Solid Waste: Assessment, Monitoring and Remediation
Twardowska, Allen, Kettrup and Lacy (Editors)

VI.6
High-volume mining waste disposal

Irena Twardowska, Sebastian Stefaniak and Jadwiga Szczepańska

V1.6.1. Introduction

Mining waste disposal permanently brings about a contamination threat to groundwater of unprotected aquifers and surfacewater in dumping sites all over the world. As was shown in Chapter III.6, even a non-hazardous coal mining waste layer of 1.5-m thickness can be a persistent source of deterioration of the aquatic environment lasting for decades. Use of mining waste in civil engineering as fill and earthworks material should thus be considered as a potential source of long-term water contamination.

The mining waste burden and management problems are particularly important due to the high volume of material disposed. According to the Central Statistical Office (2002), the total amount of mining waste (including tailings from preparation plants) generated in 2000 in Poland was as high as 73.6 million tons (Mt), i.e. 59.5% of the total waste generated in the country. Annual generation of hard coal mining waste in 2001 accounted for 38.4 Mt and waste from ore mining, almost entirely tailings, accounted for 29.9 Mt. Current percentage of mining waste use was relatively high and accounted for 91.0% of generated coal mining waste and 75.1% of ore mining and processing. The total amount of hard coal and ore mining and processing waste laying in dumps at the end of 2000 was 1253.5 Mt that was 63.4% of total amount of all wastes disposed throughout the country. Of that, hard coal mining waste comprised 53.3% and metal ore (copper) mining and processing, 46.7%. The area of each dump ranges from about 10 to >200 ha for large central dumping sites where waste from several mines has been disposed. Reused material, predominantly applied at the surface in civil engineering as common fill, e.g. for land leveling, road and embankment construction, is also exposed to the atmospheric conditions. In 1995–1996, only 12.1% of the reused coal mining waste was applied underground in mine workings, while 87.9% was used for engineering construction at the surface (State Inspectorate of the Environmental Protection, 1997). These data clearly illustrate the range of the problem (for data on the global mineral extraction and mining waste generation, see Chapter III.6).

At the stage preceding the design and construction of a dump or earthworks, the reliable evaluation of pollution potential to ground- and surfacewater from waste, as well as the prognosis of the life-cycle leaching behavior and of impact on the aquatic environment in the area of the prospective waste disposal should be the basis for the dump location permit, rejection or acceptance of a material for civil engineering constructions and protective measures to be applied. In this case, false-positive or false-negative errors in the long-term risk assessment have direct economic or environmental consequences, and thus the evaluation should be carried out particularly carefully.

Concentration of large volumes of mining waste in a relatively small area, the mechanism of pore solution formation in the anthropogenic and natural vadose zone in the disposal site, transport of contaminants from the source to the receiving aquatic environment and the resultant impact on the groundwater of the saturated zone and surfacewater in the vicinity of dumping site within its life cycle have to be considered. In general, risk management approach with respect to sulfidic mining waste is fixed on the acid rock drainage (ARD) formation and control strategies (Hutchinson and Ellison, 1992; Environment Australia, 1997; EPA/DOE, 2000), in order to prevent toxic metal mobilization at low pH. The experience of the authors with coal mining waste (Twardowska, 1981; Szczepańska and Twardowska, 1987, 1999; Twardowska et al., 1988; Twardowska and Szczepańska, 1990) discussed in Chapter III.6 shows that also non-acid leachate from the dump can cause severe degradation of the groundwater quality in the saturated zone just due to the high concentration of chlorides at the first stage of leaching and sulfates in long-term perspective. The sulfate ions are generated from the same process of sulfide oxidation that causes ARD formation and occur in leachate from buffered waste. The sulfate ions are balanced either by the Ca^{2+} and Mg^{2+} cations at the concentration level constrained by equilibrium with gypsum, or by Na^{+} ions. In this case, the TDS concentration is limited entirely by the sulfate generation rate and the vertical redistribution of loads, discussed earlier (see Chapter III.6). Therefore, not only low-buffered acid generating (AG) waste dumps, but also buffered non-acid mining waste disposal sites should not be neglected as a source of groundwater deterioration. It is obvious that ARD poses potentially a very high risk to water quality.

The general best practice principles of risk management from sulfidic waste, including coal and ore mining waste, comprise four major steps: (i) life-cycle prognosis of leaching behavior of waste material exposed to atmospheric conditions for the analyzed variants of the dump or civil engineering construction under consideration; (ii) developing preventive planning, design and disposal operation practices for the waste material, which is adequate to the assessed risk; (iii) dump rehabilitation in operational and post-closure periods; and (iv) life-cycle monitoring of the vadose and saturated zones within the area of the dump or civil engineering construction impact on groundwater quality to provide an early alert for taking remedial actions before significant degradation of recoverable groundwater resources occurs.

In this chapter, a risk management approach to the contamination caused by high-volume mining waste disposal has been exemplified in the current disposal strategies and practices with respect to coal mining waste in the Upper Silesia coal basin (USCB) in Poland. Mine waste management practices in Australia and USA also have been discussed.

V1.6.2. Long-term prognosis of leaching behavior of mining waste and its effect on the aquatic environment

V.1.6.2.1. Site selection and prognosis of leaching behavior

In every case of dumping site selection, and dump construction, an optimization analysis of pollution potential to the aquatic environment should be carried out. The site most advantageous with respect to the aquatic environmental protection requirements is often not in compliance with other optimization parameters, e.g. with the distance from the mine(s) or the system of waste transport. At the preliminary stage, several variants of the site location should be analyzed for selection of the optimum one, being consistent with all economic and technical criteria, but preferring environmental protection requirements of sustainable development.

The extent of hazard from the dump leachate to the aquatic environment should be evaluated on the basis of the long-term prognosis (no less than 25 years). The minimum time period for prognosis depends on the volume and duration of the dump construction, as well as on the occurrence in the planned dumping area of unconfined water resources of particular value. The flow diagram of evaluation is given in Figure VI.6.1. The correct assessment of input data is of a special importance for the accuracy of prognosis. The computer simulation should be thus preceded by the detailed testing of the disposed material properties, short-term and long-term static and geochemical dynamic tests discussed in Chapter III.6, and validated by long-term field experiments (e.g. lysimetric) or studies in existing old disposal sites of the same or similar waste (from the same seams) if such sites already exist. Hydrogeologic, hydrologic and climatic conditions in the planned dumping area are another set of input data for the prognosis of contaminant migration in the groundwater. The input data based on analogy can be used entirely for the preliminary assessment of the dump impact on the aquatic environment. The long-term prognosis of leaching behavior and acid generation potential of waste material, besides being of importance in deciding the dumping site location, also suggests the appropriate management strategy.

The case study exemplified in Environment Australia series (Environment Australia, 1997) as an approach to ARD assessments for new and existing mines along with waste dump planning, presents a similar procedure, with particular regard to ARD generation potential (Fig. VI.6.2). The same source (Environment Australia, 1997) identifies five main rock material types with different management prescriptions: IA, IB, IC — non-acid forming material of low to extreme salinity; II — potentially acid forming material of low risk; III — potentially acid forming material of high risk, based on evaluating net acid generation (NAG). It should be mentioned that such differentiation, though convenient, is somewhat artificial, in particular with respect to the range limits and threshold values.

The recognition of ARD potential as a key issue in the project planning and management in many areas such as mine waste testing procedure, mine planning, designing of tailing storage and waste rock dump (including water treatment facility if needed), reclamation and closure planning, or using waste material for construction was a basis of another similar approach in estimating ARD potential of waste rock by block modeling (Downing and Giroux, 1993; Downing and Madeisky, 1996; Bennett et al., 1997; Bursey et al., 1997; Downing and Giroux, 2000). The authors consider the diversity

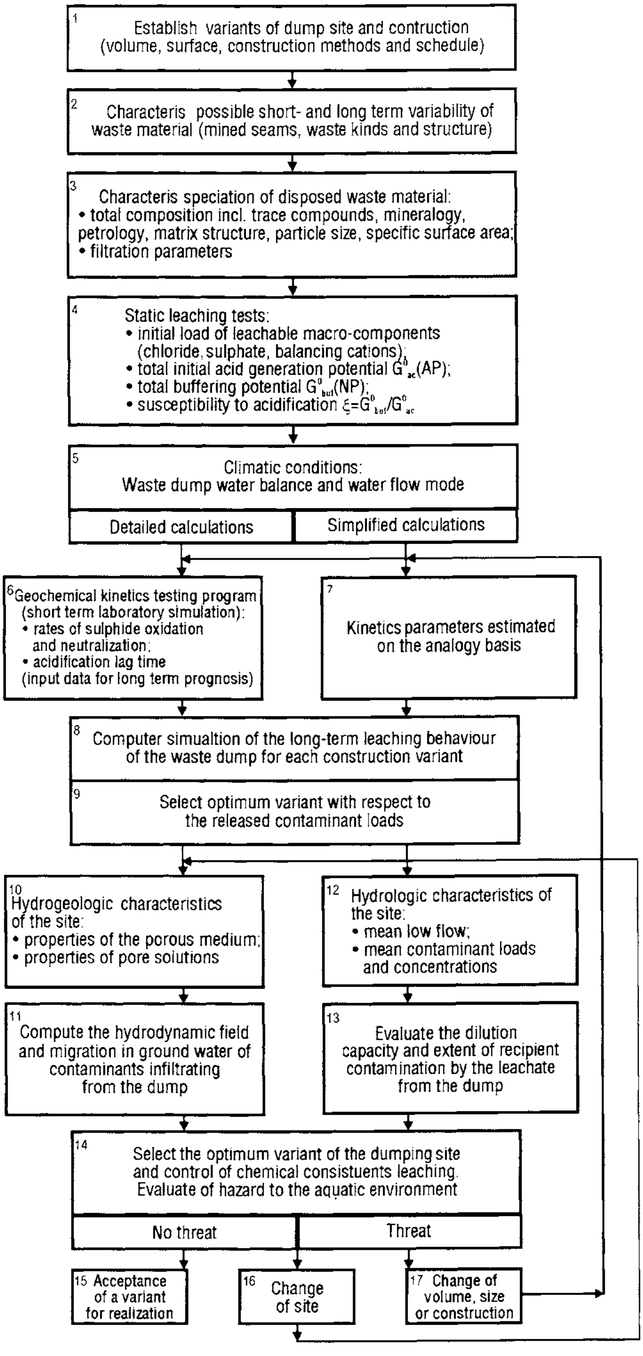
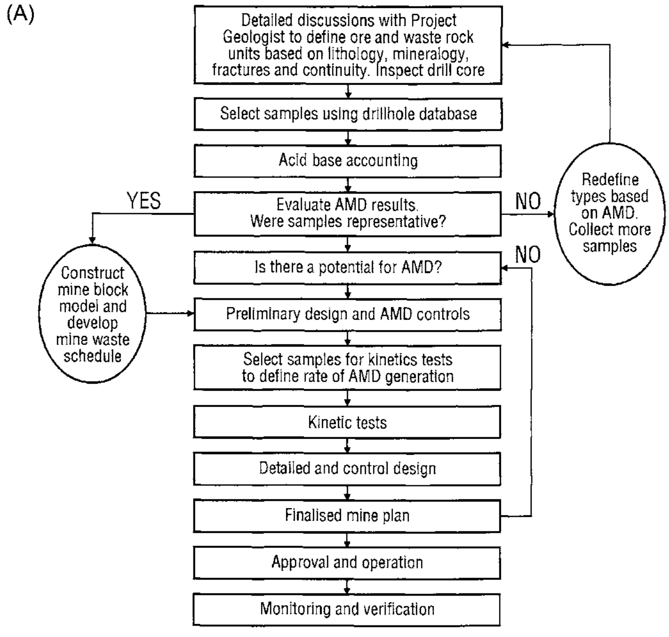


Figure VI.6.1. Flow diagram of a risk assessment to the aquatic environment from the mining waste reuse or disposal, and selecting optimum variant of a dumping site.



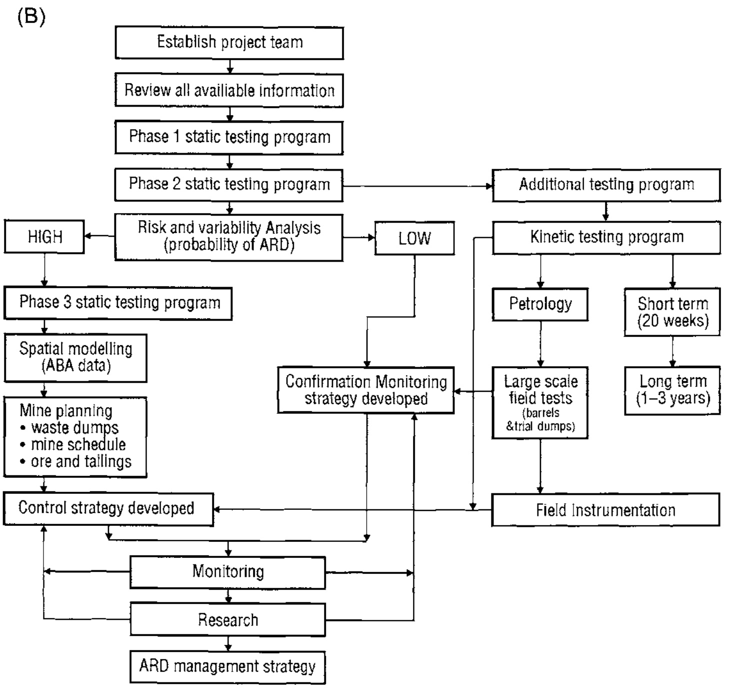


Figure VI.6.2. Approach to AMD/ARD assessments of new (A) and existing (B) mines (Placer Pacific Ltd., after Environment Australia, 1997). AMD is acid mine drainage and ARD, acid rock drainage.

of acidifying/buffering capacity of a heterogeneous rock material within the mined ore deposits and define two major components of the block model estimating the extent of hazard to the environment: (i) the acid generation potential and (ii) the metal and trace metal components of rock material that would impact the metal leaching from acid waste rock (ML/ARD). A waste material has been proposed to be classified (ABA classification) as an integral part of a geostatistical resource estimate study (through drilling in blocks $10 \times 10 \times 10\ m^3$) as AG, potentially acid generating (PAG), potentially acid consuming (PAC) or potentially neutral (PN) on the basis of a ratio Ca–Mg carbonates/total sulfur = neutralization potential/maximum potential acidity (NP/MPA). The criteria of classification were given as follows:

- PAG: NP/MPA ≤ 1.0
- PN: $1.0 >$ NP/MPA < 3.0
- PAC: NP/MPA ≥ 3.0

This classification corresponds with the criteria of evaluating acidification/buffering capacity presented in Chapter III.6 and is an attempt of its practical application on site, as it is presented in Figure VI.6.3.

ARD-generating waste material from ore mining (PAG) produces hazardous metal-rich acid leachate; though it does not imply that mine rock classified as PN or PAC is environmentally neutral.

For high-volume dumping sites active for many years before site closure and capping, and in deep mining regions often sited in subsidence-affected areas, the construction period is particularly critical and environmentally problematic. The general approach is to evaluate the risk to the aquatic environment from the waste dump as a result of: (i) geochemical characteristics of material implying contaminant generation and release rates (primary contaminant load, sulfide oxidation kinetics, equilibria constraints); (ii) dump construction, volume, development in time determining the total potential load of contaminants and its temporal and spatial distribution; and (iii) hydrogeological and hydrological conditions, which determine the ability of the vadose zone to attenuate or increase contamination and of recipients (ground- and surfacewater) to dilute the contaminants load to the environmentally acceptable level. As has been shown in Chapter III.6, not only ARD, but also leachate from buffered waste material can create a substantial problem for groundwater and make it unfit for any use for a period of decades.

V1.6.2.2. Models for long-term prognosis of contaminant leaching and transport

Long-term impact assessment of mining waste disposal sites and engineering constructions on the aquatic environment constitutes a basis for the step-by-step approach to the decisions concerning optimum dump location and construction, as well as preventive or remedial measures for control of contaminant leaching from new and old dumping sites and engineering constructions. For this purpose, the software packages have to be composed of two major integrated parts: (i) a model for hydrogeochemical calculations of generation, leaching and transport of species within the waste dump and (ii) a model of water flow and contaminant transport in

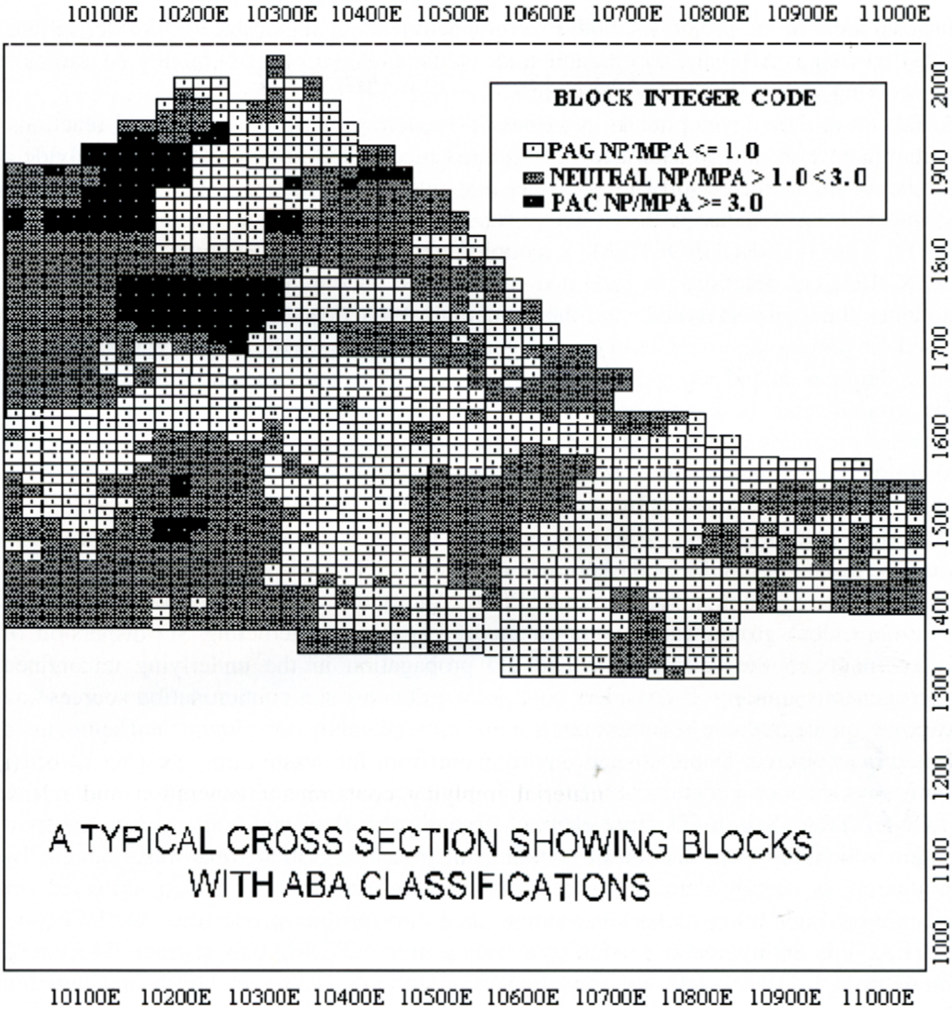


Figure VI.6.3. Example level plan showing blocks with ABA classification (after Downing and Giroux, 2000). AG is acid generating rock; PAG, potentially acid generating rock; PAC, potentially acid consuming rock; and PN, potentially neutral rock.

groundwater in the waste dump site, and eventually also in the receiving surfacewater. The local groundwater quality monitoring (LGQM) database for existing dumping sites of waste from the same mines and seams may be helpful for the prognosis and input data *in situ* verification.

Input data should include: (i) mining waste characterization related to the initial content, generation and leaching of soluble constituents; (ii) waste disposal site construction and its temporal and spatial development (consecutive waste layers formation program); and (iii) hydrologic, hydrogeological and meteorological data for the waste disposal area. In the prognosis, both macro-constituents of importance (chloride, sulfate, TDS), as well as ARD and mobilizable trace elements that may cause quality degradation of receiving waters should be considered.

The up-to-date developments in numerical modeling of hydrogeochemical reactions, contaminant transport in saturated–unsaturated media, and groundwater flow provide a number of sophisticated software packages that give an opportunity of a reliable long-term prognosis in accordance with the above scheme (SSG, 2003; Waterloo Hydrogeologic, 2003). The HYDROGEOCHEM 2 coupled model of HYDROlogic transport and GEOCHEMical reactions in saturated–unsaturated media, which contains two basic modules: the transport module and the geochemical reaction module, is particularly well suited for simulating processes of species generation, release and transport within the mine waste dump as an anthropogenic vadose zone, natural vadose zone and in saturated zone. For groundwater flow and contaminant transport modeling, several other effective software programs can be used, depending upon the complexity of the hydrogeological conditions and transport problems, e.g. from easy-to-use KYSPILL 2.0 (Anonymous, 1997; Serrano, 1997) tested by American Institute of Hydrology or FLONET/TRANS (Waterloo Hydrogeologic, 2003), through MODFLOW-SURFACT and 3DFEMFAT software packages to GMS 4.0 (Anonymous, 1999; SSG, 2003, Waterloo Hydrogeologic, 2003). Of the mentioned software KYSPILL (Anonymous, 1997; Serrano, 1997) is the scale-dependent groundwater pollution model capable of predicting 3D dispersion of contaminants in the vadose zone and 2D propagation in the underlying unconfined heterogeneous aquifers. It considers both point and non-point contamination sources and reactive or degradable contaminants, and is applicable for dump leachates as a contaminant source. Due to simplicity, it has serious limitations, e.g. does not consider continuous input of a contaminant from non-point sources, or multi-component transport. FLONET/TRANS is a 2D finite-element groundwater flow and contaminant transport model, which allows prediction of contaminant plume migration from waste dumps, by simulation of steady-state flow and time-varying transport in leaky, confined, or unconfined aquifers with heterogeneous and anisotropic properties. MODFLOW-SURFACT is the upgraded version of a widely used MODFLOW program (McDonald and Harbaugh, 1999) with the modeling capability of vadose and fully and variably saturated zone flow, and multiphase multi-component contaminant transport. It considers seepage and delayed yield conditions, parent and transformation product transport modeling, as well as linear or non-linear retardation for each species, and thus allows correct long-term flow and contaminant transport prediction from new and old dumping sites. 3DFEMFAT is a 3D finite-element model of flow and transport through saturated–unsaturated, heterogeneous and anisotropic media. This model can use a very large time step, and among other capabilities, considers spatially and temporally dependent element and point sources/sinks, a prescribed initial condition or the simulated steady-state solution as the initial condition, as well as iteratively determined infiltration, seepage, and/or evaporation boundaries, which is advantageous for application to seepage from mining waste dumps. Of these packages, groundwater modeling system (GMS) and its latest version 4.0 is the most sophisticated and comprehensive groundwater modeling environment (Anonymous, 1999; SSG, 2003). GMS supports several finite-difference and finite-element packages in 2D and 3D that provide site characterization, simulate flow and contaminant transport in saturated and unsaturated zones, bioremediation and natural attenuation and include MODFLOW, MODPATH, MT3DMS/RT3D, SEEP2D, SEAM3D (Widdowson, 2002), UTCHEM and FEMWATER. MODFLOW 2000 — the U.S. Geological Survey modular groundwater model support in GMS 4.0 is the latest and the most advanced MODFLOW version that includes the new HUF, LPF and ADV packages (Harbaugh et al., 2000; Hill et al., 2000; Clement, 2001; Mehl and Hill, 2001; Zheng et al., 2001). Due to complexity, GMS modeling system has the tools for a variety of modeling needs: the predictive analysis of mining waste dump environmental impact is one of them.

There are a number of other software packages that can be applied for predicting groundwater flow and contaminant transport from dumping sites. They have specific capabilities, e.g. Visual HELP (based on the U.S. EPA's Hydrologic Evaluation of Landfill Performance, 1984) for modeling landfill hydrology, estimating groundwater recharge rates and determining the effectiveness of capping; AquaChem package for geochemical modeling and managing water quality data with use of PHREEQC, or CVFlux 2D and ChemFlux 2D packages that are widely used for predicting the movement of contaminants from tailings, pits and earth containment facilities. Of these models, PHREEQC — a computer program for speciation, reaction-path, advective transport, and inverse geochemical calculations (Parkhurst, 1995; Charlton et al., 1997; Vrabel and Glynn, 1998; Parkhurst and Appelo, 1999; Charlton and Parkhurst, 2002; Merkel and Planer-Friedrich, 2002; Zhu and Anderson, 2002) in particular the latest version PHREEQC Interactive 2.8.0.0 (2003) is the most applicable for geochemical calculations involving reactions and contaminant transport occurring in sulfidic waste (Parkhurst, 1997; Appelo et al., 1998). The correct prediction of contaminant generation and transport with use of even the most sophisticated models highly depends upon the reliability of input data. In the case of coarse heterogeneous sulfidic waste, the critical parameters include, besides total sulfur and total carbonate contents, also spatially, vertically and temporally influenced kinetics of sulfide oxidation and availability of buffering agents under actual conditions of their exposure. The assessment of these parameters is generally particularly problematic, as laboratory exposure conditions differ substantially from those in the real field conditions.

Surfacewater quality is less impacted from mining waste dumping sites than the shallow unconfined aquifers. Nevertheless, environmental impact assessment (EIA) requirements consider evaluation of contamination threat also for surfacewater recharged by groundwater or directly from dumping site drainage system. For this purpose, several surfacewater flow and contaminant transport models might be used, e.g. GFlow 2000, flow model and solute transport system in saturated zone that supports also conjunctive surfacewater and groundwater modeling or AQUASEA upgraded software package, which consists of the hydrodynamic flow model that can simulate water level variations and flows, and the transport–dispersion model that simulates the spreading of a pollutant of any kind under the influence of the flow and existing dispersion processes (SSG, 2003; Waterloo Hydrogeologic, 2003).

The contaminant transport modeling and application of computer models for a regional prediction of the contaminants transport from the non-point source or waste disposal site in soils and groundwater has also been discussed in Chapter V and is exemplified in the case study presented in Chapter V.4.

V1.6.3. The basic tasks of mining waste dumps rehabilitation

Under the actual conditions of the thickly populated mining areas such as the USCB in Poland, the rehabilitation of mining waste dumps is a task of a particular importance. According to the definition, these tasks should be realized through the technical, agrotechnical and biological actions enabling economical and environmental restoration of the degraded areas. These measures comprise the land reshaping, dump sealing, improvement of soil properties, regulation of water balance, and installation of a monitoring network in the site area (Directive of the Minister of Environment, 2002). Polish Environment Protection Act (2001) with regard to land surface protection defines rehabilitation tasks as the optimum landscape reshaping, restoration of quality parameters of soil according to the standards and economical reuse of the derelict land. Due to the limited availability of land for locating dumps, and technical and economical limitations of waste reuse for stowing in mine workings, the requirement of land surface rehabilitation should not be treated as an exact restoration of the primary landscape and area use, but as an optimum solution, which ensures also the maximum volume of waste to be disposed in the site. In general, in the USCB conditions, it means the construction of high flat waste dumps, as a temporary (so called "forestall reclamation" in the subsidence area) or final solution. This means, in fact, the creation of a new landscape, which should fulfill the following prerequisites:

1. Life-cycle environment pollution control during dump construction and after site closure.
2. Maximum disposal volume, under the prerequisite (1).
3. Acceptable landscape planning, under the prerequisites (1), (2) and (4).
4. Optimum economic reuse of a site, under the prerequisite (1)–(3).

In general, the first two prerequisites are of superior significance, while the other two are of a subordinate weight. Independent of the gradation, all these prerequisites should be unconditionally fulfilled, though the first two (environmental protection and maximum volume) dictate the solutions for landscape planning and land use. The applied methods of biological reclamation should constitute an integral part of tasks (3) and (4), provided that the major prerequisite (1) is fulfilled. Rehabilitation of the dumping site should start from the land preparation for dumping and be continued until the dump construction is completed, finally shaped and adequately used. Rehabilitation actions should be conducted in parallel with the dump construction as temporary (transitional) and final measures.

V1.6.4. Aquatic environment protection strategies in mining waste dumping sites

V1.6.4.1. General assumptions

The availability of areas suitable for dump location due to the favorable hydrogeological conditions is generally extremely limited. Mining waste dumps are invariably located on land that is considered to have little or no value for other use, such as abandoned sand and gravel pits, areas of continuous surface deformations due to subsidence caused by underground mining, etc. These areas usually have a disturbed vadose zone and stripped soil layer that could have played the role of a protective barrier. They are therefore susceptible to groundwater contamination. A number of old dumps of a high long-term pollution potential are sited over unconfined aquifers being a valuable resource of drinking water, or along the river beds.

A high volume of sulfidic mining waste output implies the need to search for cost-effective and efficient control strategies for minimizing generation, release and discharge of contaminant loads to the aquatic environment. The most efficient and practicable way of reducing a potential risk is the prevention of atmospheric oxygen and water penetration through the waste layer, in order to:

- control sulfide oxidation, acid generation and trace metal release occurring in wastes and in the dump bedrock;
- reduce the waste volume exposed to leaching;
- increase the buffering capacity of low-buffered AG material.

The above control measures can be achieved by the appropriate construction of a dump (Twardowska et al., 1988; Twardowska, 1990, 1993; Szczepańska and Twardowska, 1999). The waste disposal strategy presented here is based mainly on the authors' experience and solutions applied in dumping sites of coal mining waste in the USCB in Poland that are not considered hazardous (2000/532/EC; 2001/118/EC). The discussion also comprises the review of management and disposal strategies for ARD generating wastes from the mineral industry (coal, uranium, base metal and precious metal projects) in Australia (Environment Australia, 1997) and in the USA.

The general principle to be followed in the mining waste disposal practice is a site-specific approach to each waste disposal site, resulting from significant differences and variability of waste rock properties, site characteristics, dump development, volume, construction period, geology, petrology, hydrogeology, hydrology and climate, in spite of the similarities in the basic mechanisms of pollutant generation and release (see Chapter III.6).

VI.6.4.2. Dump construction

VI.6.4.2.1. Minimization of the exposed surface

The dump final shape and the construction period should ensure minimization of its exposed surface, which determines the amount of infiltration water and waste rock in the uppermost layer where processes of contaminants generation and leaching are the most active. This requirement may be met by the construction of high dumps in layers of a terrace form, which provides the lowest surface area: dump volume ratio, the best utilization of a site for collection of the maximum volume of wastes and reduction of water precipitation infiltrating to the dump. To constrain side air penetration and enable fast rehabilitation and biological reclamation of the construction, the dump external batters and terraces should be formed first, in heavily compacted thin layers ($\sim$0.5 m) and shaped according to the final designed form and slope. Next, the internal parts of the dump should be filled in thicker compacted layers (up to 6 m) in sections, up to the final level, adequately constructed to prevent or reduce infiltration of precipitation water.

VI.6.4.2.2. Reduction of waste contact with infiltration water and controlled water discharge from the dump

This may be accomplished by: (i) application of a surface or shallow sub-surface drainage system at the top and terraces of the dump, placed at the low-permeable base layer of low hydraulic conductivity ($k < 10^{-10}$ m/s) and (ii) the double surface drainage ditches at the dump toe for separate uptake and discharge of unpolluted water from the adjacent catchment area and leachate/drainage/run-off water from the dump (Fig. VI.6.4).

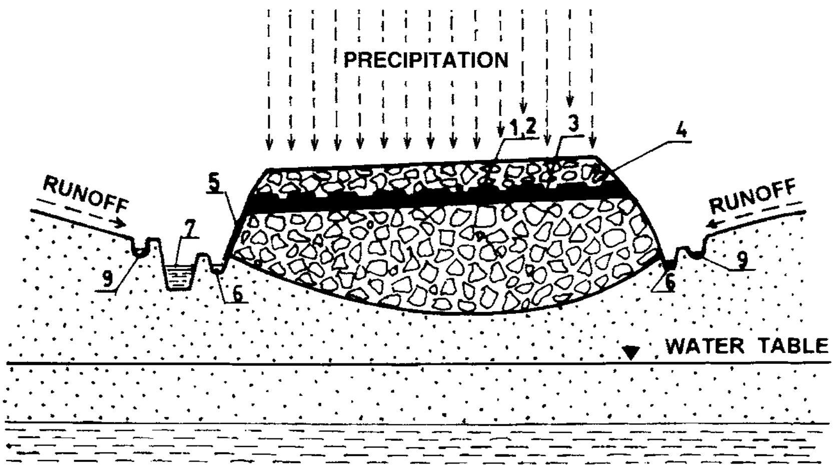
Sub-surface drainage of the dump prevents the vertical percolation of atmospheric precipitation through the dump and limits the infiltration entirely to the top cover layer. The bulk of the disposed waste will be thus eliminated from the process of contaminant leaching, which greatly reduces the pollution potential from the dump. During the dump construction, temporary open drainage ditches should be applied at the top of every intermediate layer.

The double system of surface ditches at the dump toe eliminates the contact of natural water from adjacent area with effluents from the dump and the direct contact with the disposed wastes. This reduces the volume of polluted water and provides discharge of natural waters to the nearest recipient, while polluted water, through the retention pond, can be periodically discharged to the collector of saline waters, to the closed circle of a washery or in portions to the recipient receiving water body during the period of high water flow if these waters contain no hazardous constituents in concentrations exceeding maximum concentration limit (MCL). Hazardous ARD should be up taken and treated before discharge to the receiving waters.

VI.6.4.2.3. Rendering the dump air- and water-tight

Restriction of air penetration prevents sulfide oxidation that causes generation of ARD and mobilization of pollutants from the waste rock. Reduction of water infiltration, besides control of sulfide oxidation, attenuates contaminant leaching from the dump. The top and slope protection against erosion is also of importance.

This can be achieved by the construction of the external dump batters and the internal part of the dump in compacted layers of different thickness and application of the air-tight material with high water retention capacity at the top of each internal layer (Figs. VI.6.5 and VI.6.6). Such properties have some fine-grained tailings from washeries and dense coal combustion fly ash (FA): water mixture. The dense FA: mine water mixture displays the greatest penetration resistance to air ($R = 1100-1200$ kPa/cm^2), about an order of magnitude higher than natural cohesive soils, and high residual water retention capacity, though its hydraulic conductivity is rather high ($k > 10^{-6}-10^{-8}$ m/s), within the range of non-insulating or very weakly insulating material (see Chapters III.7 and VI.8). Some fine-grained argillaceous tailings show better barrier properties with respect to water, within the range of medium permeable material ($k < 10^{-8}$ m/s). Significantly lower permeability of tailings to water and adequate buffering capacity of alkaline FA with respect to potential ARD leachate from the overlaid layers are the additional protective factors that can be utilized for rendering the mining waste dump air- and water-tight. For optimum use, the blanket layers of dense FA: mine water mixture can be placed on the intermediate internal layers of sulfidic mining waste to control ARD, while argillaceous



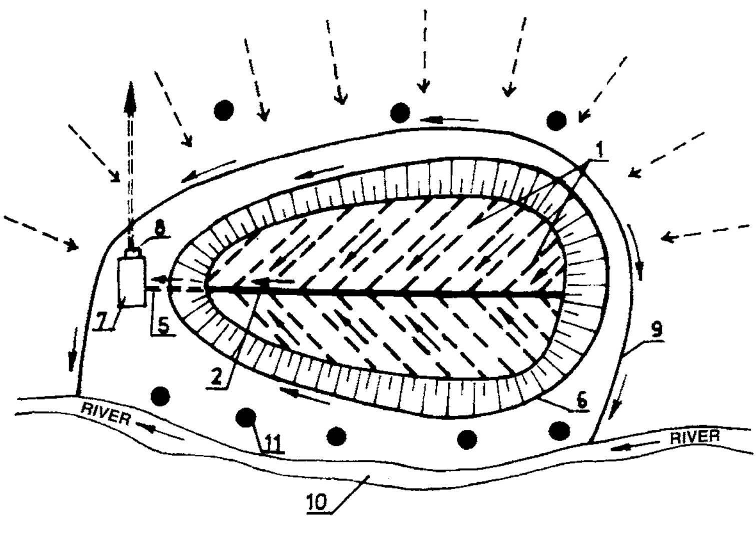


Figure VI.6.4. Schematics of mining waste dump dewatering system. Surface drainage system of mining waste dump to attenuate contaminant leaching (1,2); surface drainage system at the top layer placed at a barrier layer (3); (4) top "neutral" vegetative waste layer or topsoil; (5) drainage collector; (6) toe ditch for collecting run-off and drainage from the dump; (7) retention/evaporation pond; (8) pump; (9) exterior barrier ditch; (10) stream receiving natural runoff from the adjoining area; and (11) monitoring wells. Slope no less than 1:2.5.

tailings may be utilized as a transitional drainage-bed at the temporary top of a compacted waste layer during the dump construction. Due to susceptibility of fine-grained material to erosion, both tailings and FA mixtures should not be introduced into the external surface parts of batters. Dump closure will require construction of a permanent drainage at the dump top cover layer on a low-permeable bed, along with a system of slope ditches, toe collectors, and retention ponds.

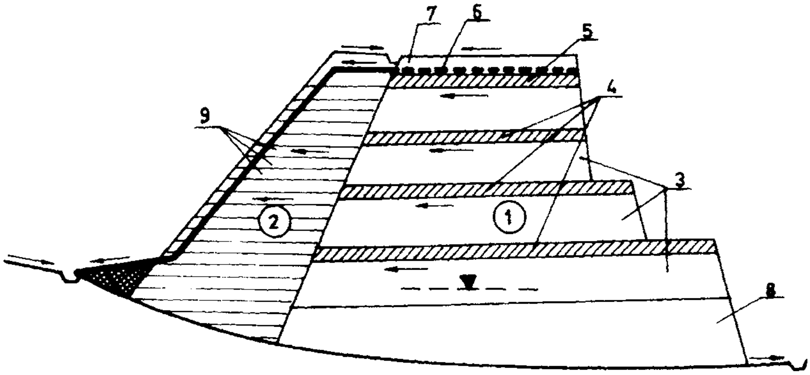


Figure VI.6.5. Mining waste dump construction with barrier layers of dewatered fine-grained tailings. (1) Internal part of the dump; (2) batters (slope min. 1:2.5); (3) compacted layers of mining waste; (4) barrier layers of dewatered fine tailings; (5) barrier layer; (6) drainage system of the top layer + capillary break (sand, gravel); (7) vegetative layer; (8) bottom layer, possible gradual inundation; and (9) batters constructed of mining waste compacted in thin layers.

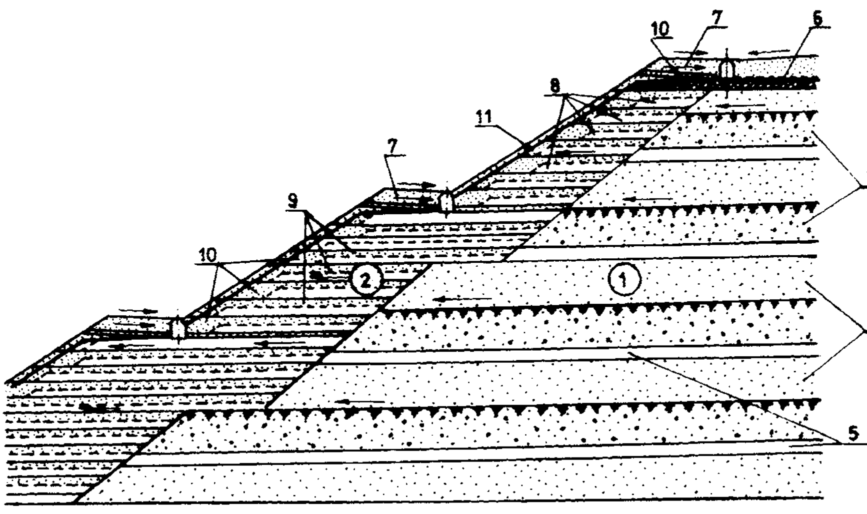


Figure VI.6.6. Construction of mining waste dump with use of dense coal combustion FA: water mixture as air-protective layers. (1) Internal part of the dump; (2) batters (slope min. 1:2.5, bench/terrace slope 1:5 to 1:10); (3) heavily compacted mining wastes; (4) moderately compacted mining wastes; (5) dense FA: water mixture; (6) surface drainage + capillary break (sand, gravel) + infiltration barrier layer (dewatered fine tailings, clay + [optionally] geomembrane); (7) vegetative layer; (8) layers of dense FA: water mixture or dewatered fine tailings $\leq$0.25 m; (9) mining waste compacted in thin layers ($\leq$0.5 m); (10) external part of batters (compacted mining waste only); and (11) drainage collector (open ditches or/and pipes).

The slopes and gradients of compacted layers should facilitate water flux to the drainage collectors and prevent vertical percolation (Figs. VI.6.5 and VI.6.6). The average slope of the dump should not be steeper than 1:2 to reduce the potential for erosion, allowing safe slope formation and convenient maintenance of vegetation.

One more protective option is the bottom lining of a dump base. In case of high-volume coarse-grained mining waste disposal in large-area dumping sites, in particular in the areas affected by subsidence, use of bottom liners is usually ineffective, unreliable (cracking), expensive and limited by the availability of a material for liners. This measure should be thus considered only for low-volume dumping sites of high-risk material, such as sulfidic metalliferous ore mining waste of high ARD potential.

VI.6.4.2.4. Minimization of contaminant loads discharged from the top cover of the dump

Despite excluding the bulk of a waste dump from generation and leaching of contaminants, the mining waste dump can still pose a serious threat to the aquatic environment, as these processes are particularly active in the surface layer. Consequently, in the case of mining waste disposal of a high contamination potential (high reactive sulfide content, low buffering capacity and thus high ARD, high heavy metal content) and unfavorable hydrogeological and hydrologic conditions in the area, additional measures for minimization of contaminant release from the exposed surface layer, which serves also for introduction of a vegetative cover, could be required. They comprise: (i) selective disposal in the top layer of low-saline and low-sulfide, low-reactive, non-acid generating waste; (ii) minimization of the vegetative layer thickness; as this parameter is determined by the depth of a root system penetration, the herbaceous carpet cover with a shallow root system is preferable; (iii) minimization of infiltration rate by the selection of plants assuring high evapotranspiration (this method is, though, limited to the vegetation period); (iv) enhancing waste properties by blending with material of high buffering capacity; and (v) reduction of air penetration to the vegetative layer, through selecting for this layer a waste material of appropriate particle size, compacted to the extent not affecting adversely the vegetation.

Due to the observed weak role of *T. ferrooxidans* in sulfide oxidation in coal mining waste dumps in the USCB area, no control measures for these bacteria have been considered. In ore mining areas of high ARD potential, these bacteria can, though, substantially accelerate and intensify the process of ARD generation (see Chapter III.6) In this case, bactericide addition or amendment of a top cover layer with high calcite or/and dolomite material for neutralization of rock and thus for suppression of the activity of these bacteria susceptible to pH changes beyond the optimum range, might be required.

VI.6.4.2.5. Minimization of contaminant concentrations in the saturated zone

In general, most dumps may be considered as an anthropogenic vadose zone. Some of these constructions are of a mixed type, the dump toe being waterlogged. In many cases of dumping site location in the area of deep coal mining, the gradual inundation of a dump toe occurs due to subsidence.

The source of groundwater contamination in the saturated zone is the release of soluble constituents from the waterlogged material, infiltration of water percolating through the upper unsaturated part of the dump and surface run-off from the batters' slopes. The critical conditions are formed in the border layer as a result of the variability of the water table.

The concept of water quality protection in the saturated toe part of the dump is based on the following assumptions:

- In the saturated zone there are practically no conditions for the generation of new contaminant loads resulting from sulfide decomposition. The kinetics of sulfide oxidation in the saturated waste layer is very low due to the negligible availability of oxygen; therefore, mining waste stored under water is chemically non-reactive.
- Release of contaminants contained in the waste material is of a short-term nature. Soluble constituents occurring in the material at the moment of inundation are transferred to the solution. After washing out the contaminant load occurring in the material during inundation, it becomes harmless to the aquatic environment provided the storage mode is not changed.
- The amount of water in the waterlogged layer is limited to the voids between the waste particles.

If the contaminants released from the gradually inundated layer pose even a short-term threat to the groundwater quality used for water supply, e.g. highly saline wastes, the pollution potential of the waste can be substantially reduced by a single thorough exchange of water in the layer. Periodic forced discharge of saline water to the saline mine water pipeline from the relief wells can be applied until the subsidence ceases and permanent reduction of chloride salinity or other contaminants to the acceptable level occurs. The dump batters along with relief wells can be used also as a protective barrier against inundation of the external area in the vicinity of the dump (Fig. VI.6.7).

VI.6.4.2.6. Further developments

The current status of groundwater protection from contaminant leaching from sulfide mining waste dumping sites cannot be yet defined as being satisfactory and needs new solutions, particularly in the most critical period of a waste dump construction that usually lasts for years. During this time, both soluble contaminant leaching and generation of new contaminant loads occur. These processes have detrimental impacts on the groundwater quality down-gradient of the dump that results in deterioration of Quaternary aquifer observed in the most cases in the USCB area, and in other dumping sites. The solutions being under development utilize ability of some other abundant waste materials to act as water-permeable barriers having required properties for attenuation of contaminant generation or migration, e.g.:

- Air-tight material for prevention of sulfide oxidation within the waste layer and thus generation of ARD.
- Material showing neutralizing properties to prevent heavy metal mobilization.
- Material of high sorption capacity for metals and some toxic organic compounds.

All these materials are considered to be used as preventive barrier layers, in particular, in the critical transitional stage of dump construction prior to capping. According to

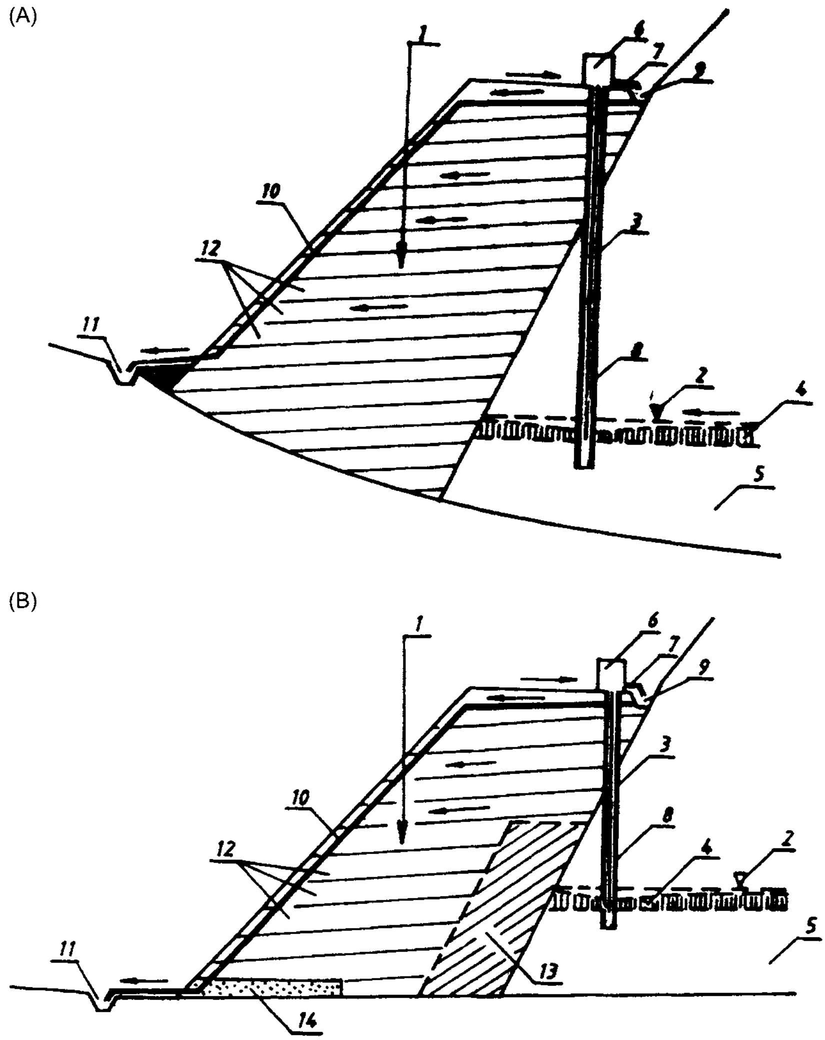


Figure VI.6.7. Construction of mining waste dump in the zone of a gradual inundation due to subsidence without (A) or with internal sealing screen (B). Batters; (2) inundation level; (3,6,8) relief wells with filter packs and a pump; (4) upper layer of inundated part of a dump; (5) inundated zone; (7) discharge from a pump; (9) collecting ditch at the batter terrace; (10) surface drainage collectors; (11) dump toe ditch; (12) mining waste compacted in thin layers; (13) impermeable screen; and (14) longitudinal strip drain.

the experience of the authors, the material of adequate air-tightness (high penetration resistance coefficient) and temporary neutralizing capacity is a dense mixture of saline water with coal combustion FA, in particular enriched in carbonatic material, e.g. containing products from lime desulfurization process (see also Chapter VI.8). To make it effective, the protection layer should be of a blanket-type placed at the top of a mine waste layer and solidified, as it has been already described (Fig. VI.6.6).

High sorption capacity for metals and organics show stabilized sewage sludge and composts rich in humic substances (HS) that can be utilized in dumped ARD-generating mine waste material as permeable barriers in a way similar to that presented in Figure VI.6.6 for dense FA: water mixture. The use of both kinds of protective barriers in one dump alternately may significantly enhance metal immobilization in waste material. The above protective measures are particularly attractive due to low capital costs and their passive character that means no running costs and no need of after-care and maintenance. Only in case of serious unpredictable surface deformations, e.g. due to accidental subsidence, when continuity of the layer is damaged, some extra repair would be needed where possible.

The possibility of solubilized HS application as permeable barriers for *in situ* remediation of contaminated aquifers from organic and inorganic pollutants has been also reported (Georgi et al., 2002). The HS barrier is proposed to be constructed by injection of HS solution, which is subsequently immobilized onto the aquifer materials optionally by: (i) coating of the mineral surfaces with Fe^{3+}/Fe^{2+} precipitates in order to produce positively charged surface sites prior to the injection of HS; (ii) flocculation of HS in the interstitial pore volume using polyvalent cations, e.g. Ca^{2+} (Fig. VI.6.8).

This active-care and much more expensive method is of a prospective use for groundwater remediation in old dumping sites where contamination of aquifer has already occurred. It has been successfully tested in a bench scale; its construction and effectiveness will be investigated in a field study.

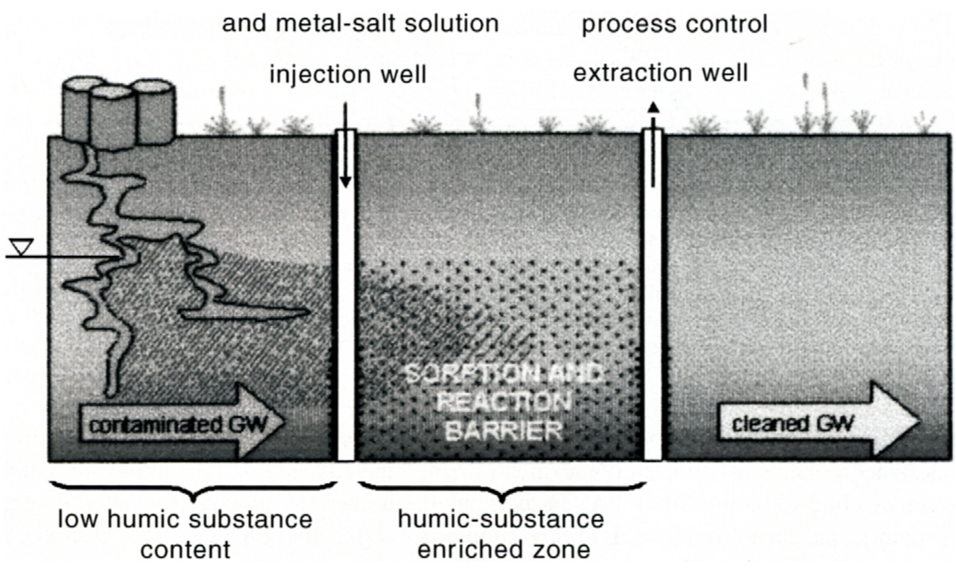


Figure VI.6.8. Construction of a humic substance barrier in a contaminated aquifer (after Georgi et al., 2002).

The dump construction and rehabilitation strategy in other countries, which are facing problems of ARD in mining areas, is based on the same approach as discussed above. The practical implementation of this approach and engineering options to manage ARD for mining waste dumps differs in details dictated by the site-specific conditions and properties of the material that exclude unified rigorous routine practices and implies the need to search for an individual optimum solution for each dumping site. A compendium of ARD management options in dumping sites is exemplified below by the practice and activity of several of Australia's best-performing minerals mining companies (Environment Australia, 1997).

VI.6.4.3. Rehabilitation strategy for mining waste dumps in Australia

The rehabilitation strategy for the waste dump construction in Australia is directed mainly to minimize the potential for ARD and considers the control of the same parameters as discussed above, i.e. sulfide oxidation and acid generation rates, water percolation through the waste layer along with control of alkalinity and acidity balance of the material. A hierarchy of appropriate management strategies is as follows:

- minimize oxidation rate;
- reduce potential for transport of oxidation products to the recipients;
- contain and treat acid drainage.

A summary of engineering options to manage ARD comprises a set of proactive preventive and reactive remedial measures (Environment Australia, 1997):

Preventive measures include (Figs. VI.6.9 and VI.6.10): (i) selective handling/encapsulation of high ARD generating waste with benign material; (ii) in-pit disposal (if available), similar to encapsulation; surface cover may be provided either by water or compacted fill cover; (iii) blending/mixing/co-disposal of ARD wastes with benign non-acid producing or acid neutralizing materials; (iv) micro-encapsulation through leaching the ARD waste with a phosphate solution with hydrogen peroxide, to form a passive surface coating of phosphate precipitate over the waste rock fragments; and (v) uncontrolled placement of low ARD generating waste with downstream collection and treatment of water; collection systems include catchment ponds, drains, trenches, groundwater boreholes; treatment/disposal systems include chemical treatment (e.g. lime dosing), controlled release and dilution by adjacent streams, evaporative disposal, process reuse and wetland filter treatment.

Remedial measures comprise (Fig. VI.6.10): (i) subsurface sealing or surface covering with a low-permeable benign material to control water and air penetration into the dump; (ii) downstream collection and treatment of water, as in option (v) of *Preventive measures*; and (iii) removal of high ARD waste, an option usually not considered because of the costs involved.

The above options, in general, are of a less complex character, and consider mainly ARD waste encapsulation and blending in order to reduce acid generation potential, without control of leached loads transported to receiving waters. They also assume availability of excess land for dumping, benign/acid consuming material for ARD waste insulation/sealing (e.g. porphyry), and also adequately high capacity of receiving waters to dilute the contaminant loads transported in non-acid leachate.

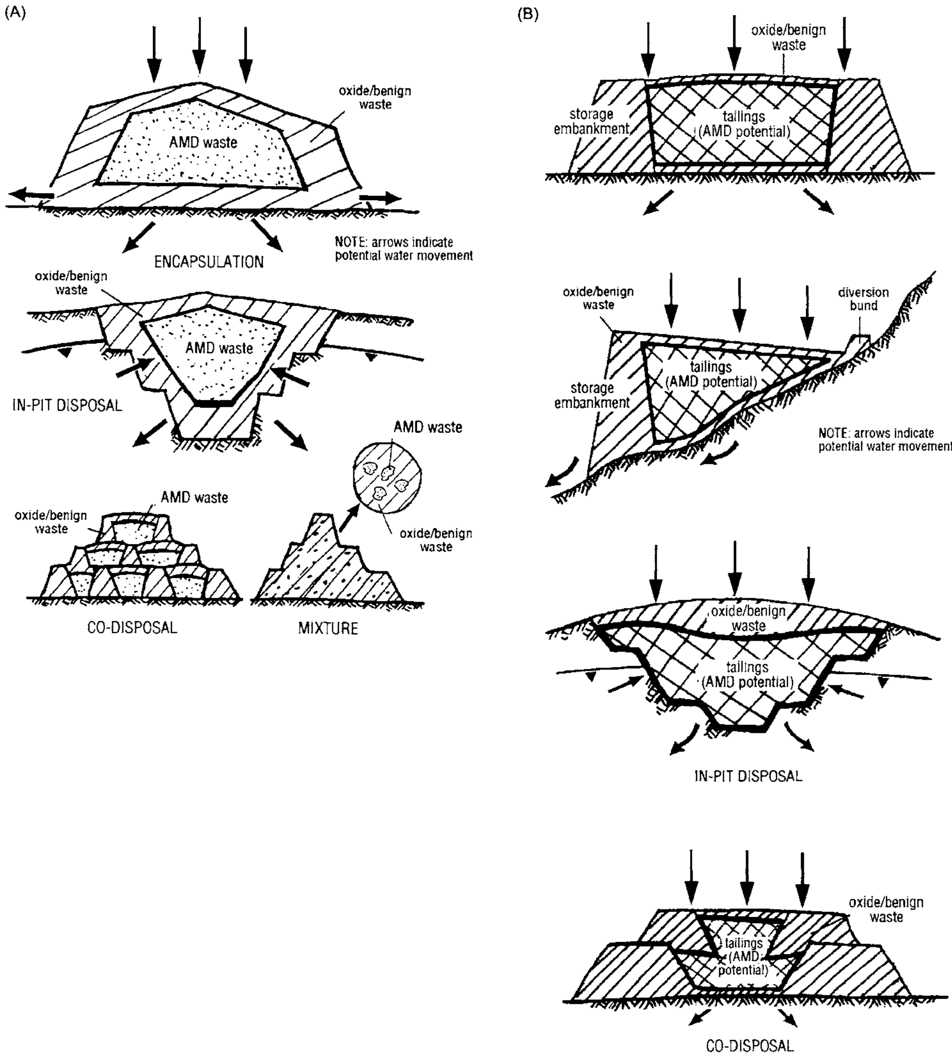


Figure VI.6.9. Methods for ARD control using a range of encapsulation and co-disposal options (A) in waste rock dumps; (B) in tailing storages (from Environment Australia, 1997, after Marszalek, 1996).

The construction of dumps using double or single insulation of ARD waste with primarily non- and low-sulfur material (Fig. VI.6.11(A,B)) prior to formation of a compacted external porphyry seal shaped as a terraced landform with an average slope angle 1:4 (Fig.VI.6.12) may be a viable option provided the feasibility analysis confirms availability of both insulating material and land for dumping.

The rehabilitation strategy focused on reducing both infiltration through the dump and convective/diffusive transport of oxygen, which is shown in Figure VI.6.13, is the closest to the approach considered by the authors of this Chapter as the most effective. Sealing

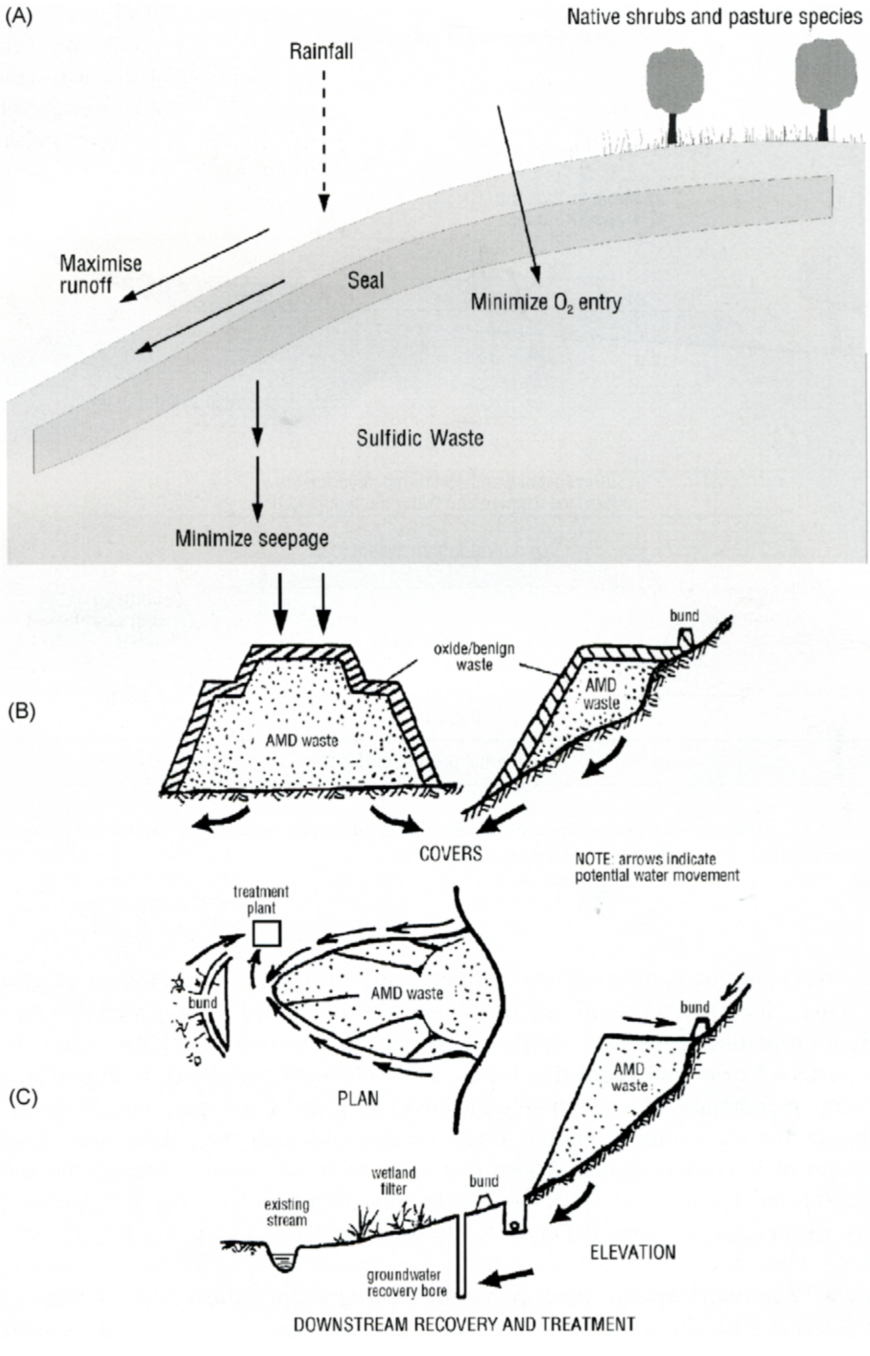


Figure VI.6.10. Methods of ARD control in waste rock dumps using subsurface air- and water-tight seal (A), surface cover with oxide/benign waste (B) or downstream collection and treatment of water (C) (after Environment Australia, 1997).

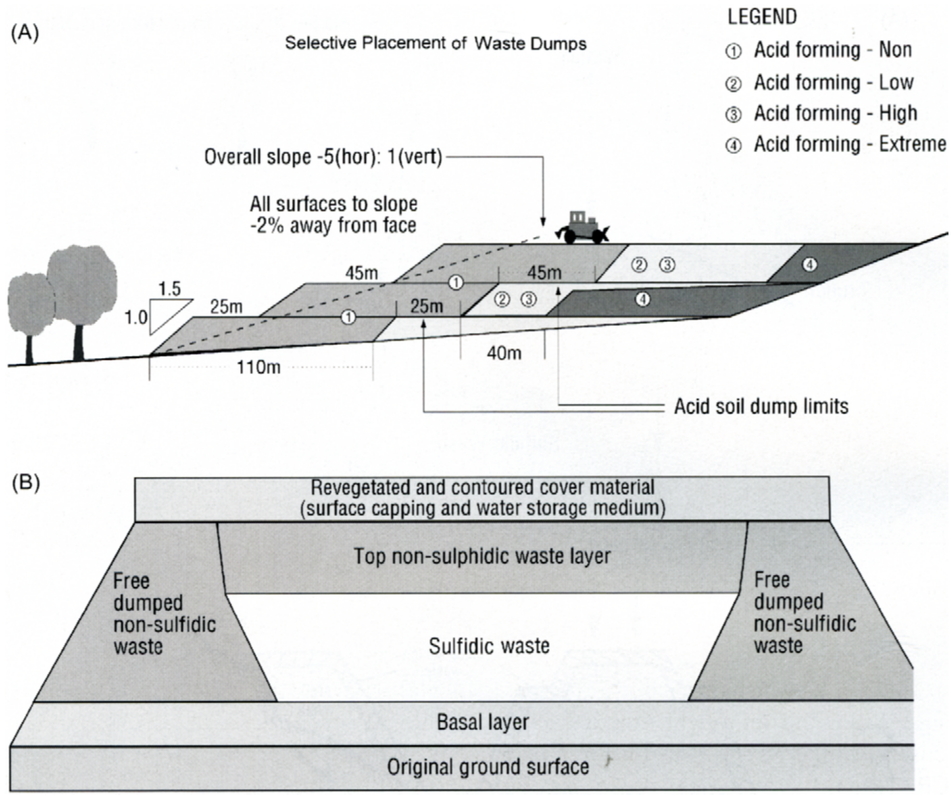


Figure VI.6.11. Formation of dumps with use of double (A) or single (B) insulation of ARD waste with non- and low-sulfur material (after Environment Australia, 1997).

covers may comprise compacted low-sulfide waste rock, oxide wastes, soil cover or multi-layer cover incorporating soil layers (sometimes combined with geomembranes) of different properties acting as a capillary break, water retention base layer, water storage zone, surface barrier and vegetation layers, as is schematically shown in Figure VI.6.13. Synthetic membranes, despite high efficiency in water exclusion, are generally not applicable for high-volume mining waste sealing although they have been used for protection of low-grade ore stockpiles and other high-risk wastes. Among the specific measures is the development of bactericides to prevent the catalytic role of *T. ferrooxidans* mainly with respect to high ARD generating waste at pH < 4.

VI.6.4.4. The rehabilitation strategy for the mining waste dumps in the USA

VI.6.4.4.1. The routine solutions

The routine strategy of mining waste management in the USA is based on a similar, but much wider general approach than ARD control and is focused on attenuation of chemical

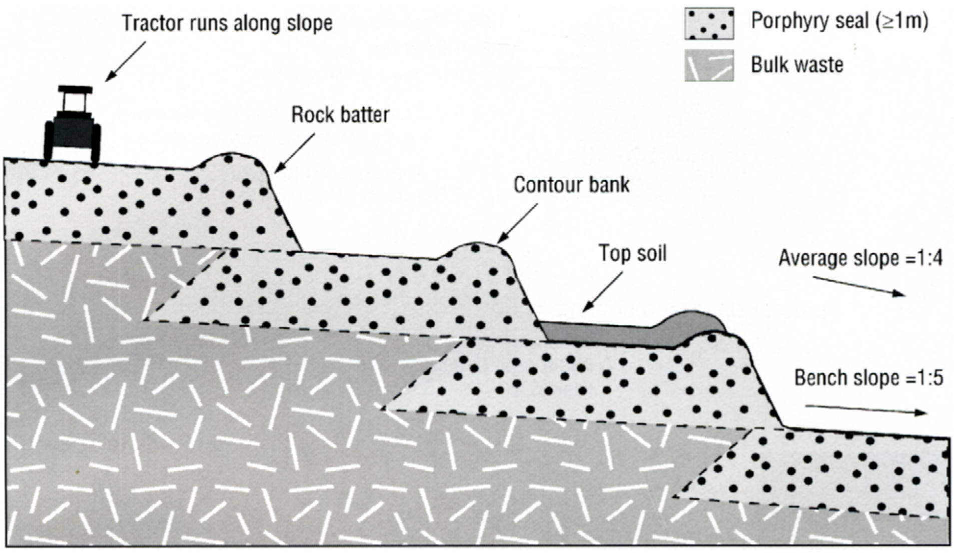


Figure VI.6.12. Detail of construction of dump batter (after Environment Australia, 1997).

constituents, as well as controlling, besides leachate, also wind and surfacewater erosion. These issues are thoroughly discussed by Hutchinson and Ellison (1992). The study focuses mainly on the safe disposal of non-hazardous mining waste regulated under RCRA, Subtitle D, but many of the aspects are applicable also to hazardous waste disposal under RCRA, Subtitle C. This study, covering a wide range of subjects concerning environmentally safe management of mining wastes, is strongly recommended as a valuable review of modern mine waste management units and requirements.

The major assumption is the disequilibration and change of mining waste properties with time. This is in agreement with the authors' approach to this material, presented in Chapter III.6 and in earlier publications (e.g. Twardowska et al., 1988; Twardowska and Szczepańska, 1990; Szczepańska and Twardowska, 1999). The geochemical changes are divided into four major stages, different for various waste materials and involving: (i) a mining process-controlled stage; (ii) acid generation-controlled stage for ARD wastes; (iii) re-solution phase for non-acid generating waste; and (iv) long-term degradation of more resistant minerals.

Attenuation of contaminant migration is thus also strongly dependent on the waste and site characteristics; its required extent is determined by federal and state regulations and groundwater quality standards with respect to the different potentially mobile chemicals of concern (macro- and trace metal and non-metallic species/complexes, asbestos, metal cations, major anions, and cyanides). In general, four types of attenuation mechanisms are considered: (i) physical (filtration, dispersion, dilution, and volatilization); (ii) physiochemical (adsorption and fixation); (iii) chemical (precipitation, hydrolysis, complexation, oxidation, and reduction); and (iv) biological (biodegradation, bacterial consumption, and cellular uptake by plants). Additional factors of importance for evaluating the applicability of the particular attenuation method on a site-specific basis

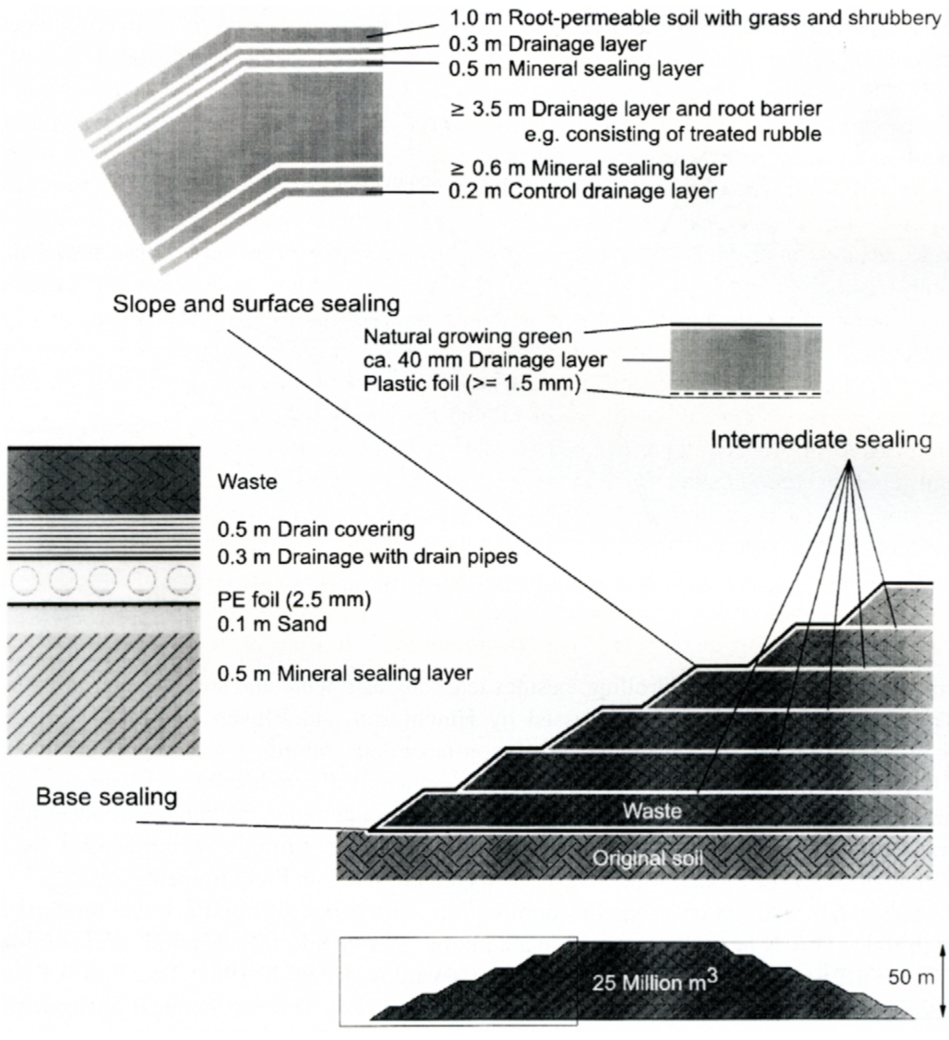


Figure VI.6.13. Schematics of insulation strategies (after Environment Australia, 1997).

include climatic and vadose zone conditions, attenuation capacity and waste unit management practices. The last factor incorporates direct and indirect activities aimed at improving the potential for attenuation, e.g. providing a lining with good attenuation properties; combining wastes with appropriately different chemical or physical properties, or locating certain wastes within specific areas having conditions adequate for attenuation to occur (direct activities). Indirect activities may include, e.g. controlling of the amount of liquid contacting the waste.

The prerequisite for selecting an optimum method for the attenuation of chemical constituents, is an application of appropriate level of verification, based on technology, prior experience and testing, as well as modeling and field demonstration to determine the attenuation capacity and verification of the predicted behavior of the chemicals of interest.

Much attention in mine waste management is paid to the bottom liner system design and closure requirements. Due to an unfavorable location of some mine waste disposal facilities close to valuable surface or groundwater resources, the liner system involving material of low hydraulic conductivity could be required to avoid an unacceptable threat to groundwater. Considering the different levels of hazard presented by different wastes, a wide variation in physical and chemical properties, as well as often very large sizes of the mine waste disposal units and long period of their development, some lining systems can be economically or technically unfeasible. As a result of the lack of natural soils of low hydraulic conductivity close to the disposal sites, the geosynthetic material application for liners in North America is increasing dramatically. It should be, though, mentioned that due to specific geological conditions of mining and location of waste dumps in areas of progressive subsidence, the efficiency of bottom liners use for large area- and volume-units could be problematic. Much more efficient and cost-effective management practices are aimed to risk reduction. In the USA, such practices comprise mainly bottom non-liner barriers, such as placement of under-drains, cut-off walls, sub-aqueous deposition, etc.

The authors' approach is focused rather on preventing generation of pollutants and their transport to groundwater, i.e. reduction of an exposed surface and disposed waste volume subjected to leaching, and limitation of operation time before the permanent closure of the dump section, along with the construction of durable surface facilities for long-term water protection.

With respect to the life-cycle protection of the aquatic environment, closure requirements for controlling the seepage migration to the receiving waters, as well as the wind and surfacewater erosion, are of particular importance. In the North American practice, the emphasis is put on "passive-care" approaches such as the provision of a durable long-life cover requiring minimum maintenance. "Active-care" approaches, e.g. collection and chemical treatment of contaminated water are considered unfavorable, and should be replaced finally by passive-care activities. The technologies and design elements, which are recommended for different solid mining waste types, comprise, in general, a similar set of alternatives, as discussed earlier. A choice of action depends upon the life-cycle evaluation of threat to the beneficial use of surface or groundwater. It may thus include several options, such as: (i) "no action"; (ii) institutional controls; (iii) conditioning or treatment (physical, chemical, biological); (iv) encapsulation (surface or subsurface); and (iv) waste removal for off-site disposal. Of these technologies, effective encapsulation appears to be the most appropriate passive-care approach.

The key routine elements of this technology include cover layers (single or multi-layered: top, drainage, capillary break), barrier and special layers, depending upon the threat posed to the environment (EPA, 1989). The EPA basic waste management unit requirements are incorporated in a vast list of guidance documents, applicable to mining wastes (EPA, 1979, 1982, 1983a,b, 1985, 1987a,b, 1989). The manner in which various types of wastes are managed depends upon the type and environmental behavior of the particular material. The detailed discussion of these requirements, as has been pointed out at the beginning of this chapter, is provided by the comprehensive study of Hutchinson and Ellison (1992).

VI.6.4.4.2. Novel mine waste technologies

Sulfidic mine wastes have been found to have severe detrimental impacts on the environment and ecosystems due to combination of acidity, heavy metals and sediments in the West of the USA (see Chapter VI.6). The technologies for ARD control being currently under testing in the USA within EPA/DOE Mine Waste Technology Program (2000) consider the following priority areas (Wilmoth, 2000):

- Source controls, including *in situ* treatments and predictive techniques to provide a permanent long-term solution.
- Treatment technologies for providing immediate (short-term) alleviation of the most severe environmental problems.
- Resource recovery (heavy metal extraction) from mining wastes in order to help offset remedial costs.

The at-source controls technologies of the first priority consider use of sulfate-reducing bacteria (SRB) method; biocyanide oxidation for heap leach piles; transport control/pathway interruption techniques, including infiltration control, sealing, grouting, and plugging by ultramicrobiological systems.

SRB: among these technologies, particular attention has been paid to use anaerobic SRB to significantly retard or prevent acid generation at affected mining sites. With respect to ARD generated in mining waste piles, it can be used to reduce the contamination of acid high-metal leachate in three ways: (1) dissolved sulfate is reduced to hydrogen sulfide through metabolic action by the SRB; (2) the hydrogen sulfide reacts with dissolved metals forming insoluble metal sulfides; and (3) the SRB metabolism of the added organic nutrient produces bicarbonate that increases pH of the solution and thus limits further metal dissolution.

In the SRB bioreactors constructed at the Calliope abandoned mine site in order to treat the metal-rich leachate with pH 2.6 from waste rock pile within EPA/DOE Program (2000) (Fig. VI.6.14A,B), a combination of organic carbon (cow manure), cobbles and crushed limestone was used as a fill to provide nutrient and stable substrate, and to adjust pH for bacterial growth. For the metal concentrations present in the ARD in the site, at SRB population from above 10^3 to 10^6 cells/ml and residence time of ARD for 4.5–5.5 days in the reactors, which were run from 1998 to 2001 throughout these years, the metals were removed to threshold levels 800 μg Zn/l, 80 μg Cu/l, and 5 μg Cd/l.

The successful application of SRB in a field scale for ARD and AMD remediation under EPA/DOE program (2000) has given rise to studies and applications of this promising and efficient technology in various projects on eliminating ARD and AMD (Diels et al., 2002; Ibeanusi and Archibold, 2002; Ngwenya et al., 2002; Zaluski et al., 2002) also with biorecovery of metals (Tabak and Govind, 2002). The obtained efficiencies of metal removal in different projects and a diverse scale of application are within the range reported above. Though most of the SRB projects are allocated in the USA, this technology becomes increasingly popular also in Europe, where several research centers are currently involved in SRB studies and upscaling of applications (Johnson and Hallberg, 2002; Piet et al., 2002; Geller et al., 2002).

The recent experiments on using an integrated mixed metal-tolerant microbial system to enhance the removal of multiple metals from coal pile run-off and their subsequent

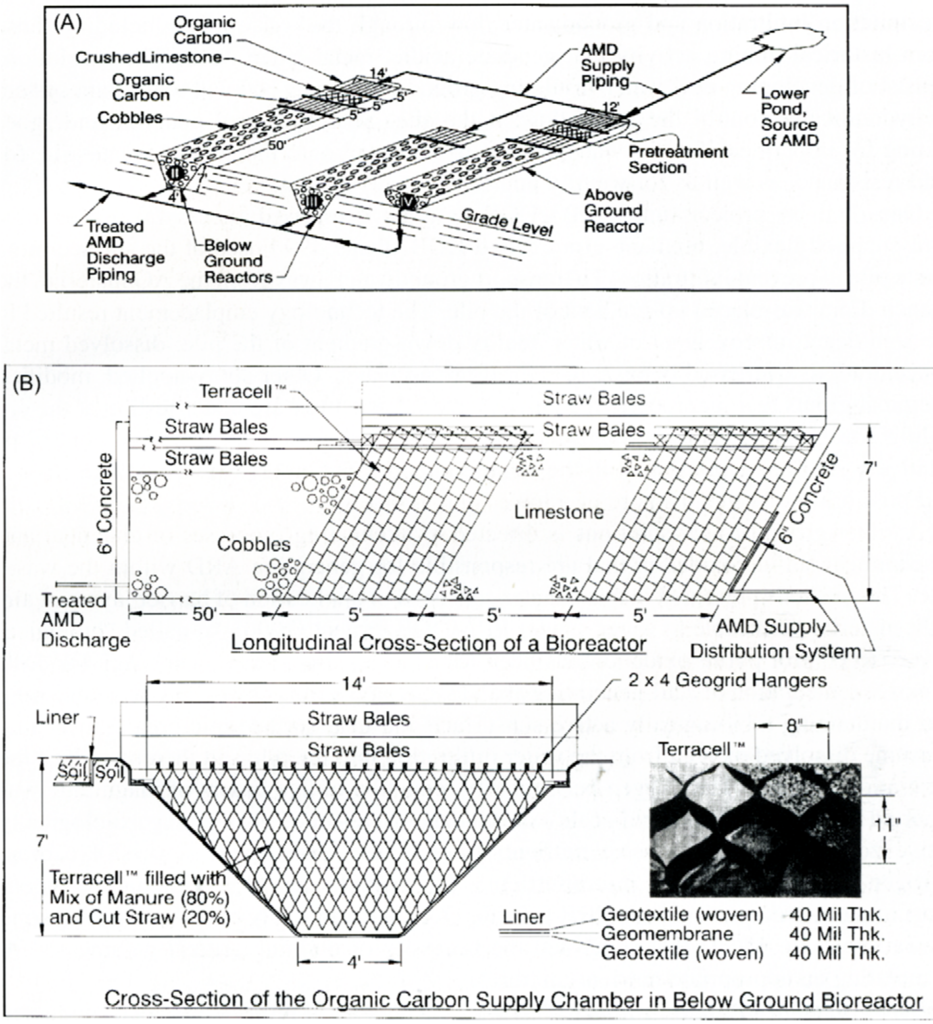


Figure VI.6.14. Layout (A) and design (B) of sulfate-reducing bacteria (SRB) bioreactors in Calliope Mine, Montana, USA (after EPA/DOE, 2000).

recovery in the bacterial biomass showed a high removal range of 83–100% and a recovery of 38–58% of Al, As, Cd, Cr, Cu, Fe, Pb, Se and Zn occurring in concentrations typical for coal pile leachate (Ni showed lower recovery rate of 15%) (Ibeanusi et al., 2003). The reported data prove high effectiveness of the mixed microbial system in metal removal and recovery process from ARD and demonstrates possibility of the prospective wide use in this field.

Source control technologies are in compliance with the methods described earlier as "rendering the dump air- and water-tight". They are focused on the new technical solutions for tasks well known for a long time, one of them is elimination or reduction of precipitation infiltration and groundwater flow through the waste piles, including those from historical mining activity that generate acidic, metal-laden ARD. These solutions consist of developing new water-insulating grouting materials coupled with drain system to hydraulically control the water flow at the site. Source control material undergoes testing for impermeability for water once it is emplaced onto the surface waste pile, for acid resistance, as well as for wet/dry and freeze/thaw cycling.

One of these projects under EPA/DOE program comprised successful testing of a spray-applied flexible, urethane grout called KOBAthane 4990 used at the surface mine waste pile to prevent infiltration. To transport groundwater away from the AG material, the French drain was placed up-gradient of the pile. The technology emplacement resulted in the significant improvement of water quality down-gradient of the pile: dissolved metal concentrations decreased below MCL for drinking water. Other spray-applied, modified chemical grouts that incorporate the tailing material as a filter are under testing at the ore tailing site. This technology might be efficient and cost-effective for source control of relatively small size piles sited in the remote unpopulated areas, where aesthetic values and possibility of vandalism are of a lower importance.

Another group of new solutions is directed to attenuating processes of chemical and biochemical sulfide oxidation that are responsible for generating ARD within the waste pile. The methods presented in this chapter consisted of application of barrier layers of air-tight material permeable to water. Under EPA/DOE Project (2000), so-called "biological cover" to control pyrite oxidation has been tested as an innovative technology for cost-effective remediation of acid-generating abandoned mine tailings by means of establishing and maintaining a biologically active subsurface and near-surface microbial barrier that consume dissolved oxygen from the water infiltration into the pile and thereby reduce the generation of ARD. The oxygen permeability reduction of five orders of magnitude was reported in laboratory- and field-scale experiments as a result of establishing a biologically active zone by adding low-cost nutrient solution that served as a source of carbon, nitrogen, phosphorus, and micronutrients for stimulation of indigenous oxygen-consuming microorganisms and SRB growth. Overall, further laboratory and field testing showed unstable results that suggested the necessity of defining critical parameters for formulating an appropriate nutrient mixture.

Heavy metal in situ stabilization technologies: a number of projects has been conducted under EPA/DOE Mine Waste Technology Program in order to identify technologies for *in situ* treatment/stabilization of particularly problematic toxic metals such as mercury and lead as a cost-effective alternative to excavation and removal of hazardous mine waste and metal-contaminated soils to a waste repository; e.g. for lead, the remedial approach consists of phosphate stabilization of mine waste contaminated soils by mixing commercial grade phosphoric acid and a trace of KCl into the soil followed by liming for pH adjustment that results in conversion of lead into a highly insoluble pyromorphite.

ARD treatment: several technologies are being focused on development of effective ARD/AMD treatment systems for removing toxic, dissolved metallic and anionic constituents from the leachate *in situ* and increasing the pH of effluents to near neutral values using biological and/or chemical treatment processes. Integrated passive biological reactor utilizes both SRB (anaerobic treatment) and aerobic bacteria (aerobic reactor) in a series of biological processes for the complete mitigation of ARD with precipitation of metal ions as insoluble sulfides (most metals) or oxides (Fe, Mn).

For removal of selenium from Se-bearing leachate to the level of 50 μg/l (ppb) under the US National Primary Drinking Water Regulation Limit, four technologies have been successfully tested: (i) best demonstrated available technology (BDAT) for coprecipitation of selenium using ferryhydrite; (ii) catalyzed cementation of Se by adsorption onto iron surface regardless of its valence state (Se^{4+} or Se^{6+}); (iii) biological Se reduction to elemental selenium by specially developed biofilms containing specific microorganisms using baffled anaerobic solids bed reactors (BASBR); and (iv) enzymatic reduction of selenium based on proprietary enzyme extraction/purification method combined with immobilization/encapsulation techniques that keep the selenium reducing enzymes in a functional arrangement within an immobilized/encapsulated matrix. All these technologies removed Se below the MCL, of them BASBR appeared to be the most consistent process tested, with the majority of results below the detection limit 2 μg Se/l (ppb).

Biological destruction of weak acid dissociable (WAD) cyanide occurring in cyanide solution heap leaching of sulfide ores of precious metals in concentrations 500–600 mg CN/l along with other contaminants such as As, Cu, Hg, Ag, Zn was found to be an efficient process for cyanide reduction to <2 mg CN/l. The indigenous aerobic and anaerobic organisms capable of effectively degrading cyanides were isolated during the bioaugmentation phase, and used in the subsequent aerobic and anaerobic reactors, and in final aerobic step. Cyanide and heavy metals were substantially removed and pH consistently neutralized in a more cost-effective way than in conventional processes. The remediation of metal-complexed cyanide has been also investigated using several photolytic methods (direct photolysis and homogenous photolysis) to enhance naturally occurring remediation processes.

To remove from ARD/AMD trivalent arsenic As^{3+} that has been reported to be more toxic than As^{5+} forms and much more difficult to remove from solution, photochemical oxidation process was used effectively for conversion of As^{3+} to As^{5+}. Removing As^{5+} was next accomplished using adsorption onto ferric iron (U.S. EPA accepted method). Other arsenic removal technologies were tested to reduce its concentration from approximately 500 to <50 μg/l that comprised: (i) mineral-like precipitation by substituting arsenate into an apatite structure; (ii) aluminum oxide adsorption; and (iii) ferryhydrite adsorption (BDAT technology). All three technologies showed favorable results.

For thallium removal from ARD/AMD to levels of <1.7 μg/l, two technologies are considered: (i) adsorption onto manganese dioxide (available as a waste from zinc electrowinning process) and (ii) reductive cementation of thallium with use of elemental iron (available in scrap form).

Under Mine Waste Technology Program, an extensive search to evaluate innovative nitrate removal technologies was undertaken. Of the 20 technologies screened, 3 were selected as the most promising economically and environmentally: (i) ion exchange with nitrate-selective resin; (ii) biological denitrification; and (iii) electrochemical ion exchange (EIX).

At present, numerous technologies are available for remediation of ARD. These technologies include biosorption, mineral/resin adsorption, chemical precipitation (e.g. lime precipitation), ion exchange, freeze crystallization, evaporation and many others. The status of development and application of these techniques can be followed in the websites of U.S. Geological Survey (USGS), U.S. EPA and some other sources given as a reference material for further information at the end of this chapter. Remediation methods are usually site-specific, thus both conventional and novel processes have to be carefully analyzed with respect to feasibility and applicability, and possibly modified to meet the requirements.

V1.6.4.5. Other rehabilitation technologies for the mining waste dumps

A potential for ARD/AMD attenuation and remediation show also other recent studies carried out in different countries.

An interesting technology of using steel manufacturing by-products incorporated into the funnel-and-gate system, which is one of the applications of permeable reactive barriers (Powell et al., 1998), for controlling mine tailing leachate with high As concentration has been proposed by Korean authors (Ahn et al., 2003). Besides elemental iron, these materials contain various compositions of Fe oxides and Ca–Fe oxides that are adsorption sites for both As^{5+} and As^{3+}, while Ca hydroxides can also neutralize acidic leachate and promote precipitation of dissolved heavy metals. Of the tested material, evaporation cooler dust (ECD) was found to be the most efficient material to remove As and dissolved metals, and to increase pH; oxygen gas sludge (OGS) and basic oxygen furnace slag (BFOS) also showed high efficiency. This technology is particularly attractive due to utilization of low-cost materials as permeable reactive barrier media in mine tailing containments.

Other investigations propose a galvanic suppression technique of pyrite oxidation and restricting further oxidation of Fe^{2+} by molecular oxygen using minerals with lower rest potentials (E^0), and *in situ* precipitation of calcite for ARD/AMD remediation (Noecker et al., 2003). The mineral with a lower potential acts as the anode (oxidation and dissolution of that mineral) and the mineral with the higher potential acting as the cathode is protected from dissolution (Holmes and Grundwell, 1995). Preliminary results showed that metallic Cr(c) in particular, and also Zn(c) and Al(c), all effectively reduced the oxidation of pyrite in aqueous solutions, while the reaction of CaO and pyrite produced a calcite precipitation when CO_2 was added. The authors have suggested that mobility of As, Se, Cu and Pb in ARD/AMD can be effectively reduced by precipitation of calcite, although current *in situ* immobilization techniques with use of FA and limestone as sorbents require extensive subsurface disturbances. At the reported stage, the feasibility of both methods, i.e. of galvanic suppression and *in situ* precipitation of calcite for AMD remediation seems to be low: introducing environmentally problematic metals or FA into acidic leachate/drainage in real systems might rather create more new hazards than positive effects, while water cover of sulfidic waste in natural conditions is in itself an effective protection against pyrite oxidation.

A substantial sorption capacity of different organic materials, including organogenic waste, for metals and metalloids has been reported by many authors (Shuman, 1999; Madejon et al., 2003; Twardowska and Kyziol, 2003; Twardowska et al., 2003). Madejon et al. (2003) found out that the affinity of metals and As for binding onto municipal waste compost, leonardite and forest litter followed the order Pb $>$ Cd $\geq$ Cu $>$ Mn $\geq$ Zn $>$ As. Of these materials, municipal waste compost appeared to show the highest sorption capacity for heavy metals, while binding of As directly related to the humic acid (HA)/fulvic acid FA ratio and was the highest for leonardite. The formation of soluble metal–organic chelates was low. These properties suggest the potential of using organogenic waste and natural material in permeable reactive barriers for metal removal from acidic leachate in mining waste dumps.

The above studies exemplify some research directions in seeking new opportunities for application of low-cost waste materials as metal sorbents from ARD.

V1.6.5. Landscape formation and land use in a dump site

The problem, which requires attention, is a need for the change of the primary landscape and land use in the dumping site. It arises on one hand from the severe land shortage in many mining areas, and on the other hand from the basic requirement of a maximum, environmentally safe use of the site area for waste disposal. As a result, high dumps are being constructed in a primarily flat area. Therefore, a new landscape and new way of land use is formed, which has to fulfill the above requirements, and not to be the aim by itself. Not always is a correct order of priorities followed. An example of an interesting, but not rational solution is the Paciorkowiec coal mining dump (Piast colliery, USCB, Poland), where the landscape became a superior target (Fig. VI.6.15). Forming the dump surface in the shape of free standing conic tips, creates high development of a surface vs. volume,

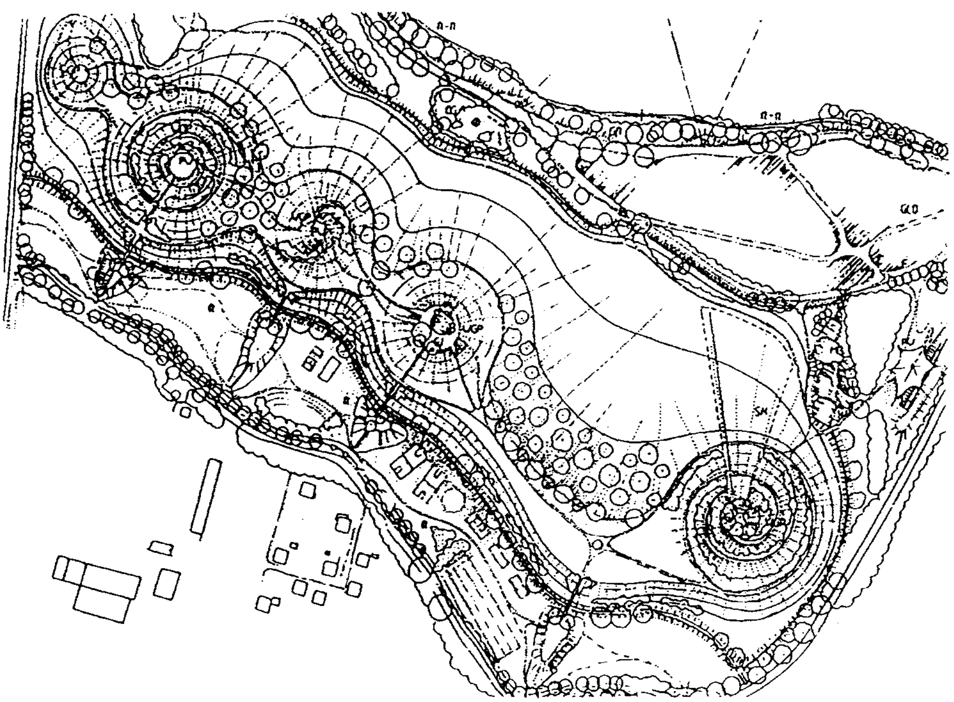


Figure VI.6.15. Paciorkowiec coal mining waste dump (Piast colliery, USCB, Poland). Construction of the dump as a recreation area with toboggan slides (architects: Bogdanowski, J.; Myczkowski, Z.).

reduces the disposal area, increases the threat of self-ignition and thus compels high-cost tips formed in thin heavily compacted layers. This solution, though, shows that the dumps can become a part of a new landscape and be used as a recreation and sports grounds, and be visually attractive and accepted by a local community. Many later landscape architecture solutions in mining waste sites designed as recreational areas avoid such extravagances (e.g. Debiensko or Bukow sites, USCB, Poland).

An unquestionably good example is an adaptation of the coal mining Janina dump as a recreation and park area (USCB, Poland). The construction of the dump was completed in 1995; its volume is 11.5 million m^3, average height 22 m, surface area 62 ha, enlarged for further 15 ha. The table-shape of a dump creates convenient conditions for disposal of high-volume wastes, is easily maintained and is suitable for siting of sports facilities (bicycle paths, tennis courts, picnic areas). The recreation and sport creates a good chance for rational and pro-ecological use of mining waste dumps, which fulfill the basic goal, i.e. waste rock disposal.

V1.6.6. Biological rehabilitation: concepts, solutions, and aims

An important role in mining waste dumps management belongs to the vegetation cover of a dump surface. For a long time this element of mine waste management was a synonym of a term "reclamation". The perception of the dump top layer as a place for growth and endurance of the introduced vegetation is still widely prevalent among specialists from this field. They usually do not consider that vegetation has to fulfill similar tasks, as other elements of rehabilitation, i.e. environmentally safe solid waste disposal, along with prevention of wind and water erosion of a dump surface. In a dump construction method, which starts from the final formation of batters and its slopes and next filling the internal part of the dump, the role of vegetation cover is particularly important both as an element of environment protection and landscape planning. Here, though, inconsistency of approaches and laws between European countries and the USA exists. The biological reclamation guidelines in Poland follow the German reclamation laws, where most frequently aforestation is the ultimate goal. Due to the lack of adequate knowledge, the complex approach to environmentally safe mine waste management and the ground and surfacewater protection as the first priority was not considered when these laws were developed, mainly by the traditional agronomists and forest specialists. The outdated guidelines, unfortunately, often correspond to the requirement of restoring primary use and high productivity of dumping areas, and are literally treated by the regional ecological administration.

In order to obtain rapid vegetative cover on mine waste, planting of trees at close spacing is still practiced and prescribed in Germany and Poland (Hutnik and Davis, 1978; Harabin and Strzyszcz, 1993; Neumann-Malkau, 1993). The emphasis on an instant exterior effect occurs also in the Czech Republic, and even in the UK, where up to the last decade of 20th century herbaceous cover had been preferred. At the same time, a number of authors report failures in establishment of trees, severe plant losses and their weak growth at the reclaimed mining waste dumps due to various reasons, such as adverse physical and chemical properties of waste material as a substrate for plant growth, low nutrient availability, unstable composition, coarse grain size, inappropriate water balance and waste acidification (Hutnik and Davis, 1978; Kerth, 1988; Kerth and Wiggering, 1990; Neumann-Malkau, 1993). This disqualifies dumping sites as areas of productive wood resources. The satisfactory insulation role of the top encapsulation against water and air penetration into the dump, as well as against water and wind erosion, and its cost-effectiveness, requires minimization of the vegetative layer thickness, which is not suitable for the root system of high plants. In general, topsoil cover for vegetation support is not available; therefore the plants have to be introduced directly on the waste material, which in the exposed, not heavily compacted top layer is particularly susceptible to adverse transformations and acidification. This suggests an application of herbaceous cover, in particular for support of natural processes of soil formation, prevention of erosion and reduction of water infiltration (through evapotranspiration). In biological reclamation practice, a parallel application of sodding and trees planting in the same area is quite frequent, which creates an inappropriate competition. The natural invasion and succession of high pioneer species, which could damage the drainage or the infiltration barrier, should be taken into account in the design and construction of cover layers. The designed vegetation should thus consider the prospective status with respect to the environmental safety and fire control requirements (for coal mining waste) towards encapsulation tightness for water and air. Unsealing of the dump cover by the deep penetrating root system threatens also the plant due to easy access of reagents, i.e. air and water, for acid generation and self-ignition of coal mining waste as a result of the exothermic process of sulfide oxidation. Some proposed methods either reduce the risk of self-ignition of coal mining waste (e.g. compaction in squares) or vegetative layer acidification by addition of buffering materials, but do not control infiltration and acidification of the deeper waste layers. In each case, the selection of plant species should be waste-, site- and use-specific and consider also both negative and positive role of the naturally invading plant species. They are generally in much better shape, than varieties planted with or without use of topsoil. In Polish climatic conditions the natural invaders comprise *Betula* sp., *Populus tremula* L., *Salix* sp. and *Robinia* sp., which effectively compete with planted species (Patrzalek et al., 1993). The present state of reclamation practice shows that stress on returning the land to a high rate of productivity and the primary mode of use is generally unrealistic, expensive and often environmentally unsafe.

The biological reclamation regulations and practice in the United States are, to a much greater degree, incorporated into the complex mine waste management activities focused on minimization of the damage to the environment. The restoration of high productivity or a primary use is not a priority. Vegetation is considered as a component of the top layer of the surface encapsulation; in the inappropriate climate conditions the alternative can be an armoring layer of gravel-size material.

The EPA provides information concerning plant species, cultivation and areas of adaptation (EPA, 1983c) that are still in force. The general requirements reflect the approach to tasks and the function of the introduced vegetation (Hutchinson and Ellison, 1992):

- perennial, locally adapted and resistant to the unfavorable conditions (temperature extremes, low-nutrient soil, little or no maintenance),
- of sufficient density to minimize cover erosion,
- with the shallow root system that will not disrupt the drainage or the infiltration barrier.

For this reason, the use of tall plants (shrubs or trees) is not recommended due to deep penetration of a root system and a threat of drainage and barrier layer damage. In contrast to planting trees at close spacing practiced in Germany and still being in wide use in Poland, the perennial herbaceous cover, sometimes in combination with wider spacing of trees where applicable, is a common practice in mining waste dumping sites in the USA. It should be added, that also in Poland, the establishing herbaceous cover at the dump surface (predominantly without topsoil) as a cost-effective, efficient and environmentally sound solution gradually became a common dump re-vegetation practice, though still not without problems of a different kind.

V1.6.7. Monitoring strategies

Monitoring is an essential part of management of mining waste units. As ground and surfacewater are the most endangered compartment of the environment in the area of dumping sites, every mining waste dump or other industrial waste disposal facility, in particular the dumps sited in the area of the major groundwater basins (MGWB) and usable horizons of groundwater (UHGW) and in the river valleys, should have an adequate groundwater and/or surfacewater life-cycle monitoring program. Such programs should be capable of early detection of the threat to the beneficial use of water resources and provide documentation about the extent of the threat before it actually occurs. This concept provides an early alert for taking remedial actions, which greatly reduces the potential for loss of recoverable water resources. Besides water, other compartments of the environment should also be included in the monitoring program, if a proven risk occurs.

Essential components of the aquatic environment monitoring program are: (i) background studies to identify the primary environmental parameters and define the environmental values to be protected in the dumping site area, in particular water resources. The results of the studies serve as reference values in case of newly established dumping sites; (ii) the vadose zone life-cycle monitoring/screening to provide early means to detect a risk, and subsequently, undertake a remedial action before the contaminants degrade the recoverable water resources; and (iii) the saturated zone life-cycle monitoring up-gradient and down-gradient of the dump to determine site impacts, validate short- and long-term prognosis and effectiveness of prevention/remediation strategies in the dumping site. The scope of parameters to be incorporated into the monitoring program should be based: (i) on the waste characteristics determining potential for the receiving water deterioration from macro- and trace component generation and release from the disposed mining waste and (ii) the pathways of the constituents generation, interaction and migration in the anthropogenic (dump) and natural (bedrock) vadose and saturated zones in the actual hydrogeological and hydrologic conditions.

The guidelines for the design and operation of the monitoring systems for the vadose and saturated zones presented in comprehensive handbooks, are recommended for further reading (EPA, 1986; Hutchinson and Ellison, 1992; Sara, 1994; Wilson, 1995; Wilson et al., 1995; Looney and Falta, 2000; Nielsen, 2000; Boulding and Ginn, 2003). These books discuss basic principles of vadose and saturated zone hydrology, prevailing monitoring techniques and installation of monitoring devices, as well as operational and analytical details. The design of direct (sampling) and indirect (non-sampling) monitoring systems specific for mine waste management units, including in-waste, vadose zone and groundwater monitoring is discussed in the guidelines edited by Hutchinson and Ellison (1992). The monitoring issues addressed there focus on locating monitoring points in the monitored media, selecting monitoring equipment and determining monitoring parameters required to provide an early alert for assessment of threat and taking remedial actions, along with limitations of the methods. The prevalent methods are focused on obtaining a profile of soil moisture content, suction and other details of water balance and transport within the dump. A limitation of the in-waste and natural vadose zone monitoring is a high probability of false-negative readings in all point- and non-point sampling methods. False-positive readings are evaluated as of a low probability, or even unlikely (for direct point sampling methods). Also field sampling, which entails drilling for core samples and subjecting them to laboratory tests to determine density, void ratio, soil moisture content and soil moisture suction is considered as an alternative. To determine chemistry of seepage, batch or column leach laboratory testing methods are recommended.

The authors of this chapter (Twardowska et al., 1988; Twardowska and Szczepańska, 1990; Szczepańska and Twardowska, 1999) have, for a more than decade, routinely used drilling and core sampling along the dump for screening the water balance, and also for pressure extraction (under nitrogen) of pore solution from the core. The pore solution is analyzed for its chemical composition by ICP-OES or ICP-MS. For the high-volume non-hazardous waste dumping sites, where no surface synthetic liners are used, we regard this method as highly reliable and informative. It provides direct data on the vertical redistribution of contaminant loads in the vadose zone and transformations of chemical composition of pore solution vs. water exchange rate in the waste layers. Several examples of basic hydrochemical profiles of pore solutions at different characteristic coal mining dumping sites have been presented and discussed in Chapter III.6. On the basis of own experience, the authors recommend this method for wide use for in-site screening (see also Chapter IV.5).

The application of commonly used biological test systems for the environmental monitoring of waters affected by the leachate from mining waste is not possible due to often high acidity and iron content, and lack of nutrients. Nevertheless, successful development of new bioassays based on organisms native to acidic mining lakes has been recently reported (Picki et al., 2003). This opens the prospects of extending biological testing on this specific kind of waste, provided that these organisms are adequately sensitive to typical pollutants occurring in ARD-affected waters.

With respect to the major parameters to be analyzed in sulfidic mining wastes, pH and sulfate concentrations in the pore solution are the basic indicators of acid generation, while concentrations of calcium and magnesium indicate buffering capacity of the material. Chloride balanced by sodium is the major indicator of the water-exchange process in the dump constructed from waste of high or moderate salinity. Aluminum is a common ion in the first stage of waste acidification, while silica along with re-appeared alkalis indicate deep acidification of the material. Of trace metals, iron, manganese and zinc are abundant components of the pore solution of poorly buffered waste. Zinc frequently becomes a macro component of ARD. Other compounds commonly present in acid pore solution of mining waste in higher concentrations include lead, copper, cadmium, arsenic and selenium, and other heavy metals specific for a mined metalliferous ore or coal. These compounds, in addition to pH, Eh and conductivity should be analyzed in a life-cycle monitoring of the vadose and saturated zone in the vicinity of a dump. The practice of monitoring shows that the highest pollution potential from the mining waste dump to the aquatic environment caused by acidification and high heavy metal release commonly lags behind the start of acid generation and sulfate release and occurs in the post-closure period. This confirms the requirement for life-cycle vadose zone and groundwater monitoring in the vicinity of the mining waste dump.

Formerly, this delayed adverse environmental impact was not taken into consideration in Polish mining areas, therefore monitoring terminated after the dump closure. The monitoring itself was rare, if any, in the vicinity of non-hazardous mining waste dumps, which were considered harmless on the basis of compound concentrations in 1:10 water extract from freshly generated waste. Currently, as a result of the growing consciousness of the delayed and non-linear increase of pollution potential from these facilities, the life-cycle monitoring becomes an obligatory component of the dumping site project and maintenance that comprises also 30 years' post-closure period (Directive of the Minister of Environment, 2002).

More detailed discussion of monitoring issues related to mining waste is addressed in Chapter IV.5.

V1.6.8. Public opinion

Due to the precondition of acceptance of a local community for getting a localization permit for a waste disposal site from the Environmental Departments of the district administration, the role of public opinion has a substantial impact on siting a project (Environment Protection Act, 2001). Thickly populated mining areas and severe shortage of places for waste landfilling brought about the necessity of dump location in close proximity to settlements and farms. Though the members of the local communities are traditionally connected with mining, the "not-in-my-backyard" syndrome makes requirements for siting, constructing and managing dumping areas more stringent. Public opinion is extremely sensitive to aesthetics and to the way a facility is used. In general, there is no interest and firm rejection by the former owners, and lack of potential new candidates for continuation of the primary production in the rehabilitated formerly agricultural land. The final shape of batter re-vegetated as a first stage with herbaceous cover and a target use of a disposal area for recreation or even as aesthetic barrens become the preferable way of high-volume mining waste disposal site management in Poland.

V1.6.9. Underground disposal and reuse

V1.6.9.1. Disposal strategies

In recent years, a shortage of available land for high-volume mining waste disposal, and its environmental burden, as well as high economic and social costs, have resulted in an increasing pressure of administration and public opinion against the surface methods of waste landfilling. Simultaneously, a concern about severe surface deformations in a thickly populated area caused by subsidence brings about the need of stowing underground mine workings. This promotes a long-known method of mining waste use for filling goafs, disused mine workings and abandoned mines as an ideal solution. The additional pro-environmental aspect of this method is a decrease of the land damage by quarrying natural material (sand, gravel) used for backfilling (stowing), besides an adequate reduction of mining waste disposal at the surface and control of subsidence. In the Polish environmental strategy, increase of stowing of underground workings with use of coal mining waste is considered as a priority for mining waste management in the USCB. The amount of mining waste used underground by the end of 1994 was estimated to be at least 100 Mt. The annual amount of mining waste utilized underground as an additive to the sand stowing in 65 collieries of the USCB in 1990, 1993 and 1994 was 4.2–5.1 Mt or ~23% of the total mining waste reuse and it did not show an increasing trend (State Inspectorate of Environmental Protection, 1995). In 1994–1995, a regress of this way of coal mining waste use occurred. The annual rate of mining waste use underground dropped to 3.1–3.8 Mt, which amounted to 12.1% of the total amount utilized. The rate of coal mining waste directed for stowing did not exceed 7.4% of the total amount generated by coal mining, while the amount of mining waste utilized at the surface was continuously growing (State Inspectorate of Environmental Protection, 1997). In 1998, the rate of mining waste utilization reached 69.5% of the total amount generated (Central Statistical Office, 1999), while in 2001 it was already 89.1%, of this coal mining waste were used in 91.0% (Central Statistical Office, 2002). The predominant field of mining waste use is in engineering construction as common fill, where these wastes are often even more vulnerable to the adverse weathering processes than in the dumping sites. The major reason for limited use of mining waste underground is that the coal companies responsible for the environmental strategy of mines, consider costs of mining waste use for backfilling of mine workings still too high compared to surface utilization and disposal (State Inspectorate of Environmental Protection, 1997). An additional technical reason for limited use of mining waste for backfilling is its relatively high compressibility compared to sand (Skarżyńska, 1995); therefore it can be used as an additive to sand in proportions up to 30–40% wt.

V1.6.9.2. Legislative and regulatory framework

In general, Polish regulations and the Geological and Mining Law of 1994 amended in 2001 do not restrict use, disposal and storage in mine workings of any waste that is not qualified as hazardous, if the environmental and technological requirements are fulfilled, a legal permit for use, disposal or storage is obtained and adequate fees for disposal and storage paid (waste use underground is free of charge). Technological requirements are defined in the Polish Standards PN-93/G-11010 "Mining. Materials for Hydraulic Filling. Specifications and Tests" (PKN, 1994). Environmental requirements, also with respect to the mining waste, comprise the compulsory EIA. Unfortunately, up to now there is no standard testing procedure for evaluating a potential risk to the environment during the reuse or disposal of waste materials. This results in a frequently simplified approach to the evaluation of waste leaching behavior, which is based on the analysis of water extracts according to compliance tests (EN 12457-1/2/3/4). An up-to-date approach to characterization of waste leaching behavior discussed in Chapter III.1 considers the use of a complex testing procedure, which much better reflects the short- and long-term risk from waste to the aquatic environment. After final development by CEN/TC 292 of a standard testing procedure for environmental risk assessment from granular wastes and its approval as a European Standard, it will have the status of a national standard without any alteration in the CEN members and affiliated countries, among them Poland. This should greatly improve the reliability of the risk evaluation, provided these Standards will be acknowledged to be obligatory for use by an adequate governmental Directive. At the present stage, the confidence in the environmental risk assessment from the disposed waste based on the water extract is highly problematic.

Mining waste disposal at the surface is covered in Poland by the Waste Act of 2001, and the relevant regulations.

According to the USA regulations, the mining waste backfilling falls under the US-EPA Underground Injection Control (UIC) Regulatory Program of 1981 and the SDWA — Safe Drinking Water Act of 1974.

At the European Union, there is no specific Mining Law or legislation on waste from mining and processing of minerals, thus all relevant directives and regulations on waste are applicable also to mining waste, while the Member States use their national legislation. Recent mining accidents in Spain (Aznalcòllar accident in 1998) and in Romania (Baia Mare accident in 2000) that endangered aquatic and terrestrial environment, have induced the European Commission to issue a document COM(2000)624 final that sets priority actions related to the safety of mines, management of mining waste and integrated pollution prevention and control. These actions comprise the following initiatives:

- Amend the Seveso II Directive to include mineral processing of ores and in particular, tailing ponds and dams used in connection with mineral processing of ores.
- Develop guidelines on management of mining waste covering the environmental issues as well as the best practices, which could prevent environmental damage during the waste management phase.
- Develop a best available techniques reference document (BREF) on waste management to reduce current pollution and to prevent or mitigate accidents in the mining sector.

On the basis of these initiatives, the EC Environment Directorate General started a public consultation process on a working document related to the management of waste resulting from prospecting, extraction, treatment and storage of mineral resources (EC DG ENV A2/LM, 2002).

More information on these initiatives can be found in the EU websites listed in the references. It should be added that though the EU plays a modest role in the global mineral mining industry, it still has 1872 inventoried mining sites, of that 347 for non-ferrous metal extraction, most of them in France, and 578 coal mines, mostly in Belgium (all closed) (BRGM, 2001). Despite the fact that only 917 sites, i.e. 49% are still under operation, closed mines and waste dumping sites often continue to generate contaminants.

The EU legislation relevant to mining and environmental aspects of these activities, and the regulations of several other countries with developed mining industry (Canada, the USA, Australia, Mexico, Malaysia) are referred in BRGM report (2001).

V1.6.9.3. Environmental implications

Proponents of utilization mining waste for mine backfilling, besides stressing the environmentally beneficial protection of the surface against damage due to the structural support of undermined areas and reduction of surface waste disposal, also stress on the returning of the rock material to its original environment. It should be taken into consideration that during mining and processing, rock material undergoes significant physical and chemical transformations (decrease of grain size, increase of the exposed surface and hydraulic conductivity, temporary exposure to oxic conditions and high humidity). These can result in a higher degree of decomposition of unstable minerals (e.g. sulfides) and enhance material susceptibility to contaminant leaching.

The results of studies reported by Levens and Boldt (1994), indeed showed definitely higher concentrations of almost all dissolved elements in leachate from lead–zinc mine waste backfill compared to the recharge water quality. Some of them occasionally or permanently exceeded MCL, set forth in the Safe Drinking Water Act (As, Fe, Mn, Pb, SO_4), but were lower than concentrations detected in the worst acid drainage of the mine. Concentrations of major ions (SO_4^{2-}, HCO_3^-, Ca^{2+}, Mg^{2+}) in leachate, in conjunction with near-neutral pH values (pH 6.89–7.79) gave evidence of sulfide oxidation and acid generation, buffered by carbonate dissolution. Due to buffering, only SO_4^{2-} concentrations (340.0–1140.6 mg/l) consistently exceeded the MCL values (250 mg/l), while contents of heavy metals were low. The authors assume, that after the mine closure when the backfilled stope is flooded, the rate of oxidation of sulfide minerals and associated mineral dissolution will be much lower. Therefore, the contamination potential of the backfill will be greatly reduced, though metals already contained in secondary minerals may be released after the backfill is submerged. The overall impact of this specific backfilled stope has been evaluated as small.

Another example referred to the dual beneficial effects of using colliery spoil rock paste to fill the collapsing limestone mines, which has given rise to a general subsidence or localized surface damage (Jarvis and Braithwaite, 1994).

These examples indeed prove the benefits from reuse of mining waste material for the bulk backfilling (stowing) of underground mine workings. At the same time they also show the need of the site-specific approach to the reuse of mining waste underground, to avoid any threat of contamination of the groundwater by the material, which might be geochemically unstable.

V1.6.10. Conclusions

The environmental implications and practices of mine waste disposal lead to the following general conclusions:

1. Mining waste disposal sites, due to the concentration of large volumes of geochemically unstable (mainly sulfidic) material in relatively limited areas, and usually a long period of construction, should be treated as a potential source of a long-term aquatic environment contamination, which may display non-linear, time-delayed maximum release of contaminants in the post-closure period. The dumping site design should thus be based on the long-term prognosis and ensure reliability and persistence

of environmental protection measures during construction and after closure with little or no maintenance for an adequately long period.
2. The design of waste disposal units should be waste-specific and consider also site-specific conditions, such as climate, hydrogeological conditions and site factors, defining the categorical design criteria and environmental protection measures.
3. The sulfidic waste dump design during construction and after closure should be focused on the prevention of air penetration and water infiltration through the dump and limitation of waste exposure to the atmosphere. The management practices of all kinds, from the methods of placement and schedule to the top layer re-vegetation have to be in concert and consider predominantly surface interception, a passive-care approach and use of adequate chemical or/and physical properties of the waste to reduce risk to the environment. Bottom liners and active-care approach should be applied only in exceptional situations.
4. Engineering constructions from mining waste exposed to the atmospheric conditions should be treated the same way as waste disposal units with respect to EIA, preventive measures and monitoring requirements.
5. Monitoring systems have to provide an early alert for taking remedial actions: the potential costs and degradation of recoverable water resources can be thus greatly reduced. One of the best direct sources of information on actual contaminant generation and transport is quantitative and qualitative analysis of pore solution along the dump and a bedrock profile (anthropogenic and natural vadose zone).
6. The reuse of mining waste for backfilling of underground mine workings provides a triple benefit of reducing surface waste disposal, protecting against deterioration of recoverable groundwater resources and conservation of the undermined land endangered by subsidence. Due to transformation of physiochemical properties of the rock material due to crushing and exposure for some time to the atmospheric conditions, even in the case of returning waste rock to the original excavation, the assessment of potential impact of backfilling material on the groundwater quality should precede its reuse.

References

Ahn, J.S., Chon, C., Moon, H., Kim, K., 2003. Arsenic removal using steel manufacturing byproducts as permeable reactive materials in mine tailings containment system, pp. 244–245. In: Cobran, G.R., Lepp, N. (Eds), Conference Proceedings 7th International Conference on the Biogeochemistry of Trace Elements, Uppsala, Sweden, 2003. Vol. 1. Scientific Programs II, SLU Service/Repro, Uppsala, Sweden, p. 287.

Anonymous, 1997. User Manual, KYSPILL v 2.0. A Groundwater Pollution Forecasting System, HydroScience Inc., Lexington, Kentucky, p. 63.

Anonymous, 1999. User's Manual, GMS Groundwater Modeling System, Boss International Inc., Madison, WI and Brigham Young University, Provo, UT, p. 610.

Appelo, C.A.J., Verweij, E., Schafer, H., 1998. A hydrogeochemical transport model for an oxidation experiment with pyrite/calcite/exchangers/organic matter containing sand. Appl. Geochem., 13, 257–268.

Bennett, M.W., Kempton, H.J., Maley, J.P., 1997. Applications of geological block models to environmental management. Fourth International Conference on Acid Rock Drainage, pp. 293–303, Vancouver.

Boulding, J.R., Ginn, J.S., 2003. Practical Handbook of Soil, Vadose Zone, and Ground Water Contamination. Assessment, Prevention, and Remediation, Lewis Publishers/CRC, Boca Raton, p. 664.

BRGM: Management of mining, quarrying and ore-processing waste in the European Union. Final report on the study made for EC DG Environment, 2001, p. 79, 7 Figs, 17 Tables, 7 annexes, 1 CD-ROM (Collected data). Website: http://europa.eu.int/comm/environment/waste/studies/mining/index.htm.

Bursey, G.G., Mahoney, J.J., Gale, J.E., Dignard, S.E., Napier, W., Reihm, D., Downing, B.W., 1997. Approach used to pit filling and pit lake chemistry on mine closure — Voisey Bay, Labrador. Fourth International Conference on Acid Rock Drainage, pp. 257–275, Vancouver.

Central Statistical Office, 1999. Environment 1999. Information and Statistical Papers, Central Statistical Office, Warsaw, pp. 510, in Polish.

Central Statistical Office, 2002. Environment 2002. Information and Statistical Papers, Central Statistical Office, Warsaw, pp. 501, in Polish.

Charlton, S.R., Macklin, C.L., Parkhurst, D.L., 1997. PHREEQCI—A Graphical User Interface for the Geochemical Computer Program PHREEQC. Lynn University, Lakewood, CO, 1997.

Charlton, S.R., Parkhurst, D.L., 2002. PhreeqcI — A Graphical User Interface for the Geochemical Model PHREEQC. U.S. Geological Survey Fact Sheet FS-031-02, April, p. 2.

Clement, T.P., 2001. A generalized analytical method for solving multi-species transport equations coupled with a first-order reaction network. Water Resour. Res., 37, 157–163.

Diels, I., Gemoets, J., Bastiaens, L., Hooybergs, L., Simons, Q., Vos, G., Vito Geets, J., 2002. *In situ* bioprecipitation of heavy metals in acid mine drainage or acid groundwaters: from feasibility testing to pilot scale. P765, pp. 309–309. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Directive of the Minister of Environment of 9th December, 2002 on the scope, time, methods and conditions of conducting monitoring of waste landfills. Dz.U. 220.1858.2002, in Polish.

Downing, B.W., Giroux, G., 1993. Estimation of a waste rock ARD block model for the Windy Craggy massive sulphide deposit, Northwestern British Columbia. Explor. Mining Geol., 2 (3), 203–215.

Downing, B.W., Giroux, G. 2000. ARD Waste Rock Block Modeling, website: http://www.enviromine.com/ard/introduction/BlockModel.htm.

Downing, B.W., Madeisky, H.E., 1996. Acid rock drainage study of the Voisey's Bay Ni–Cu–Co massive sulphide deposit, Newfoundland. International Conference on Acid Rock Drainage, May.

EC: Commission Decision 2000/532/EC of 3 May 2000 replacing Decision 94/3/EC establishing a list of wastes pursuant to Article 1(a) of Council Directive 75/442/EEC on waste and Council Decision 94/904/EC establishing a list of hazardous waste pursuant to Article1(4) of Council Directive 91/689/EEC on hazardous waste. OJ L 226 06.09.2000, p. 3.

EC: Communication from the Commission to the Council and European Parliament COM(2000)624 final of 23 October 2000 on safe operation in mining activities: a follow-up to recent mining accidents, 2000.

EC: Commission Decision 2001/118/EC of 16 January 2001 amending Decision 2000/532/EC as regards the list of wastes. OJ L 203 28.07.2001, pp. 18–19, with Annex: List of wastes pursuant to Article 1(a) of Directive 75/442/EEC on waste and Article 1(4) of Directive 91/689/EEC on hazardous waste.

EC DG ENV A2/LM, 2002. The Management of Waste from the Extraction Industry. Working Document No. 3. Brussels, 5 June 2002.

EEC. Directive 75/442/EEC on waste, OJ L 194, 25.7.1975, p. 39. Directive as last amended by Commission Decision 96/350/EC, OJ L 135, 6.6.1996, p. 32.

EN 12457/1/2/3/4, 2002. Characterization of waste — leaching — compliance test for leaching of granular waste materials and sludges — Parts 1/2/3/4. CEN, Brussels, September 2002.

Environment Australia, 1997. Managing sulphidic mine wastes and acid drainage. Series on Best Practice Environmental Management in Mining, Commonwealth of Australia, p. 84.

Environment Protection Act of 27th April, 2001. Dz.U. 62.627.2001, in Polish.

EPA/DOE, 2000. Mine Waste Technology Program. 2000 Annual report, MSE Technology Applications, Inc., Butte, Montana, p. 67.

Geller, W., Wendt-Potthoff, K., Koschorreck, M., 2002. *In-situ* bioremediation of acid mining lakes by addition of organic substrate and lime, P774, pp. 311–311. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Geological and Mining Law, Act of 4 February, 1994, Dz.U. 27.96.1994, as last amended by Directive of 21st December, 2001, Dz.U. 110.1190.2001, in Polish.

Georgi, A., Balcke, G.U., Kopinke, F.-D., 2002. Application of humic substances for *in-situ* groundwater remediation, pp. 16–17. Use of Humates to Remediate Polluted Environments: From Theory to Practice. NATO Advanced Research Workshop, Zvenigorod, Russia, September 23–29, 2002, MSU, Moscow.

Harabin, Z., Strzyszcz, Z., 1993. In: Skarzyska, K.M. (Ed.), Usability of some species of trees and shrubbery for forest reclaiming of colliery spoil dumps, pp. 901–910. Proceedings of the fourth International Symposium on the Reclamation, Treatment and Utilization of Coal Mining Wastes, Vol. II, Univ. of Agriculture, Krakow, pp. 986, Krakow, Poland, September 6–10, 1993.

Harbaugh, A.W., Banta, E.R., Hill, M.C., McDonald, M.G., 2000. MODFLOW-2000, the U.S. Geological Survey Modular Ground-Water Model — User Guide to Modularization Concepts and the Ground-Water Flow Process. U.S. Geological Survey Open-File Report 00-92, Denver, p. 121.

Hill, M.C., Banta, E.R., Harbaugh, A.W., Anderman, E.R., 2000. MODFLOW-2000, the U.S. Geological Survey Modular Ground-Water Model — User Guide to the Observation, Sensitivity, and Parameter-Estimation Processes and Three Post-Processing Programs. U.S. Geological Survey Open-File Report 00-184, Denver, p. 210.

Holmes, P.R., Grundwell, F.K., 1995. Kinetic aspects of galvanic interactions between minerals during dissolution. Hydrometallurgy, 39, 353–375.

Hutchinson, J.P.G., Ellison, R.D. (Eds), 1992. Mine Waste Management, Lewis Publishers, Boca Raton, p. 654.

Hutnik, R.J., Davis, G., 1978. Reclamation of coal mining land in the United States as compared with the Ruhr, pp. 71–83. In: Goodman, G.T., Chadwick, M.J. (Eds), Environmental Management of Mineral Wastes. NATO Advanced Study Institutes Series, Series E: Applied Science — No. 7, Sijthoff and Noordhoff, Alphen aan den Rijn — The Netherlands, p. 367.

Ibeanusi, V.M., Archibold, E., 2002. On-site remediation of metals from acid mine drainage. P769, pp. 310–310. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Ibeanusi, V.M., Phinney, D., Thompson, M., 2003. Removal and recovery of metals from a coal pile runoff. Environ. Monit. Assess., 84, 35–44.

Jarvis, S.T., Braithwaite, P.A., 1994. Colliery spoil rock waste. Mining Environ. Manag., 2 (4), 21–24.

Johnson, D.B., Hallberg, K.B., 2002. Novel biological systems for remediating acidic, metal-rich wastewater, P767, pp. 310–310. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Kerth, M., 1988. Die Pyriterwitterung in Steinkohlenbergematerial und Ihre umweltgeologischen Folgen. Dissertation, Univ. — GHS Essen, p. 182, in German.

Kerth, M., Wiggering, H., 1990. The weathering of colliery spoil in the Ruhr — problems and solutions, pp. 417–425. In: Rainbow, A.K.M. (Ed.), Reclamation, Treatment and Utilization of Coal Mining Wastes, A.A. Balkema, Rotterdam-Brookfield, p. 527.

Levens, R.L., Boldt, C.M.K., 1994. Mine waste landfill. Mining Environ. Manag., 2 (4), 16–20.

Looney, B.B., Falta, R.W. (Eds), 2000. Vadose Zone Science and Technology Solutions, Battelle Press, Columbus, Ohio, p. 1500.

Madejon, E., Perez de Mora, A., Puente, P., Cabrera, F., 2003. Heavy metals and arsenic adsorption by organic materials, pp. 266–267. In: Gobran, G.R., Lepp, N. (Eds), Proceedings of the 7th International Conference on the Biogeochemistry of Trace Elements, Uppsala, Sweden, 2003. Vol. 1. Scientific Programs II, SLU Service/ Repro, Uppsala, Sweden, p. 287.

Marszalek, A.S., 1996. Preventive and remedial engineering measures to control acid mine drainage in Australia. In Preprints of Papers, Engineering Tomorrow Today. The Darwin Summit, National Engineering Conference, Darwin, Northern Territory, 21–24 April 1996. The Institute of Engineers Australia, Barton Act 2600, Australia.

McDonald, M.G., Harbaugh, A.W., 1999. Modflow — A Three-Dimensional Finite-Difference Ground-Water Flow Model, USGS, Denver, Colorado, p. 530.

Mehl, S.W., Hill, M.C., 2001. MODFLOW-2000, The U.S. Geological Survey Modular Ground-Water Model — User Guide to the Link-Amg (LMG) Package for Solving Matrix Equations Using an Algebraic Multigrid Solver. U.S. Geological Survey Open-File Report 01-177, Denver, p. 33.

Merkel, B.J., Planer-Friedrich, B., 2002. Grundwasserchemie Praxisorientierter Leitfaden zur numerischen Modellierung von Beschaffenheit, Kontamination und Sanierung aquatischer Systeme, Springer, Berlin, p. 219.

Neumann-Malkau, P., 1993. Acidification by pyrite weathering on mine waste stockpiles, Ruhr District, Germany. Engng Geol., 34, 125–134.

Ngwenya, E.G., Walsh, M.M., Porter, R.J., 2002. Treatment of acid mine drainage sediments/leachates by an anaerobic biotreatment approach, P764, pp. 309–309. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Nielsen, D.M., 2000. Practical Handbook of Ground-water Monitoring, Lewis Publishers, Boca Raton, p. 728.

Noecker, B.L., Reddy, K.J., Brown, T.H., 2003. Galvanic suppression/*in-situ* precipitation of calcite for acid mine drainage remediation, pp. 270–271. In: Gobran, G.R., Lepp, N. (Eds), Proceedings of the 7th International Conference on the Biogeochemistry of Trace Elements, Uppsala, Sweden, 2003. Vol. 1. Scientific Programs II, SLU Service/Repro, Uppsala, Sweden, p. 287.

Parkhurst, D.L., 1995. User's Guide to PHREEQC — A Computer Program for Speciation, Reaction-Path, Advective Transport, and Inverse Geochemical Calculations. U.S. Geological Survey Water-Resources Investigations Report 95-4227, p. 143.

Parkhurst, D.L., 1997. Geochemical mole-balance modeling with uncertain data. Water Resour. Res., 33 (8), 1957–1970.

Parkhurst, D.L., Appelo, C.A.J., 1999. User's Guide to PHREEQC (Version 2) — A Computer Program for Speciation, Batch-Reaction, One-Dimensional Transport, and Inverse Geochemical Calculations, U.S. Geological Survey, Water-Resources Investigations Report 99-4259, Denver, Colorado.

Patrzalek, A., Twardowska, I., Szczepańska, J., 1993. In: Skarzyńska, K.M. (Ed.), Biological reclamation of coal mining waste tip as an essential factor in its resultant environmental impact, pp. 807–816. Proceedings of the Fourth International Symposium on the Reclamation, Treatment and Utilization of Coal Mining Wastes, Vol. II, Univ. of Agriculture, Krakow, pp. 986, Krakow, Poland, September 6–10, 1993.

PHREEQC I Version 2.8.0.0. (April 15, 2003). USGS Website: http://wwwbrr.cr.usgs.gov/projects/GWC_coupled/phreeqc.

Picki, C., Moser, H., Fomin, A., 2003. Development of biological test systems using organisms native to acidic mining lakes, MOP 107, pp. 99–99. SETAC Europe 13th Annual Meeting, Hamburg, Germany, 27 April–1 May 2003. Abstracts, SETAC Europe, Brussels.

Piet, L., Marcus, V., Look, H.P., 2002. Bioreactor design for high-rate biological sulfide production. P771, pp. 310–310. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

PKN — Polish Committee of Standardization, Measures and Quality. Polish Standard PN-93/G-11010: Mining. Materials for Hydraulic Filling. Specifications and Tests. Wyd. Normalizacyjne ALFA-WERO, 1994, in Polish.

Powell, R.M., Puls, R.W., Blowes, D.W., Vogan, J.L., Gillham, R.W., Schultz, D., Powell, P.P., Sivavec, T., Landis, R., 1998. Permeable Reactive Barrier Technologies for Contaminant Remediation. Office for Research and Development, Office of Solid Waste and Emergency response, U.S. EPA, EPA/600/R-98/125.

RCRA, 1984. Resource Conservation and Recovery Act of 1976, Public Law 98-616, November. 8, 1984.

Sara, M.N., 1994. Standard Handbook for Solid and Hazardous Waste Facility Assessments, Lewis Publishers, Boca Raton, p. 976.

SDWA, 1986. The Safe Drinking Water Act, 1974, with amendments in Section 1412.

Serrano, S.S., 1997. Hydrology for Engineers, Geologists and Environmental Professionals — An Integrated Treatment of Surface, Subsurface and Contaminant Hydrology, HydroScience, Lexington, KY, p. 480.

Shuman, L.M., 1999. Effect of organic waste amendments on zinc adsorption by two soils. Soil Sci., 164, 197–205.

Skarżyńska, K.M., 1995. Reuse of coal mining wastes in civil engineering — part 2: utilization of minestone. Waste Manag., 15 (2), 83–126.

SSG: The Scientific Software Group, 2003. Environmental Software: Groundwater, Surface Water, Flood Hydrology, Geotechnical, Air Pollution, Bioremediation, Environmental Graphics, SSG, Sandy, Utach (Ad.).

State Inspectorate of the Environmental Protection, District Inspectorate of the Environmental Protection in Katowice, 1995. In: Jarzębski, L. (Ed.), Report on the State of the Environment in Katowice District in 1994, Library of the Environmental Monitoring, Katowice, p. 231.

State Inspectorate of the Environmental Protection, District Inspectorate of the Environmental Protection in Katowice, 1997. In: Jarzębski, L. (Ed.), Report on the State of the Environment in Katowice District in 1995–1996, Library of the Environmental Monitoring, Katowice, p. 371.

Szczepańska, J., Twardowska, I., 1987. Coal mine spoil tips as a large area source of water contamination, pp. 267–281. In: Rainbow, A.K.M. (Ed.), Reclamation, Treatment and Utilization of Coal Mining Wastes, Proceedings of the Second International Conference, Nottingham, 1987, Elsevier, Amsterdam, p. 667.

Szczepańska, J., Twardowska, I., 1999. Distribution and environmental impact of coal-mining wastes in Upper Silesia. Poland Environ. Geol., 38 (3), 249–258.

Tabak, H.H., Govind, R., 2002. Advances in biotreatment of acid mine drainage and biorecovery of metals. P775, pp. 311–311. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.

Twardowska, I., 1981. Mechanism and Dynamics of Coal Mining Waste Leaching at Dumps. Polish Academy of Sciences, Institute of Environmental Engineering, Works and Studies No. 25, Ossoliński Publishers of the Polish Academy of Sciences, Wroclaw-Warsaw-Cracow-Gdansk-Lodz, pp. 206, in Polish.

Twardowska, I., 1990. Buffering capacity of coal mine spoils and fly ash as a factor in the protection of the aquatic environment. Sci. Total Environ., 91, 177–189.

Twardowska, I., 1993. In: Tedder, D.W. (Ed.), Pollution abatement from coal mining wastes, pp. 341–344. Emerging Technologies in Hazardous Waste Management V, Vol. I of III, American Chemical Society, Washington, DC, p. 368.

Twardowska, I., Kyziol, J., 2003. Sorption of metals onto natural organic matter as a function of complexation of adsorbent–adsorbate contact mode. Environ. Int., 28 (8), 783–792.

Twardowska, I., Szczepańska, J., 1990. Transformation of chemical composition of pore solution in coal mining wastes, pp. 177–187. In: Rainbow, A.K.M. (Ed.), Reclamation, Treatment and Utilization of Coal Mining Wastes, Proceedings of the Third International Conference, Glasgow, 1990, A.A. Balkema, Rotterdam Brookfeld, p. 527.

Twardowska, I., Szczepańska, J., Witczak, S., 1988. The Impact of Coal Mining Spoil on the Aquatic Environment: Evaluation of Risk, Prognosis, Prevention. Polish Academy of Sciences, Institute of Environmental Engineering, Committee of Environmental Engineering, Works and Studies No. 35, Ossoliński Publishers of the Polish Academy of Sciences, Wroclaw-Warsaw-Cracow-Gdansk-Lodz, pp. 251, in Polish.

Twardowska, I., Kyziol, J., Schmitt-Kopplin, P., 2003. Organic matter as a sorbent of cadmium for prevention and control of environmental pollution from wastewaters and leachates, pp. 222–223. In: Gobran, G.R., Lepp, N. (Eds), Proceedings of the 7th International Conference on the Biogeochemistry of Trace Elements, Uppsala, Sweden, 2003. Vol. 1. Scientific Programs II, SLU Service/Repro, Uppsala, Sweden, p. 287.

U.S. EPA, 1979. Design and Construction of Covers for Solid Waste Landfills. EPA-600/2-79-165. NTIS PB 80-100381, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1981. Underground Injection Control (UIC) Regulatory Program.

U.S. EPA, 1982. Evaluating Cover Systems for Solid and Hazardous Waste. EPA SW-867. NTIS PB 87-154894, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1983a. Hazardous Waste Land Treatment. EPA-SW-874. NTIS PB 81-182107, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1983b. Lining of Waste Impoundment and Disposal Facilities. EPA SW-870. NTIS PB 86-192796, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1983c. Standardized Procedures for Planting Vegetation on Completed Sanitary Landfill. EPA-600/2-83-055. NTIS PB 83-241018, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1984. The Hydrologic Evaluation of Landfill Performance (HELP) Model. Vol. 1 and 2, User's Guides Version 1. EPA/530-SW-84-009 and EPA/530-SW-84-010. NTIS PB 85-100840 and PB 85-100832, Municipal Environmental Research Laboratory, Cincinnati, OH.

U.S. EPA, 1985. Final Covers for Uncontrolled Hazardous Waste Sites. EPA-540/2-85-002. NTIS PB. 87-119483, Hazardous Waste Engineering Research Laboratory, Cincinnati, OH.

U.S. EPA, 1986: RCRA Groundwater Monitoring Technical Enforcement Guidance Document (TEGD). OSWER 9950.1.

U.S. EPA, 1987a. Engineering Guidance for the Design, Construction and Maintenance of Cover Systems for Hazardous Waste. EPA-600/2-87/039. NTIS PB 87-191656/AS, Hazardous Waste Engineering Research Laboratory, Cincinnati, OH.

U.S. EPA, 1987b. Geosynthetic Design Guidance for Hazardous Waste Landfill Cells and Surface Impoundments. EPA/600/2-87/097. NTIS PB 88-13263/AS, Hazardous Waste Engineering Research Laboratory, Cincinnati, OH.

U.S. EPA, 1989. Technical Guidance Document: Final Covers on Hazardous Waste Landfills and Surface Impoundments OSWER. EPA/530-SW-90-047, 39.

Vrabel, J., Glynn, P.D., 1998. User's Guide to PHRQCGRF — A Computer Program for Graphical Interpretation of PHREEQC Geochemical Transport Simulations. U.S. Geological Survey Open-File Report 98-281, p. 30.

Waste Act of 27th April, 2001. Dz.U.62.628.2001, in Polish.
Waterloo Hydrogeologic, 2003. Groundwater and Environmental Software Catalog, Waterloo Hydrogeologic, Waterloo, Ontario, Canada, p. 28.
Widdowson, M.A., 2002. SEAM3D: A Numerical Model for Three-Dimensional Solute Transport Coupled to Sequential Electron Acceptor-Based Biological Reactions in Groundwater, The Via Department of Civil and Environmental Engineering, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, June.
Wilmoth, R.C., 2000. Vision statement for the Butte Mine Waste Technology Program, pp. 1–2. EPA/DOE MineWaste Technology Program. 2000 Annual Report, MSE Technology Applications, Inc., Butte, Montana, p. 67.
Wilson, N., 1995. Soil Water and Ground Water Sampling, Lewis Publishers, Boca Raton, p. 208.
Wilson, L.G., Everett, L.G., Cullen, S.J. (Eds), 1995. Handbook of Vadose Zone Characterization and Monitoring, Lewis Publishers, Boca Raton, p. 752.
Zaluski, M.H., Trudnowski, J.M., Park, B.T., Bless, D.R., 2002. Designing sulfate-reducing bacteria bioreactors. P772, pp. 310–310. SETAC 23rd Annual Meeting in North America Abstract Book, 16–20 November 2002, Salt Lake City, Utah, SETAC, Pensacola.
Zheng, Ch., Hill, M.C., Hsieh, P.A., 2001. MODFLOW-2000. The U.S. Geological Survey Modular Ground-Water Model-User Guide to the LMT6 Package, the Linkage with MT3DMS for Multi-species Mass Transport. U.S. Geological Survey Open-File Report 01-82, Denver, Colorado, p. 44.
Zhu, Ch., Anderson, G., 2002. Environmental Applications of Geochemical Modeling, Cambridge University Press, Cambridge, p. 284.

Web Sites for Further Information

http://europa.eu.int/comm/dg03/publicat/emy/index.htm
http://europa.eu.int/comm/environment/waste/mining/020624workingdocument3.pdf
http://eippcb.jrc.es/
http://europa.eu.int/en/comm/eurostat
http://www.ul.ie/~edc/stat.html
http://amli.usgs.gov.amli
http://mine-drainage.usgs.gov/mine
http://water.wr.usgs.gov/mine/coal.htm
http://www.epa.gov/r10earth/offices/oea/qaindex.htm
http://www.state.sd.US/state/executive/denr/DES/mining/acidrock.htm
http://gils.doe.gov:1782/cgi-bin/w3vdkgw/