beneficial treatments with respect to biological development (Nagawiecka and Krzanowski 1985; Kostuch and Nagawiecka 1993).

Pollution of the surface and groundwater occurs due to the dissolving of harmful substances in rain or thawed snow and ice, which flows off the structure, as surface water, or which infiltrates through the structure. The extent of water pollution by compounds derived from the body of a dam or embankment depends on the structure type and on the local water system. The dissolution of harmful substances occurs mainly near the surface of the bank, at a depth of 0.2–0.3 m, and is due to the weathering of the material. Leaching will be more rapid during the first 2–3 years of the life of the structure and it will decrease during the next 3–5 years until approaching a negligible amount after 10 years.

In order to prevent the leaching of harmful substances by rain and thawed waters, hydraulic structures made of colliery spoils should be: (1) constructed so that the water will quickly run off the surfaces of the crest and slope; (2) covered with turf so that the surface of the slope and the crest has been utilized for biological development; and (3) if necessary, constructed using hydrotechnical engineering methods to protect the structure on the waterside of the embankment. The intensity of leaching of harmful substances by surface water infiltrating through the body of the structure depends on the quantity of water flow. When the hydraulic gradient of infiltration is high during overflowing of rivers, pollution will be greater than during the regular, normal flow. The concentration of these substances will decrease with the time of structural life.

To evaluate the degree of pollution of surface or groundwaters, the following cases should be considered (Skarzynska and Michalski 1985; Skarzynska *et al.* 1993):

1. Flood embankments where the normal water level of the river is below the surface in the adjacent area (Fig. 12.4a). In this case, the groundwater movement is normally directed toward the watercourse and occurs, usually, below the embankment. Only periodically, during flooding, is the groundwater flow through the embankment. In view of this fact, the amount of water infiltrating from the watercourse to the adjoining groundwater is very low because the period of time of hydraulic loading on the structure is short when the material is being compacted in the embankment. In this case, it is not necessary to use any special protective measures.
2. Surface embankment where the water flows through ground depressions caused by mining subsidences in order to protect adjoining areas against flooding (Fig. 12.4b). In this case, the free surface water flows mostly over the groundwater level and constant filtration of water takes place through the body of the embankment. These waters stagnate or infiltrate the subsoil, thereby raising the groundwater level. Due to a constant hydraulic load on the embankment body, it is necessary to minimize infiltration by reducing the coefficient of permeability of the material in the embankment to a value lower than $k = 10^{-6}$ m s^{-1}. This can be done by increasing compaction or by using additional sealing on the vulnerable section of the embankment.
3. River embankment which simultaneously functions as a dike for sediment or deposit ponds (Fig. 12.4c). In this case, the free surface of flowing water in the river can be below or above the free surface of water in the nearby pond, and according to this, filtration can be toward the flow or toward the pond. If the direction of water filtra-

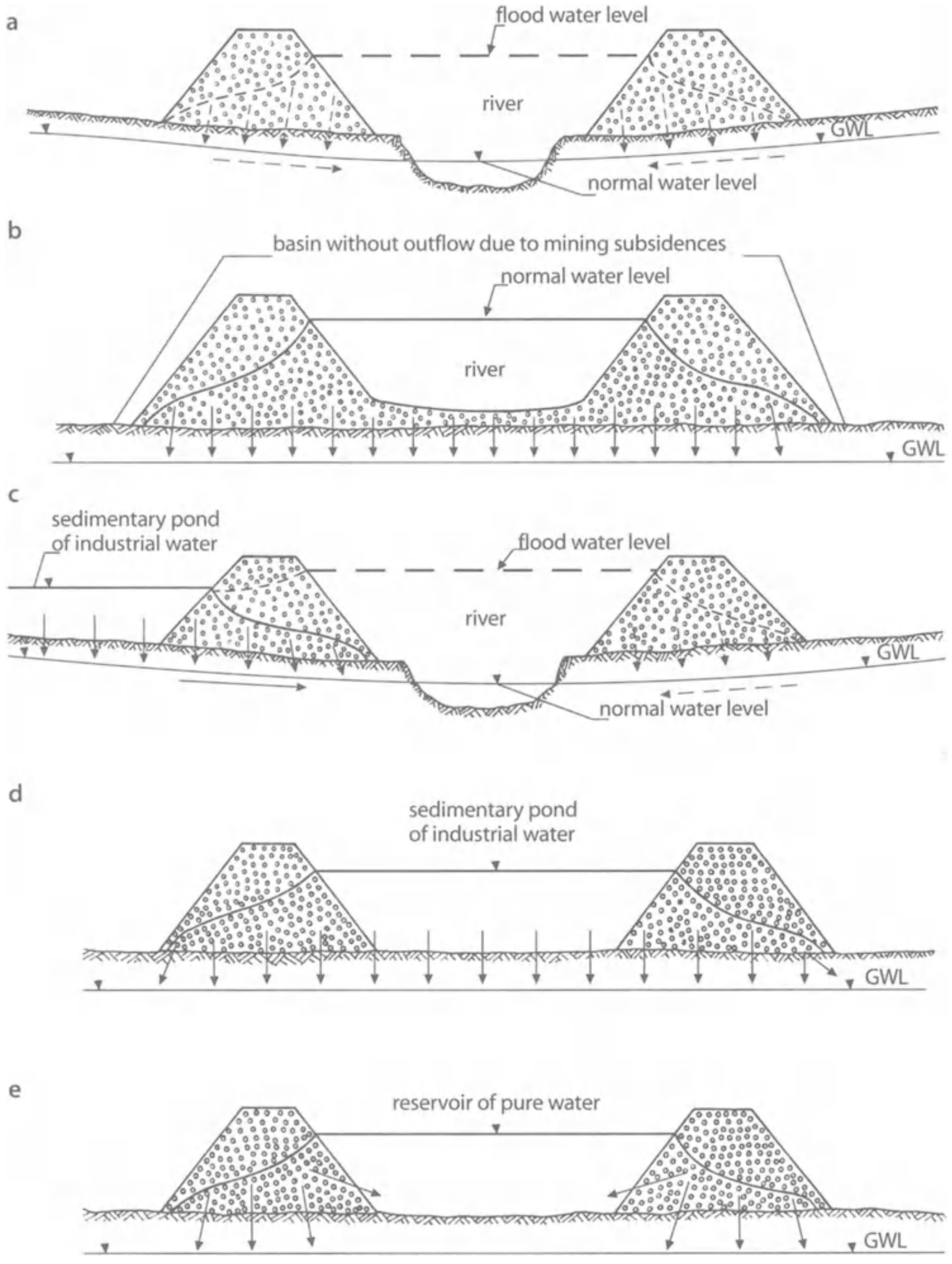

Fig. 12.4. Possible pollution of surface and groundwaters by hydraulic embankments made of colliery spoils. *GWL* groundwater level

tion is from the river flow to the sediment, conditional protection is unnecessary because sludge waters are as polluted as, or even more than, the water filtration through the embankment body. If water filtration is from the sludge sediment trap to the river flow it is necessary to limit infiltration by decreasing embankment permeability. An additional factor to consider is the effect of the fine sludge material sealing the slope on the pond side, which decreases embankment permeability.

4. Dikes of mining water reservoir or of industrial wastes which are dumped "wet" (Fig. 12.4d). In this case, the free surface of the water in the reservoir is always above the groundwater level, and thus infiltration often occurs through the dike body as well as through the bottom of the reservoir to the groundwater beneath. The permeability of the reservoir body should be limited because of constant hydraulic load. The silting-up of the waterside of the dike by the material collected in the reservoir is beneficial because it decreases the permeability of the fill. Usually, in such cases, drainage is designed to remove infiltrating waters.
5. Embankments for pure water reservoirs (Fig. 12.4e). In this case, the free surface of the water in the reservoir lies above the groundwater level and, thus, infiltration to the groundwater may constantly occur through the embankment body as well as from the bottom of the reservoir. Contamination of the groundwater and of the reservoir itself occurs continuously under the permanent hydraulic load of the fill. Therefore, separation of the water in the reservoir from the material used to form the body of the embankment should be considered when the structure is designed. For instance, application of sealing material, such as concrete slabs on the surface of the reservoir side, should be incorporated. Regardless of the type of drainage system, it is necessary to determine the concentrations of the various components in the collected water and, if necessary, use suitable purification equipment in order that concentrations will be within permissible levels.

In the case of river embankments, which are used simultaneously as dikes for sludge traps for mining water or for industrial waste dumped "wet", groundwaters are endangered mainly by infiltration through the bottom of the traps or the reservoir. Water present in these structures is usually polluted and its influence on the environment is much greater than the washing-out of harmful substances from the body of the dike.

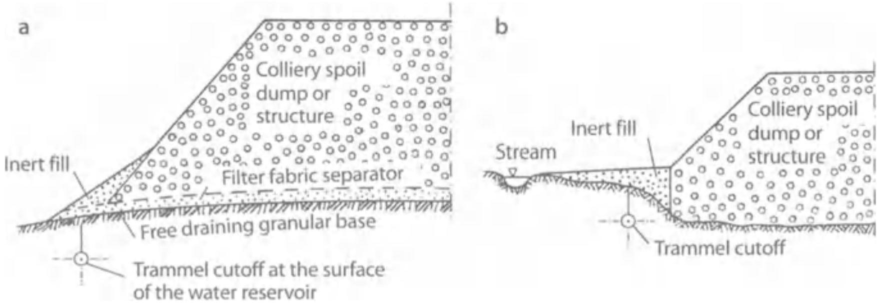

Fig. 12.5. Separation of contaminants of colliery spoil dumps or structures: **a** drainage leachate cut-off: **b** drainage as a separator at the base of a dump

When designing new dumps or structures from colliery spoils, polyethylene sheeting or clay screens should be incorporated along with drainage to remove the infiltrating water (Fig. 12.5). To prevent the contact of underground and filtrating waters through the existing dumps or structures, a drainage cut-off should be placed lengthwise or around the structure. In both of these cases, the flow of the water from the drainage should be directed to the water treatment station or to the watercourse to ensure that the contaminants are diluted to an acceptable level (Skarzynska 1995b).

12.5.3
Risk of Nuclear Radiation

Detailed investigations of the radioactivity of colliery spoils and of areas adjacent to stockpiles have been conducted in Poland to determine the risk of nuclear radiation. The investigations were comprised of laboratory determinations of the contents of the radioactive elements in samples taken from colliery spoils, soil and water as well as field measurements of background gamma radiation. In addition, radioactivity was determined in the laboratory to evaluate the relationship between the dose rate of gamma

Table 12.8. Radioactive element content of colliery spoil stockpiles and surrounding areas

Sample no.	Material	^{226}Ra	^{228}Th	^{40}K	^{137}Cs	Safety factors[a]	
		(Bq kg^{-1})				f_1	f_2
Samples taken from stockpile Przezchlebie							
1.	Colliery spoil	43.3 ± 2.5	42.4 ± 4.0	590 ± 10	<0.5	0.46	43.3
2.	Colliery spoil	48.5 ± 2.3	48.3 ± 2.0	625 ± 10	<0.5	0.51	48.5
3.	Soil	5.7 ± 1.8	6.5 ± 1.0	170 ± 5	36.6 ± 2.0	0.09	5.7
4.	Water	<1.5	<1.5	<1.5	<1.5		
5.	Water	<1.5	<1.5	<1.5	<1.5		
Samples taken from transport vehicles Colliery spoils from:							
6.	Krupiński mine	47.0±4.0	42.8±3.8	530±10	<0.5	0.52	47.0
7.	Zabrze mine	42.4±1.6	31.8±1.5	530±5	<0.5	0.44	42.4
8.	Makoszowy mine	41.1±1.6	39.8±1.5	615±5	<0.5	0.45	41.1
Samples taken from Anna colliery spoil stockpile and surrounding areas							
9.	Fresh wastes	117±14	69±4	730±48	<0.5	0.81	117
10.	Fresh wastes	121±15	60±4	650±44	<0.7	0.76	121
11.	One-year deposited wastes	117±14	55±3	590±37	1.0± 0.6	0.71	117
12.	Two- to three- years deposited wastes	115±14	55±4	620±42	<0.5	0.72	115
13.	Soil from recultivated part of stockpile	26±5	32±2	410±27	1.0±0.5	0.32	26
14.	Soil from meadow near Oder river	38±6	36±3	550±35	67±3	0.40	36

[a] Safety limits (ITB 1980): $f_1 < 1.0$; $f_2 < 185$ Bq kg^{-1}.

radiation measured by thermoluminescence in the field and by the radionuclide activity method in the laboratory.

The concentration of radioactive elements in colliery spoils is shown in Table 12.8 (Waligórski *et al.* 1994; Skarzynska 1995a). Included in this table are the calculated safety factors based on the Guidelines given by the Institute of Structural Engineering (ITB 1980) for production of building materials in constructions where humans will work or live. All waste samples from the stockpiles (with the exception of the Anna mine) as well as samples of materials transported to the dumping ground show that concentrations of the radionuclides are in the range of values reported for the lithosphere, i.e. ^{226}Ra 0–100 Bq kg^{-1}, ^{228}Th 0–100 Bq kg^{-1} and ^{40}K 0–730 Bq kg^{-1}. The concentration of cesium (^{137}Cs) in the soil samples is high compared to the other waste samples. ^{137}Cs comes from the Chernobyl radioactive fallout and is adsorbed by the silty soil particles (Eisenbud 1987). However, it is not present in the samples of the waste materials transported recently from the mines. As indicated in Table 12.8, the content of the natural radionuclides in all tested coal mining wastes and the adjacent areas does not exceed the acceptable safety factors of the ITB Guidelines for construction materials. Hence, these wastes are not radiologically hazardous for the human environment.

12.6
General Remarks

Data described in this chapter and from other sources allow a few remarks concerning the mitigation and remediation measures preventing the environmental impact of coal mining wastes:

When colliery spoils are not utilized at the time of mining, they have to be dumped. The vast majority of coal mining wastes have been tipped mainly on the earth surface. Recently, more of these wastes have been stored below the earth surface as an infill of abandoned sand and gravel quarries, or as a fill for land leveling (mining sub-sidences).

At the present time, a very popular and strongly recommended method of utilization of coal mining wastes is as a bulk fill for all types of embankments. Coarser grained wastes with higher permeability are more suitable for road and railway embankments, whereas finer material of lower permeability is more suitable for hydraulic embankments.

The most frequently recommended method of utilization of colliery spoils is to pack them back to mine excavations as a fill. Unfortunately, only a small proportion of wastes are utilized in this manner. Emphasis should be placed on further development of this method.

When colliery spoils are deposited on dumps, or used for land leveling, protection measures against spontaneous combustion and leaching of the pollutants should be undertaken. High compaction and covering of the dumping surface with natural soil are the main remedial measures which diminish, to a considerable degree, the risks of self-ignition and leaching of harmful substances. If leveled land contaminated with colliery spoils is to be used in an urban area, compaction is necessary for foundation purposes. In cases where groundwaters can infiltrate the colliery spoil, the sealing with clays or geomembranes should be applied to cut off the contact.

When colliery spoils are used for road and railway embankments, they should be protected against the rainwaters infiltrating through the embankment body to the groundwaters.

When colliery spoils are used for hydraulic embankments the prevention measures should be applied as indicated in Section 12.5.2.

Radioactivity of colliery spoils according to Polish and UK tests and regulations does not create a danger for human health. Nonetheless, since the data on radioactivity of coal mining wastes are not available in the literature, controlled measurements of nuclear radiation should be carried out when colliery spoils are to be used for fillings or earth structures.

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Section C

Study Cases

Chapter 13

The Swedish Acid Mine Drainage Experience: Research, Development, and Practice

Hans E. Gustafsson · Tom Lundgren · Manfred Lindvall · Lars-Åke Lindahl
Nils Eriksson · Hans Jönsson · Per G. Broman · Torbjörn Göransson

13.1
Introduction

Mining has been going on in Sweden for more than 1000 years. For many centuries Sweden used to be one of the main suppliers of steel, copper, and silver to central Europe. Over the years more than 1000 sulfide mines have been mined out and left without further attention. Most of the abandoned mines were small with waste deposits of fairly high grade. Concentration by flotation started in Sweden in the 1920s making it feasible to mine large ore bodies of lower grade which has resulted in a number of large deposits of tailings and waste rock from open pit mining. In 1994, the total volume of accumulated waste from sulfide mining was estimated at 300 million tons of tailings and 200 million tons of waste rock. A national survey on mine tailings and waste rock deposits identified about 70 sites causing severe acid mine drainage (AMD) problems in 1994. Out of these, 29 sites are considered of highest priority and are supposed to be attended to within 5 years. The total cost for the reclamation program was estimated at 300 million US$ in 1995. Today, 14 sulfide mines are in operation resulting in a yearly production of 20 million tons of tailings and 17 million tons of waste rock.

The Swedish Environment Protection Act came into force in 1969. The Act states, among other things, that the producer is responsible for remediation of installations and deposits in operation after 1969 when the Act came into force. A new act is under preparation and will require the producer to set aside funds for future remediation as early as during the production period. Mining is regarded by the Swedish EPA as one of the major metal hazards to the Swedish environment. Today, about 60% of all metal discharges to water from Swedish industry originates from mining and mine waste. It has been estimated that if no attention is taken leakage from mine waste deposits will increase tenfold until the year 2100, which will lead to increased metal impact on many environmental compartments. The Swedish government set some targets in 1990 to limit discharges of heavy metals to water: "Decrease by 1995 the discharges of Hg, Cd, and Pb by 70% and of other heavy metals by 50%".

13.1.1
Research and Development

The AMD problem in Sweden was first described and reported by Jacks (1976). The Swedish EPA carried out a research program in 1983–1988 called "Waste Deposits from the Mining Industry". Among the outcomes of the program were a better understanding of the importance of limiting oxygen diffusion through sulfide containing material, and the recommendation to use dry covers to limit the diffusion and percolation of oxygen and water.

Several models were developed for this purpose (Magnusson and Rasmuson 1984, 1985; Collin and Rasmuson 1986) and a full scale pilot research project was carried out under this research program – the Bersbo pilot project (described in Sect. 13.2.1).

The experiences from the research program including the pilot project and other international experiences, such as the Canadian Mine Environment Neutral Drainage (MEND) Program (see Chap. 9, this Vol.), resulted in a new research program now financed by the Swedish Waste Research Council (AFR). It was found important to continue research in the following areas: to increase knowledge of the weathering processes of sulfide minerals; to improve the tools for prediction of the transport processes in waste materials; to study the effects from alternative remedial actions; to develop cost-effective techniques of remediation.

Out of about 20 applications, the following seven proposals from Swedish research groups were accepted to be carried out from 1994 to 1996 with a total budget of 1 million US$:

1. Hydrochemical processes in mine waste deposits and drainage water – heavy metal speciation, sorption, and sedimentation processes (Lövgren, L. and Sjöberg, S., University of Umeå)
2. Long-term release of heavy metals from waste deposits at the atmosphere–geosphere interface (G. Destouni and S. Banwart, Royal Institute of Technology, Stockholm)
3. In situ investigation of a dry cover with biologically dense material at Galgberget, Falun (Hallberg, R. and Granhagen, J., University of Stockholm)
4. Geochemical studies of flooding as a treatment method of mine waste: in situ studies of weathering and transport processes at Stekenjokk (Öhlander, B., Ljungberg, J., and Holmström, H., University of Luleå)
5. Microbial speciation of metals in mine waste deposits – Fe-, Mn-, and sulfate-reducing bacteria may have a key role (Pedersen, K. and Ledin, M., University of Gothenburg)
6. Development of a method for investigation and monitoring of the integrity of sealing on mine waste deposits (Elming, . and Friborg, J., University of Luleå)
7. Effects of iron concentration on the adsorption of trace elements in systems with low pH and high iron concentrations (Håkansson, K. and Ledin, A., University of Linköping).

The projects have been reported and are available from the Swedish EPA, Stockholm.

Besides government funding, a number of research projects have been funded by mining enterprises. The following two projects are of special interest. First, the measurement of bulk physical parameters in waste rock at the Aitik mine resulting in specifications of dry covers on large waste rock deposits at the Boliden Aitik mine in Swedish Lapland (ANSTO, Australia). Second, a research program of the dry covers at the Boliden Saxberget mine in central Sweden co-financed by Boliden and the Swedish government (Terratema AB and Boliden Mineral AB).

The Swedish Foundation for Strategic Environmental Research (MISTRA) has decided to support a new research program called "Mitigation of the Environmental Impact from Mining Waste (MiMi)". The program will be carried out as a joint effort by six universities and two mining companies and will have a budget of 5.3 million US$ from MISTRA and an additional 2.7 million US$ from the mining industry. The program is planned to cover a period of 4 years, 1997–2000. The work within the MiMi program will be carried out as five research projects: (1) field studies and characterization; (2) laboratory studies of key processes; (3) predictive modeling; (4) prevention and control; and (5 communication and commercialization. The major goals of the program are:

- To establish a scientific basis for the development of economic and environmentally efficient prevention and control measures for mining waste disposal
- To establish an understanding of the fundamental governing processes leading to releases from disposed mining waste by combining available techniques for field studies, controlled experimental investigations, and development and improvement of predictive instruments in order to support future decisions regarding remediation
- To closely follow the development of new methods for mining operation which might have implications for the disposal of mining wastes
- To be able to predict the long-term evolution of the physical, geochemical, and biogeochemical processes governing the release of metals in disposed mining waste
- To develop strategies for finding efficient combinations of different prevention and control methods to achieve a high efficiency in each part of the near field system, e.g. combinations of physical and hydraulic barriers (covers), geochemical control measures, and suitable biochemical conditions.

13.2
Major Reclamation Projects

13.2.1
Ranstad Uranium Mine

Uranium bearing slate was mined at Ranstad in central Sweden during the 1960s. Rehabilitation of the tailings took place in 1990–1993 and was made with a multi-layered dry cover (Fig. 13.1). The area covered was 25 ha and the total costs for the dry cover was 7.7 million US\$ or 31 US\$ per m^2. The project was financed by the Swedish Nuclear Fuel and Waste Management Co.

13.2.2
Stekenjokk Base Metal Mine

The Stekenjokk high sulfide base metal mine in the Swedish Lapland was in operation from 1976 to 1988. The lack of suitable cover materials nearby resulted in the decision to cover the tailings by flooding. The rehabilitation of the 150 ha tailings area was completed in 1992 and the total cost for the wet cover was 2.3 million US\$ or 1.5 US\$ per m^2. The project was financed by Boliden Mineral AB.

13.2.3
Galgberget, Falun Mine

Mining had been going on in the Falun mine for about 1000 years until the mine was closed in 1993. Galgberget is an old tailings deposit that was rehabilitated (1990–1996)

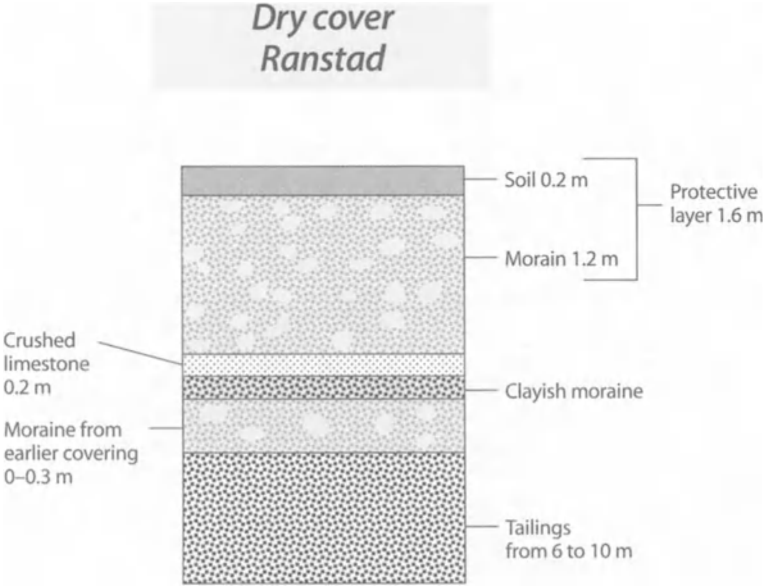

Fig. 13.1. Design of the dry cover applied in Ranstad. Tailings were covered with 0.3 m of moraine, 0.2 m of crushed limestone and 1.6 m of protective layer with soil and moraine

Fig. 13.2. Design of the dry cover applied in Falun. Tailings were covered with biosludge and fly ash, and 0.5 m of protective layer with moraine and forest waste

utilizing biological sludge from a nearby paper mill mixed with fly ash to form a dense layer under a protective cover of coarse moraine and wood waste (Fig. 13.2). The area covered was 20 ha and the total cost of the cover was 2.3 million US$ or 11.5 US$ per m^2. The project was financed by STORA Kopparberget AB.

13.2.4
Kristineberg Mine

A complex sulfide ore at the Kristineberg mine in northern Sweden has been mined since 1940 until 1992 when the concentration plant was closed. The mining operation left a number of tailings areas of which three were rehabilitated in 1995–1996. Besides dry covers with clayish moraine for the dense layers and coarse moraine for protective covers, Boliden has developed a new method called "saturated cover" (Fig. 13.3). The tailings where the method is being applied have only the upper 1–3 m unsaturated. By applying a 1-m layer of moraine the unsaturated parts will be saturated by help of the capillary forces according to the plan presented. The saturated covers are being monitored and the results will be reported.

- Tailings area #1: 8.6 ha, 1.0 m of moraine, 4.8 US$ per m^2 (saturated cover); 6.4 ha, 0.3 m dense layer of clayish moraine, 1.5 m of moraine, 7.2 US$ per m^2. Total cost 1.0 million US$
- Tailings area #1B: 4.1 ha, 0.3 m dense layer of clayish moraine, 1.5 m of moraine, 7.1 US$ per m^2. Total cost 0.3 million US$
- Tailings area #2: 7.6 ha, 1.0 m of moraine, 4.6 US$ per m^2 (saturated cover). Total cost 0.6 million US$.

The cost per square meter is determined by several factors, among them the year of rehabilitation, the area rehabilitated, and the location of the deposit. The inflation on

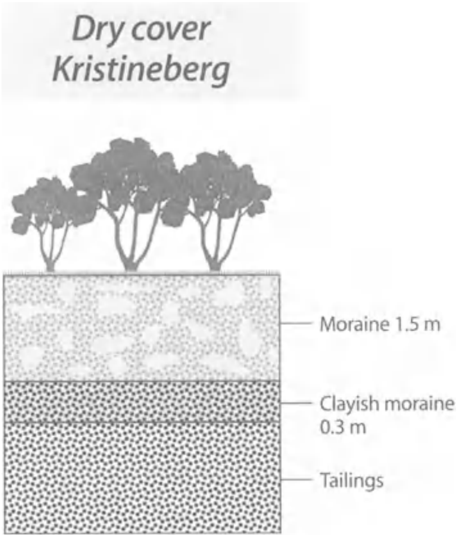

Fig. 13.3. Design of the dry cover applied in Kristineberg. Tailings were covered with a 0.3 m layer of clayish moraine and 1.5 m of moraine

Table 13.1. Major rehabilitation projects, indicating location of deposit, area rehabilitated, and cost per square meter

Location	Year of rehabilitation	Area (ha)	Cost (US$/m^2)
Bersbo	1987–1990	5.7	28
Ranstad	1990–1993	25	31
Stekenjokk	1990–1992	150	1.5
Falun	1990–1996	20	11.5
Saxberget	1994–1996	50	12
Kristineberg	1995–1996		
Dense layer		14	7.2
Saturated cover		15	4.8

earth moving in Sweden from 1987 to 1996 was about 50%, but the inflation indexes have not been considered when comparing the costs of the different projects. Nevertheless, these figures give a fair indication of the costs of the different covers used (Table 13.1).

Generally, the efficient function of a cover depends on the permeability of the materials used for the cover. To give some general indications from the results obtained, monitoring of oxygen inside the covered tailings showed effective oxygen diffusivity values in the range of $5 \cdot 10^{-9}$ to $2 \cdot 10^{-10}$ m s^{-1} at Falun, Bersbo, and Saxberget. The reclamation projects carried out in Bersbo, Saxberget, Aitik, and Stekenjokk are presented in detail in the following sections.

13.3
Bersbo Pilot Project

13.3.1
General Considerations

The sulfidic waste rock piles in Bersbo, southern Sweden, were subjected to remediation measures during 1987–1989. This was the first remediation project addressing old sulfidic mining wastes in Sweden and one of the first in the world. The project cost about 5 million US$ and was financed mainly by government funds, but the municipality of Åtvidaberg and the landowner (Barony of Adelsvärd) also took part in the financing.

The overall objective of the project was not only to reduce the discharges of AMD but also to perform a full scale covering project, including the selection of design criteria, construction, quality control, and monitoring. Thus, the Bersbo project became a pilot project, not only in the decommissioning of mining wastes but also in the decommissioning of waste deposits in general.

The performance and efficiency of the measures have been followed up since 1989. As no monitoring technique was established at the start of the project, the follow-up studies and the monitoring work, in this respect, also became a development project.

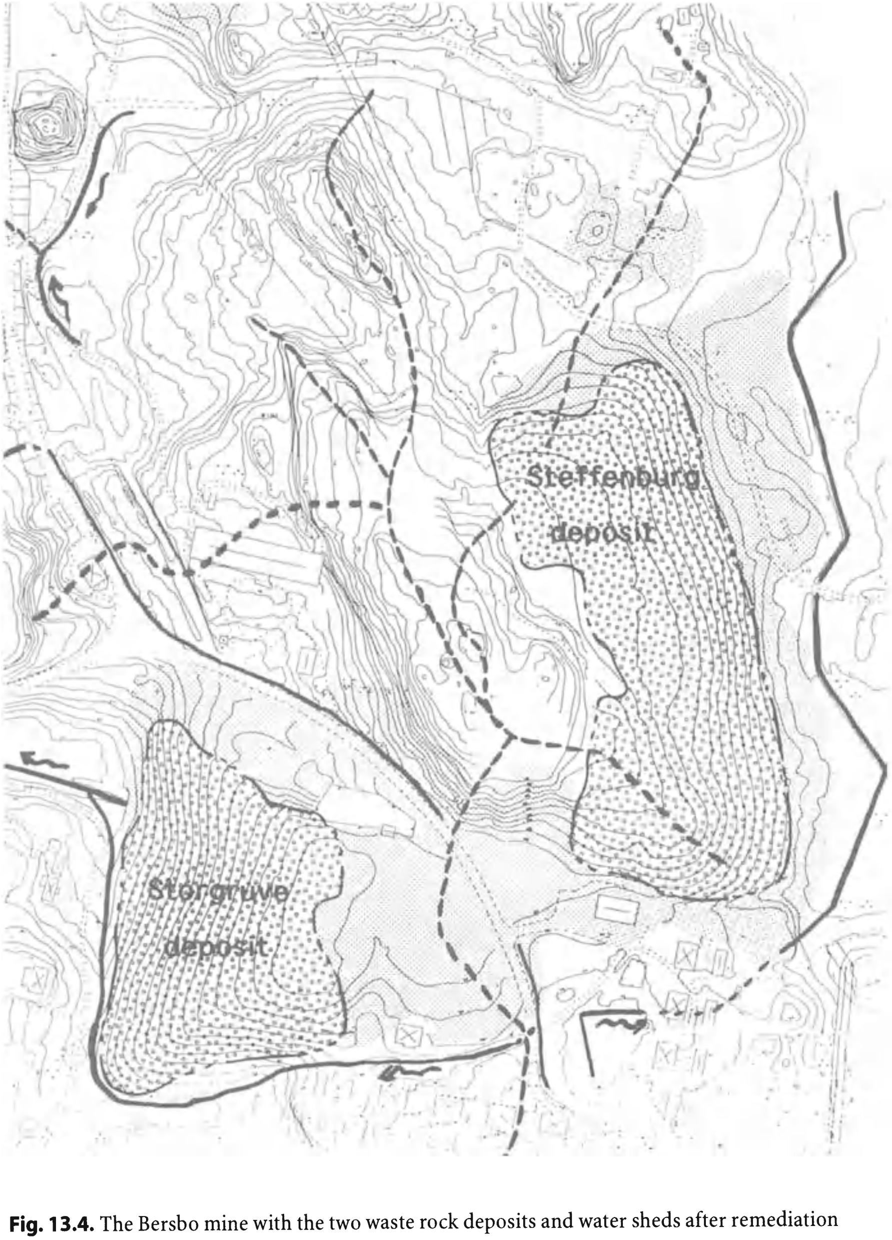

Fig. 13.4. The Bersbo mine with the two waste rock deposits and water sheds after remediation

Based on the monitoring results, minor supplementary sealing operations took place in 1995–1996. The project and follow-up studies have been regularly reported (Södermark and Lundgren 1988; Lundgren and Lindahl 1991; Håkansson *et al.* 1994).

The remediation works in Bersbo can be divided into three main actions: (1) size reduction and remediation of waste rock pile areas ("reshaping and rinsing"); (2) disposal of waste rock below the groundwater level in existing mine shafts ("wet cover"); and (3) covering two concentrated waste rock piles with a water and oxygen barrier ("dry cover"). This section concentrates on the third action.

Before covering, all waste rock (700 000 m^3) was concentrated in two deposits, the Storgruve deposit (23000 m^2) and the Steffenburg deposit (34000 m^2) (Fig. 13.4).

13.3.2
Capping Measures

About one third of the waste volume was disposed off in the shafts and the remaining part in two deposits, both of which were covered with a sealing layer and a protective layer (Fig. 13.5). On the Storgruve deposit, the sealing layer was constructed with a concrete-like product (Cefyll®) based on pulverized coal fly ash, cement, and water which was blended into a paste and grouted into a 0.25 m thick layer of crushed rock aggregate. On the Steffenburg deposit, the sealing layer was constructed from a local dry crust (native) clay found some 2 km from the site. The clay was applied on a thin filter layer of tailings sand and was compacted in three lifts. Both sealing layers were protected by a 2 m thick layer of a local glacial till which was found very close to the clay pit.

Before capping, the net percolation on the coarse waste rock deposits was estimated at 500 mm $year^{-1}$. After capping, the percolation through the cover has been calculated at 60 mm $year^{-1}$ on the Storgruve deposit and 15 mm $year^{-1}$ on the Steffenburg deposit. The cover is estimated to change the total annual water balance as shown in Table 13.2. On the Steffenburg deposit it has been possible to measure the collected discharge from the deposit through a siphon tube installed in the waste rock at the lowest point of the basement of the deposit. Steady state in the discharge was recorded about 4 years after

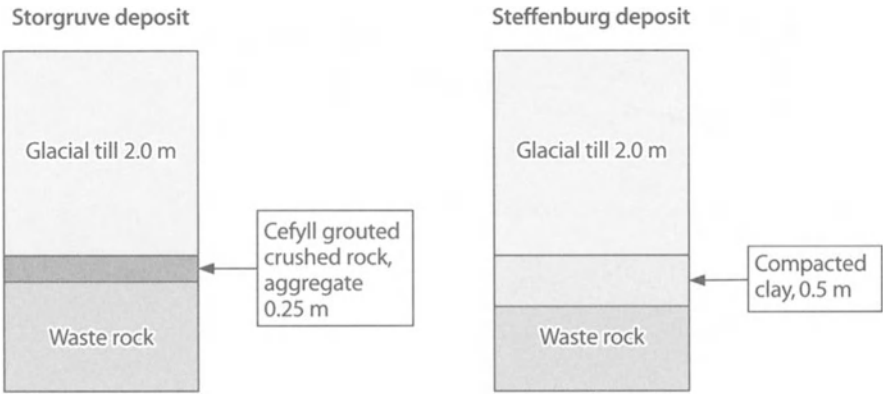

Fig. 13.5. The cover concept for the two waste rock deposits in Bersbo

Table 13.2. Estimated annual water balance (mm $year^{-1}$) for the two waste rock deposits in Bersbo

	Precipitation (corrected)	Evapo-transpiration	Run-off	Percolation
Storgruve deposit (with Cefyll® liner)				
Before remediation	570	70	0	500
After remediation	570	380	130	60
Steffenburg deposit (with clay liner)				
Before remediation	570	70	0	500
After remediation	570	335	220	15

the cover was completed. During that period, the groundwater level in the deposit and above the native clay layer on which the waste was dumped rose about 1.5 m. On the Storgruve deposit it has not been possible to separate the discharge from the waste deposit from that of the mine. The joint outlet is arranged through a water lock to prevent air penetrating the deposit. During the supplementary sealing operations in the winter of 1995–1996 pumping was carried out to access the mine and during that period the water level in the mine dropped below the discharge level, decreasing the discharge from 0.4 to 0.05 l s^{-1}.

13.3.2.1
Oxygen Transport

Oxygen was measured in thin tubes that were installed in the deposit prior to capping. Most of the tube ends were located only 0.5–1 m below the waste rock surface. In both deposits, two to three reference tubes end at deeper levels. However, the recordings showed no significant variations in oxygen concentration with depth. Before covering, the oxygen concentration was close to 21% in both deposits and its transport was governed by advective transport processes. The oxygen supply did not become a limiting factor for oxidation until the cover was applied, and 1 or 2 months after all the cover was applied, the oxygen concentrations were below the detection limit of the instrument.

In 1991, the oxygen concentration was found to have increased in both deposits and also to vary during the season. Since then, monitoring of the oxygen concentration in the deposits has been carried out regularly on a monthly basis and the seasonal fluctuation has been established (Fig. 13.6). The variations are greater with time and area than with depth. The mean oxygen concentration in 1991–1995 was 3.2 and 0.44% for the Storgruve and Steffenburg deposits, respectively. The difference is due to the higher hydraulic conductivity and lower saturation degree of the cementitious liner on the Storgruve deposit.

In both deposits, there is a relatively small area with a high variability in oxygen concentration. As a matter of fact, all peak values in the Steffenburg deposit are related to a small, circular area with an electric power supply pole. This area had to be covered with Cefyll® before any other remediation activity was started on this deposit and before the clay pit was opened. In the Storgruve deposit, the high concentrations and

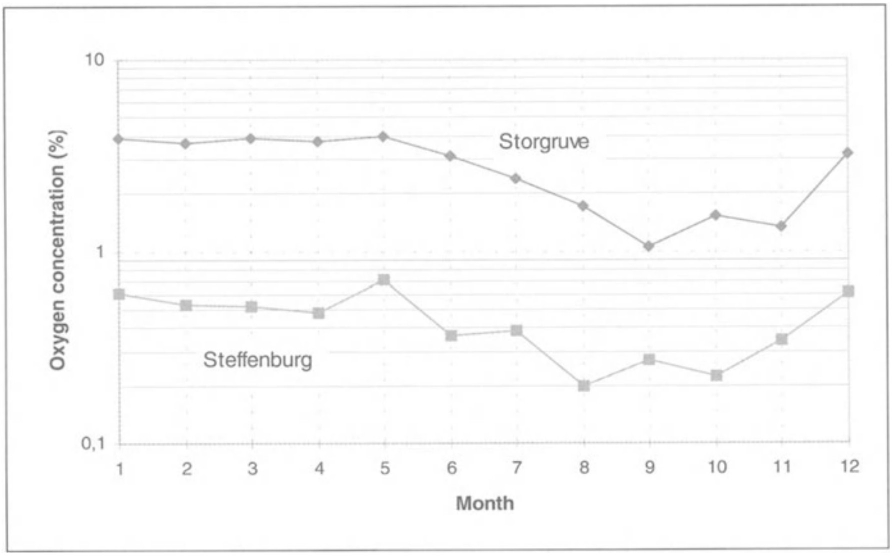

Fig. 13.6. Monthly mean values of oxygen concentration in the two deposits, expressed as the average of all monitoring points (1992–1996)

heavy fluctuations are related to the contact zone between the deposit and the mine. Intensive geophysical investigations took place in Bersbo during 1995–1996, indicating a narrow linear anomaly close to the Cefyll® plate on the Steffenburg deposit (Bergström 1995).

13.3.3
Discussion and Conclusions

13.3.3.1
Water Balance

As indicated by the lower saturated hydraulic conductivity of the material, the results suggest that the compacted clay liner ($K_s = 1 \cdot 10^{-10}$ m s^{-1}) provides a better oxygen and water barrier than the Cefyll® liner ($K_s = 1 \cdot 10^{-9}$ m s^{-1}). The relatively smaller difference for the two materials in water sealing properties than in oxygen sealing properties probably rests with the fact that the Cefyll liner is never completely saturated which results in lower conductivity to water. The unsaturated conditions in the Cefyll liner also result in a higher diffusivity to oxygen which explains the higher oxygen concentrations under this cover. All conditions including pore pressure measurements indicate that the clay liner is more or less constantly saturated.

Dry covers on waste rock piles with high slope gradients are subjected to more runoff than are those on flat tailings impoundments. Waste rock is almost never saturated with water while substantial parts of tailings often are. Hence, in general, dry covers on waste rock piles need to be more efficient as both water and oxygen barriers than dry covers on tailings. The results from the Bersbo pilot project indicated that waste rock piles, in the Scandinavian type of climate, should not have a saturated permeability much lower than that of the Storgruve deposit. Even with permeability of about $3 \cdot 10^{-9}$ m s^{-1} ($1 \cdot 10^{-8}$ m s^{-1} in full scale) the water barrier effect is almost zero and the oxygen barrier is probably only effective during short time periods in nordic climates.

13.3.3.2

Oxygen Transport and Oxidation Rates

Table 13.2 shows that the discharge flow rate of AMD has been reduced by about one order of magnitude. From predictive calculations, the reduction of oxygen transport due to the capping should be several orders of magnitude (Rasmuson and Collin 1986). This should be reflected by lower oxidation rates. The annual oxidation rate of pyrite in the covered deposits has recently been calculated at less than 1 g m^{-2} surface area of the Steffenburg deposit and less than 5 g m^{-2} surface area of the Storgruve deposit (Collin 1997). This represents only 0.1–0.5% of the estimated oxidation rate prior to covering (Collin 1987) and corresponds to an oxygen concentration of 0.05–0.5% in the deposit depending on the assumed reaction rate constant ($1.4\text{–}17 \cdot 10^{-8}$ m^3 gas m^{-3} waste s^{-1}).

Since the oxygen concentration in the Storgruve deposit has often been higher than 0.5%, the oxidation rate in this deposit (as well as in parts of the Steffenburg deposit) could be underestimated by the calculations. However, in this deposit it is obvious that air leakages have occurred during monitoring and that they play an important role for the oxygen transport to the waste. The resulting effect of the supplementary sealing operations will be interesting to follow. Variability in oxygen concentrations with time in covered mining waste deposits has been reported in the Rum Jungle rehabilitation project (Northern Territory Department of Mines and Energy 1986; Ritchie 1995). In Rum Jungle (tropical climate) a waste rock deposit was covered with a 0.5 m of compacted clay liner. The first paper reports a significant and sudden general drop in oxygen concentration which, after about 1 year, rose again to high levels. Diurnal changes in oxygen concentrations in the order of 7–8% units are reported to occur at a few points in the Rum Jungle deposits and are considered to be due to variations in air pressure relative to the atmospheric pressure. Efforts are currently being made to study the influence of air pressure variations on oxygen transport to the waste rock in Bersbo.

13.3.3.3

Generation of AMD

By using the actual field monitoring values of the water balance and oxygen concentrations, the effective field oxygen diffusivity was calculated at $<2\text{–}4 \cdot 10^{-9}$ $m^2 s^{-1}$ for the cover of the Storgruve deposit and $<2 \cdot 10^{-10}$ $m^2 s^{-1}$ for the Steffenburg cover as dry period values (Collin 1996). This value is approximately one order of magnitude higher than what was recorded in oxygen diffusion cells 1–2 years after completion of the cover. The cells were installed beneath the cover prior to the installation of the sealing layer. The field values are recordings integrated over almost the whole year and should therefore be significantly lower than values recovered from dry periods.

The total efficiency of the covering operation should eventually be measured as the reduction in the short- and long-term transport of contaminants. This section covers only the physical parameters, while water quality conditions have been described by Håkansson *et al.* (1994). Water quality terms could roughly be mirrored by the physical parameters if the reductions in AMD flow rate and oxidation rate are integrated. However, there are two mechanisms that are difficult to integrate: leaching kinetics and oxidation kinetics. Oxidation can take place also in the anoxic state whereas leaching is a complex process which may change due to factors such as pH, reduction potential, microbial activity, and available complexing agents. Thus, the actual reduction in metal concentrations in the discharge will be element specific, site specific, time dependent, and not necessarily proportional to the reduction in oxygen transport rate or even to the reduction in oxidation rate.

The oxygen transport rate through the clay liner is low and therefore also the oxidation rate of the waste rock under this cover must be low. This is verified by a substantial decrease in metal concentrations in the discharge from this deposit. The oxygen transport rate through the Cefyll® is higher than through the clay liner which is caused by the not fully moisture saturated conditions in the sealing layer and the discrete air leakages via the mine galleries. The water discharge from the Storgruve deposit is a common outlet for drainage the waste rock deposit and the mine. This water contains leachate from oxidized waste rock dumped of the ferric iron which is reduced into ferrous iron resulting in a continued AMD formation. Hence, the high metal concentrations in the discharge from this deposit are reflecting not only the situation in the deposit.

Significant seasonal variations occur in the oxygen concentrations just below the sealing layers on both deposits. The average amplitude is in the order of 400% of the minimum value for both deposits. Irregular peaks in oxygen conentration which do not correlate with the variability of the groundwater levels in the protective layers, indicate that diffusion is not the only mechanism for the oxygen transport through these covers. Advective transport mechanisms related to the identified air leakage paths could be involved.

Despite different problems with air leakages, the AMD discharge has dropped from both deposits. So far, the compacted clay liner constitutes a better barrier than the Cefyll®. However, a much lower saturated permeability (1×10^{-9} m s^{-1}) than that of this cementitous liner is not recommended in climates similar to that in Sweden, especially not on sloping surfaces where run-off is substantial and capillary barrier effects could be active.

Reduction of discharge flow rates from the two Bersbo deposits has been estimated at 97% in Steffenburg and 88% in Storgruve. The metal concentrations of the primary discharge from the Storgruve deposit are still about the same as before the remediation, while those of the Steffenburg deposit are reduced significantly (Sandén, pers. comm.). Hence, in items of metal transport, the efficiency of the cover at the Storgruve deposit should be in the order of 90% and for the Steffenburg deposit >99%.

Both the cover based on a compacted clay liner and the cover based on a Cefyll® liner act as good water barriers. The clay liner also acts as a good oxygen barrier. The water percolation has been reduced to less than 3% for the clay liner and around 12% for the cementitous liner. The full scale permeability is about five times higher than the small scale permeability for the clay liner but, surprisingly, almost two times lower for the cementitous capping, the latter indicating capillary barrier effects causing unsaturated conditions in some parts of the sealing layer. When setting the standards for a dry cover and specifying the sealing material properties as the saturated permeability, the difference between full-scale conditions and small-scale conditions is evident.

13.4
Reclamation Project at Saxberget Mine

13.4.1
General Considerations

At the time of the closure of the operations in 1988, the Saxberget deposit, which was discovered in 1880, had been mined for just over 100 years. Although limited in tonnage, it was one of the richest ores ever mined in Sweden, with combined Zn and Pb grades reaching up to 15%. In addition to complying with the operation permit, valid during the production period, responsibilities for reclamation had to be considered. The interpretation of the current law (Environment Protection Act of 1 July 1969) was subject to negotiations between Boliden and the Swedish EPA. The outcome was an agreement stating that remediation of the waste deposits and other objects generated or maintained in use after 1969, the year in which the above-mentioned act became effective, should be financed by Boliden and older remnants should be the responsibility of the community, i.e. funds available within the Swedish EPA.

A steering committee for the reclamation work was formed, with representatives from Boliden, the EPA, and the town of Ludvika. According to the statutes, the chairmanship of the steering committee was alternated between the company and the authorities. It was agreed that the costs up to a certain level should be shared equally between the company and the community. A project budget of about 4.2 million US$, well within the limit for the cost-sharing agreement, was set up for the reclamation of the tailings ponds.

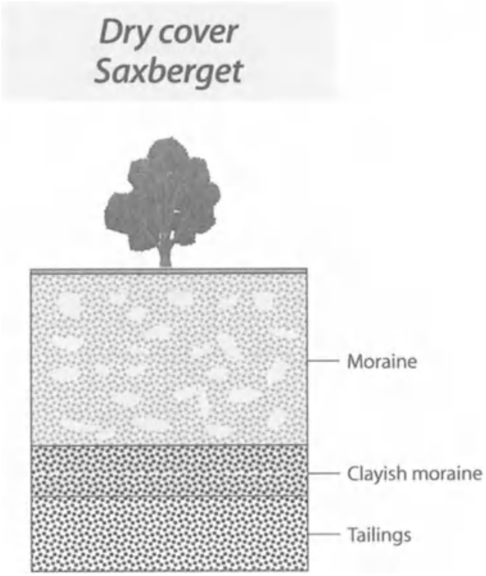

Fig. 13.7. Design of the dry cover applied in Saxberget. Tailings were covered with a layer of clayish moraine and one of moraine

Although limited in tonnage, the older waste proved to be responsible for the majority of the metal transport to Lake Saxen at the time of the mine closure. The old deposits in the Saxdalen community consisted mainly of mill tailings and roasting residue along the creek with metal content exceeding 1% of Pb and Zn. About 100 000 m^3 of old waste rock was removed and deposited in the mill tailings pond. Some quantities of contaminated soil from gardens established on old tailings and metallurgical waste were also included in the reclamation project. After clean-up, the area was covered with fresh soil.

Since 1930, the mill tailings have been deposited in a nearby valley north of the mine. Two separate ponds have been used for different periods, the West pond (18 ha) from 1930 to 1958, and the East pond (35 ha) from 1958 to 1988. In total, the tailings amount to 4 million tons, with a composition of about 2% S, <1% Zn, and 0.5–1% calcite. This mineral composition suggests that the material is potentially acid-generating even though the tailings produce a circum-neutral pH drainage at present.

The ponds are located on a permeable glacial formation, which was predicted to cause a rapid fall of the water table as soon as the supply of tailings slurry ceased. Large amounts of tailings would then be exposed to atmospheric oxygen. During the production period, the mobilization of Zn was estimated at 3 tons per year. Studies showed that after depletion of readily available buffering minerals, the pollution load would increase considerably if the oxygen supply to the material could not be controlled.

13.4.2

Reclamation of Tailings Ponds

Modeling of the future mobilization of metal estimated an annual mobilization of up to 600 tons of Zn in the ponds. Due to precipitation and adsorption processes at neutral pH levels, the amount was estimated to stay at 3 tons per year as a net transport for several years to come. However, the predicted high pollution load called for remedial actions. As the hydrogeological situation excluded flooding of the ponds, the only realistic option remaining was a cover designed to reduce the oxygen transport to the tailings.

As the proposed project would be the second one of its kind in Sweden, and certainly the largest, there was no real practical experience at the time of developing the remediation plans. Therefore, a number of options had to be investigated. In general terms, the cover was designed in accordance with principles defined within the Swedish EPA's investigation program aiming at long-term, low maintenance remediation solutions for mining waste. This called for a cover with at least two components, one low permeability sealing layer, and one protective layer on top of the sealing layer (Fig. 13.7). The key component was the sealing layer. For this purpose, a number of solutions were considered. One of them was compacted municipal sewage sludge, which was found to possess favorable hydraulic properties but for practical reasons, mainly the time factor, this alternative was rejected.

Modeling of oxygen and water transport coupled to solubility calculations permit the estimation of metal transport. Based on these calculations, the specifications for the permeability of the sealing layer were established: 0.3 m with a permeability of $5 \cdot 10^{-9}$ m s^{-1}. The extent of the protective layer was subject to discussion. Boliden claimed that 1 m of unclassified till would constitute sufficient protection against frost and root penetration. The EPA argued in favor of a thicker cover, and finally it was agreed that a 1.5 m protective layer should be used. The lay-out of the tailings area was designed to adjust to the surrounding landscape as much as possible. Surface run-off water is led in a small creek winding along the West pond. The drainage from the West pond overflows to the East pond and forms large areas of shallow wetlands. In this way, water saturation is maintained in the sealing layer and it gives the area an attractive and varied appearance. Excess water is discharged through a stone paved outlet down the former dam slope.

13.4.2.1

Cover Application Experience

The sealing material was applied in two 0.15 m layers, to eliminate the influence of variations of hydraulic properties of the till. Originally, three individual layers of 0.1 m each were planned, but field measurements had shown that a less complex procedure would suffice. At times, the application had to be interrupted due to extensive rain, which affected the compaction procedure. The target value, 95% Modified Proctor, was initially difficult to reach at high moisture contents, but it was found that compaction at a higher moisture content required lower compaction rates to reach the corresponding hydraulic conductivity. This enabled work to continue also during rainy periods.

The protective layer was applied as soon as the sealing layer was completed. As protective material, unsorted till from nearby sources was utilized. To protect the cover, large boulders were scattered over the area, constituting obstructions for future traffic that otherwise might cause damage to the cover. Finally, the cover was vegetated by sowing grass and birch seeds to rapidly protect the surface against erosion.

13.4.3

Monitoring Program and Results

A monitoring program was designed comprising the hydraulic behavior of the cover, the oxygen transport, the temperature, and the drainage water quality development. The key factor to minimizing the oxygen transport through the cover was to maintain a high degree of water saturation in the sealing layer. Therefore, great effort was put into designing a monitoring program aimed at describing the hydraulic conditions in this layer. To accomplish this, besides monitoring general hydrologic information, a number of groundwater monitoring wells were installed in the cover itself. Additionally, a number of infiltration lysimeters were installed under the cover. To quantify the effect of the cover on the oxygen transport, a total of 30 oxygen probes were installed in a network covering various depths in both ponds. The probes were also designed for temperature measurements. In addition, oxygen lysimeters were installed, i.e. cells tightly installed under the cover in which the pore gas can be replaced with nitrogen, and the hereby following increase in oxygen concentration quantifies the oxygen transport through the cover. A general surface water sampling program, enabling the verification of the results, completed the monitoring program.

The groundwater measurements verified that the cover had the ability to capture enough water to create a more or less continuously water saturated zone in the sealing layer covering the tailings ponds. The calculations of the infiltration rate, based on lysimeter readings, yielded figures close to or better than the target value for the permeability, $5 \cdot 10^{-9}$ m s^{-1}. As expected, a large spatial variation in the permeability was observed, almost two orders of magnitude. Some initially confusing results were found to be caused by lateral flow of groundwater in the cover.

Temperature measurements for the period, which presented examples of both low winter temperatures and a limited snow layer, indicate frost-free conditions under the cover. This is positive since the first year is the most critical before the vegetation has been fully established.

The surface water monitoring results, so far, serve as a basis for future follow-up of the project. Representative steady-state will not be achieved for a number of years, and, as a consequence, no significant change in the metal concentrations has been detected yet. Infiltration has been considerably reduced and flow measurements have started to enable an estimation of the mass flow of pollutants in the future.

The economic follow-up shows that the project has fulfilled the cost budget with a unit cost for the cover of about 11 US$ per m^2.

13.4.4
Conclusions

The project has served as Boliden's first major composite cover construction, generating experience for future, similar measures. Important experience has been gained to guide such projects towards more cost-efficient solutions. Besides the important practical aspects of the application of the cover, detailed technical knowledge has also been attained. As a general conclusion, the key parameter is the performance of the sealing layer. This can be controlled by the material used, which more or less eliminates the need for extensive follow-up of the groundwater behavior in the cover. Based on the results of in situ oxygen measurement results so far, the performance of the cover seems to be even better than expected. In the future, it might be possible to reduce the requirements of the cover. However, such a conclusion can only be made after an extended observation period.

Estimates of future pollution loads indicate that as little as 500 kg Zn per year will be mobilized from the ponds over a period of a few thousand years. Thereafter, when the calcite has been depleted, the pollution load may increase to just below 3000 kg per year, limited by the rate of oxygen transport through the cover. If no action had been taken, the pollution load was expected to increase to 600 tons per year.

One important experience is that to avoid leakage the installation of oxygen probes must be done in accordance with rigorous procedures to avoid leakage. Furthermore, the type of instrument used for oxygen measurements is essential, as the pore gas volume is limited. The instrument must be capable of analyzing small gas quantities, to avoid creating too high under-pressure in the probe.

The thickness of the protective layer appears to be sufficient. Also, this detail needs to be monitored for an extended period, with fully established vegetation and with a sufficient statistical variation of the climatic conditions, to ensure that the protection effect of the snow layer is unaffected by the measurement activities.

13.5
Design of Decommissioning Plans at Boliden Mineral Aitik Mine

13.5.1
Site Description and Project Outline

In 1970 Boliden obtained its formal license, from the National Licensing Board for Environment Protection, for the operations in Aitik. At that time, acid rock drainage and closure plans were not discussed. Since then, Boliden has applied for permission to increase the level of production on four different occasions. With a growing awareness from both the authorities and the industry, the decommissioning issue has become the most important question to be dealt with in the latter licensing procedures. In the 1989 license, Boliden was obligated to develop a decommissioning plan for the entire area and to submit this plan before the end of 1993 to the Licensing Board for approval. This was the starting point for a number of studies on different topics in order to make it possible to develop and evaluate different remedial options for the Aitik waste rock and tailings. These investigations have been planned and carried out with full insight from the environmental authorities, i.e. the Swedish EPA, the County Administration of Norrbotten, and the local community of Gällivare. Intermediate results and plans for the future work have been presented and discussed in depth with the authorities on a regular basis, normally twice a year. After the plan was presented in 1993, Boliden was required to carry out some additional work that was presented in late 1994 and in June 1996. A final approval of the plan was given in 1997, enabling Boliden to start covering parts of the waste rock dumps.

The open pit mine occupies an area 2.5 km · 0.8 km with a maximum depth of 270 m and a planned final pit depth exceeding 400 m. Approximately 13 million tons y^{-1} of waste rock is deposited on dumps on both sides of the pit and about 50% of it has been found to be potentially AMD producing. Fresh Aitik waste rock produces neutral pH leaches, as long as there is readily accessible calcite available for acid consumption. A transition time of several years has been observed from neutral to acid pH leaches.

Six pilot columns containing 1500 kg waste rock each have been operated since 1992 under field-like conditions. Studies of the transition from neutral to acid leaches in conjunction with geochemical and transport modeling have provided a better understanding of the development of AMD on a field scale. Geochemical heterogeneity has been identified as an important parameter to account for, especially during the initial development.

During the early stages of production, marginal ore was deposited in a separate dump aiming at a future metal extraction. Today, this relatively small part of the dump generates more than 50% of the dissolved metal in the AMD. Before the acid generating properties of the rock were known, a bulk management strategy was applied. Today's strategy is to avoid expanding the stockpile area, unless additional area is used for nonreactive rock.

Downstream of the tailings pond is a separate settling pond that was constructed in 1986 with an area of 4 km^2 to ensure efficient sedimentation of the very fine particles. From the settling pond, process water is diverted back to the mill through a spillway and a channel system. On an annual basis, only about 20% of the water passing the tailings pond system is discharged into the Una River. This discharge occurs in the late summer, following the melting of snow and ice in the tailings pond. Regular environmental impact investigations are carried out in the surrounding areas including fish, bottom fauna, and moss investigations. Some elevations in the metal content in moss have been observed in the proximity of the mine as a result of fugitive dust and emissions from the dryers, but no signs of impact have been found on the fauna.

The two main objectives of the decommissioning program were identified early: the waste rock dumps and the tailings pond. However, to provide background information, investigations of the regional geology, as well as of the hydrologic and geochemistry situation were also carried out. The investigations of the waste rock comprised characterization of the different rock types, field measurements, surface water investigations, and pilot scale tests in columns. Due to the location of the dumps, the leading remediation option was a cover. Suitable modeling technology was introduced in the project, aiming to select a glacial till cover as an option. For the tailings pond, similar measures were taken, as earlier investigations indicated acid generating properties also for the tailings. Sampling of the pond was conducted to quantify segregation, and a groundwater model was apply to predict the long-term situation. Extensive laboratory and pilot tests were carried out to determine the weathering properties of the tailings and to quantify the possibilities of removing pyrite to favor the ratio between carbonates and pyrite.

The field measurements were carried out using PVC containers which were carefully inserted in the tailings to minimize disturbance of the material. The oxygen flux through the surface was determined by measuring the development of oxygen content in the air volume captured above the tailings. The measurements were conducted in summer, during weather conditions enhancing pyrite weathering. The hydrogeological investigations showed that the dumps are not hydraulically connected with the pit. The whole area, on which the dumps are located, is covered with a 10 m layer of low permeable glacial till. Virtually all the infiltrated water leaves the dumps at the toe, and is easily collected in ditches. Field investigations estimated the annual amount of Cu leaving the dumps to be 80 tons, of which 55 tons originate from the old marginal ore stockpile. The corresponding overall amount of sulfate is 4000 tons annually.

One section of the dump, including the marginal ore, was equipped with an extensive set of sampling devices. In total, 13 drill-holes perforated the dump to the underlying till, over an area of 300 · 200 m. They were used for installation of probes for pore gas sampling, at 1-m intervals along the drill-hole. Two different kinds of waste rock were identified based on repeated sampling of oxygen concentration gradients. The diffusion coefficient was determined by injection of tracer gas, and the in situ oxygen consumption rate could be calculated. A comparison between measured and calculated pollution loads indicated that approximately 20% of the waste rock was responsible for the major part of the effluent release.

Physical tests of the glacial till in the area, i.e. the stockpiles and the overburden that has been or will be removed in the future, indicated that this material would be suitable for engineering of a gas diffusion barrier of relevant quality. The results were used for computer simulation of the performance of a number of possible cover alternatives, including tailings sand as one possible component, due to its abundance.

13.5.2
Results and Discussion

The main issue with the tailings was the acid generating properties of the material. Due to an early assumption that the material would produce AMD, a number of options to change the composition of the material were investigated. In its crude form, the tailings has an ABA value of –13 kg $CaCO_3$ per ton, ruled by the pyrite content (0.9% S). Flotation tests, as well as sampling of various products in the concentrator, yielded a range of samples with sulfur content ranging from 0.12 to 31%. These samples were then subjected to humidity cell tests in three different campaigns, each with gradually improved and expanded methodology.

Still, after 30 months of testing the samples of the third campaign, no sample with representative composition has shown acid properties (Fig. 13.8). To create a tool to evaluate these somewhat unexpected results, and to make a final interpretation, chemical modeling of the reactions was carried out. The results from this modeling indicated that the silicates involved represented a substantial acid consuming capacity. However, the principal factor is the sulfide oxidation rate in the field. The dissolution of silicates is capable of consuming the acid produced by pyrite oxidation up to a certain rate. Below that rate, the carbonates are slowly consumed, but above that rate the carbonates are depleted after some time, after which the silicates alone are unable to neutralize the acid generated. In some of the samples undergoing humidity cell test, the acid production rate was in the same order of magnitude as the silicates acid consumption rate, which made verifying tests in the field necessary. The field results indicated acid production corresponding to the silicate acid consumption capacity of only the top 20 cm layer of tailings.

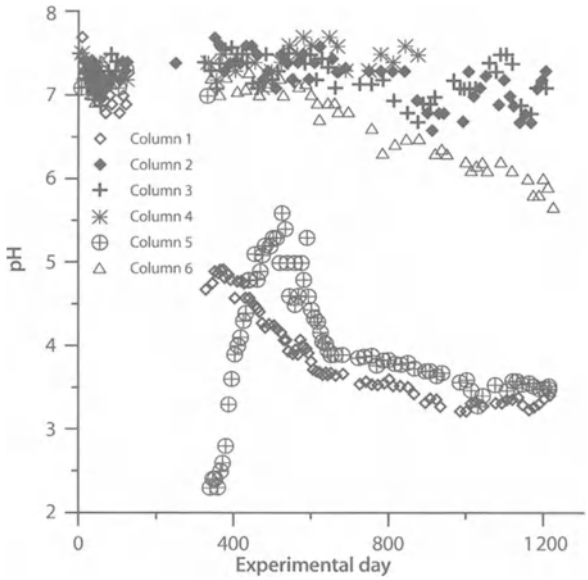

Fig. 13.8. Time dependence of pH in humidity cell tests of Aitik mill tailings. After 30 months of leaching at room temperature, neither de-pyritized nor representative tailings samples produce acid leaches

In Aitik, where frost conditions prevail for 7 months of the year, the kinetics differ significantly from the conditions in the laboratory and during the actual field test. To verify that the tailings do not possess AMD capabilities, a large scale column test was commissioned, under conditions which are representative for the unfrozen period at Aitik. In this test, the measured oxygen consumption rate was 50% below the lowest oxygen consumption rate calculated from sulfate export in the humidity cell experiments.

Parallel to those tests, the hydrogeological modeling of the groundwater showed that over 90% of the volume will be water saturated, which is equal to an underwater disposal. Only minor areas at the upstream and downstream dikes may become unsaturated. As the watershed crosses the pond in its upper part, Boliden considered it essential to extinguish any doubt about the situation in the lower parts. Therefore, a solution was suggested in which a wetland would be arranged by storing the snowmelt water in the spring. With this arrangement, the unsaturated areas in the lower parts of the pond would be avoided, leaving the solution for only a few percent of the tonnage, at the upstream dike, unverified.

The investigation is still waiting for steady-state conditions to establish in the column experiments, which will result in element budgets of the column drainage. With these results in hand, it will finally be possible to determine whether the silicates alone will be able to consume the acid in the long term. The option remains to apply a till cover on the unsaturated areas close to the upstream dike. The need for this can be judged when the column test is completed.

The dikes, which to a large extent are built with sulfidic waste rock, are proposed to be covered with till similar to the solution suggested for the waste rock dumps. Special attention will be focused on stabilizing the slopes against erosion. Industrial buildings will be demolished, and the industrial area will be soil-covered and vegetated according to practices developed within a separate investigation project.

The results of the computer simulation indicated that a 1-m layer of compacted till would reduce the oxygen transport into the dump to less than 1% of the reference case without cover. From this result, it was estimated that the reduction in Cu pollution load would be in the same order, i.e. at a level below 1000 kg year^{-1}. In addition, modeling results indicated that very little was to be gained by using tailings as an additional cover layer below the till and, if used on top of the till, problems with dust spreading and erosion could be foreseen.

An estimation of the influence of freezing, which could possibly affect the performance of the cover, indicated that frost would penetrate the cover to a depth of 0.7 m. To enhance the establishment of vegetation and to secure resistance to frost penetration, Boliden suggested an additional top layer of 0.3 m of uncompacted till. An illustration of the decommissioned waste rock dump and the proposed cover is shown in Fig. 13.9.

One remaining question is the infiltration and the properties of the drainage water. As the cover will divert surface water and increase evaporation, the infiltration will be dramatically reduced and the water retention time in the dump will increase significantly. Consequently, the AMD production will be practically interrupted until steady-state has been established in the dump. It is likely that the drastic reduction in the infiltration and the corresponding extended retention time in the dump, together with the anoxic conditions, will benefit the balance between buffering and acid producing reactions.

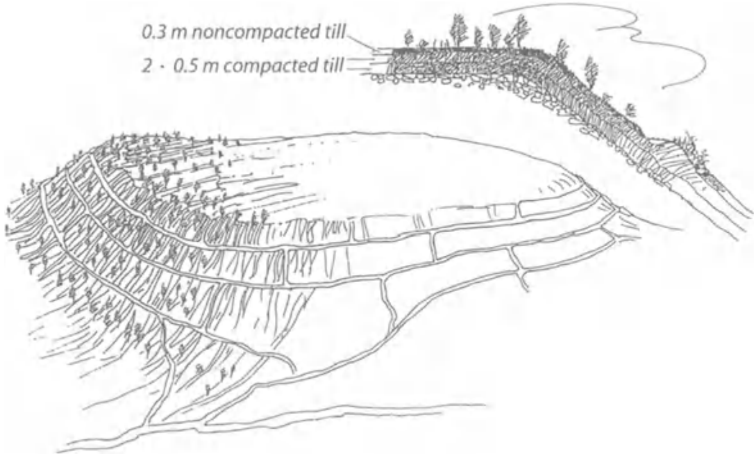

Fig. 13.9. Waste rock dump at closure. Final design of the water collection system and erosion stabilization will be determined by full scale experiments on parts of the eastern waste rock dump during 1998–1999. These investigations will also include vegetation studies

13.5.3
Conclusions

Covering of the east dump started during 1997. The final design of the run-off collection system and slope stabilization are determined by full-scale experiments on parts of the dump during 1998–1999. These investigations also include vegetation studies with selection of appropriate species and methods for transplanting. After covering the dumps, the small amounts of generated leachate will be diverted into the water filled open pit. The final pit, converted to a 400 m-deep lake, will then act as a precipitation reactor, polishing the leachate before it finally, after about 60 years retention time, is discharged into the Una River.

The investigations have been carried out in close co-operation with the authorities, with emphasis on developing the technical solutions required. Meanwhile, Aitik, as a major industrial site in a remote and undisturbed area, has the constant attention from the public being the subject of frequent articles in the press debating the risk of future pollution. It is Boliden's experience that the information flow to the public should be regarded as a matter of equal importance to the technology itself.

13.6
Decommissioning of Tailings and Waste Rock Areas at Stekenjokk

13.6.1
Site Description and Objectives

Stekenjokk is located in Sweden's southern Lapland Mountains. Copper and Zinc mineralizations were localized in this area about 80 years ago. From 1976 to 1988, Boliden Mineral AB mined and concentrated approximately 8 million tons of ore in this area, of which the value components were Cu (1.5%), Zn (3.5%), and Ag (40 g ton^{-1}). At its peak, 630 000 tons of ore were mined annually.

The mining facilities were situated at approximately 800 m above sea level in naturally beautiful but barren surroundings. The buildings were specially designed with the climate and environment in mind. Practically all mining occurred underground by means of cut-and-fill operations. Following autogenous grinding, the ore was subjected to flotation to produce separate Cu and Zn concentrates. A substantial part of the tailings coarse fraction was used as backfill, with the slimes fraction deposited in a tailings and clarification pond immediately downstream from the concentrator.

During the mine operation a range of protective measures were adopted, including water recycling and strict control of pH values in the tailings pond. Accordingly, this limited the environmental impact, enabling the ecological balance of the sensitive surroundings to be maintained. Several facilities remained in Stekenjokk after closure of the mine, such as a small open pit, a 110 ha tailings and clarification pond, a dike enclosing Stekenjokk´s raw water pond, rock dumps formed from ramp drifting and overburden stripping, and various aboveground facilities (Markland and Eurenius 1989).

The decisive prerequisites of the decommissioning were that the tailings pond and segments of the rock dump contained significant amounts of sulfur-rich material (up to 20% S and <1% of buffer minerals) (Arnesen 1987). Consequently, the residue products were potentially strong AMD sources. Other important considerations were that 2 years of operations remained at the time that detailed planning of the decommissioning started, the localization in a climatically harsh environment, and the limited access to covering material (e.g., moraine).

Objectives of the decommissioning were largely the following: (1) preventing the area from becoming a major source of migrating metals and acidic components; (2) removing facilities that could be hazardous to humans; and (3) adapting the area to its natural state.

13.6.2
Studies of Alternatives

The basic phenomenon that the decommissioning must address is the oxygen-induced weathering of sulfides. There are two principal methods of overcoming weathering: (1) elimination of access to oxygen, thereby inhibiting the reaction, and (2) manipulation of the composition so that weathering can occur without consequences to the surroundings. The detailed decommissioning plan began with a study of both these alternatives.

The best method of eliminating access to oxygen is using water as a barrier. The material can be either permanently immersed in water or sealed by a film of water. Such a water film can be attained in connection with a fine-particle cover layer as shown in Fig. 13.10. The capillary forces in the cover layer prevent it from being drained of water as well as reduce evaporation. Manipulating the composition to form a buffering mineral system was evaluated through pyrite removal or mixing in a suitable quantity of buffering material.

The technical measures representing the different principles can be summarized as follows: (1) flooding; (2) dry moraine covering; (3) depyritization; and (4) buffering. As the term implies, flooding involves damming the water so that it covers the entire

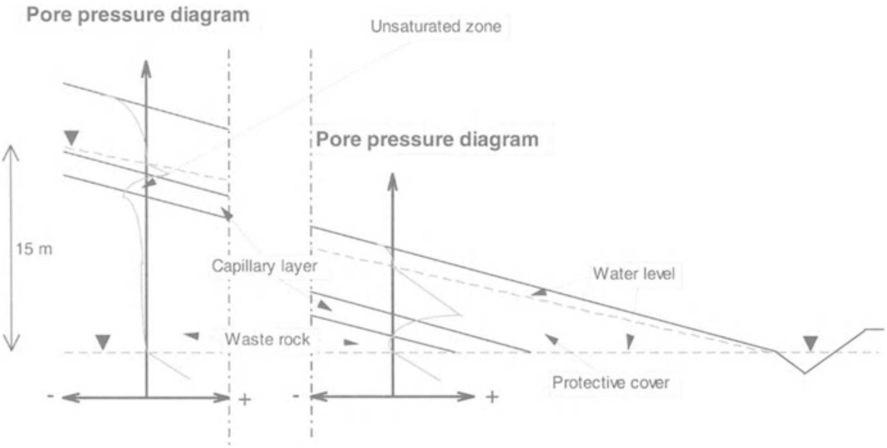

Fig. 13.10. Possible capillary barrier conditions in the cover of the Storgruve deposit (Cefyll liner)

tailings area. The flooding must be permanent and the water level is not allowed to fall and thereby drain parts of the tailings pond, nor can any physical forces be permitted to resuspend the fine tailings particles in the water cover (Kristiansen and Dahl 1989). Dry covering involves covering the entire tailings area with moraine of appropriate thickness and quality, but Stekenjokk is remote from suitable moraine deposits (Schönfeldt 1989a). In the comparative study, it was supposed that moraine of dike-construction quality would be applied to a thickness of 0.5 m. Depyritization, or the removal and separate disposal of pyrite, involves an extra step in the ore treatment process in the form of pyrite flotation. The open-pit mine would be a suitable site for dumping of the pyrite. Finally, buffering involves opening an olivine quarry and adding coarse olivine to the concentrator's autogenous system in quantities corresponding to 15% of the ore volume.

In comparing the four options, all conceivable and relevant parameters were taken into consideration. A central parameter was, of course, the environmental protection effect. Other important parameters were:

- The requirement for methods development and other studies
- The requirement that final dumping sites be determined
- The time required for implementation
- The necessity for supplementary protective measures
- The consequences for other landscape preservation measures
- The uncertainties and limitations of the various methods
- The most critical parameter cost

A comparison of the different options showed that neither manipulation alternative depyritization and buffering was feasible due simply to the fact that the remaining time of operation was too short. It is essential that enough time is available to form a sufficiently thick layer of buffering tailings and thereby provide an adequate protection. Ultimately, the choice was between the flooding and moraine-cover alternatives. In practically all respects, the evaluation favored the flooding method. Simply stated, flooding was judged to be safer and more effective, and above all, eight times more cost-effective. The two manipulation options, which were eliminated, were two to three times more expensive than flooding.

The decision was reached, in consultation with the appropriate government agencies, that the ongoing work should focus on the detailed planning of the flooding option. Flooding rests on two fundamental conditions: (1) flooding must be permanent to prevent any droughts of the area, and (2) the water-immersed tailings cannot be subjected to any movement or disturbance causing resuspension which would break the thin film of oxidized material at the tailings surface and the shielding of other tailings particles.

Permanent flooding is, of course, a matter of water balance and is determined largely by precipitation and evaporation effects on the water surface, the run-off of surface water from adjoining land, inflow and outflow of groundwater, and seepage through the dikes (Knutsson 1989). A hydrogeological study indicated an assured water availability and, in the event of extreme drought occurring once every 1000 years, that the water level would fall a maximum of 20 cm (Axelsson and Ekstav 1989).

The question of the stability of the tailings surface is somewhat more complicated. This is almost exclusively a matter of the shear force exerted, i.e., the wave-induced lateral forces that affect the tailings layer. The shear force diminishes if the water cover thickness is raised, i.e., the water depth is increased, or if the wave fetch is decreased. The resistance to shear force increases if the tailings surface is stabilized by superimposing a layer of coarse material.

The selected solution required that all stabilizing possibilities be used. This comprehensive action plan includes a large number of interim steps. First, the water level was lowered, and the dikes were raised 3 m, the extent permitted by the availability of construction material, mainly moraine. Attaining the required water depth required more than raising the height of the dikes, it also proved necessary to lower the surface of the upstream part of the tailings area. To obtain a decrease in wave force, it was required that a breakwater system be constructed. It was also deemed necessary to superimpose coarse material in the shallow parts for added resistance to shear forces. Other important features of the plan included the design and construction of erosion-stable dikes having a long-term safety factor and able to withstand an earthquake with an intensity of 6 on the Richter Scale, with an epicenter directly under the dikes. Of equal importance was the design of erosion-stable spillway arrangements. Prior to plan approval, we were also required to demonstrate the effect of the lake's water freezing into a complete ice cover.

13.6.3
Implementation of Flooding

The work was initiated in the summer of 1990 and was completed by late summer 1991. The work on the dikes, mainly the nearly 2-km-long downstream dike, provided an excellent opportunity for eliminating the existing sulfide bearing waste rock dumps. Large amounts of sulfide-free waste rock were required to raise the dike and reduce its slope to a 1 : 3 proportion. An erosion-inhibiting, breakwater shelf was set in place on the dike's inner side. Here, large amounts of sulfide-bearing waste rock were deposited to ensure safe underwater disposal during a long period. The breakwaters were constructed using similar type waste rock and, preferably, such material that required underwater deposition for safe, long-term disposal. The breakwaters were constructed somewhat in a checked pattern, which covered the shallow part of the future lake. The tailings were excavated and removed using the breakwaters as working and transport surfaces for excavators and hauling vehicles. The tailings ground water level proved to be higher than originally calculated, resulting in water saturation of the tailings surface occurring prior to the final depth being reached (Schönfeldt 1989b). Driving vehicles over such material was impossible. In this instance, the area's plentiful supply of crushed waste rock turned a drawback into an advantage. By using waste rock, the surface could actually be made sufficiently stable for transit. Approximately 100 ha, or one tenth of the surface, was thus covered with waste rock. About 90 000 m^3 of tailings was excavated from other locations, hauled to the lower-lying areas, and deposited. An erosion-stable spillway was built in the land area adjoining the dike.

The water has risen to a level covering the adjoining land area on the lake's western shore. The surface of the water remains undisturbed, except when strong winds blow, wherein the waves are broken along the breakwaters. In the event of a once in a 1000 years' drought, the breakwaters would be visible, even during a calm. To the casual onlooker, this lake is indistinguishable from any other in the area. Eventually, normal biological life should develop in the lake.

The simulations of the lake's water quality assume that a maximum Zn load of about 800 kg will be transported by overflow from the lake annually. This is the same level as when operations were underway. Ultimately, this transport will decline to very low levels. The effect of the decommissioning will be followed by a monitoring program which primarily is focused on a 5-year period. Results to date have been positive. The Zn contents from samples taken from spillway water are well within the anticipated range.

The monitoring has demonstrated that a certain level of metal migration occurs at some areas of the dike base. A review of the causes of this diffuse metal transport is presently underway and, to date, two explanations have been formulated: (1) the transport originates from the "flushing" of the partially weathered rock that has been deposited on the inside of the dike; (2) the transport results from acid-generating rock fractions in the dike's outer support filling. The survey now in progress shall determine the extent to which actions are necessary, as well as those which are feasible and suitable.

The decommissioning at Stekenjokk has taken place largely according to plan. The overall cost has amounted to about 3.3 million US$ in 1991, of which the tailings pond accounts for 2 million US$.

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Chapter 14

Mining in the Arctic: Mitigation and Remedial Measures

Sukumar Bandopadhyay

14.1
Introduction

Permafrost, or perennially frozen ground, underlies about one-fifth of all land surface of the earth, and much of this area is in the northern hemisphere. Most of Alaska, at least one-half of the Canadian land area and more than a third of the territory of Russia, lie within the continuous and discontinuous permafrost zones. Permafrost, by definition, is any soil that has been continuously frozen for at least two consecutive years. The continuous permafrost zone is totally underlain by permafrost except for areas concomitant with large water bodies. The discontinuous zones are characterized by permafrost and thawed ground existing together.

Arctic (Antarctic) is that portion of the world underlain by a continuous layer of perennially frozen ground, and its effects on human activities are profound. Tundra vegetation occupies vast treeless regions of the arctic, where large deposits of mineral resources are found (Fig. 14.1). The arctic and subarctic cover such a large area that generalization on the environmental impacts of mining activities are difficult to make. The only generalization that has meaning is that mining in the arctic will create irreversible changes in the soil and vegetation cover which will alter the permafrost regime. Therefore, the scope of this chapter is limited to surface coal mining in the arctic, and mitigation of the environmental impacts due to surface mining. The features of permafrost that influence mining activity include the active layer, which thaws during the summer and freezes again in the winter, ice lenses, ice wedges, and ice-rich ground, in which volume of ice exceeds the void volume of the thawed soil. The freezing and thawing of mine spoil is a complicated process that includes heat and mass transfer at the surface of the spoil and below, and involves three phases of water and phase changes among them (Bandopadhyay *et al.* 1988). The behavior of water in permafrost terrains requires special attention because of effects of frozen ground on water movement and hence the hydrologic consequence of mining and reclamation.

In order to suggest the specific mining and reclamation approaches one must, however, define in advance the types and degree of modification the permafrost landscape can sustain without adversely affecting its original quality. This definition should include both human or aesthetic values and the physical and thermal constraints imposed by the permafrost. An approach must recognize the problems of low temperature for plant growth and development, short frost free periods, and the presence of permafrost on a vast territory with specific hydrologic, thermal, saline and nutritional regimes of the soil (Bandopadhyay and Maneval 1986).

The objective of this chapter is to establish the interrelationships between the problems due to permafrost and the factors in reclamation of post mining land in order to

Fig. 14.1. Main mines in the Arctic region. (After CIA Polar Region Atlas 1978)

find ways to transform nature in a rational and purposeful manner and to suggest specific reclamation approaches that may be appropriate in these regions. To do this job efficiently, among other things, data must be established for the following:

- Water balance in the seasonal thaw layer due to redistribution of moisture when the ground alternately freezes and thaws
- Major characteristics of arctic soil and changes in composition and properties of soil during freeze–thaw cycles
- Approaches in mine planning to reduce slope movement, especially when soils are saturated with melt water at a time when the permafrost layer is at a depth favorable for sliding
- Effect of organic layers for restoration or revegetation purposes
- Proper species of plants for revegetation when plants roots are restricted only to the active layers
- Impact of low temperature in the root zone

Owing to the almost non-existent data base and sparse published research in this area, it is not possible to describe all the interactions and specific reclamation approaches that might have applications. Instead, this chapter is directed at establishing the problems in post mining reclamation due to the presence of permafrost and suggesting a specific representative approach in a unified way for the purpose of drawing some general conclusions. In the following, some general problems in reclamation of post mining lands in the arctic are described.

14.2

General Problems of Reclamation of Surface Mined Land in the Arctic

In the Arctic, as elsewhere, reclamation efforts to achieve a permanent vegetative cover must deal with a considerable range of conditions. Permafrost is undoubtedly the most unusual environmental condition in the Arctic. It is of particular concern in mining and reclamation because of the profound changes that sometimes take place in the physical behavior of earth materials as a result of thawing, which is accelerated by mining operations and related disturbances of the thermal regime. Reclamation and restoration in areas of continuous permafrost, therefore, would face problems associated with thaw behavior under climatic conditions where permafrost may be redeveloped. Two principal difficulties might be anticipated. First, permafrost presents a formidable challenge to reclamation efforts aimed at achieving a stable surface that is compatible with adjoining areas. Surface mining can commonly result in either an excess or a deficiency of backfill material depending on the stripping ratio. Grading to merge with adjoining lands may form a topographic high or basin. If excess spoil should form an area of higher ground, runoff might form a connected drainage system, and this might lead to accelerated erosion (Walker 1973). On the other hand, where the spoil is deficient, a basin is created and thawing could take place at the margins of adjoining undisturbed lands. These man-made thaw basins may continue to grow laterally to uncertain limits.

Ice is a nearly universal constituent of permafrost. It is present as horizontal and vertical masses or as filling of pore space between mineral grains. Because ice may melt and change to water during mining operations, knowledge of its distribution in the ground is important in any mining activity. The frozen zone at the surface thaws each summer to a shallow depth. This thawed zone is known as the active layer, a name suggestive of the role of this zone as the site where surface processes, including plant growth, take place. Water in the active layer cannot drain downward because the underlying permafrost is impervious. Hence, permafrost terrain is commonly a waterlogged landscape in the summer. On the North Slope, Alaska, the flat coastal terrain is dotted with innumerable shallow, water-filled basins called thaw lakes. Virtually all of them freeze solid in winter. Thaw lakes form by progressive subsidence brought on by the melting of ground ice and have been a part of the arctic landscape for thousands of years (Hopkins 1949; Black 1969; Sellmann *et al.* 1975). In coastal areas alternate melting and freezing tends to produce pockets of briny ground water. Arctic streams and lakes are notably clear, but the lack of streamflow in winter can markedly increase the concentration of dissolved solids.

Local geology determines the behavior of ice-rich materials when they thaw. Gravel saturated with ice, for example, is relatively stable when thawed, but silt and clays are not. Since the fine grained sediments commonly are interbedded with coal and therefore, an important component of most overburden, their instability when thawed may be critical to the mining operation. Permafrost in undisturbed areas is protected from deep thawing by a surface mat of semi-decayed organic matter. This mat is the growth medium for tundra vegetation.

The most conspicuous surface processes in areas of permafrost take place when the active layer thaws in the summer (Brown and Sellmann 1973). On sloping ground, even on very gentle slopes, the active layer may creep or slide slowly downhill by the process of mass wasting, termed solifluction. The result is a smooth topography dominated by convex forms (Pèwè 1974). In the event that thawing progresses to greater depths because of disturbances of the surface organic layer, or because of even deeper disturbances by surface mining, a greater amount of earth material eventually may become unstable.

Surface drainage is generally poorly developed in permafrost terrain, although some water in the active layer usually seeps out and collects in drainage channels. Some extensive tracts of permafrost, however, do not have exterior drainage. Most of the summer runoff is from snow melt (Brown *et al.* 1968; Church 1974), but any melting of interstitial ice below the active layer would result in additional outflow and some loss of volume, hence a somewhat lower terrain. From the above discussion, it is clear that uncontrolled erosion of permafrost areas when disturbed is primarily a consequence of the unusual hydrologic and thermal properties of frozen ground (Williams 1970; Dingman 1975). From the perspective of surface coal mining, a matter of fundamental importance is that local hydrologic and thermal properties probably would change greatly if surface mining was done. That is, ice in the ground would melt and fluvial erosion might become dominant. This consequence could be destructive to the landscape, leading to the further melting and erosion and the new processes of thawing, outflow, and fluvial erosion. These processes would continue until perennially frozen conditions were once again established (Brown *et al.* 1969).

The arctic tundra is the second major problem for surface mining of coal because of the difficulty of replacing its distinctive vegetation (Benninghoff 1974). The difficulties of building a tundra landscape using materials excavated by mining pertain to the problems of sorting, handling and preparing the mined spoils, and to shaping the surface in a manner adjusted to the surface processes of permafrost regions. The reclamation of land in the Arctic must normally be done immediately after mining it, or in any case during the same summer. Otherwise, the mass of stockpiled earth will freeze. In as much as the thickness of the winter freezing layer is usually greater than the thickness of the summer thawing layer, this freezing will progress until the underlying permafrost completely merges with the spoil. If this occurs, revegetation of the land becomes almost impracticable.

The presence of a regional, frozen, water impervious bed only a short distance below the surface, combined with a negative mean annual air temperature and very low winter air temperature gives rise to unique hydrochemical situations in the Arctic. These factors also lead to the extensive occurrence of processes in cryogenic metamorphization of the ground water, including the water of the active layer, and the supra-permafrost and intra-permafrost taliks or unfrozen zones. The overall trend of these processes is towards an increase in groundwater mineralization and changes in the relation between the basic ions, which ultimately leads to an increase in the sodium chloride and magnesium sulfate content of the water and to salinization of the soil with hydrocarbonates and sulfates of calcium and magnesium. Under natural conditions these processes are in equilibrium with the leaching processes and do not usually cause irreversible changes in the ground water composition or salinity of the soil profile.

The occurrence of permafrost close to the surface makes for an extremely thin aeration zone thus greatly reducing its capabilities for natural purification of water percolating from the surface. Thus, the upper aquifers of the talik water remain unprotected against surface pollution and the squeezing of polluted water from the active layer into them. This imposes special requirements on the application of fertilizers, the percolation of which below the soil profile becomes almost impossible. In the absence of a sufficiently rapid surface runoff it is inevitable that there will be a gradual increase in fertilizer concentration to the point where this leads to a reduction in soil fertility. The application of various types of fertilizers will be determined by the specific nature of the permafrost hydrological situation, the absence of aeration zone, the climate, the cryogenic processes, and the delayed course of the oxidizing and exchange reaction (Grasimov 1980).

Low temperature for plant growth, short frost free periods and the presence of permafrost on a vast territory with specific hydrologic, thermal, saline and nutritional regimes of the soil are the basic characteristics of reclamation problems. Even a partial reduction of the gap between these soil conditions and those considered optimum for stabilization and revegetation can only be achieved by development of efficient methods for optimizing the hydrological and hydrothermal regime of the disturbed area and artificially changing or improving the natural equilibrium of the frozen-thawed rock system.

14.3
Reclamation Approach: Stabilization of Spoil

Excavation during construction, mining and subsequent building of spoil piles causes intensive surface disturbances. These activities influence all permafrost zones and particularly sites with a high content of ground ice. During mining, not only are the soil and plant covers destroyed, but shocks and vibration from machinery disturbs the soil structure. Waste rock piles during mining activities are also heat insulators. In areas where permafrost temperature is relatively high, taliks may form in a few years. The taliks form as a result of stagnant waters which grow together into one body with those formed under waste piles.

The primary objective of reclamation and revegetation is to provide erosion control, cover for stabilization and to aid natural plant succession processes for the re-establishment of a permanent, stable, natural plant cover. The wastes derived from mining sometimes are difficult to stabilize. The removal and even partial disturbance of soil covering of ground underlain by icy fine-grained material causes a progressive thawing of the frozen ground. Seasonal thawing during the first year increases considerably, however, in the absence of a significant volume of water the process is stabilized. After 3 or 4 years of heat exchange between the soil and the atmosphere, there is a new stable equilibrium. The accumulation of surface water in such a situation creates the necessary conditions for the progressive development of thermokarst because the heat transfer in shallow water bodies is several times greater than in ground. The development of methods for adequate drainage of water is one of the means for preventing thermokarst in areas with icy ground. Thus, the means of solving the problem of reclamation can be seen to be primarily the development of efficient methods for optimizing the hydrological and hydrothermal regime of the disturbed area and to artificially changing or improving the natural equilibrium and to establish a stable vegetation cover.

The spoil which thaws can be undersaturated, saturated, or supersaturated with respect to ice (water). Thawing of undersaturated or saturated soils will release no excess water. In supersaturated icy soils the thaw rate will decrease as the ice content rises because of the retarding effects of the latent heat of ice fusion. The presence of icy soil is very widespread in arctic regions (Hussey and Michelson 1966). Less well know, but just as important, are the supersaturated soils of the western Arctic, occurring at depths below about 2 m. When thawed, the volume ratio of excess water to saturated thawed soil is often 5 : 1 or even 10 : 1 (Mackay 1970).

Because of natural humidity conditions, soils of disturbed lands in the arctic have either excessive or insufficient moisture content (Bogushevskii 1975). Excessive moisture is a local characteristic due to additional surface flow from an outlying drainage basin. If there is excess moisture, it is necessary to remove water from the entire thickness of the thawed ground in order to improve the thermophysical properties of the soil.

As pointed out earlier, permafrost terrain is commonly a waterlogged landscape. The characteristics of water recharges in these areas are: (1) atmospheric, (2) surface flow and alluviation, and (3) ground flow or supra-permafrost water. The principal type of water recharge can be estimated quantitatively on the basis of water balance computation. If W denotes a total change of water in the seasonally thawed layer of spoil over a specified period; W_p and W_n are the variation of water supply in the surface layer and the supra-permafrost layer, and W_{pw} is the variation in the supply of surface water, then the water balance equations can be written as:

$$W = (\text{Precipitation}) + (\text{Inflow–Outflow of surface water}) + (\text{Inflow–Outflow of suprapermafrost water}) - (\text{Evaporation+Evapotranspiration});$$

W_{pw} = (Precipitation) + (Inflow–Outflow of surface water)
- Evaporation - Absorption;

W_n = Absorption - Evapotranspiration
± Water interchange between subsurface layer and suprapermafrost layer; and

W_p = (Inflow - Outflow of suprapermafrost water)
± Water interchange between subsurface layer and supra-permafrost layer.

If the total variation in moisture content of the soil exceeds the overall outflow of surface water and supra-permafrost water, an approach for reclamation of this type of land would be to accelerate the discharge of surface water and increase the depth of the seasonal thaw layer in order to increase water retention. Such a method can be implemented in practice by the construction of a network of open drains and top soil treatment.

In moisture deficient regions, the major portion of surface runoff comes in the spring time. Hence, the possibility of water accumulation in the soil depends on the depth of ground thaw during that period. Deeper thaw is ensured by changing the relationship between heat balance components on the surface of the soil in the layer of seasonal thaw. Thus, a new thermal balance component can be obtained by altering the surface characteristics of waterlogged meadows by the elimination of hummocks and leveling the micro-relief.

Locally, waterlogged areas will form in depressions caused by insufficient spoil where surface runoff is considerable. The predominant recharge in such locations is a combination of alluvial (surface flow) and ground recharge. The regulation of surficial drainage is an indispensable component of the overall design and stabilization of disturbed slopes in permafrost areas; especially for fine grained ice-rich soils. These slopes are vulnerable to the combined thermal and mechanical erosion surface runoff. Hydrologic balance in such areas would consist of protecting them from inflows of surface and supra-permafrost water and accelerating the removal of excess surface water from reclamation areas. In practice, these requirements can be met by the construction of a protective system of intercepting channels and selectively located open drains to draw water from stagnant depressions.

Under the conditions discussed above, one should not, however, exclude the possibility of supra-permafrost water forming inside the layer of seasonal thaw and saturating the entire thickness of this layer. Infiltrating surface water seems to be the only source of recharge of supra-permafrost water in such cases. If surface runoff is removed at a faster rate, that portion of it which ultimately recharges the supra-permafrost water will be reduced. The gradual decrease of supra-permafrost water due to evaporation and the lowering of its table caused by the increasingly deeper thaw will stabilize the supra-permafrost water table at a level suited for revegetation (Bogushevskii 1975).

It is possible theoretically to use other methods of removing excess moisture. It has been known that water migrates towards subsoils with lower temperatures. If pipes (with holes on top and on the sides) are laid in the lower part of the active layer, free moisture will flow down these pipes in the summer, thus draining the soil layer. The drainage of excess water in tundra areas always raises the soil temperature. To obtain a better effect, soil drainage can be supplemented by mulching with peatmoss to keep the soil warm during the revegetation cycle. Thermal coatings of mineral (slag and cinders, etc.) and organic (peat and moss, etc.) have the advantage of low cost when compared with other coverings. Summer warming of the soil can also be combined with winter insulation.

The concept of using a synthetic mulching layer and film covers on the surface of the rock to control the thermal balance component was advanced in the former Soviet Union. According to their observations, polyethylene film 0.06 mm thick reduced the effective radiation of soil by 2–14% during the summer season. Despite the decrease in effective radiation, there is a decrease in radiation balance when film covers are spread on the soil surface due to sharp increase of reflectivity. It has been demonstrated that the use of film permits a reduction of both evaporation and turbulent heat transfer. In particular, the value of evaporation during the summer season on sites with film covers decreased by 26–33%.

The thermo-physical effectiveness of film covers is in the increase of the surface temperature of the soil compared to natural conditions. Film covers increase accumulation of heat by the soil and increase the depth of seasonal thawing for the following reasons (Pavlov 1975): a decrease of heat losses to evaporation of moisture; a decrease of the heat transfer coefficient of the surface due to the variation of the structure of the boundary layer; an increase in the heat transfer coefficient of the seasonally thawing layer of soil as a result of an increase in its moisture content compared to natural conditions.

Control of the thermal balance components during the summer season to intensify thawing of the soil or raise its temperature is one integral part of the problem of thermal balance of soil and rock in the frozen zone and the second is to protect the soil against freezing and cooling. A method of protecting the thawed rock mass system against freezing by flooding to a depth exceeding the depth of freezing the water in closed reservoirs might have applications. The technical possibilities of accomplishing this procedure on a broad scale, however, have not been confirmed experimentally.

14.4
Revegetation

The consequences of vegetation removal in permafrost areas is well addressed in the published literature. Investigations from different locations in the tundra and forest-tundra of western Siberia during 1974 and 1976 (Brown and Grave 1978), have shown that the difference in moisture content of the ground and the depth of seasonal thaw are not so much conditioned by zonal climatic factors as they are local ecological conditions. These ecological conditions are factors which determine the degree to which vegetation and soil cover disturbance will affect ground temperature, moisture, and summer thaw (Table 14.1). As can be seen from Table 14.1, in the forest-tundra and northern taiga zones of western Siberia, the removal of the surface moss cover from hummocky peat land and muskeg underlain by frozen sand and silt did not lead to a significant increase in ground temperatures or depth of seasonal thaw. This is due to the low thermal conductivity of the peat, which usually dried out at the beginning of the summer. However, once the peat is removed, the sand and silt thaw deeper and thermokarst occurs.

Removal of the tree cover in the southern taiga, on the other hand, decreased the depth and increased the density of snow cover on the treeless sites, which resulted in a lowering of the ground temperatures, and as a result, short term permafrost appeared

Table 14.1. Temperature-moisture regime of soil and ground under natural and disturbed conditions (Brown and Grave 1978)

Natural ecological complex	Average ground temperature (°C)	Average moisture in 50-cm soil layer (mm)	Maximum summer thaw (m)
Southern tundra (1974)			
1. Spotty tundra on sandy loam deposits			
a) Sandy loam spot	-6.2	179	1.15
b) Grass–shrub cover	-5.7	167	1.06
2. Shrub–moss tundra on peaty sandy loam deposits	-6.8	193	0.52
3. Polygonal tundra on peaty sandy loam			
a) Shrub–lichen–sphagnum polygon	-7.3	224	0.35
b) Grass–sphagnum trough	-7.2	–	0.43
Forest tundra (1976)			
4. Spotty polygonal tundra on sandy deposits			
a) Sandy spot	–	35	1.8
b) Shrub–lichen polygon	-1.3	49	1.7
5. Spotty larch shrub on peaty loam			
a) Loamy spot	–	171	1.49
b) Shrub–lichen cover	-2.3	165	1.3
c) Loam with vegetation removed	–	160	1.62
d) Peaty loam with vegetation removed	-2.5	172	1.52
Northern taiga (1976)			
6. Shrub–sphagnum–lichen peat moss			
a) Hummock	-0.8	260	0.63
b) Depression between hummocks	–	285	0.57
c) Same surface with vegetation removed	-0.8	248	0.7
7. Peat–mineral mound with shrub lichen cover			
a) Mound	-0.7	200	0.92
b) Depression	–	220	0.65
c) Same surface with vegetation removed			
mound	-0.7	160	1.43
depression	–	180	1.3

where there was none previously. Given the consequences of vegetation removal, it follows that revegetation of disturbed areas in permafrost may constitute an important design tool in stabilization approaches in that it may allow for a rapid return to original conditions (Linell 1973).

In Alaska this is largely a problem of re-establishing the tundra plant cover. It dominates the Arctic region north of the Brooks Range where Alaska's most extensive coal resources are found. Although there are diverse vegetation types and a wide variety of plant associations depending on soil, moisture, and other site conditions, in a general way, tundra plant association can be correlated with the broad physical features of the landscape. Plants characteristic of wet tundra are generally found in the poorly drained coastal plain of the arctic region, whereas plants characteristic of moist tundra are generally found in the better drained foothill areas.

Experience with revegetating tundra areas in permafrost terrain is limited to disturbed areas along roads and pipelines and at oil fields (Johnson *et al.* 1977; Brown 1978; Lawson *et al.* 1978; McKendrik and Mitchell 1978; Mitchell 1979). The establishment of a self regenerating vegetative cover on disturbed areas is generally regarded as the most effective reclamation practice. Regardless of how revegetation is to be accomplished, there are a number of factors which must be considered. Site conditions are important regardless of the region where revegetation occurs. On the other hand, there are aspects of mineral nutrition and plant adaptation peculiar to the Arctic which must be understood in order to ensure the success of a vegetation scheme. It is the physical environment that controls most plant growth and establishment in the Arctic. Certain biological factors are important locally and with some species, but these are pale beside the rigors of winds, snow, and low temperatures of the atmosphere and soil. The arctic lite zone for plants is only a shallow layer of soil and air. As an environment for plant growth, reproduction, and establishment, it is ephemeral and available only from thaw to freeze-up (Billings 1987). As it can be seen from Table 14.2 there are numerous factors that influence revegetation efforts in the arctic. These include soil type, soil temperature, site conditions, nutrient regime, plant adaptation, etc. The influence of these factors on revegetation efforts is briefly presented here. A detailed discussion on the effect of these factors can be found elsewhere (Johnson and Van Cleve 1976).

Arctic tundra soils in general are poorly developed. The soil physical environment can be characterized by temperature, water, aeration, mechanical impedance, and spatial variability. The soil physical environment required for the development of an efficient roof system must be free from stresses caused by mechanical, water, oxygen and cold temperature conditions. Soils, because they vary texturally, structurally, in porosity, and pore size distribution, differ greatly, (1) in their resistance to root penetration, (2) how they collect, store, and release water for root uptake, and (3) how they maintain aeration by gas exchange with the atmosphere (Brar and Reynolds 1996). According to Tedrow *et al.* (1958) soils in the foothills of the Arctic region are generally silty, whereas

Table 14.2. Principal environmental stressors in the Arctic (Billings 1987)

Physical
- Low air and soil temperatures
  - Below freezing most of year
  - Frosty, or slightly above, during summer
- Very short growing season
- Strong and chilling winds
- Long-lasting snowdrifts
- Permafrost and shallow active layer
- Thin life zone between permafrost and wind
- Flooding at thaw
- Drought
- Thermokarst erosion
- Nutrient-deficient soils
- Shortening day-length signaling end of summer

Biological
- Plant-to-plant competition
- Pollination problems
- Grazing ("point-intensive" vs. "wide-ranging")
- Trampling
- Slow decomposition

some of the coastal plain are likely to be sandy (Table 14.3). Parent materials range from normal bedrock to wind blown or water transported materials. Well developed soils on bedrock are often thin, but frost-shattered bedrock may be as much as 2 m deep in some places. It is recognized that tundra soils have developed under the dominating influence of low temperatures and high moisture conditions, and soil-forming proc-

Table 14.3. Names of mapping units – Arctic Slope of Alaska and percentages of soils that occur there (Fibich 1977)

Soil number		Description	Percentage on Arctic Slope
IA7	IQ2	Histic Pergelic Cryaquepts, nearly level to rolling, loamy	
IA8	IQ3	Histic Pergelic Cryaquepts and Typic Cryofluvents, nearly level, loamy	
IA9	IQ5	Histic Pergelic Cryaquepts, nearly level to rolling, loamy and Pergelic Cryorthents, hilly to steep, very gravelly	
IA11	IQ6	Histic Pergelic Cryaquepts, nearly level to rolling, loamy and Pergelic Cryofibrists	
IA12	IQ7	Histic Pergelic Cryaquepts, nearly level to rolling, loamy, and Pergelic Cryaquepts, nearly level to rolling, very gravelly	27
IA13	IQ8	Histic Pergelic Cryaquepts, nearly level to rolling, loamy, and Pergelic Cryaquepts, hilly to steep	
IA15	IQ11	Histic Pergelic Cryaquepts, nearly level to moderately sloping, loamy, and Pergelic Cryumbrepts, hilly to steep, very gravelly	
IA5	IQ18	Histic Pergelic Cryaquepts and Typic Cryochrepts, hilly to steep, very gravelly	
IA26	IQ20	Pergelic Cryaquepts and Pergelic Ruptis Histic Cryaquepts, nearly level to rolling, loamy	
IA20	IQ21	Pergelic Cryaquepts and Pergelic Cryopsamments, nearly level to rolling, sandy	
	IQ22	Pergelic Cryaquepts, nearly level, very gravelly	4
IA21	IQ24	Pergelic Cryaquepts and Pergelic Cryorthents, hilly to steep, very gravelly	
IU2	IU2	Pergelic Cryumbrepts and Histic Pergelic Cryaquepts, hilly to steep, very gravelly	
MA3	MA1	Pergelic Cryaquolls and Histic Pergelic Cryaquepts, nearly level to rolling loamy	
MA1	MA2	Pergelic Cryaquolls, nearly level to rolling, very gravelly	
MA2	MA3	Pergelic Cryaquolls, nearly level to rolling, very gravelly and Pergelic Cryoborolls, hilly to steep, very gravelly	6
MB2	MB2	Pergelic Cryoborolls and Pergelic Cryaquolls, hilly to steep, very gravelly	
RM2	RM2	Rough mountainous land and Lithic Cryorthents, hilly to steep, very gravelly	30
		Water	4

esses that are common to areas of less severe climates have been restricted. The rate of chemical and biological action is low and the depth to which soil-forming processes are active is restricted by permafrost.

Rieger (1974) has divided arctic soils into two broad groups, namely, (1) those that are poorly drained and usually water saturated and (2) those that are well drained. Poorly drained soils dominate the arctic landscape. Permafrost limits the downward percolation of water, thus the soils remain wet during almost all the summer thaw period. Limited areas of well-drained, coarse-grained soils are found on some steep slopes, on elevated parts of the flood plains, and in other areas where drainage is facilitated. Because moisture is able to migrate through these soils, oxidizing conditions prevail, and mineral and organic materials are redistributed into distinct horizons (Rieger 1974).

General observations (Palazzo 1989) have shown that plants germinate and grow more rapidly when a cover keeps the soil surface temperatures higher than the ambient air temperature. In some situations, soil covers are used in the late fall to conserve heat, thereby lengthening the growing season and permitting plants to germinate before winter. This technique may help prevent injury to seedlings when the seeds are sown late in the season. Late seeding under a cover, however, produce succulent plants that may be more susceptible to cold injury because they do not go through the hardening-off process.

The timings may be crucial for the successful establishment of grasses and legumes for soil erosion control. There is a 1- to 2-month transition period between latest date of permanent seeding and the earliest date of dormant seeding. If seeding occurs between these dates, there is presumably an increased risk of seeding mortality due to low fall and winter temperatures. In the case of dormant seeding, spring temperatures and moisture conditions are also important when the growing season begins again (Racine *et al.* 1990).

Arctic soils, and to a somewhat lesser extent sub-arctic soils, are acknowledged to be nutrient deficient (Johnson and Van Cleve 1976). In some cases in the Arctic, lack of nitrogen has been recognized to be the limiting factor for plant growth, whereas in other instances phosphorus was probably a limiting nutrient. Van Cleve (1975) has recognized the two key aspects of the nutrient regime: the amount of nutrients and the rate of nutrient cycling. It is the interaction between these two which determines the nutrient availability for the plant. The nutritional status of the soils from the plant's perspective is very much interconnected with the low soil temperatures characteristic of soils in permafrost areas. Since the utilization of soil nutrients requires the translocation of energy rich substrates from photosynthesizing tissues, the organic nutrient status of plants under a physiological nutrient stress caused by low temperature may be important.

Productivity of tundra vegetation is considered in general nutrient limited and nutrient availability has major consequences for the distribution and abundance of native tundra plants. As in virtually all terrestrial ecosystems, physical disturbance to tundra disrupts the normal patterns of nutrient availability. Shaver *et al.* (1984) studied the effect of an increase in nutrient availability by fertilization on improvement of the recovery of native plant and in the later stages of recovery of the effect of manipulation of nutrient availability on the composition of the native plant community.

Results of their research show that fertilization has various effects on native plant growth, depending on species, time since disturbances, and type of vegetation. The lack of a long-term fertilization effect on native plant recovery on disturbed sites is consistent with the long-term vegetation responses on unmanaged disturbances. The nutrient availability is non-limiting after about 5 years on a disturbed site if the site is relatively moist and covered with an organic soil. Fertilization thus may improve native plant recovery in the first year or two, but there is no need to continue the application except perhaps on dry mineral soils (Shaver and Chapin 1980).

In areas of ice-rich permafrost, recovery from disturbance is likely to be slow, because soils and hydrology are disturbed by thermokarst and erosion. Soil nutrients and water regimes are relatively favorable for adapted native species, so seed supply is the most likely limitation to initial colonization. Once the first plants are established, competitive interactions probably outweigh facultative effects on later-successional species. These points are clearly illustrated in recovery after blading of tussock tundra, in cases where at least part of the organic mat remains (Cargill and Chapin 1987).

In marginal arctic environments the development of vegetation is determined almost entirely by the individual physiological and genetic attributes of the invading species. In such environments it is impossible to trace any emergent property because there is virtually no linkage between the widely separated individuals. The widely scattered plants fail to achieve a critical mass which would trigger the "ecological warming-up" of the system; or in other words, would result in competitive interactions. These species depend largely on their individualistic strategies for survival (Svoboda and Henry 1987).

Once an adequate cover is established, restoration of a natural community must involve manipulation of the relative abundance of species and growth forms. Nutrient availability is a key variable in this process, the major problem being that nutrient cycling rates are very rapid on old disturbances. The fertilization experiments have shown that when nitrogen and phosphorus are added to undisturbed tundra, the vegetation becomes more like that on old disturbances. Thus, management of older disturbances should be focused on slowing down nutrient movement and decreasing nutrient availability (Shaver *et al.* 1984). These results suggest that in the revegetation of arctic disturbed sites by native plants, the single most important factor is the preservation or the replacement of the upper organic layer of the soil. The organic layer is important because it contains a large readily germinable seed pool and the germination rates of native seeds are higher in organic than in mineral soils (Chapin and Chapin 1980). The organic layer reduces soil thaw and thermokarst erosion. Because the organic mat contains buried seeds, as well as some living below ground stems of shrubs, this recovery is essentially a secondary successional sequence. Initially, the disturbed sites are dominated by the sedges from the buried seed pool, and some shrubs that sprout from below the ground stems. Further invasions of shrubs by seed is slow, so at least 50 to 100 years may be required to completely restore the original community (Cargill and Chapin 1987). In later stages of plant recovery, it may reduce nutrient movement and nutrient losses from the soil. Fertilization improves native plant recovery immediately after disturbances, but after 5 to 10 years it may prevent re-establishment of a normal species composition.

Any attempt to speed up this recovery process, except soil stabilization, are likely to be counter productive. In disturbed tussock tundra sites with some remaining organic mat, seeding of native and exotic grasses has either inhibited the growth of sedges establishing from the buried seed pool, or had no effect on them. Because the initial site conditions are suitable for the native sedges, there is little opportunity for facilitation by the sown species, and their effects are mainly competitive.

In cases where there is no buried seed pool, seeding of native species may be desirable. For example, when the entire organic mat is removed by blading, few buried seeds remain, and initial seedling establishment is slow, depending entirely upon current seed rain (Chapin and Shaver 1981). It is important to recognize that seeding of exotic grasses may be beneficial for restoration in some cases, despite its inhibitory effects on natural recolonization. The ideal solution may be to sow low densities of exotic grasses that are known to have poor survival or competitive ability in the arctic environment; this could provide initial cover without strongly inhibiting later establishment of native species (Cargill and Chapin 1987). Revegetation techniques encompass a wide variety of technologies that vary from very simple methods (natural vegetation) to quite sophisticated schemes. Some of these techniques are described elsewhere (USDA 1972). Much of this technology is being utilized and evaluated in ongoing revegetation studies.

As discussed previously, temperature is a major detriment of plant community. Research has demonstrated that cold hardening is a process by which some plant species increase their cold tolerance during exposure to low, nonfreezing temperatures. Like many engineering terms, the phrase cold tolerance includes a condition (cold), an object subject to the condition (plants), and a response of concern (tolerance, that is their relative ability to survive and prosper). The central focus of many cold-tolerance ongoing studies, what is the minimum temperature (and the duration of temperature) that the plant can tolerate when it has been conditioned through a natural process called cold hardening? Efforts to transfer cold-tolerance expressive genes has had only limited success. New techniques in molecular genetics will likely improve our understanding of this nontranslocatable, cellular process (Reid and Palazzo 1990).

More specialized techniques such as hormone treatment (Mitchell and McKendrik 1974), nurse crop (Hernandez 1973a) and seed mixing have also been evaluated. Techniques using native species such as replacing the organic mat (Hernandez 1973b) and fertilization of native communities for seed bed production (Mitchell and McKendrik 1974) also have been tried. The relatively short time since revegetation research began in the arctic is readily apparent since results are only tentative. The complexity and diversity of the problems involved in revegetation in this region ensure that one method can not meet all of the needs for revegetation. Therefore, the need in defining a revegetation approach is not the search for an optimum method but rather is the elucidation of a complex set of strategies which can be adapted to a multitude of situations and goals.

14.5
Conclusions

Mining on a scale large enough to justify mine development in the Arctic would have severe environmental effects. Reclamation efforts to achieve a permanent vegetative cover must deal with a considerable range of conditions. Permafrost is undoubtedly the most unusual environmental condition in the arctic. Permafrost is a major concern in mining and reclamation because of the profound changes that sometimes take place in the physical behavior of earth materials as a result of thawing, which is accelerated by mining operations and related disturbances in the thermal regime.

The means of solving the problem of reclamation was seen to be primarily in the development of efficient methods for optimizing the hydrological and hydrothermal regime of the disturbed area to artificially change or improve the natural equilibrium and to establish a stable, natural plant cover. The approach would involve characterization of physical aspects of the processes that create a water balance as well as determination of technical measures to develop an optimal heat and water balance in the soil for the germination and growth of plants. The foundation of the suggested approach is provided by the logical sequence: type of water recharge, drainage method (moisture control), technical measures for a system of water management, and proper effort of revegetation.

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Chapter 15

Pollution from Mining in Greenland: Monitoring and Mitigation of Environmental Impacts

Poul Johansen · Gert Asmund

15.1
Introduction

In Greenland, mining started as early as the 1850s and has taken place for copper, graphite, coal, cryolite, lead, zinc, and silver, but at present no mines are in operation. Environmental protection in connection with these operations was not an issue dealt with until the early 1970s, when the largest mine in Greenland was to start operating. Since then a legal framework has been developed for environmental protection in connection with exploration and exploitation of minerals in Greenland with standards similar to those used in North America and Western Europe. Monitoring and assessment of environmental impacts have been part of this process, in which the authors of this chapter have been deeply involved.

The chapter describes the environmental impacts and issues of mining operations in Greenland. Historically only three major mines have been in production, all of which will be dealt with: a cryolite mine in South Greenland (Sect. 15.2), a lead-zinc mine in East Greenland (Sect. 15.3), and a lead-zinc mine in West Greenland (Sect. 15.4). The location of these is shown in Fig. 15.1. The focus is on heavy metal pollution, which has been by far the most important environmental problem of these mines. In Sect. 15.5 the legal framework relating to mining and environmental protection is discussed briefly.

15.2
The Cryolite Mine at Ivittuut

The cryolite mine is located at Ivittuut in Arsuk Fjord in South Greenland (Fig. 15.1). A small Greenlandic community, Arsuk, with ca. 300 inhabitants is located 15 km to the west of Ivittuut, and Greenlanders from this community use Arsuk Fjord for fishing and hunting. Kangililinnguit (or Grønnedal) 5 km to the north-east of Ivittuut is a Danish naval station with ca. 200 inhabitants, mainly Danes who use the area for recreation.

The following presentation is based on studies conducted by the authors and by mining companies exploring and producing in Ivittuut. A detailed compilation of all information is presented in a Danish report by Johansen *et al.* (1995), and some of this has been published by Johansen *et al.* (1985) and by Asmund *et al.* (1988, 1991).

15.2.1
Mining Operations and Pollution Sources

The cryolite mine at Ivittuut was an open pit operation and ore was mined from 1854 to 1987 by the Danish company Kryolitselskabet Øresund A/S. Approximately 3.3 mil-

Fig. 15.1. Location of three major mines in Greenland (Maarmorilik, Ivittuut and Mestersvig) and the largest prospect (Citronen Fjord)

lion tons of cryolite ore was shipped out during the lifetime of the mine, most of this in the last 50 years, when about 50 000 tons was shipped out annually. The cryolite was located in one orebody, ca. 115 · 50 m in plan and ca. 70 m deep. The main part of the orebody consisted of siderite-cryolite rock, i.e. cryolite with about 20% siderite and 1–2% quartz and sulfides. The formation was located very close to and partly in the fjord, and a dam was built towards the fjord to prevent seawater from filling the pit. Figure 15.2 shows the pit during mining in the early 1980s.

At Ivittuut the ore was handled only mechanically and shipped out after crushing and sorting. Flotation of the ore was not carried out in Greenland but at a plant in Copenhagen, Denmark. During the first years of mining only almost totally "clean" cryolite with a very low content of other minerals was shipped out. Therefore, large quantities of unsalable "unclean" cryolite were produced and used as landfill in quay and road building. Later, the methods to extract cryolite from the ore were improved, so that ore with a cryolite content down to 40% could be used. Thus, it became profitable to use the old landfill areas containing cryolite, and in the later part of the lifetime of the mine extensive digging, sorting, and shipment of cryolite from landfill areas took place. Low-grade ore originally left in the pit was also produced during the last few years of the mine's life.

The first environmental studies at Ivittuut were carried out in 1982 and showed that lead and zinc had polluted seaweed and blue mussels in the intertidal zone of a large part of the fjord (see Sect. 15.2.2). This was unexpected, and further studies were then

Fig. 15.2. The mine at Ivittuut, showing the open pit, landfill areas at the coast and a ship being loaded with cryolite

initiated to identify pollution sources. The by far most important source was shown to be the landfill areas at the coastline, which released dissolved lead and zinc under influence of the tidal water, whereas the possible pollution sources, dust dispersal and water release from the pit, were not important. The chemical composition of the landfill areas has been studied as part of an exploration project to mine them. On average they contain 12% cryolite, 2200 mg kg^{-1} lead, and 2040 mg kg^{-1} zinc. Copper and cadmium also occur, but in lower concentrations, and seem not to have affected biota in the fjord.

A study was initiated to quantify the transport of dissolved lead from the landfill areas at the coast by sampling and analyzing water from five different levels at different times over the tidal period. This was repeated several times from May 1985 to April 1986. The general finding was that seawater with a lead concentration of about 0.1 µg l^{-1} entered the landfill areas with the tide. In the landfill, lead was released to the entering tidal water resulting in dissolved lead concentrations between 20 and 5000 µg l^{-1} dissolved in the seawater that returned to the fjord at falling tide. It was estimated that the amount of lead released from the landfill areas was between 1 and 3 kg per day corresponding to approximately 400 and 1000 kg per year. A similar but not as extensive study was conducted for zinc, and it was concluded that the release of zinc was in the same order of magnitude as for lead.

15.2.2
Environmental Impact and Mitigative Measures

A major environmental study was carried out in 1983 covering lichens, seawater, marine sediments, seaweed, blue mussels, prawns, and six fish species. Samples were analyzed for lead, zinc, copper, and cadmium. Fish and prawns did not accumulate significant amounts of metals. On the other hand, seaweed and blue mussels, from some localized areas at Ivittuut, had elevated levels of copper and cadmium. After 1983 a program for monitoring the heavy metal pollution in the area has included only seaweed and blue mussels.

Examples from the monitoring program are shown in Figs 15.3–15.5. Figure 15.3 shows the lead concentration in blue mussels in Arsuk Fjord at the most recent study carried out in 1995. The highest lead levels are found at the pollution source in Ivittuut, with

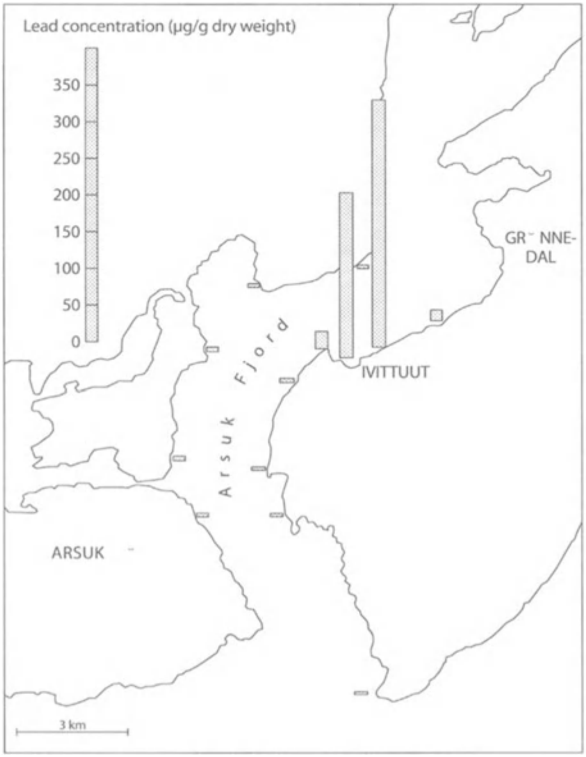


Fig. 15.3. Lead concentration (µg g^{-1} dry weight) in blue mussels in Arsuk Fjord, 1995

Fig. 15.4. Zinc and lead concentration ($\mu g\ g^{-1}$ dry weight) in brown seaweed from three groups of localities in Arsuk Fjord from 1982 to 1995

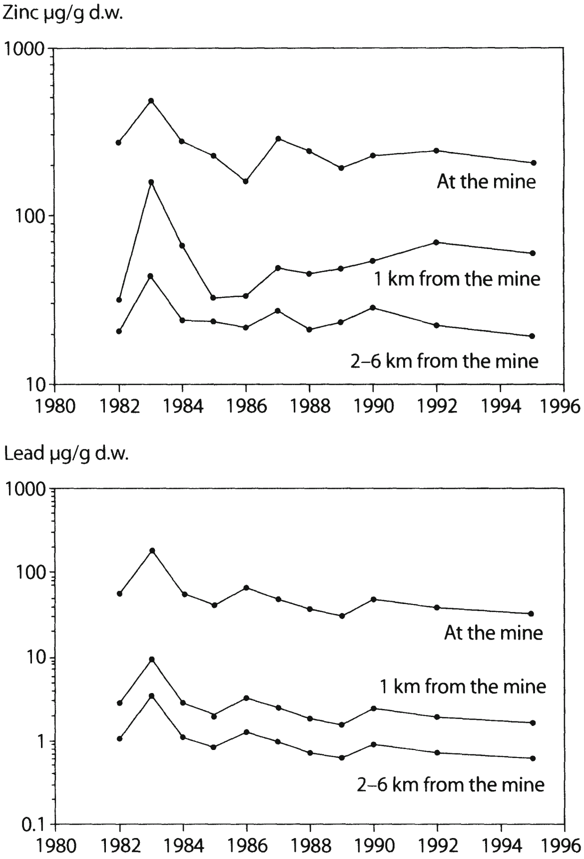
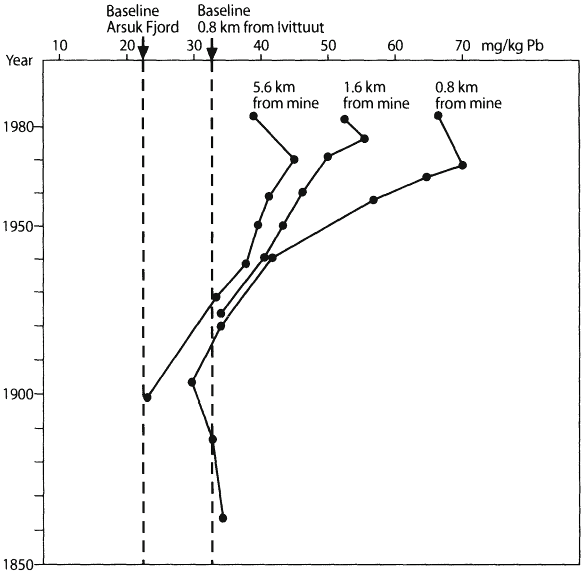


Fig. 15.5. Lead concentration ($\mu g\ g^{-1}$ dry weight) in dated sediment cores from Arsuk Fjord



decreasing values away from the source. At all sampling stations lead is elevated above background, which means that a stretch of coastline of more that 50 km is affected. Figure 15.4 shows the zinc and lead concentration in brown seaweed in three groups of localities of the fjord, and a decreasing trend over the monitoring period is observed. A possible explanation of this trend could be that lead and zinc gradually are washed out by the tidal action and removed from the landfill areas.

Sediment cores from the fjord were collected in 1983, and sliced every 1 cm, for dating (Pb-210 method) and metal analyses. On this basis an historical record of the lead pollution in the fjord was constructed as shown in Fig. 15.5. It may be seen that the lead concentration in the sediments did not increase above baseline until the late 1930s. There is an interesting correlation between the data shown in Fig. 15.5 and the amount of cryolite shipped out from Ivittuut. From 1854 to 1937 this amount was quite constant and small, about 6000 tons per year, but in 1937 it rose sharply to about 50 000 tons and remained at this level until mine closure.

15.2.2.1
Mitigative Measures

For many years the mining operation was conducted without environmental regulations, and it was unexpected when the first environmental studies in 1982 showed a widespread lead and zinc pollution of the fjord. To significantly mitigate impacts from the dominating source, the landfill areas at the coast, it would be necessary to remove 1 million tons of landfill or to shield it from the action of the tide water. This would be an operation so large that it could not be legally imposed on the mining company. The source, therefore, still exists and will continue to release lead and zinc to the fjord for many years. However, a slow decrease in the release rate has been observed in the monitoring program.

15.3
The Lead-Zinc Mine at Mestersvig

The lead-zinc mine at Mestersvig is located at Kong Oscars Fjord in East Greenland (Fig. 15.1). It is far from human settlements, the closest one being the small Greenlandic town Illoqqortormiut (or Scoresbysund) about 200 km to the south-east with ca. 500 inhabitants. They live mainly by hunting marine and terrestrial animals. However, they only rarely visit the Kong Oscars Fjord area at the mine.

The following presentation is based on studies conducted by the authors and the Greenland Technical Organization and on information from the mining company producing in Mestersvig. Most of this information is only available in reports published in Danish, but some has been published by Johansen *et al.* (1985).

15.3.1
Mining Operations and Pollution Sources

The mine and a concentrator were underground, located ca. 10 km inland from Kong Oscars Fjord (Fig. 15.6). The mine was operated by the Danish company Nordisk Mineselskab from 1956 to 1963 when the orebody was depleted. By then, 554 000 tons of

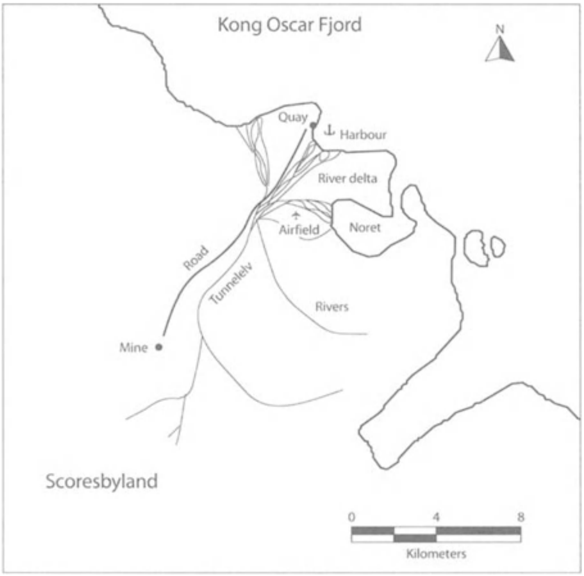


Fig. 15.6. The area at the lead-zinc mine at Mestersvig, East Greenland

ore had been mined, and 58 000 tons of lead concentrate and 75 000 tons of zinc concentrate produced. A road was constructed between the site and the fjord where concentrate was shipped out. Tailings from the concentrator were discharged on a nearby mountain slope at the mine, from where most has slid into the river Tunnelelv flowing to Kong Oscars Fjord.

The operation seemingly was carried out without any concerns about environmental impacts. The first environmental studies were carried out in 1979 and showed that tailings were an important pollution source. It was estimated that only 15% (66 000 tons) of tailings remained in the original tailings disposal area, and the rest of the tailings had been washed into the river, the river deltas, and the fjord. The tailings remaining had high concentrations of lead (0.66%) and zinc (2.13%).

The environmental studies also identified the loss of concentrate in connection with loading operations in the local harbor at the fjord as an important pollution source. During the operating period, concentrate was transported in bags from the mine to the harbor. From there, the bags were loaded on barges, which were hauled to a ship anchored a few hundred meters off the coast. The bags were then opened and emptied into the ship. It is known that significant amounts of concentrate were spilled in the fjord during loading, including whole 4-tons bags and whole barge loads of about 30 tons of concentrate.

15.3.2
Environmental Impact and Mitigative Measures

As expected, the results from the first environmental studies, carried out in 1979, showed high levels of lead and zinc in the sediments of Tunnelelv and in the river deltas at Kong Oscars Fjord. Concentrations of up to 13% of Zn and 0.7% of Pb were measured in the finest sediment fraction (<50 µm). Also high concentrations of lead and zinc were found in the water of Tunnelelv downstream of the mine (zinc values up to 150 $\mu g\,l^{-1}$ dissolved and 397 $\mu g\,l^{-1}$ particulate; lead values up to 12 $\mu g\,l^{-1}$ dissolved and 25 $\mu g\,l^{-1}$ particulate). At the mine site the lead and zinc concentrations in lichens were 500 $mg\,kg^{-1}$ d.w. (dry weight) and 140 $mg\,kg^{-1}$ d.w, respectively. These values are about 100 times above natural background levels. In lichens, elevated zinc and lead levels were also found up to 10 km from the mine site, demonstrating that some airborne transport of metals had taken place.

The most significant impact was found in brown seaweed from the intertidal zone. Elevated lead and zinc levels were found on a significant part of the south coast of Kong Oscars Fjord (Fig. 15.7). The figure shows that the heavy metal pollution of the fjord originates from three main areas: the northern and the southern delta of Tunnelelv and the harbor area in-between, reflecting the transport of tailings in Tunnelelv and the spill of concentrate at the harbor.

Further environmental studies have been conducted in 1985, 1986, 1991, and 1996, and have included seaweed in all cases. In 1985 also bivalves, fish, and seals were included, and some of these also in 1991 and 1996. The studies have shown that cadmium and copper are not elevated in marine biota, and that zinc is only elevated in seaweed, whereas lead is elevated in several species, including three bivalve species and the liver and bone tissue of sculpin (Hansen and Asmund 1986; Agger *et al.* 1991; Asmund *et al.* 1997). In seals, levels in muscle, liver, and kidney are not higher in Kong Oscars Fjord than elsewhere in Greenland (Agger *et al.* 1991).

The time trend of the pollution of the area is illustrated in Fig. 15.8 showing lead and zinc levels in seaweed. It may be seen that both lead and zinc concentrations have decreased over the monitoring period as a whole. However, lead levels had increased significantly in 1991. This was caused by the erosion of an area in the harbor severely polluted with lead, resulting in exposure and release of old concentrate spillage.

Overall levels are expected to continue to decline, as the contaminated material becomes more and more dispersed and gets mixed up with or covered by nonpolluted sediments. However, Kong Oscars Fjord will be affected by heavy metal pollution for many years. Lead and zinc concentrations in the marine environment may be expected to fluctuate, with high levels following coastal erosion of major contaminated land areas, followed by rapidly decreasing levels in periods with no erosion.

15.3.2.1
Mitigative Measures

While the mining operation was carried out, environmental impacts and protection were not issues dealt with. This first happened in the 1980s, after it became known that part of Kong Oscars Fjord was affected by lead and zinc pollution from the mine. Studies were initiated by the Danish government agencies to assess whether the pollution could be mitigated. One possibility was to partly seal the coastal area that had been severely polluted by loss of concentrate by covering the most contaminated area with stones and gravel and shielding the covered area from the sea with a bentonite layer. However, the costs of this operation would be very high and would only solve the problem to some degree. It would significantly reduce the lead pollution, which mainly seems to originate from concentrate spillage, but it would not affect the pollution from the

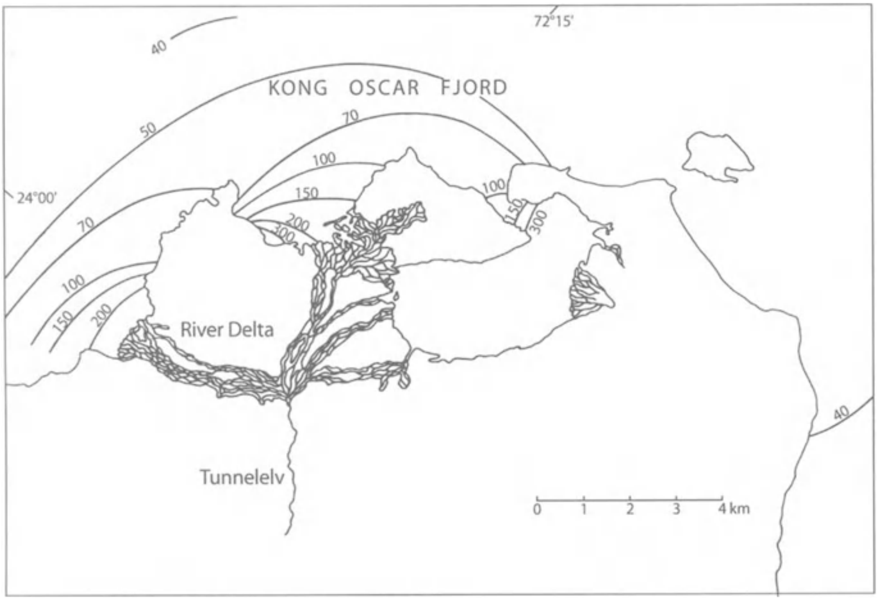
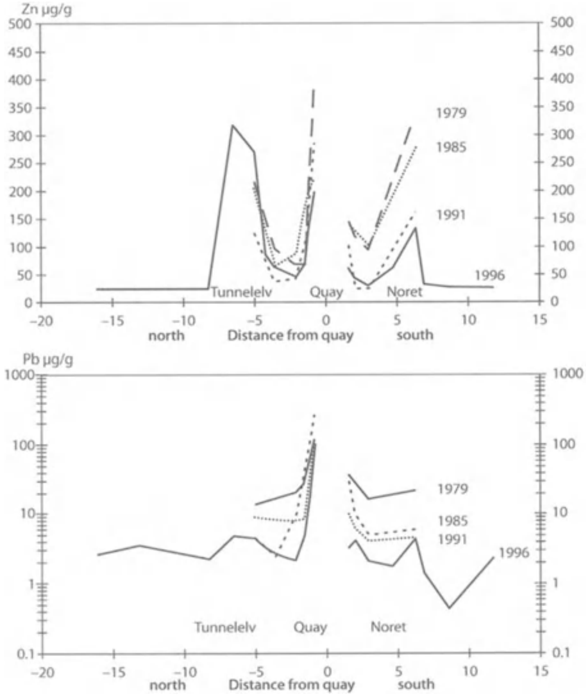


Fig. 15.7. Zinc concentration (μgg^{-1} dry weight) in brown seaweed at Mestersvig, 1979. Lines combine points at the coast with equal concentration

Fig. 15.8. Zinc and lead concentration (μgg^{-1} dry weight) in brown seaweed at Mestersvig 1979–1996



other main source, the tailings that have been dispersed over such vast areas. It can be concluded that no realistic abatement measures are possible.

15.4
The Lead-Zinc Mine at Maarmorilik

The lead-zinc mine was located at Maarmorilik in the inner region of the large Uummannaq Fjord system in north-west Greenland (Fig. 15.1). A small Greenlandic settlement, Ukkussissat, with ca. 200 inhabitants is found about 25 km from the mine. People in this region of Greenland mainly live by fishing and by hunting marine mammals and birds, but while the mine operated also several local people worked here.

The following presentation is based on studies conducted by the authors and by the company operating the mine. Some information is available in Danish reports only, but much has been published in English (Johansen *et al.* 1985; Asmund *et al.* 1988, 1991, 1994; Johansen *et al.* 1991; Asmund 1992a,b).

15.4.1
The Mining Operation and Pollution Sources

The mine called the Black Angel was the largest mining operation in Greenland and operated from 1973 to 1990 by the Danish company Greenex. About 11 million tons of ore with ca. 12% zinc and 4% lead was mined and milled at Maarmorilik. The general outline of the operation is shown in Fig. 15.9. The main orebodies were found in the Black Angel mountain 600 m above sea level. Crushed ore was brought in cable cars across a fjord to a flotation plant in Maarmorilik, where on average 135 000 tons of zinc concentrate and 35 000 tons of lead concentrate were produced annually and shipped out in the open water season (June to December). Tailings from the plant were discharged into the small adjacent fjord Affarlikassaa. Figure 15.10 shows a picture with a view from the mine towards Maarmorilik and Qaamarujuk Fjord.

Before the mine started operating in 1973, the environmental impact was assessed and environmental baseline studies were conducted. Shortly after the mine began producing, studies showed that the environmental impact was much more severe than anticipated, and since then the impact of the mine has been an important issue followed closely by the public and government agencies. Environmental monitoring and regulations became part of an ongoing process to mitigate the impact.

The main pollutants were identified as lead and zinc. Trace amounts of mercury, cadmium and copper also occurred, but not in concentrations so high as to cause important elevations of levels in marine biota (Johansen *et al.* 1991). Several sources of heavy metal pollution were identified: tailings, waste rock, ore crushing, and transport of concentrates were the most significant ones. Polluted water from the mine also occurred, but only for a short period (Johansen *et al.* 1985).

In terms of quantities of pollutants released to the environment, tailings were by far the most important source. Tailings were discharged at 25-m depth into the fjord Affarlikassaa, where solids settled, as a sill at the mouth of the fjord prevented tailings solids from entering the adjacent larger fjord Qaamarujuk (Fig. 15.9). However, soon after mining started in 1973, elevated concentrations of dissolved lead and zinc were measured in the waters of Affarlikassaa. When the fjord water mixed during winter,

Fig. 15.9. The Maarmorilik area. *w.d.* water depth

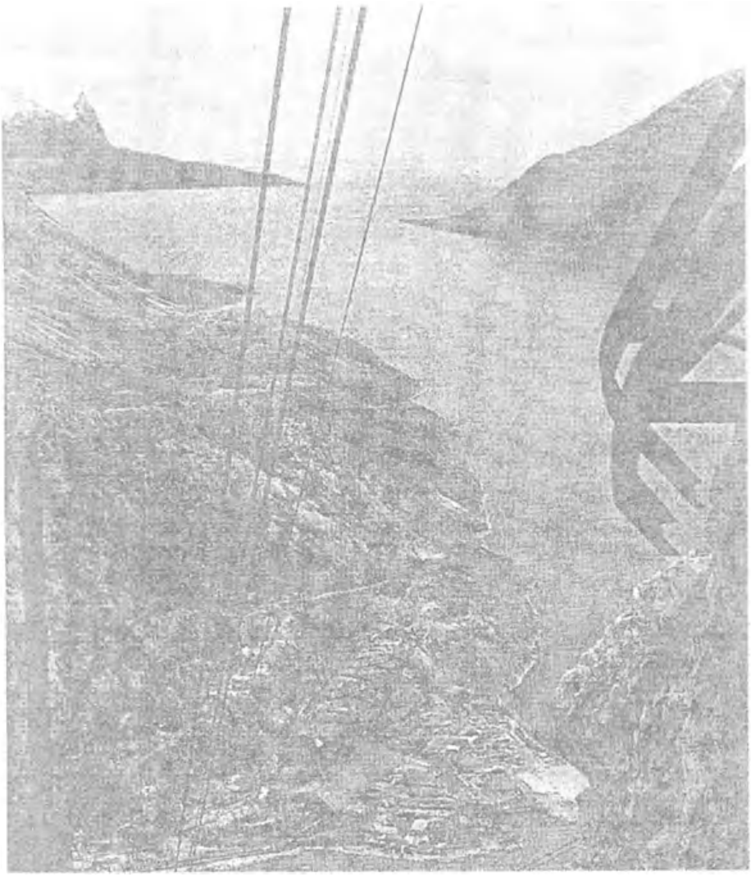


Fig. 15.10. View towards Maarmorilik and Qaamarujuk Fjord from the Black Angel Mine, 600 m above sea level

dissolved metals were transported from the lower to the upper water column and over the sill into the surface waters of Qaamarujuk. It is likely that lead and zinc carbonate in the tailings were the source of dissolved metal, although most of the metals in the tailings occurred as sulfides. It was estimated that between 10 and 30 tons of lead and between 30 and 55 tons of zinc dissolved per year while the mine operated (Asmund 1992a). After settling of the tailings on the fjord bottom, only very small amounts (if any) of lead were released to the overlaying water, whereas zinc was released in significant amounts, about 10 tons per year (Asmund 1992a).

A second important source of pollution is waste rock dumps. During the first 10 years of operation, disposal of waste rock outside the underground mine workings was not regulated, and 2–3 million tons of waste rock with 0.1-0.8% lead and 0.3-2.3% zinc was left in four dumps on the slopes of steep mountains in the area. Metals are released to the sea mainly in the form of sulfide particles in rain and meltwater from glaciers. An average annual input of 8 tons of lead has been estimated from this source. Most of this, however, settles close to the river outlet in the fjord.

One of the waste dumps was located directly at the Qaamarujuk Fjord, partly in and below the coastline. This caused a high mobilization of lead and zinc from the waste, which was in direct contact with seawater and was exposed to wave and tidal action. The highest lead and zinc levels in seaweed and blue mussels were found at this dump. As part of a plan to mitigate metal pollution in the area after mine closure, this dump was removed to the extent possible as described below.

A third significant source of metal pollution was dust. An annual input to the environment of 2 tons of lead and 5 tons of zinc from all dust sources has been estimated during the last years of the mine operation (Johansen *et al.* 1997). The main source of dust was the ore crusher, and because it was located ca. 600 m above sea level, dust and metals from it could be dispersed over long distances. Another important source of dust was the conveyor belts transporting concentrates from the concentrate storage building to the bulk carriers. Other pollution sources also existed, e.g. mine water and ventilation from the mine and the concentrate storage building. However, these sources do not seem to be important compared to tailings, waste rock dumps, and dust from the ore crusher (Greenex Environmental Action Plan, see Asmund *et al.* 1994).

15.4.2
Environmental Impact and Mitigative Measures

As mentioned, the environmental impact turned out to be much more severe than anticipated shortly after the start of the mining operation. For many years the impact has been monitored closely, both by the mining company and the regulating agency. The results of the monitoring program were used to assess the effect of mitigative measures taken by the mining company. The studies have included lichens (as an indicator of atmospheric pollution), seawater, marine sediments, seaweeds, blue mussels, several fish species, prawns, seabirds, and seals. A thorough review of the environmental studies has been presented by Johansen *et al.* (1991).

The amount of lead dispersed with dust has been monitored systematically since 1983 by sampling and analyzing a lichen species over a large area at the mining site. The main results are shown in Fig. 15.11, from which it can be seen that significant reductions have occurred as a result of mitigative measures taken mainly after 1984. After

Fig. 15.11. Estimated amount of lead dispersal with dust at Maarmorilik, 1983–1996

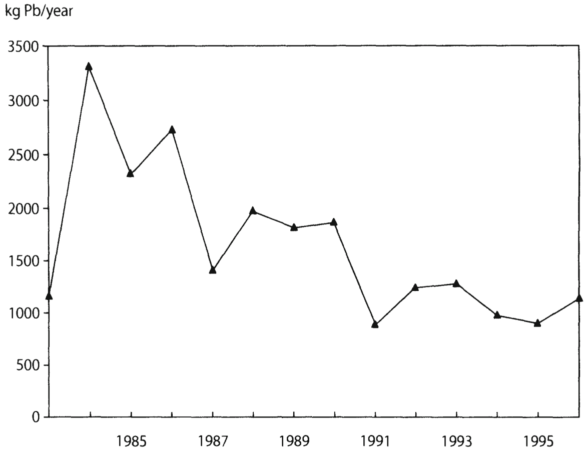
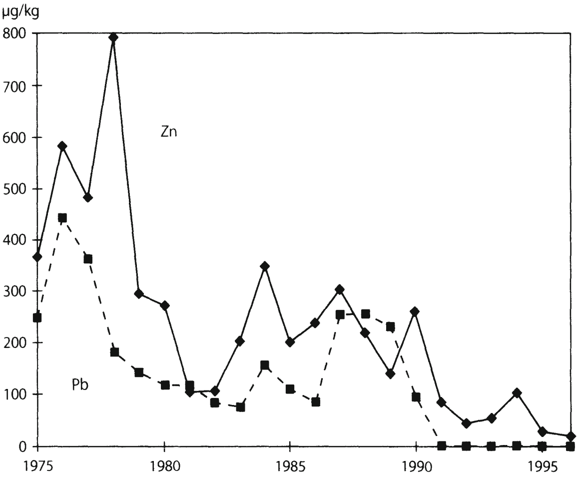


Fig. 15.12. Lead and zinc concentration in bottom water of Affarlikassaa in autumn 1975–1996



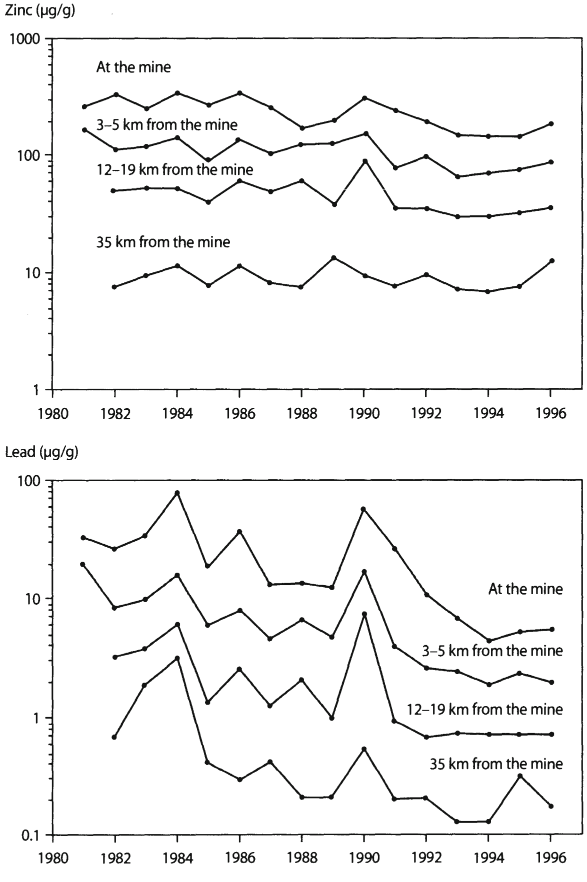
mine closure in 1990 the amount of metals in dust has remained at an unexpectedly high level (Fig. 15.11). It is likely that the dust dispersal after mine closure originates from particles which have been spread over large areas while mining took place.

Seawater from the fjords at Maarmorilik has been analyzed for lead and zinc on a regular basis. The highest metal levels in the bottom water of the Affarlikassaa Fjord into which tailings were discharged were found at the beginning of the period (Fig. 15.12). In the late 1970s both the lead and the zinc concentrations decreased in the fjord as a result of increased flotation efficiency and other mitigative actions. After that, zinc concentration remained at the same level until mine closure, after which it has declined significantly, despite the release of zinc by deposited tailings and waste rock on the bottom of Affarlikassaa. The lead concentration remained at a lower level until 1987 when a new orebody which was more difficult to flotate was brought into production. After mine closure in 1990 the lead pollution of seawater has changed drastically, and only very small amounts of lead are released from settled tailings and waste rock on the bottom of Affarlikassaa (Fig. 15.12). While mining took place, lead and zinc levels were also elevated in Qaamarujuk Fjord, particularly in winter when mixing of the water masses in Affarlikassaa caused transport of dissolved lead and zinc into the surface waters of Qaamarujuk (Asmund 1992a).

Brown seaweeds and blue mussels were used as biological indicators of the pollution in space and time. The main results for seaweed are shown in Fig. 15.13. Seaweeds from a large part of the coast of the fjords have elevated zinc and lead concentrations and these levels have decreased over the monitoring period, particularly for lead. Nevertheless, zinc levels have remained rather high after mine closure.

Lead levels in blue mussels increased significantly in all three fjords at the mine after the start of the mining operations. Lead concentrations above 3000 $\mu g\,g^{-1}$ (on a dry weight basis) were observed at the most affected sites. This is more than 3000 times higher than the concentration found in unaffected areas. For many years it has been recommended not to collect and eat blue mussels, in an area up to 30 km from the mine site, because of high lead concentrations. Studies in the area have shown that only about half of the lead taken up by the mussels in the polluted area at the mine is excreted within a year of been transplanted to a clean environment, and after this first year no

Fig. 15.13. Zinc and lead concentration ($\mu g\,g^{-1}$ dry weight) in brown seaweed at Maarmorilik from 1981 to 1996



further excrease takes place (Riget *et al.* 1997). This means that mussels which have been exposed to elevated lead levels near the mine will be affected all their lives. To assess the yearly impact on blue mussels we therefore have introduced a method of transplanting mussels from a clean site to polluted sites at different distances from the mine and have successfully been able to use the first year of lead uptake in the transplanted mussels as an indicator of impact (Fig. 15.14; Riget *et al.* 1997). Transplantations after mine closure confirm that the intertidal zone is still affected by lead pollution, as mussels after one year of transplantation to the Maarmorilik area obtain a higher lead concentration. However, the transplantations indicate that the impact after mine closure has declined to about 20% of the level found while mining took place, and that this level is slightly declining since the mine closed.

A number of marine fish species and a prawn species have been monitored regularly and no elevations of zinc in these have been observed. Lead levels in the meat have remained low whereas in liver and bone tissue elevated lead levels have been found in many cases. The lead levels in fish and prawns have generally decreased since the mid 1980s and particularly after mine closure. Figure 15.15 shows lead concentration in the liver of shorthorn sculpin, a rather stationary fish species. After the mine closure

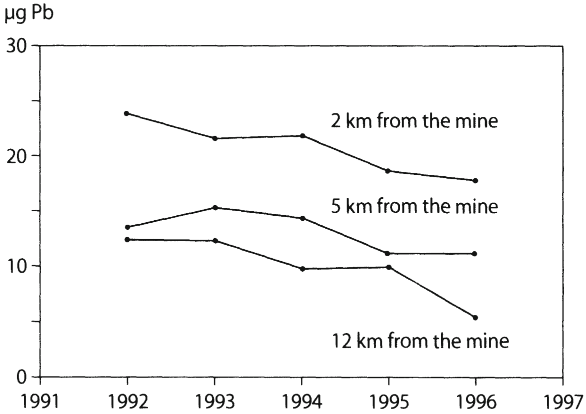


Fig. 15.14. The lead content in blue mussels (µg Pb/mussel) after 1 year of transplantation to the Maarmorilik area

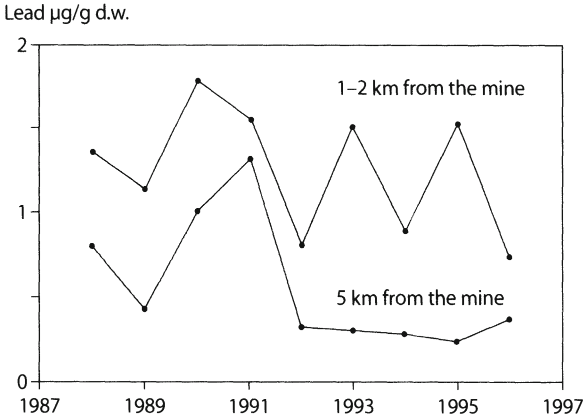


Fig. 15.15. Lead concentration (µg g^{-1} dry weight) in liver of shorthorn sculpin at Maarmorilik from 1988 to 1996

lead levels in sculpin remained high, and only fish living 4–5 km from the mine site showed a significant decrease.

The most common seal species (the ringed seal) and five marine bird species (common eider, Iceland gull, glaucous gull, and black guillemot) also have been studied. Metal levels were not elevated in blubber, meat, liver, and kidney of the seals. In the most recent bird study (1991) elevated lead levels were only found in bone tissue of the birds and not in the meat, liver, or kidney as in earlier studies. Lead concentrations in seals, birds, and in fish meat have all been below the guidelines for human consumption.

15.4.2.1
Mitigative Measures

When it became obvious in the mid-1970s that the impact of the mining operation was more severe than expected, work was initiated to abate the pollution. The mining company organized a task force which included external experts to formulate an action plan. This plan was discussed with and approved by the regulatory agency and became the basis for several mitigative actions that were implemented throughout the lifetime of the mine. It included a number of studies and action plans and was regularly reviewed. Changes and amendments were made and approved as experience was gained and as a response to the results of the environmental monitoring program. The plan and the mitigative measures introduced are described by Asmund *et al.* (1994), and the main elements are mentioned below. In the production phase the most important measures taken were as follows:

1. Reducing the total amount of metal in tailings. Studies showed that the predominant metal quantity was within the particulate fraction of less than 0.01 mm. A number of measures were introduced into the process to reduce this fraction in the tailings, including improvements in grinding and classification, introduction of grinding-circuit flotation (unit flotation), and improved routines for reagent additives and process control. These measures resulted in a significant reduction of the lead content in the tailings, from 0.40% in 1973 to 0.18% in 1983.
2. Reducing the dispersal of particulates from the tailings discharge. The most important measure taken was the introduction of pulp dearation of the tailings slurry at the point of discharge into the fjord. This led to a considerable reduction of lead emissions, which earlier had been traceable to mineral-laden air bubbles. Chemical treatment of the tailings with lime, aluminum sulfate, and a flocculating agent enhanced the settling of particles in the fjord and reduced the dissolution of soluble compounds. Finally, to limit the spreading of the tailings stream in the fjord, the tailings pulp density was increased by raising its salinity with salt from a desalination plant operated to produce freshwater at the mine site.
3. Reducing the dust generation. Mine generated dust was significantly reduced after the installation of a bag filter for cleaning ventilation air from the ore crusher. Similarly, dust generated by transport of concentrates was reduced by enclosing the conveyor belts and implementing new routines for ship loading of concentrates.

When the mine stopped production in 1990, the mining company started a large decommissioning project with the objective of leaving the site in the condition it was in before mining started to a degree possible, and in particular to limit heavy pollution from sources left (Asmund *et al.* 1990). The most comprehensive measure included the relocation of the waste rock dump partially located in the sea. About 400 000 tons of waste rock was dumped on top of the tailings in Affarlikassaa and 200 000 tons was used to cover waste and scrap deposited in an old marble pit where concentrate had been stored. All production facilities and living quarters in Maarmorilik were torn down, sold, and shipped out, burned or left underground in the mine and below the waste rock in the old marble pit.

Environmental monitoring of the area has continued after mine closure and funds from the mining company are available to continue this program 15 years after mine closure.

15.5
Conclusions

The three major Greenlandic mines all have created heavy metal pollution. The most severe of these appears to be pollution with lead and zinc, when tailings, waste rock, or concentrate have come into contact with seawater. In particular the simultaneous action of waves, tidal water, and air seems to enhance dissolution of lead and zinc. In contrast, waste rock and tailings deposited in deep water do not cause lead to be released, whereas this is the case for zinc. This knowledge and other experience from environmental monitoring at mine sites in Greenland are very important and will be used when dealing with future mine projects. Environmental protection is an important issue when new projects are evaluated and is part of a legal framework for approving and regulating these.

With new projects the operators are responsible for assessing associated environmental impacts and designing their operations in such a way that impacts are avoided or limited to a degree acceptable to the regulating agency. Environmental baseline and monitoring studies are important tools in this process. At the end of operations sites must be restored to the extent possible and with no significant risks of pollution from deposited waste. Plans for restoration when closing an operation must be prepared as early as the application stage of a project.

In recent years exploration for minerals in Greenland has increased, but most projects are not at such an advanced stage that opening a new mine is considered. However, environmental studies recently have been carried out at one of the prospects, a zinc deposit in North Greenland at Citronen Fjord (Fig. 15.1) explored by the company Platinova. Significant natural zinc elevations have been found in the fjord at the orebody. It has been estimated that 1.5 tons of zinc enters the fjord during the three summer months dissolved in a river passing through extensive weathered sulfide outcrops (Glahder *et al.* 1996). The results of these studies have demonstrated the importance of conducting thorough baseline investigations before mining operations start.

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Chapter 16

Strategies for Remediation of Former Opencast Mining Areas in Eastern Germany

Ulrich Stottmeister · Walter Glässer · Helmut Klapper · Erika Weißbrodt
Bernd Eccarius · Christopher Kennedy · Martin Schultze · Katrin Wendt-Potthoff
René Frömmichen · Peter Schreck · Gerhard Strauch

16.1
Introduction

Lignite was for decades the main source of energy and the primary raw material used by the chemical industry in the now German Federal States of Saxony, Saxony-Anhalt, and Brandenburg. Except during the initial phases of extraction, lignite was exclusively recovered by opencast mining. The close ties between industry and mining led to the establishment of major industrial operations such as petrochemical, organochemical, and electrochemical plants located near the mining centers. In terms of its quality and composition, lignite in eastern Germany can be classified into West and East Elbian formations, corresponding to the Central German and the Lusatian mining districts.

Lignite is still mined in several regions in Germany (Fig. 16.1). The most important of these are the Central German district situated in Saxony and Saxony-Anhalt, the Lusatian district in Brandenburg, and the Lower Rhine district west of Cologne. However, the volume of lignite mining has generally decreased over the past decade and lignite has lost its significance as a primary raw material for the chemical industry. In fact, within the space of just 5 years (1989–1994), annual lignite extraction in central Germany and Lusatia has dropped from about 330 million tons to some 100 million tons. The current output target in these two districts is around 70–90 million tons per year.

16.1.1
Geological Background of Lignite Formation in Eastern Germany

All the main lignite deposits in Germany are of Tertiary age. In eastern Germany they were formed between the Lower Eocene and the Upper Miocene. Two different types of lignite accumulations occur in the Central German mining district (Fig. 16.2): (1) locally restricted lignite occurrences of Eocene age (occasionally reaching thickness of up to 100 m) accumulated in erosional structures such as channels and pockets caused by karst phenomena; and (2) stratiform coal seams of Eocene to Lower Miocene origin (sometimes separated by epicontinental sediments and up to 15 m thick) found between the southern margin of the White Elster basin and the area of Bitterfeld in the north.

Two workable seams are also located in the southern White Elster basin around Borna and in the Bitterfeld area. Lignite mining occurs in three areas in Lower Lusatia: in the ice-marginal valley of Baruth (open pit mines at Jänschwalde and Cottbus), at Seese-East and Welzow, and in the ice-marginal valley of Lusatia (open pit mines at

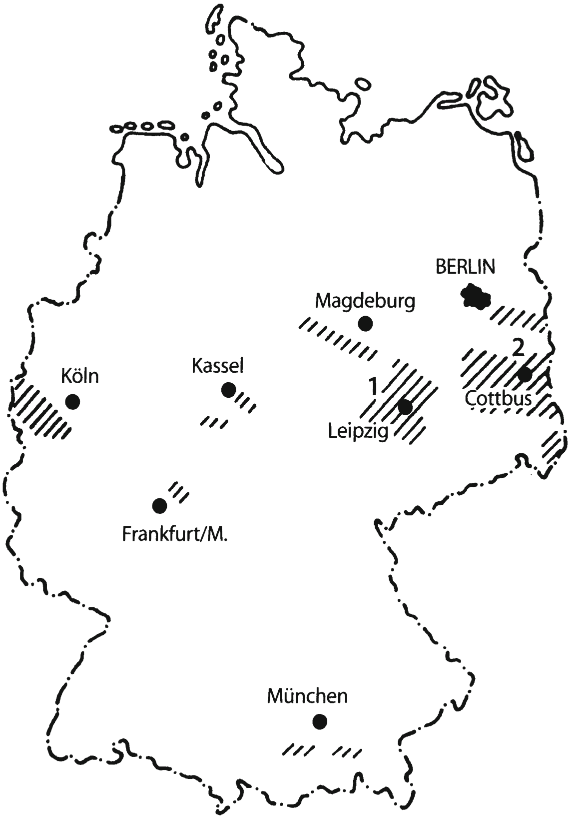


Fig. 16.1. Lignite mining areas in Germany, especially northwest Saxony (White-Elster basin and northern part of Leipzig bay, 1), and Lusatia (2)

Senftenberg and Lauchhammer). All major lignite deposits are of Miocene age; only the Calau seam at Seese-East (Oligocene) and the Nichtewitz seam (Eocene) are older (Fig. 16.3). In Upper Lusatia, only one deposit is mined at present: the open pit of Berzdorf near Görlitz, where a lignite seam up to 100 m thick is found in Tertiary basins formed by tectonic subsidence resulting from the tectonic uplifting of the Lusatian granodiorite massif in the Oligocene/Miocene.

16.1.2
Lignite Production as Feedstock of Industrialization of Eastern Germany

Beginning in the mid nineteenth century, lignite production increased continuously in eastern Germany. After World War I, opencast lignite mining was carried out on an international scale (at Geiseltal near Halle) and enabled the development of carbochemistry based on lignite pyrolysis including gasification, carbonization, coking, and hydrogenation. Large lignite-fired power stations contributed to the development of electrochemical industries in central Germany.

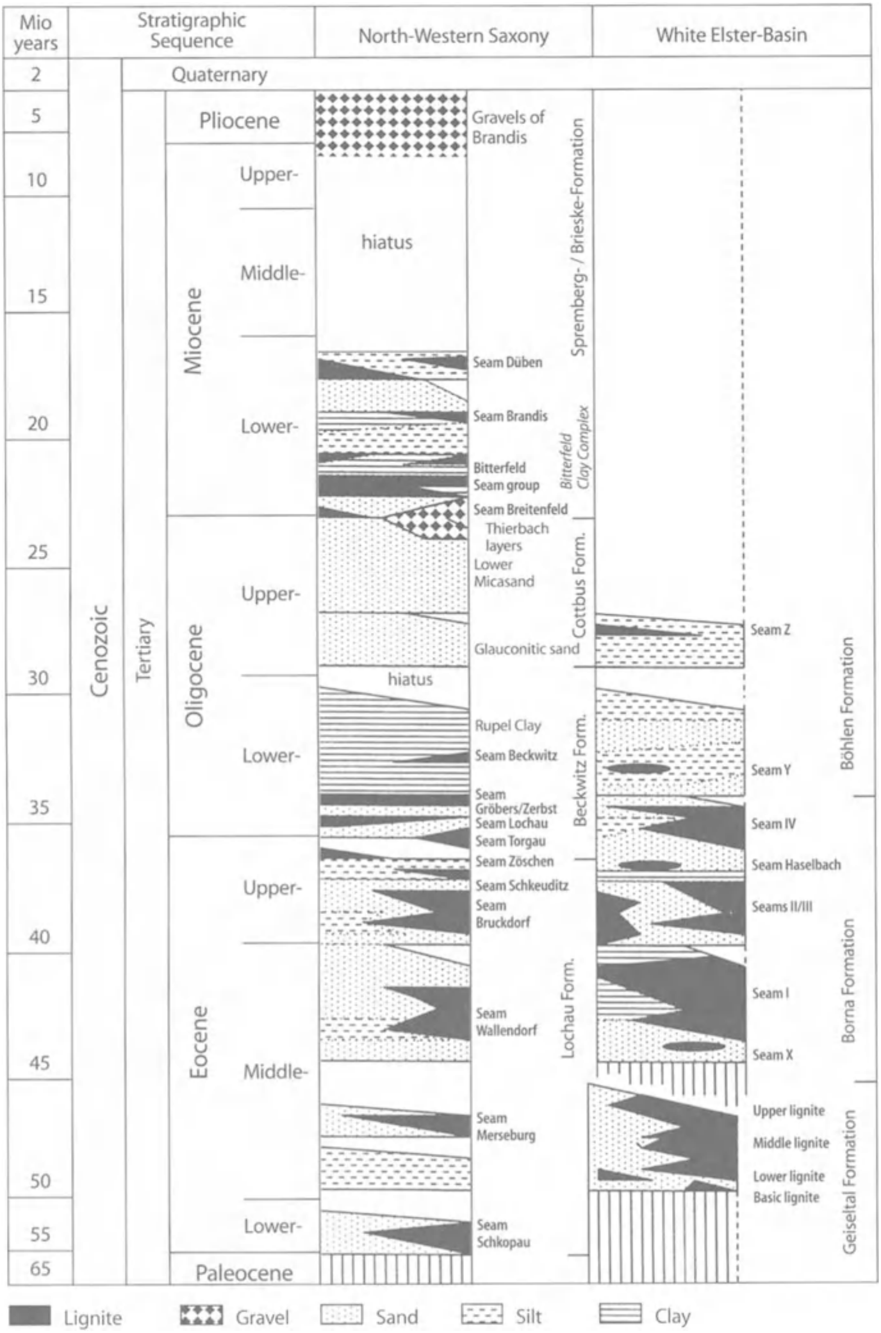


Fig. 16.2. Lithostratigraphic situation in the Tertiary in north-west Saxony

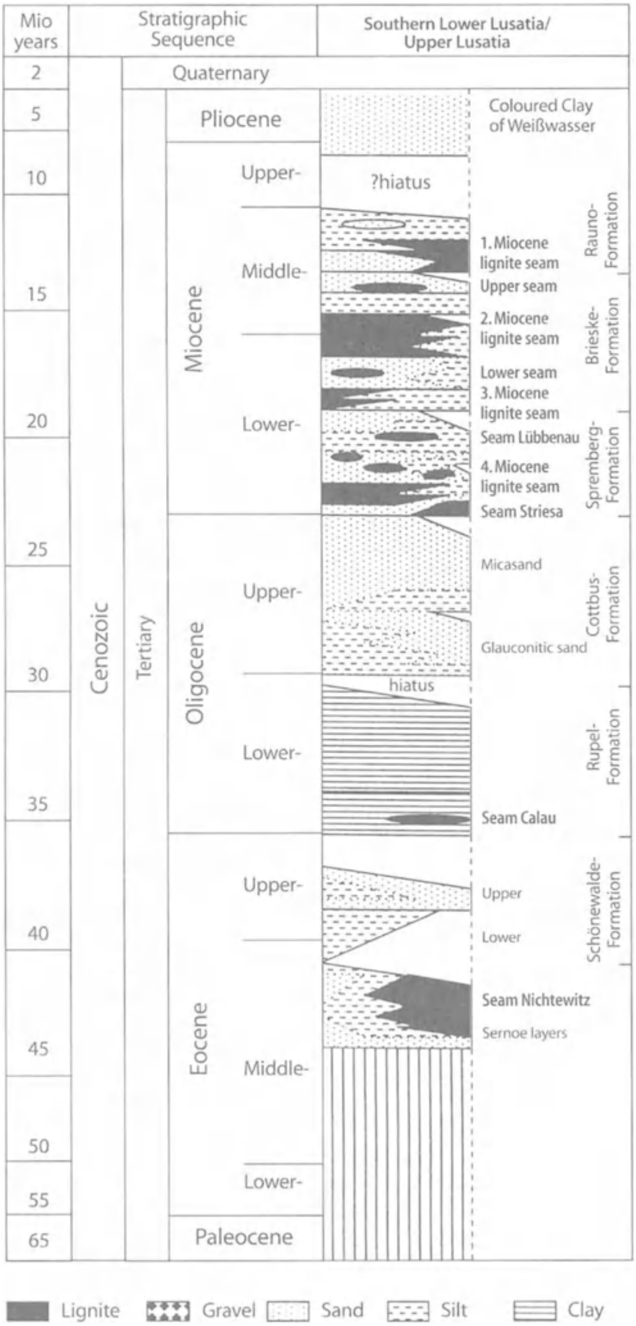


Fig. 16.3. Lithostratigraphic situation in the Tertiary in Lusatia

In the 1920s and 1930s, the Central German lignite mining region became highly industrialized, featuring, among others, lignite pyrolysis-based chemical industry and fuel production, aluminum and magnesium electrometallurgy, and chlorochemistry comprising pesticide and polymer production, thereby earning the epithet "the Black Triangle". After World War II the newly founded East German State (the DDR) built upon this industrial legacy by intensifying mining and industrial activities. Until its economic collapse in 1989, the DDR was the world's largest lignite producer, mining about 300 million tons per year from 33 open pits. Lignite was mined in the areas surrounding the industrial centers. In the east, the Lusatian mining district extended about 60 · 60 km, while in the west the central German lignite-mining district, in particular the area around the cities of Leipzig and Halle, extended about 90 · 40 km.

Lignite mining activities had an enormous ecological and sociological impact in eastern Germany, modifying entirely rural landscapes including towns and villages, and caused the water table to drop by more than 30 m. These mining operations adversely affected the water balance and destroyed the landscape, with no prospects of adequate restoration in sight. Moreover, emissions from obsolete industrial plants impaired the population's health and quality of life. Sulfur dioxide and dust from power plants and from domestic stoves, emissions from chlorinating plants, and organo-sulfuric and organo-nitrogenic compounds from pyrolysis were all widespread, while the rivers served as sewage pipes for untreated industrial effluents.

16.1.3
Situation After German Reunification in 1989

After German reunification, lignite mining was drastically reduced and lignite processing abandoned, causing an immediate improvement of the environmental situation. A restoration program was set up by the Federal German Government to assess the consequences of the former lignite opencast mining activities. An initial analysis of the situation presents the following scenario:

- Open pits are being flooded to form recreational lakes. The problems include mechanical slope erosion and the acidification of the water by natural processes (forming "acidic lakes"). This acidification is a general problem which has not yet been solved. Although the examples described below are rather specific, the remediation strategies developed could be used in similar cases, especially in Eastern Europe, where comparable situations are often encountered.
- Opencast mines are being flooded, but the water is contaminated by waste deposits adjacent to the lake. Problems include the fact that the process of the migration of leachates from the dump to the lake and to the groundwater is not fully understood and must be studied before proposals for remediation and protection can be elaborated.
- Abandoned opencast mines were used for uncontrolled waste disposal, especially industrial waste. Studies have to be carried out to assess the chemical interactions and the formation of metabolites so that proposals for remediation and protection can be drawn up.

16.2
Remediation of Acid Lakes from Former Opencast Mines

16.2.1
Description of the Problem

When a mining lake begins to form, the water quality depends on the origin and quality of the flooding water, the geological and lake-morphological conditions, and the anthropogenic pollution. Filling the lake with ascending groundwater is not a viable option in many cases because it leads to the geogenic acidification of the developing water body. Prior to mining activities in the eastern German coal fields, the subsoil was anoxic and the groundwater table was high. Under these conditions, pyrite and marcasite were stable. During the course of opencast mining these minerals were exposed to air and oxidized forming sulfuric acid and ferrous sulfate, the latter of which hydrolyzes to ferrihydroxide with additional proton generation. While low pH values are encountered in the groundwater of the overburden heaps, the lowest pH values are found in the mining lakes. Acidification causes metals to be leached from mineral phases, increasing the concentration of dissolved toxic metals in lake waters.

Most geogenically acidified water bodies are far more acidic than those affected by acid rain. In the Lusatian region of eastern Germany, pH values in mining lakes ranged between 2.6 and 3.8. Only very few mining lakes and most lakes affected by acid rain (all with relatively high aluminum levels) have pH values between 4 and 6. Some other lakes in Lusatia and the majority of mining lakes in the Leipzig–Bitterfeld region belong to the circumneutral pH range governed by the bicarbonate buffer system (Fig. 16.4).

The high metal concentrations often present in these acidic water bodies have a significant ecological importance. Relations between pH and metal content indicate that an increase in pH by neutralization measures may be the most promising way to overcome toxic concentrations (Fig. 16.5). Metal concentrations in hard waters are not

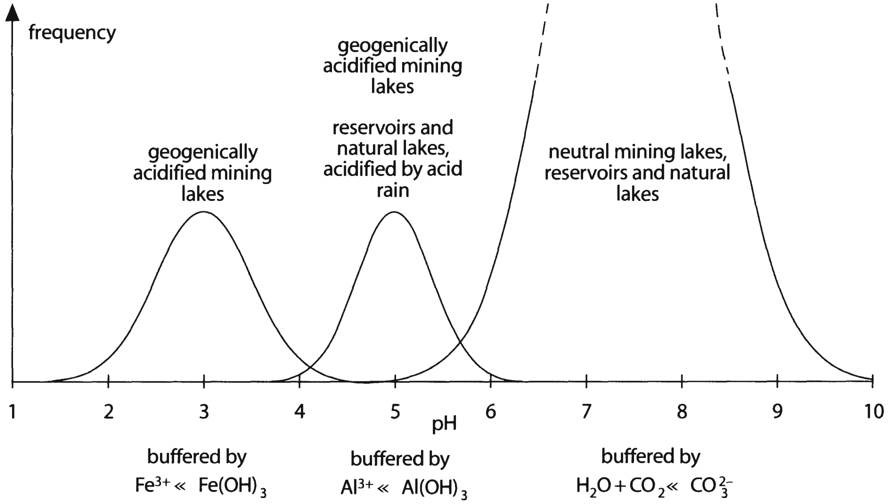


Fig. 16.4. Generalized frequency of lakes with different acidity in Germany (according to Klapper *et al.* 1996)

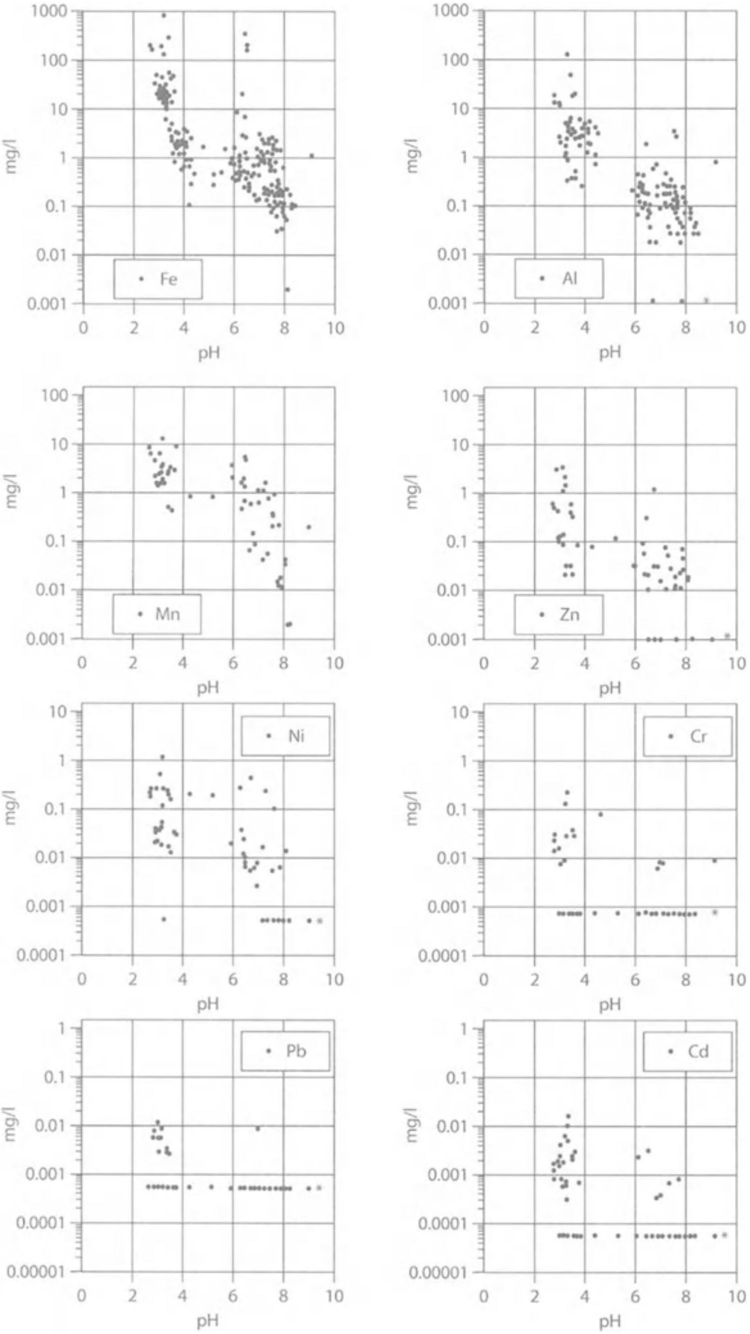


Fig. 16.5. Metals in the mining lakes investigated, as a function of pH. Plotted values are total concentrations

as toxic as those in the very soft waters of rain-acidified water bodies; the high calcium concentrations evidently decrease toxicity. In the youngest stages of geogenically acidified mining lakes, the water is dark red but clear due to the presence of dissolved iron. However, even this unfavorable habitat is colonized by acid-tolerant species, some of which occur in high abundance because of the lack of competition. Inorganic carbon is only available in low concentrations as CO_2. Bicarbonate is absent, so that the *Scenedesmus* type of photosynthesis (based mainly on bicarbonate) cannot be performed.

The first phytoplankton to appear are members of the same taxa found in bog waters: *Chloromonads, Cryptomonads,* and *Dinoflagellates.* Major productivity occurs at the level of pico- and bacterioplankton. The total algal biomass is low, and this criterion alone complies with the classification "oligotrophic" (Nixdorf *et al.* 1995). This low algal production is unexpected given the relatively high phosphorus supply. Instead, carbon limitation of bioproductivity appears to prevail. The low respiration rates and low solubility of CO_2 in acidic waters seem to be the reasons for this (Ohle 1981; Schindler 1994). The typical pioneer plant *Juncus bulbosus* occurs as the first macrophyte, followed by some mosses and emergent littoral plants (Chabbi *et al.* 1996, 1998).

Animals which need calcium carbonate for their skeletons or shells such as fish, amphibians, snails, mussels, and higher crustaceans cannot exist in an acidic environment. Moreover, zooplankton is also absent from the filling stages in acid lakes. The rotifers (especially *Brachyonus urceolaris)* inhabit lakes with a pH < 3, as does *Chydorus sphaericus* and *Cyclops. Daphnia* only occur in circumneutral lakes. Whether their absence is caused by acidity or by the frequency of invertebrate predators such as *Corixidae* is unclear (Steinberg *et al.* 1998).

16.2.2
Remediation Strategies

The first general question that has to be answered is whether remedial measures need to be taken. As outlined above, in most cases, especially with large water bodies, the aim is to develop more natural lakes which are populated by fish and can be used as recreational areas. The very acidic mining lakes are interesting subjects for limnological research and some smaller mining lakes with a highbase-binding capacity should be selected for long-term investigation. The alleviation of acidification should be considered and implemented right from the initial planning stages of mining until the recultivation of the overburden heaps. To obtain sustainable neutralization results, a complex, long-term abatement program must be set up covering the whole area affected by mining activities.

In order to minimize the contact of pyritic minerals with oxygen, the temporal and spatial dewatering of the lignite to be mined must be limited. The layers of overburden with the highest pyrite contents should be dumped in the deepest part of the mine, thus enabling these materials to become submerged as early as possible. The diffusion coefficient for oxygen in water is only 0.0001 of that in air.

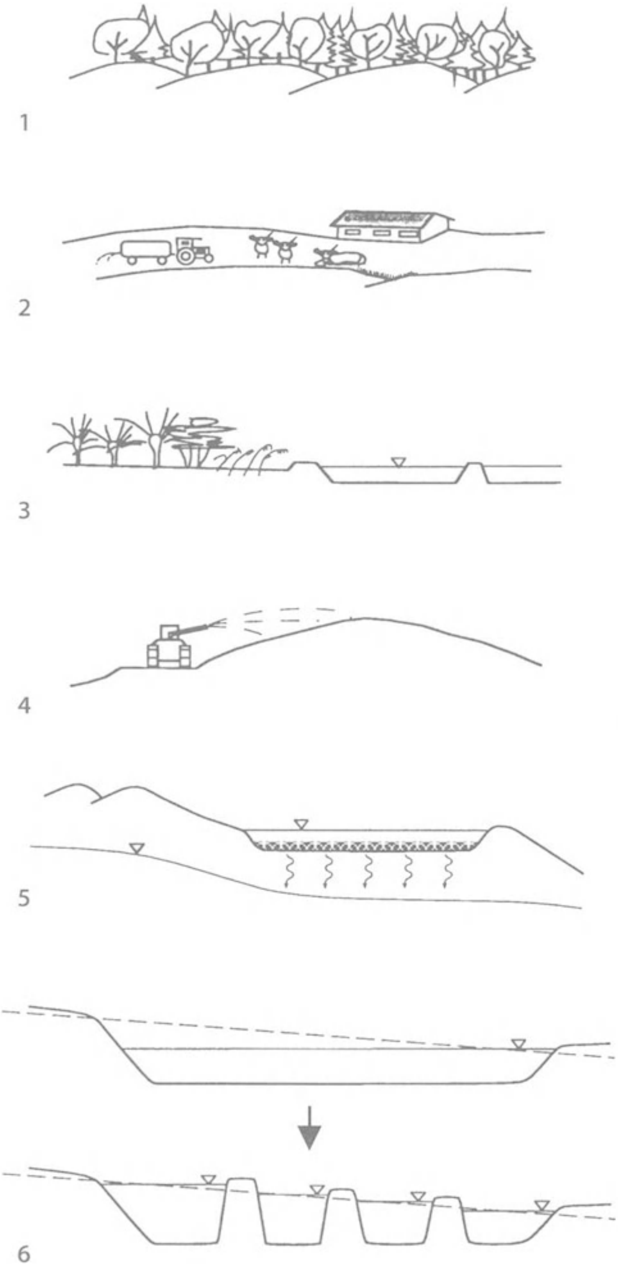
Groundwater from the undisturbed rock is mostly neutral, whereas that from the overburden heaps is acidic as a result of the oxidization of all the sulfidic ores during the mining process. Mining lakes with a long extension in the direction of groundwater flow act as groundwater pumps. At deeper levels, such lakes pull water from surrounding aquifers upstream while at higher levels downstream causes surface water to infiltrate downwards. In order to avoid intensive flow and mass transport, such long-shaped lakes should be subdivided, thereby decreasing the slope (Fig. 16.6, part 6).

Revegetation of the acidic heaps can be boosted by chemical treatment with alkalinity-delivering materials. Due to the low precipitation (about 500 mm $year^{-1}$) and high evapotranspiration rates in established mixed forests of the lignite mining area of eastern Germany, groundwater formation can be reduced to nearly zero. The soil and humus cover of the forest decreases unwanted oxygen penetration to the subsoil by leached organic substances (Fig. 16.6, part 1). Similar conditions are observed with permanent grassland and after the addition of organic fertilizers (Fig. 16.6, part 2). Wetlands and fish ponds can be re-established in the lowlands along the creeks and rivers. These were typical in parts of the Lusatian landscape before the onset of lignite mining, and the wet covers prevented the aeration of the ground (Fig. 16.6, part 3).

The application of tensides, which kill the bacteria responsible for acid formation, is appropriate for smaller heaps from sulfidic ores. Although toxic to *Thiobacillus*, these materials provide a degradable substrate for heterotrophic microorganisms which consume oxygen (Kleinmann *et al.* 1981; Fig. 16.6, part 4). High-alkalinity ponds are recommended to neutralize groundwater outlets to mining lakes. With limestone on the bottom they promote the infiltration of alkalinity into the acidic ground (Fig. 16.6, part 5).

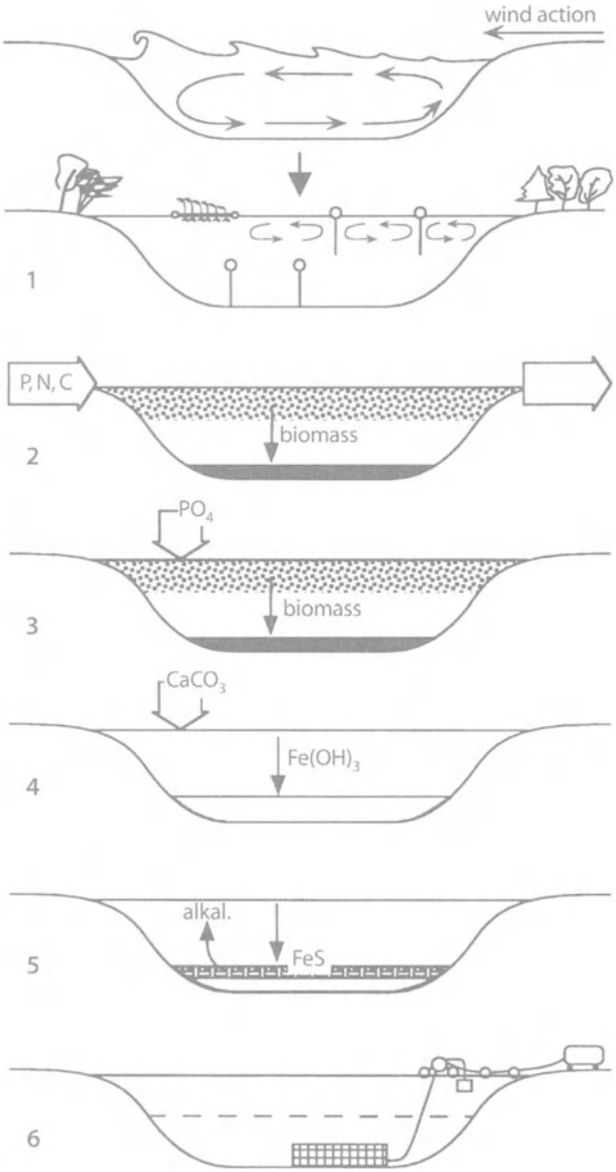
Figure 16.7 illustrates a number of in-lake approaches for neutralization. These ideas stem from the literature, our own field experience (controlling nitrate), and from laboratory experiments (Fichtner 1983; Benndorf 1994; Kepler and McCleary 1994). Many relatively small and shallow lakes have remained acidic for decades while others with stable stratification and organic loading have been neutralized within a few years by internal mechanisms. Anoxic hypolimnia, the monimolimnia of meromictic lakes, anoxic sediment layers, and the decaying plant matter of macrophyte stocks are habitats where the pH increases due to microbial activities. Ecotechnologies acting on the basis of microbial sulfate reduction require the absence of oxygen and nitrate and the availability of a carbon source for sulfate respiration. Both stratification and mixing in a lake depend on the length of free wind action and the wave fetch. Stratification is a desired condition for acid mitigation. To show how efficient stratified systems can be, Lake Fuchskuhle, a small shallow moor lake in eastern Germany, was divided into four limnocorrals with help of plastic sheeting. The corrals became stratified with anoxic deep water and sediments; as a result, the pH changed to nearly neutral while nutrients and bioproductivity increased (Babenzien 1996). Possible tools to induce such stratification in mining lakes include oil barriers, floating reeds, and submerged foils positioned from the bottom into the free water like a fishing net. The provision of wind protection by the shore is only effective where lake areas are small (Fig. 16.7, part 1).

We suggest that the main strategy for obtaining neutral mining lakes in the Lusatian region will be first to flood the pits rapidly with surface water. Although the necessary pipelines or ducts are expensive, this approach also provides surplus benefits for shore stability. Running waters contain neutralizing and buffering bicarbonate, nutrients, and dissolved and particulate organic matter, all of which constitute required ingredients for the abatement of acidification. In order to avoid excessive nutrient input, at the end of the flooding process the respective river should function only as a bypass and the minimum possible amount of surface water should be used (Fig. 16.7, part 2).



- Fast recultivation of the heaps by neutralization with lime and ashes. Afforestation with mixed forest
- organic or nitrate fertilization and intensive cropland and pasture
- abatement of pyrite oxidation by wet covers: wetlands and fish ponds
- Spreading of tensids as bactericides to inhibit the acidifying metabolism of *Thiobacillus* spp. on the heaps
- high alkalinity ponds to stop acid sources
- subdividing of long-shaped mining hollows for adaption to the ground water and surface water level

Fig. 16.6. Abatement of acidification by measures in the drainage basin (Klapper and Schultze 1997)



- Changing of polymictic mining lake into a stable stratified dimictic lake by swimming and submerged barriers, and by shore afforestation
- Controlled surface water filling and throughflow: bicarbonate, nutrients and organic load stimulate alkalinisation of the free water and sediments

or

- Fertilization by phosphorus for controlled eutrophication
- Liming for iron flocculation or utilisation of neutralized groundwater from a groundwater treatment plant
- Capping the sediments with degradable organic matter to support the sulfidic binding of heavy metals and acidity
- Installation of an upgrowth reactor and addition of a carbon-source for sulfate reduction in the anoxic hypolimnion

Fig. 16.7. Abatement of acidification using in-situ technologies (Klapper and Schultze 1997)

The addition of phosphorus to a rain-acidified soft water upland lake in England led to neutralization due to the above-described mechanisms (Fig. 16.7, part 3). A total of 5.9 m^3 of phosphate solution had the effect of 34 tons of calcium carbonate (Davison *et al.* 1995; George and Davison 1998). In addition, to restore the pelagic food web in acidified and limed lakes, gentle fertilization with treated sewage containing nutrients and organic carbon would enhance the success of the treatment. One good example of neutralization following the addition of sewage is the case of Laubusch Lake. During stagnation periods sulfur and iron are transported into the black sediment as iron sulfides, thereby fixing acidity. With pH values of 3–4 in the epilimnion and 6–7 in the hypolimnion, a good stock of fish has developed (Klapper and Schultze 1995).

Liming has been successfully applied in Canada, the USA, and Sweden to neutralize rain-acidified lakes. About 5000 surface waters have been neutralized since 1988 under a government program in Sweden (Olem 1991). These soft-water bodies are quite different from hard-water mining lakes. The latter are strongly buffered by iron at a far lower pH and the alkalinity demand for neutralization is about tenfold higher. In the Lusatian region some groundwater treatment plants exist for the neutralization of effluents. These may now be used to produce limed and cleaned water to fill newly emerging lakes. Of course, this ex-situ treatment is expensive (Fig. 16.7, part 4), which is why restoration research focuses on natural processes of alkalinization by microbiological metabolisms and their limnophysical preconditions.

The addition of organic matter may provide a way of enhancing the sulfide-forming properties of the sediments. Promising experiments are currently being performed with potatoes as the carbon source (Fyson and Nixdorf, pers. comm.), and with the addition of liquid organic substrates (UFZ Laboratories). Another proposal has been to dredge the bottom sediments from eutrophic lakes and spread it on acidic mining lakes. This method of restoration is designed to counteract eutrophication in the case of natural lakes and acidity in mining lakes (Nixdorf and Schöpke 1996; Fig. 16.7, part 5).

Another example of an effective ecotechnology under anoxic conditions has been developed for heterotrophic nitrate dissimilation in Zeulenroda Reservoir (Thuringia, Germany). A steel cage (20 · 60 · 1.50 m) filled with 13 000 bales of straw and equipped with distribution pipes was positioned on the hypolimnic bottom near the dam. Nitrate-rich surface water was pumped through the straw reactor together with fatty acids as a carbon source. The dissolved oxygen was quantitatively consumed, after which nitrate was utilized for respiration and nitrogen escaped as dinitrogen gas. This technique made the hypolimnion anoxic and nitrate-free within 8 weeks. The water was reaerated along a stretch of 3 km of turbulent flow between Zeulenroda and a terminal reservoir, and proved to be a suitable source of raw water for drinking water after treatment (Klapper 1991; Fig. 16.7, part 6).

16.2.3
Laboratory Experiments and Initial Results

With biological reduction processes such as iron and sulfate reduction, alkalinity is generated during the oxidation of organic matter:
Iron reduction:

$$FeOOH + CH_3CHOHCOO^- + H_2O \rightarrow Fe^{2+} + CH_3COO^- + CO_2 + 1{,}5\,H_2 + 2\,OH^-.$$

Sulfate reduction:

$$2\,CH_2O + SO_4^{2-}\ \text{fi}\ \ H_2S + 2\,HCO_3^{2-}.$$

Several studies have been performed to understand the role of these processes in nature and to use them for neutralization of acid mine waters in various types of bioreactors (for review see Wendt-Potthoff and Neu 1998). We followed a similar strategy with material from a mining lake in Lusatia with a pH value of 2.7 (RL 111 near Plessa). To support microbial reduction processes, a laboratory-scale approach was used with original water and sediment and the addition of soluble carbon sources and a degradable substratum (straw) to allow biofilm formation. Appropriate carbon sources were selected by monitoring pH, E_h, sulfur, and iron species, as well as iron- and sulfate-reducing bacteria and their activities. Successful carbon substrates are currently tested in laboratory columns with a larger volume which allow the temperature-controlled stratification of the water column to be simulated. Parallel to this, in-lake enclosures are being developed to test biological neutralization on a larger scale under field conditions.

The bottle experiments were performed with 2000 ml lake water, 255 g lake sediment, 15 g straw, and 30 mmol carbon source l^{-1} incubated for 28 days in the dark at 21 °C. The chemically pure carbon sources used as electron donors were sodium pyruvate, sodium succinate, D-glucose, sodium-DL-lactate, ethanol, and sodium acetate. Molasses, whey, sugar beet/molasses granula, and carbonation-lime from sugar production were also used as complex industrial carbon sources.

Sampling of bottles at the end of an experiment was conducted under anoxic conditions in a glove bag and E_h, O_2, pH, and conductivity were measured immediately. Water chemistry was investigated using standard analytical methods. Simplified carbon budgets were derived from TOC, DOC, and TIC measurements. Most Probable Number (MPN) determinations of iron-reducing bacteria were carried out in modified *Geobacter* medium (DSM) with ferric citrate under anoxic conditions. Sulfate reduction rates were determined after incubation with $^{35}S\text{-}SO_4^{2-}$. Whole cell hybridizations were performed following the protocol of Amann *et al.* (1990). Ferrous iron and microbially reducible ferric iron in bulk sediment were determined using the method of Lovley and Phillips (1987). Biofilm formation on straw was observed after nucleic acid staining of live cells with a confocal laser scanning microscope (Table 16.1).

The preliminary results may be summarized as follows:

- Sulfate elimination from the water column was most efficient with lactate and pyruvate.
- Pyruvate, lactate, acetate, and ethanol exhibited alkalinity generation, whereas in the case of acetate this was already present after dissolution of the substrate.
- Only a small fraction of the added organic carbon was respired during the 28-day experiment.
- Glucose and succinate bottles did not display any sulfate reduction.
- Iron-reducing bacteria multiplied during incubation, especially with lactate, ethanol, acetate, and complex industrial carbon sources. Their abundance was at least one order of magnitude higher in the sediment than in the water phase.

Table 16.1. Relative intensity of sulfate reduction and detection of sulfate-reducing bacteria with fluorescent in situ hybridization (experimental results of J. Meier, UFZ, Dept. of Inland Water Research)

	Intensity of reduction	Detection δ-Proteobacteria	SRB genera
Autoclaved	–	–	–
Straw + sediment only	+	+	+
Pyruvate	+++	+	n.d.
Glucose	–	–	n.d.
Succinate	–	–	n.d.
Lactate	+++	+	–
Ethanol	+	–	+
Acetate	++	–	+

n.d.; Not yet determined; *SRB*, sulfate-reducing bacteria.

- Sulfate-reducing bacteria (δ-proteobacteria- and genus-specific SRB probes tested) were present in very low numbers in the water phase, so only presence/absence could be recorded. They were probably much more abundant in the sediment (see iron-reducing bacteria), but we were unable to perform in situ hybridization with sediment samples.
- Biofilm from lactate-amended bottles had higher biomass compared to bottles with straw alone.
- Complex substrates containing the same monomers are currently being tested and provide good results in sulfate reduction.
- Fe (III)-reduction also plays a role in alkalinity generation since iron was completely turned into the Fe (II)-form even in bottles with no organic carbon addition or sulfate reduction.

16.2.4
Conclusions Regarding Management of Acid Lakes

Acidification is the most severe water quality problem in lakes originating from lignite mining activities in Germany. Only a few smaller bodies of water should be left acidic for conservation purposes and limnological research: most should be treated to form more natural and less acidic environments. The abatement of acidification includes measures to combat pyrite oxidation, steps to decrease groundwater and acidity transport, and in situ neutralization measures in the mining lakes.

Chemical neutralization is often impracticable because of the large amounts of alkali required and the costs of treatment. Large mining basins should primarily be flooded with surface water containing bicarbonate. A temporarily higher trophic level must be tolerated. The most promising alternative for chemical neutralization is the encouragement of microbial processes such as Fe (III)-reduction and dissimilatory sulfate reduction.

Field and laboratory experiments have shown that under reduced conditions, alkalinity is generated and the content of sulfur and heavy metals in the water phase decreases. Phased scale-up from small laboratory experiments to larger stratified reaction columns and in-lake enclosures will show whether this strategy is applicable for the restoration of entire mining lakes. Aerobic treatment with macrophyte systems such as in artificially constructed wetlands is suitable for completing water treatment via the flocculation of the heavy metals contained as hydroxides. These strategies can in principle be applied to the restoration of nearly all mining lakes with pyrite-containing rock such as hard coal, quartz, and slate, as well as sulfidic ores (especially uranium).

16.3
A Recreational Lake near a Mixed Waste Deposit (Lake Hufeisensee)

16.3.1
Description of the Problem

16.3.1.1
Situation

In addition to the existence of recreational lakes created in opencast mines, other pits were used to dump domestic or industrial waste (Glässer 1995; Schreck 1998). The problems we face nowadays stem from our inadequate knowledge of the interactive transfer processes between disposal sites and adjacent lakes, which may impair the quality of groundwater and surface water. One such typical waste disposal site in a former lignite mining area in central Germany is the old lignite mine at North Bruckdorf 5 km to the east of Halle/Saale, which was once the scene of both open pit and underground mining. The subhydric landfill at Kanena covers an area of about 25 ha and is bounded by the horseshoe-shaped Lake Hufeisensee, which itself has an area of about 70 ha. Groundwater flooding began in 1968 and was completed more than 15 years ago. Today, Lake Hufeisensee is 29 m deep (Fig. 16.8) and is a popular place for swimming and waterskiing (Glässer *et al.* 1994).

Lignite mining was carried out at North Bruckdorf between 1919 and 1941, and was followed by gravel extraction until 1964. The environmental problems are caused by an old waste deposit located on lignite mining overburden in the open cut formerly exploited. The waste deposit comprises about 3.3 million m^3 of domestic refuse, building material, hazardous waste from the chemical and metallurgical industries, waste from the munitions industry, and power plant ash (Dermietzel *et al.* 1995). This landfill was operated until 1984 and does not contain a basal liner system. A dam was erected between the deposit and the lake to prevent the seepage of leachates from the waste deposit into the lake. While waste material was still being deposited, the basal part of the dump was flooded by rising groundwater and the hydrochemical conditions in the landfill changed from aerobic to anaerobic. In 1984 the landfill was covered with a thin layer of soil and building refuse.

In 1990 an initial assessment of the potential environmental hazard was carried out (Glässer 1990, internal report), after which the surface and groundwaters were monitored. Over the past 2 years, a surface cover and drainage system for the landfill has been installed to reduce the formation of groundwater in the landfill.

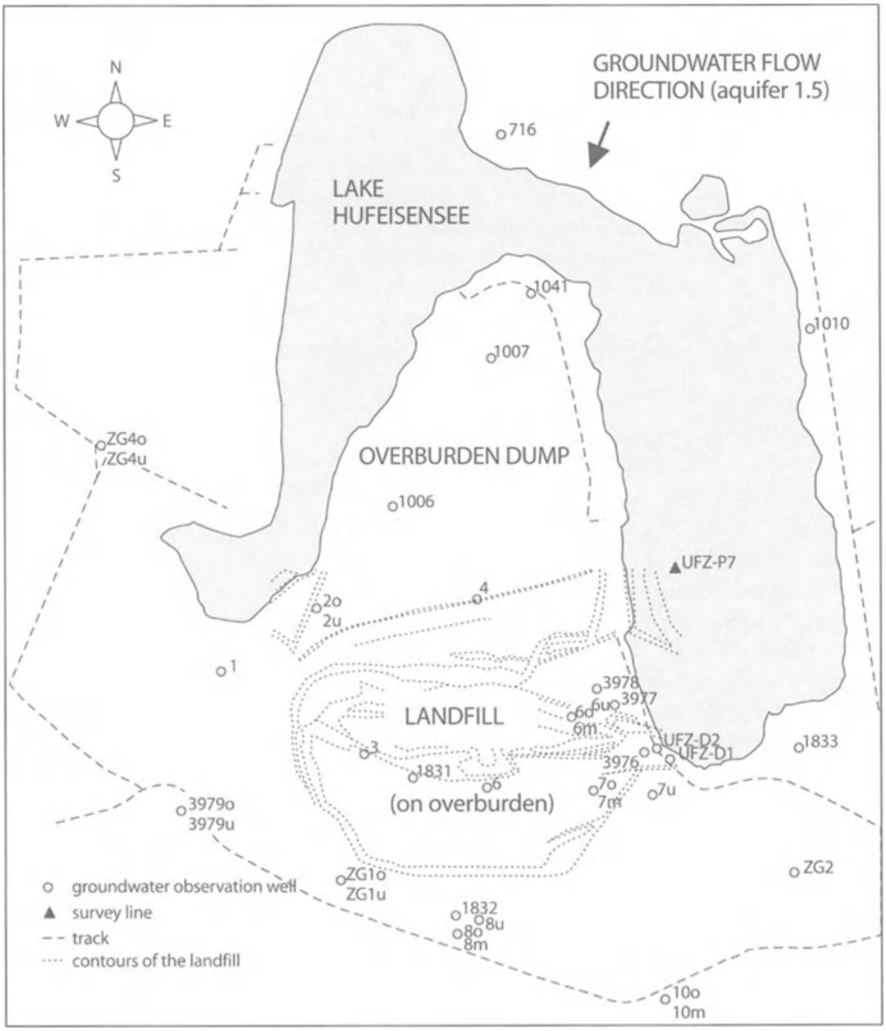


Fig. 16.8. Map of the investigation site: Kanena landfill situated on lignite mining overburden, Lake Hufeisensee, inner part of overburden dump, and aquifer GWL 1.5

16.3.1.2
Geology

The geological setting of the area is shown in a simplified diagram (Fig. 16.9):

- Above the Tertiary sediments, 10–15 m of Quaternary sediments occur. They consist of layers of till and glacial meltwater sand, sandy loam, and river gravel, and include the Saale main terrace aquifer (GWL) 15.
- The lignite layer (Bruckdorf seam) is interbedded in a sequence of sand and clay.
- The subjacent beds are composed of clay and pre-Tertiary altered sediments of Triassic age (Buntsandstein).
- The overburden from lignite mining consists of Quaternary and Tertiary sediments containing minor amounts of coal particles and pyrite (mean transmissivity, kf, $= 5 \cdot 10^{-5}$ to 10^{-6} m s^{-1}).

The hydrogeological situation is characterized by the upper aquifer Saale main terrace, GWL 15, and the Tertiary aquifer. The direction of groundwater flow (GWL 15) is from N to S. The groundwater level in the landfill is higher (up to 93 m NN) than that of the lake (91 m NN) and the surrounding groundwater (92 m NN). Hydraulic links are thought to exist between the aquifers and numerous wells are situated around the landfill and its outer parts to monitor groundwater quality at various depths.

Open pit lignite mining and subsurface mining go hand-in-hand with enormous changes in the local and regional hydrogeology, with former aquifers being disrupted or even eradicated. The flooding of the open pits results in the formation of renewed groundwater levels similar to those prior to mining (Glässer 1995). Consequently, flooding must be considered as a compromise between the current hydrogeological situation and the viable technical and financial options for ensuring the future quality of the groundwater and its usage. The danger often emanates from the uncontrolled disposal of waste material at unsuitable places and from a lack of monitoring (Schreck 1998).

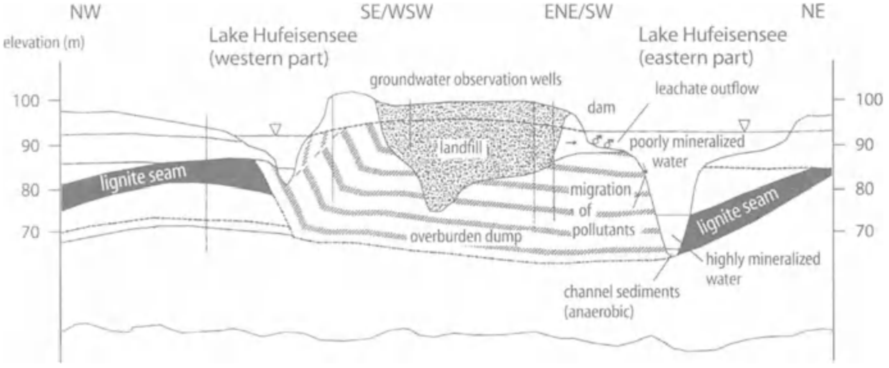


Fig. 16.9. Schematic section of the mining area with compartments Lake Hufeisensee, landfill, overburden, aquifers, and lignite seam

Fundamental research into this typical situation in a mining landscape, namely a lake located next to a mixed waste deposit (Fig. 16.8), comprises investigations of the following aspects:

- The effect of pollutants and their metabolids on the groundwater
- Problems of fixation and the mobilization of noxious substances
- Alteration processes in waste disposal sites occurring in disused open pits flooded by groundwater
- Process identification for ongoing reactions in water bodies by means of isotopes and chemical tracers
- Evaluating lake development in the recultivation process of former mining areas.

16.3.2
Remediation Strategy

The most hazardous problems at Lake Hufeisensee are the input of leachate from the waste dump into the lake at the south-eastern dam and the contamination of the Quaternary and Tertiary aquifers. The first remediation step in this area was to cap the landfill and to install an effective surface drainage system to minimize groundwater recharge. Other remediation measures aim at reducing the input of contaminants into the groundwater and the lake, and monitoring the quality of the groundwater downstream, and of the lake water. An integrated program of hydrological, geochemical, and geophysical investigations was implemented to evaluate the environmental situation in the mining area and to quantify mass transfer processes and metabolisms in and between the various compartments involved, i.e. the groundwater, Lake Hufeisensee, and the waste disposal site.

The investigations include monitoring of organic and inorganic geochemistry, isotope measurements, and the application of microbiological and geophysical methods in wells and surface water. Several biocenotic investigations have been carried out to evaluate the possible environmental impacts of waste deposition and a highly mineralized lake water on the flora and fauna in the surrounding area.

16.3.3
Experimental Results and Discussion

16.3.3.1
Flow Paths Around the Landfill

The most serious threat to the environment caused by a landfill which is located within the regional groundwater system comes from seepage. At Lake Hufeisensee the most effective leakage paths of seepage into the surrounding water system are via the dam site to the lake and diffuse input into the groundwater main flow from N to S. The sealing function of the dam as a barrier for contaminated water has been investigated by geophysical measurements (Dermietzel *et al.* 1995; Gläser 1995).

16.3.3.2
Geophysical Investigations and Field Measurements

To determine the effective flow paths of leachate from the landfill to the lake and the groundwater, an area of 450 · 200 m was surveyed by direct geoelectrical and seismic measurement. The low apparent resistance strongly suggests a seepage soaking the lower parts of the dam below 18 m over a width of 110 m. The main flow of highly mineralized water is located in the SW area of the dam. One likely trouble spot for water penetration is the area around well 70/7 m. Additional direct current tomography measurements have identified minor flow paths in the dam region down to about 11 m.

Isotope tracer measurements yielded a filter velocity of about 11 cm day^{-1} in the most endangered part of the dam. Samples collected in this area (observation well UFZ-D2) showed a highly mineralized seepage (>50 g l^{-1}) indicating an anoxic milieu (redox value <-400 m V) in the landfill. Following the morphology of the former open pit, seepage from the landfill is thought to flow mainly to the S and only partly to the E (i.e. to the lake). This assumption is supported by characteristic cation/anion contents in water samples from the wells 3976, 60, and 70.

16.3.3.3
Three-dimensional Flow Modeling in the Dam Region

One of the important problems when evaluating the impact of Kanena landfill on Lake Hufeisensee is the speed of the groundwater inflow to the lake via the dam range. Geoelectrical measurements indicate the existence of flow paths through the dam. Flow and transport simulation was calculated using the FEFLOW hydrodynamic program (Diersch 1993). The conditions of the stationary flow and transport modeling are a volume of 200 · 200 · 25 m with a saline solution of 30 g l^{-1} and a transmissivity k_f of about 10^{-6} m s^{-1}. Working on the basis of the morphologic profile, the calculation simulates the transport of the saline solution in the upper stream directed to the lake. The model calculations correspond well to the tracer measurements showing that after 15 years the bulk solution only moved about 100 m.

16.3.3.4
Groundwater Types and Characteristics

Water of different origin and development occurring in the mining area can be distinguished by its characteristic hydrochemical and isotopic signature (Table 16.2). Tritium measurements and stable isotope distribution (^2H and ^{18}O) indicated that groundwater flow within the landfill seems to be stagnant with only minor exchange with the groundwater environment (Table 16.3). The results indicate that the residence times of the body of water are more than 20 years: the groundwater in the landfill accumulated even before its closure and the groundwater renewal rate was calculated to be 85–90% of the precipitation input before covering.

Above-average PAH concentrations, attributed to coal particles, occur within the landfill (well 3 = 8 µg l^{-1}) and in the deep layers of the dumped overburden (well 7u = 15 µg l^{-1} at 34 m; Holland B = 10 µg l^{-1}). Slightly increased PAH concentrations have been found in the Tertiary aquifer at the southern groundwater flow (well 10u), indi-

Table 16.2. Groundwater classification: major ions, pH values, and electrical conductivity (Strauch 1994)

Compartment	pH	Electrical conductivity (μS cm^{-1})	Hydrochemical signature
Groundwater inflow: Quaternary aquifer 15	6.7–7.1	3100–7100	Calcium–magnesium–sulfate
Groundwater downstream: Tertiary aquifer (Eocene)	6.6–7.0	2200–4300	Sodium–chloride dominant
Southern border of landfill	6.4–7.2	9600–22 900	Sodium–sulfate–chloride
Landfill	6.2–8.9	24 000–65 000	Sodium–sulfate dominant
Hufeisensee down to 24 m	7.6–7.7	1600–2900	Calcium–magnesium–sulfate dominant
Hufeisensee below 24 m	6.8	17 300–28 900	Sodium–chloride dominant

Table 16.3. Compilation of hydrochemical parameters in the various compartments

Parameters	Groundwater (Quaternary aquifer)	Groundwater (Tertiary aquifer)	Groundwater (below overburden dump)	Groundwater (landfill)	Lake water (epilimnion)	Lake water (deepest part, monimolimnion)	Lake sediments (deepest part, channel)[a]
Conductivity (mS cm^{-1})	2.2 – 3.1	4.2	15 – 32	20 – 73	2.6	27–28	–
pH	6.8 – 6.9	6.7	6.6 – 6.9	6.4 – 8.4	7.5 – 7.7	6.8	–
Cations (g l^{-1})							
Ca	0.2 – 0.7	<0.2	0.3 – 0.8	0.2 – 0.8	0.2 – 0.4	0.8	–
Na	–	0.2 – 0.7	3.0 – 5.0	<70	–	6.0	–
Mg	<0.2	–	–	–	0.1	–	–
Anions (g l^{-1})							
SO_4	0.5 – 2.0	1.0 – 2.0	0.7 – 2.0	3.0 – 30	1.2	0.3	–
Cl	<0.3	0.2	4.0 – 10	2.0 – 13	0.3	8.0	–
PAH (μg l^{-1})	<0.1	<0.1	0.3 – 2.0	0.7 – 11	<0.1	0.8	16 – 48
isotopes:							
δ^{2}H (‰ VS MDW)	−61 – −69	>−40	−50 – −62	−47 – −59	−31 – −61	−56 – −62	–
δ^{18}O (‰ VS MOW)	−8.3 – −8.8	–	−6.2 – −8.2	−7.0 – −9.1	−3.3 – −8.6	−7.8 – −8.4	–
Tritium	28 – 35	1 – 3	19 – 48	3 – 1200	10 – 20	±4	–
Trace elements (mg l^{-1})							
Zn	14 – 110	2 – 16	2 – 74	2 – 57	27	14	71 – 233
Cu	1 – 26	16 – 25	2 – 8	1 – 45	23	2	23 – 80
Pb	30 – 102	2 – 35	2 – 83	7 – 146	11	14	23 – 74
Ni	10 – 60	1 – 18	2 – 19	1 – 80	16	5	n.d.
Cr	0.3 – 27	0.3	1 – 9	12 – 85	1.4	3	n.d.
Cd	1 – 20	0.3	0.3 – 9	1 – 24	1	4	n.d.

[a] Trace elements in sediment expressed as mg kg^{-1};
PAH, polycyclic aromatic hydro carbons;
Data from Strauch, Trettin, Haendel, Kowski, Dermietzel, Glässer (UFZ Leipzig-Halle, Department of Hydrogeology).

Table 16.4. Average heavy metal content in different wells (µg l^{-1})

Compartment	Well	Ni	Cr	Cd	Cu	Zn	Pb	As	Al	Fe	Mn
Upstream	3979o	55	17	20	5	88	102	7	283	70	50
	3979u	34	12	10	1.5	86	29	6	220	100	490
	ZG 4u	16	0.3	0.3	16	19	28	14	800	1740	280
	ZG 4o	11	0.3	0.3	26	14	43	12	60	320	200
Landfill site	6o	37	32	5	7	57	83	457	3050	9560	2100
	6u	60	12	18	11	28	141	37	315	8280	8930
	7o	37	57	5	11	31	68	160	350	470	90
	3977	79	52	24	24	50	146	581	495	86x10^{3}	2720
	3976	nd	nd	1	45	24	7	760	nd	1100	150
	7u	97	21	15	34	238	91	32	530	37x10^{3}	5140
Downflow	8o	13	2	4	14	20	14	0.9	nd	8150	530
	8u	29	22	14	14	110	35	34	3190	17x10^{3}	510
	ZG 1o	11	1.3	0.3	21	8	13	13	3	8740	1240
	ZG 1u	18	0.3	0.3	16	16	35	0.9	200	1080	90
	10o	60	4	1	1	110	49	nd	550	700	1440
	10u	10	27	2	3	42	30	nd	7900	1900	56x10^{3}
	ZG 2	11	45	30	14	30	7	3	80	640	3320
Lake Hufeisensee	Lake <24m	16	1.4	1	23	27	11	0.9	4	330	340
	Lake >24m	5	3	4	2	14	14	50	4	660	3860
	Holland A	20	20	1	20	50	20				
	Holland B	50	50	205	50	200	50	30			
	Holland C	200	200	10	200	800	200	100	200^{a}	200 · 10^{3a}	50 · 10^{3a}

[a] Limiting values for potable water; *nd*, Not determined.

cating a geogene PAH origin. Although the landfill is strongly anaerobic (–350 m V in well 3), increased sulfate and chlorine levels (30 g l^{-1} and 13 g l^{-1}, respectively) of the landfill groundwater are due to sulfurous ash, chemicals, and building waste material. The pH values vary from 6.4 to 8.9 and microbial activities have increased. Sulfate precipitation (gypsum) in the dam region suggests flow paths from the landfill to the lake, which have also been detected by geophysical methods.

As expected, the highest concentrations of heavy metals were found in the landfill groundwater and in some wells in the surroundings of the disposal site (Table 16.4). The concentrations are in the range of the Holland B values for Ni and Pb and above the Holland C values for Cd. These metal concentrations in the range of Holland B require high attention and groundwater monitoring.

16.3.3.5
Lake water

The upper part (<10 m) of Lake Hufeisensee comprises 87% of the lake volume. During the summer/autumn stagnation (thermocline 10–12 m), the lake perceptibly evaporates, clearly reflected by more positive $\delta^{2}H$ and $\delta^{18}O$ values (Table 16.3). In the winter/spring season, the lake is well mixed down to 22 m. However, the deepest zone (volume <0.2%) is stable under anaerobic conditions, as indicated by a DOC content of 45 mg l^{-1} and an H_2S concentration of up to 300 mg l^{-1} and CO_2 and methane degassing (Asmussen and Strauch 1998). This monimolimnion is fed by ascent sodium chloride dominated saline water from the deeper Buntsanstein aquifer. The lake is mainly fed by groundwater (GWL 15) and only secondarily by precipitation (about 500 mm $year^{-1}$). Seepage from the landfill can be neglected for the water balance (Table 16.3).

Significantly increased metal levels were only observed in the deep water zone below 24 m. The organic PAH pollution in the lake water does not exceed the geogene background. *Dreissena* mussels were used as indicators for organic pollutants in surface waters, showing PCB enrichment up to a factor of 4000. Heavy metal contents in lake sediments vary locally. The highest concentrations of Pb, Zn, and As were found in the northern bay sediments (0–50 cm) where industrial sewage water was poured into the lake for many years. Measured metal concentrations of 90 mg Pb kg^{-1}, 300 mg Zn kg^{-1}, and 30 mg Ni kg^{-1} do not exceed the Holland B level. The sediments from the endangered dam zone (eastern part of the lake) and the deep zone of the lake are, however, more polluted under anaerobic conditions. One very important fact is that because of the strong reducing conditions in the sediments and in the deep zone of the lake, these represent a "natural geochemical sink" and prevent the migration of toxic compounds and elements due to their fixation as sulfides.

16.3.3.6
Dynamic Development of Water Quality

Christoph (1995) drew up a mass balance for the lake and concluded that the relative annual increase in dissolved components in lake water is 6.2% and that the final composition of the water, 1.92 times the current values, will be reached after 12–36 years. The model calculations yielded a yearly water exchange of the upper epi- to mesolimnion of about 35% and an input : output ratio of the groundwater of about 2.5 : 3.

On the other hand, isotopic data showed that the anaerobic deep water zone in the lake is renewed much faster because of seasonal mixing processes and the erosion of the interface with the deep water zone. Using the isotope data and the isotope mixing relation $M_{gw} = (\text{đ}_{\text{deep zone}} - \text{đ}_{\text{lake}})/(\text{đ}_{gw} - \text{đ}_{\text{lake}})$, the deep water zone can be calculated to consist of 79% of lake water from the Quaternary aquifer and 21% of deep groundwater. This implies a renewal time of about 5 years. Therefore, the anaerobic zone is a mixture of lake water with Tertiary groundwater and deep saline groundwater rising from the Buntsandstein aquifer (Tables 16.3 and 16.4).

16.3.3.7
Microbial Processes and Degassing

In the chemical reactive zone of the landfill (around wells 3 and 5), gas exhalations occur which contain varying proportions of methane, CO_2, and nitrogen (Nitzsche *et al.* 1997). The highest temperature measured here is 28°C. Such exothermic effects are caused by intensive degradation processes, like sulphate reduction (Asmussen and Strauch 1998) continuing even after the closure of the landfill in 1984. In order to determine the microbial activities in the landfill and in the surroundings, sulphate-reducing anaerobiants, nitrificants, Fe-Mn-specific microbiota, and *E. coli* bacteria were estimated. The amount of bacteria colonies varies, depending on the environment. Investigations focused on the following compartments: groundwater inflow, landfill, landfill south border, and groundwater downstream (Krauss and Birger 1996).

16.3.4
Conclusions for Site Management and Fate

From the results of our monitoring we conclude that:

- The hydrodynamic system groundwater–open pit-lake–landfill–overburden is connected by the Quaternary–Tertiary aquifers. Each system has developed independently and there is only a minor exchange of compounds.
- Within the landfill, only a small lateral flow gradient is assumed. Therefore, the release of pollutants seems to be a long-lasting process which especially occurs along the groundwater flow path. The vertical migration of pollutants and the contamination of the pre-Tertiary groundwater has not yet been detected.
- Saline water and seepage containing heavy metals from the landfill are diluted and their load fixed in the upper sediments in an anaerobic environment.
- Microbial degradation processes in the landfill and in the deep zone of the lake suggest self-cleaning mechanisms in the groundwater system landfill–lake.
- The water balance of Lake Hufeisensee is characterized by high evaporation and relatively quick short-term water exchange. Modeling of the long-term (element) saturation equilibrium yields a time frame of about 30 years.

We may conclude from these findings that despite the existing landfill being in close contact with the lake, the water quality of the upper layer of the lake will not markedly change over time.

16.4
Pyrolysis Waste Water Deposit in a Former Lignite Opencast Mine (Lake Schwelvollert)

16.4.1
Description of the Problem

16.4.1.1
History of Site and Geological Background

Lake Schwelvollert is a lignite pyrolysis wastewater deposit situated in an abandoned opencast lignite mine, located near the villages of Trebnitz and Deuben in Saxony-Anhalt, Germany, about 60 km southwest of Leipzig (Fig. 16.1). A number of other surface mines also exist in the vicinity of Deuben and Trebnitz. Currently Lake Schwelvollert is about 530 m long, 360 m, wide and 27 m deep; it has an approximate volume of 2 million m^3, and the area of the deposit is about 10 ha (Wießner *et al.* 1993; Stottmeister *et al.* 1996).

Mining at Schwelvollert began in 1883 and continued until 1944. The abandoned mine was then used from 1952 until 1969 as a deposit for condensation waters from lignite pyrolysis from the nearby Deuben carbonization plant. The waste water was stored in the pit before being injected into the local Zechstone formation (dolomite) at a depth of around 100 m as an "alternative" to surface disposal. However, these injected waters are not expected to have any bearing on the analysis of groundwater contamination from the lake. More relevant local sources of pollution include four small waste deposits near the village houses which have now been removed, sewage from this same village, which was formerly discharged into the lake, and the likely usage of the lake as an illegal waste dump. The use of fertilizer by local agriculture is also thought to be a source of nitrate contamination in the surrounding aquifers.

16.4.1.2
Chemical Characterization

The vertical profile of total phenols, chemical oxygen demand, dissolved organic carbon, and ammonium-N from Lake Schwelvollert is shown in Table 16.5. Additionally, more than 90 different organic compounds were detected by GC-MS. The water was black with a visibility of 3 cm, completely anoxic and toxic (with bacterial luminescence test-Microtox: EC50 value (30 min) of surface water = 1.3–4.0%; water from a depth of 20 m = 0.2–0.4%). The highly toxic deposit water is an ecological hazard. Moreover, the smell of the deposit (organosulfur compounds) was a constant nuisance for the local residents. The remediation of the deposit was also necessary owing to the geological situation and the expected long-term groundwater contamination.

16.4.1.3
Geological Situation

Groundwater seepage into and out of the pyrolysis waste water lake was investigated by using an experimental deep water seepage meter. As a result the situation can be described as follows: the lake water is surrounded, like a bowl, by the coal layer (comprising several seams separated by silt and sand layers) and spoil material. However, the current lake water level transcends the coal layer from 0 m at the northern side up to 7 m at the southern side (Fig. 16.10). The Quaternary aquifer 18 situated directly above the coal layer could, therefore, be connected to the lake at shallow water depths where landsliding has not occurred. Fluid potentials in this aquifer indicate the possibility of inflow from the southern and western sides, with outflow to the east. However, phenol and ammonium concentrations at wells on the eastern side are not consistent with a significant outflow of lake water. Furthermore, contamination at the east side (well P11 248

Table 16.5. Chemical analysis (mg l^{-1}) of Lake Schwelvollert. (Wießner *et al.* 1993)

Depth (m)	Phenols	NH_4^+-N	COD	DOC
0	7	79	954	263
5	23	85	980	274
10	86	134	350	405
15	202	221	2220	627
20	228	239	2290	689
24	219	250	2245	690

COD, chemical oxygen demand; *DOC*, dissolved organic carbon.

screened above lake level, not shown in Fig. 16.10) indicates the existence of a further source. The possibility of an aquifer 18 outflow from the lake in the north-east corner requires further investigation.

Mean ammonium (14 mg l^{-1}) and phenol (1 mg l^{-1}) concentrations at the well UFZ2/96 in the Tertiary aquifer 52 indicate that the aquifer is contaminated by lake water. The aquifer 52 lies beneath the silty "Luckenauer" clay layer, which lies beneath the coal. An approximate hydraulic analysis shows that the estimated dilution of seeping lake water into the aquifer water is consistent with the ammonium concentration at UFZ2/96. However, there is considerable uncertainty in the analysis primarily due to the unknown hydraulic conductivity of the Luckenauer clay; this requires further measurement.

16.4.1.4

Fundamental Research Necessary for the Development of a Remediation Strategy

Standard methods of waste treatment and experiences of natural lake remediation were not applicable for Lake Schwelvollert deposit for the following reasons:

- The stored water is much older than the newly produced condensation water from former carbonization plants (Ringpfeil *et al.* 1988; Martius and Stottmeister 1993).
- The microbiological degradation processes are limited by toxic effects and are nutrient-deficient (phosphate) (Wießner *et al.* 1994).
- The season-dependent water circulation typical of natural lakes is inhibited by the black color of the water resulting in a nontypical temperature behavior.
- The actual oxygen concentration in the water is near zero even at the surface.
- Standard photometric analytical methods cannot be used owing to the black color and the negative redox potential of the water. Standard methods of gas chromatography exhibit inexplicable deviations or effects.

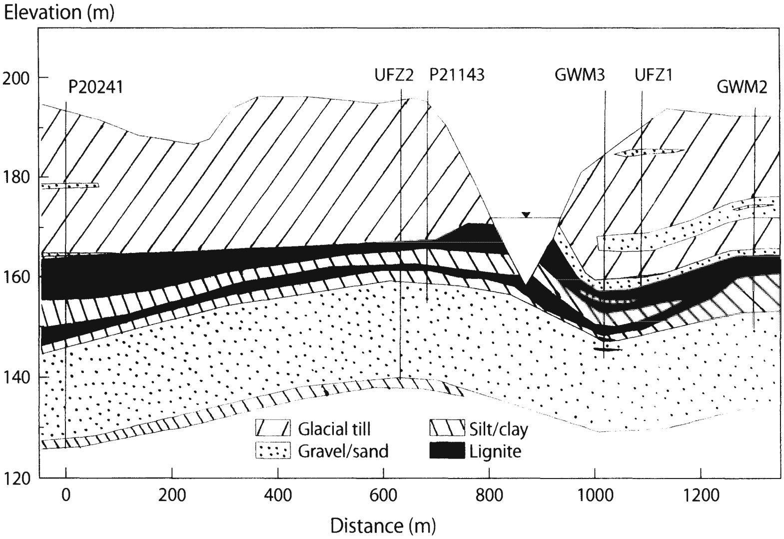


Fig. 16.10. Geology of the Schwelvollert site

For this reason, the key point of interest for research appeared to be the black color-bearing organic compounds: anthropogenic humic substances (AHS) from lignite pyrolysis, a group of xenobiotics with ecological relevance. Freshly produced carbonization waters were colorless, but a rising formation of black color was observed due to abiotic processes after air contact (Ringpfeil *et al.* 1988; Martius and Stottmeister 1993). The high content of phenolic compounds in these waters was responsible for the formation of colored polymeric compounds. Autoxidizable polyphenols can be used to model the formation of humic substances (Ziechmann 1980). The phenolic polymers in the deposit water showed the reactions similar to humic substances. Standard methods for refractory organic acids allow differentiation into fulvic and humic acid fractions.

16.4.1.5
Toxicity, Sorption, and Masking Effects

The polymeric black-colored substances are not microbiologically degradable, and some fractions are toxic (Martius *et al.* 1996). The AHS are able to "mask" other organic molecules like natural humic matter. Pörschmann and Stottmeister (1993) showed the reversibility of this effect by pH decrease. A model equation was developed to estimate the sorption–desorption of organics on humic matter using the K_{OW}-values (Kopinke *et al.* 1995a, b)

16.4.1.6
Development of New Analytical Methods

The determination of organic compounds was performed with respect to undisturbed balances in dissolved humic matter by using solid phase microextraction (SPME) (Pörschmann *et al.* 1996, 1997). However, for the fast field determination of phenolics in dark solutions a biosensor was developed based on the tyrosinase reaction with mediator-supported electrochemical detection (Gründig *et al.* 1992; Kotte *et al.* 1993). Biosensors for nitrate and nitrite are currently being developed (Gründig *et al.* 1993; Strehlitz *et al.* 1994, 1996).

As a result of the basic research performed on the physicochemical behavior of AHS, conclusions can be drawn about its impact on the sorption–desorption of low-molecular organic compounds and thus on masking with respect to analytical determination and bioavailability. Newly developed analytical methods allow the pollutants of interest (including in this medium, which disrupts the standard methods) to be precisely determined.

16.4.2
Remediation Strategy

The important role of the dissolved phenolic polymers in the interaction of biotic and abiotic reactions can be summarized as follows:

- Autoxidation of the phenolic components in the deposit is responsible for the lack of oxygen in the water.

- Autoxidation products are black-colored and responsible for the daylight absorption by the waterbody and its atypical temperature behavior in comparison to natural lakes.
- Autoxidation products react like humic substances, therefore, we call them anthropogenic humic substances (AHS).
- AHS formation is a dynamic process in the deposit and they are able to mask small organic compounds.
- AHS could be bound by sorption to autochtonic microorganisms without degradation.
- AHS could inhibit the metabolic activity of autochtonic microorganisms.
- Just like natural humic substances, AHS from Lake Schwelvollert react with metal ions (especially with Fe^{3+} and Al^{3+}) under an acidic pH (approx. 4–5) to form metal salts (fulvates, humates) with low solubility.

These specific aspects must be taken into consideration when developing a remediation strategy for the deposit.

Removing the AHS from the water in the dump is the key to every approach. Two methods are conceivable in principle: (1) ex situ treatment on the bank of the lake with flocculation in plants as already used for sewage treatment, and (2) in situ treatment of the entire body of water. A pump and treat technology was not a viable option owing to the volume (2 million m^3) of the water deposit, the enormous costs calculated for a multistage water treatment plant including nitrification/denitrification stages, and the special nature of the deposit (being neither waste water nor a natural lake). The treated water would have to be recycled into the lake, causing long-term dilution after external cleaning. The diversion of the treated water is not realizable either, for the sloping sides of the lake shore would be threatened and the seepage of uncontaminated groundwater into the deposit could not be avoided. Therefore, in situ treatment was the only feasible alternative and the first task was to match the remediation target to what was technically possible and required by legal considerations.

The objective of remediation was defined as a harmless ecosystem with no negative impact on ecological compartments and no future groundwater contamination. The proposed remediation process was a combination of cheap, simple technologies and the initiation of natural self-cleaning processes. Stottmeister *et al.* (1996) proposed the following strategy derived from our fundamental research and supported by laboratory experiments (Table 16.6):

- The AHS are flocculated by Fe(III) salts at pH 4.5. The resulting colorless water with the decreased inhibition of biological processes forms the basis of the start of the natural water circulation and rising oxygen concentrations.
- The natural microbial biocenoses must be reactivated. Limiting factors impeding the degradation of organic compounds (e.g. lack of nutrients) must be eliminated.
- Microbial nitrification starts after the exhaustion of the biodegradable DOC.
- The flocculated AHS-iron complex remains in the deposit and Fe(II) sulfide is formed (H_2S concentration 150 mg l^{-1}) deep in the lake, binding the hydrogen sulphide.

The later option of cleaning the sludge by exhaustion can be applied in the case of unexpected effects such as resolution or resuspension.

Table 16.6. Pollutants removal and decolorization of the deposit water by flocculation with iron(III) chloride

Parameter	Decrease (%) Laboratory experiments	 Enclosure experiments
Aromatic compounds[a]	78	75
Color[b]	95	93
Turbidity[c]	98	95
COD	55	47
TOC	60	55
Ammonium nitrogen	6	7
Water steam-volatile phenols	0	0
Dissolved sulfide and sulfur hydrogen	100	100

[a] optical density at 270 nm.
[b] optical density at 440 nm.
[c] optical density at 650 nm.
COD, chemical oxygen demand; *TOC*, total organic carbon.

16.4.3
Experimental Results and Discussion

Enclosure systems (four plastic tubes fixed on a floating platform, measuring 1 x 10 m, filled with water depending on the concentration gradient of the deposit) were installed in the deposit after positive runs under laboratory conditions to check the steps flocculation and activation of the microbiology. The results of these enclosure experiments support the basic laboratory results (Table 16.7). However, the natural oxygen input by waves and wind was not detectable for modeling the aerobic microbiological processes. Therefore, a larger enclosure system (volume 20 000 m^3, water surface 855 m^2, depth 26 m, diameter 33 m) was constructed and installed (Fig 16.11 and 16.12).

The plastic tube (2 mm HD-PE foil) was fitted on two ring steels. The enclosure was filled with a mixed deposit water (corresponding to original water from a depth of about 10 m) (Table 16.7). The flocculation of the AHS was carried out with 45 tons of 40% $FeCl_3$ solution with a final pH of 4.5. After 80 days it was neutralized with 41 tons of 20% lime suspension to pH 7.0. Flocculating the dump water with $FeCl_3$ removed about 50% of the DOC. This results in a clear water with a visibility depth of up to 2.5 m max. Figure 16.13 shows the changes of concentration in DOC and the development of the visibility depth over a period of 550 days.

The drop in organic pollution is mainly due to the flocculation of the phenolic autoxidation and polymerization products. Although the phenolic monomers are not removed, they are now able to undergo autopurification by means of microorganisms. The reduction of the oxygen depletion potential and the hydrogen sulfide binding results in an increase of the oxygen level. After the flocculation process, up to 75% air saturation was measured in the surface water, while air saturation at a depth of 6 m reached was as high as 4%. The decolorization and the rise in dissolved oxygen boost

Table 16.7. Concentration of different pollutants in the mixed water of the enclosure (according to depth of 10 m of deposit)

Depth (m)	pH	Conductivity ($\mu S\ cm^{-1}$)	Sulfide ($mg\ l^{-1}$)	NH_4-N ($mg\ l^{-1}$)	Water vapor volatile phenols ($mg\ l^{-1}$)	COD ($mg\ l^{-1}$)	TOC/DOC ($mg\ l^{-1}$)
0	7.80	1.52	19.0	94.5	21.1	993	328/271
10	7.64	1.55	34.3	99.2	21.5	1009	322/271
20	7.66	1.59	35.4	105.7	22.3	1093	323/281

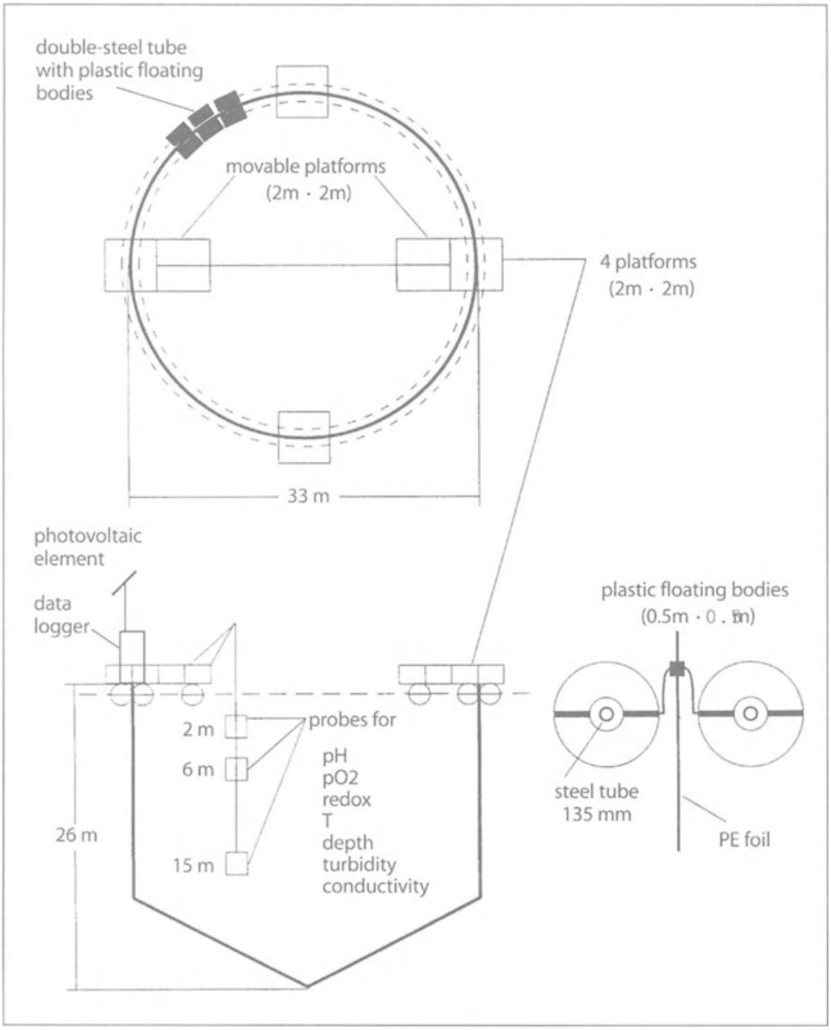


Fig. 16.11. Enclosure used for flocculation/intrinsic bioremediation experiments

Fig. 16.12. View of enclosure

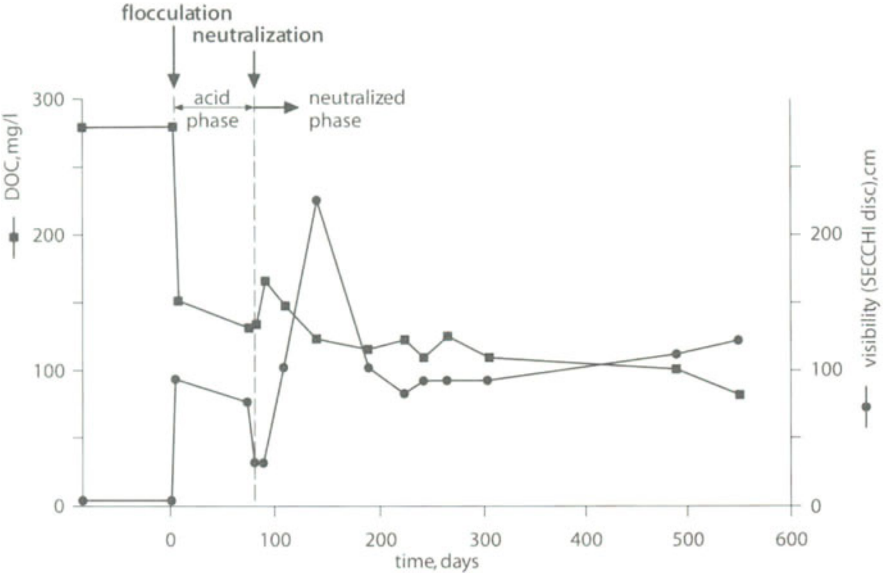


Fig. 16.13. Course of DOC (dissolved organic carbon, *squares*) and visibility (*dots*) after 550 days in enclosure experiment after flocculation and neutralization

microbiological activity. The slow reduction of the DOC (Fig. 16.13) is an expression of this autopurification process (Becker *et al.* 1997). An additional program is devoted to detailed analysis and studying both the microbiological degradation potential and the development of the hydrobiology.

16.4.4
Conclusions

The promising results from the enclosure experiment indicate the possibility of the in situ treatment of the whole deposit. Technical solutions for the injection of the flocculant and the lime suspension in layers of different depths within the water column are currently being developed. The complete remediation of the entire lake according to the results of the enclosure experiments is expected to be accomplished in 1998.

16.5
Concluding Remarks

The remediation of opencast mines on the scale described here is not possible at the current state of the art (even when available) for economic and technical reasons. Remediation research focuses on applying environmental technologies and "soft" remediation strategies such as bioremediation, bioattenuation, and bioaugmentation. Three different examples are given:

- By creating conditions for microbiological sulfate reduction, in situ measures can be taken to counteract or even eliminate the acidification of flooded opencast mine pits. Geological surveys allow prediction of the natural processes and of how they can be influenced. Laboratory investigations provide information on the subsequent approaches which can be adopted. In the case of biologically mediated neutralization, the customary control of eutrophication has to be adapted into a reverse strategy of organic and nutrient loading in order to bring about the anoxic conditions necessary for sulfur reduction and acidity binding.
- Exact study of the geological conditions (Hufeisensee) highlights the development of stable, safe hydrological conditions in a lake to be used for recreational purposes. Thus, despite its close proximity to a landfill, elaborate remediation is unnecessary, and the further management of the dump and the lake can be steered towards this goal.
- Natural autopurification processes can be started in water from an industrial dump with a high risk factor and a complex composition. However, this requires identifying the reasons for inhibition, and so new analytical methods had to be developed.

In situ experiments in enclosures support the methods worked out in the laboratory of the flocculation of blackening phenolic polymers (AHS, anthropogenic humic substances). This triggers biological activity in a clear, colorless water high in oxygen, which initiates further autopurification.

The interdisciplinary research pursued at the UFZ (Center for Environmental Research Leipzig-Halle) takes into consideration both the natural sciences and social and economic issues. The remediation of large former industrial areas as an integrated approach is able not only to form new landscapes but also to assist the economic and social revitalization of the regions concerned.

Environmental contamination on the scale described in this chapter in a region with a population of a few million people has resulted in social repercussions such as a diminished quality of life and impaired health. In the past this resulted in an exodus of inhabitants from the region and even in social unrest, in particular in 1989. Following German reunification and the economic changes, one of the first steps taken was to close down the uneconomic and worn-out carbochemical and electrochemical plants. This quickly brought about the reduction of air and water pollution, and the situation has since continued to improve. However, industrial closures naturally also led to severe unemployment. A new economic and social structure must be built up, with remediation and the restoration of the ruined landscape forming a framework for such new developments.

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