



TAILSAFE

Sustainable Improvement in Safety of Tailings Facilities
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Report

Tailings Management Facilities – Risks and Reliability

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Tailings Management Facilities – Risks and Reliability

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List of reports will be updated according to availability

Table of Contents

1. INTRODUCTION.....	7
1.1 GENERAL REMARKS ON FAILURE, RISK AND RELIABILITY	7
1.2 USUAL FEATURES OF TAILINGS FACILITIES	10
1.3 EMBANKMENT DAMS VERSUS TAILINGS DAM	11
2. OPERATIONAL OR DISUSED TAILINGS FACILITIES INTERACTING ‘NORMALLY’ OR ‘AS INTENDED’ WITH THEIR ENVIRONMENT	14
2.1 TRANSPORT LOSS	14
2.2 EROSION AND SILTATION.....	15
2.3 WATER CYCLE IN TAILINGS FACILITIES	16
2.4 UNAUTHORISED REMOVAL AND USE OF TAILINGS FOR BUILDING OR LANDFILL	18
2.5 STRUCTURAL FAILURE OF TAILINGS EMBANKMENT	19
3. OPERATIONAL OR DISUSED TAILINGS FACILITIES IMPAIRED BY ADVERSE FACTORS, NATURAL OR HUMAN	21
3.1 GENERAL REMARKS ON DANGEROUS IMPACTS AND FAILURE MODES	21
3.2 ENVIRONMENTAL IMPACTS / ADVERSE NATURAL IMPACTS.....	23
3.2.1 Ageing and internal erosion.....	23
3.2.2 Influence of water / external erosion and siltation by water	24
3.2.3 Influence of wind / external erosion and siltation by wind	28
3.2.4 Influence of frost.....	28
3.2.5 Weathering	29
3.2.6 Anthropogenic damage	30
3.2.7 Climatic variability and substrate deterioration	30
3.2.8 Topography.....	30
3.2.9 Material risk potentials.....	32
3.2.10 Natural hazards	32
3.2.11 Extraordinary human factors.....	34
4. BODIES OF FINE MATERIAL, WITH PARTICULAR ATTENTION TO THE MOISTURE CONTENT, STRUCTURAL STABILITY, FLOW DEFORMATION BEHAVIOUR AND LIQUEFACTION	35
4.1 GENERAL.....	35
4.2 3-PHASE SYSTEM OF SOIL.....	36
4.3 SOIL CHARACTERISTIC PROPERTIES	37
4.3.1 Void Ratio.....	37
4.3.2 Grainsize distribution	38
4.3.3 Permeability.....	39
4.3.4 Shear strength.....	45

4.3.5	Effective stress.....	47
4.3.6	Density	49
4.3.7	Consolidation characteristics	51
4.3.8	Plasticity of fine grain soils.....	54
4.3.9	Spatial variability.....	55
4.3.10	Water content.....	55
4.3.11	Chemical properties	55
4.4	FLOW DEFORMATION BEHAVIOUR AND LIQUEFACTION POTENTIAL	56
4.5	VARIATIONS OF THE PROPERTIES OF FINE MATERIAL BY DYNAMIC EVENTS AND DIFFERENT NATURAL HAZARDS.....	63
4.6	STRUCTURAL STABILITY WITH RESPECT TO THE SOIL PARAMETER	66
4.6.1	Common failure modes	66
4.6.2	Failure modes for embankments	67
4.7	TYPICAL RESIDUES AND ABNORMALITIES.....	70
4.7.1	Coastal and field type tailings ponds.....	72
4.7.2	Abnormal situations in the internal structure of the tailings pond	73
4.7.3	Mud pockets.....	73
4.7.4	Subsidence funnels.....	73
4.7.5	Clay content.....	74
4.7.6	Liquefaction of the pond material.....	74
5.	BODIES OF SUCH MATERIALS CONTAINING TOXIC/OR HAZARDOUS SUBSTANCES, WITH PARTICULAR ATTENTION TO THE RELEASE OF CONTAMINANTS.....	75
5.1	MIGRATION PATHWAYS OF CONTAMINANTS AND RELATED PARAMETERS	75
5.1.1	Seepage	75
5.1.2	Liners.....	76
5.1.3	Surface run-off water.....	79
5.1.4	Wind erosion	79
5.2	TAILINGS CONTAINING SULPHIDE	81
5.2.1	Oxidation–acid generation reactions.....	82
5.2.2	Acid neutralisation reactions	83
5.2.3	The role of bacteria	84
5.2.4	Stages in AMD generation.....	84
5.2.5	Dissolution of other contaminants.....	85
5.2.6	Factors affecting the AMD generation	87
5.2.7	AMD prediction methods.....	87
5.3	TAILINGS CONTAINING CYANIDES.....	95
5.3.1	Origin and amounts.....	95
5.3.2	Cyanidation process	96
5.3.3	Fate of cyanides (mechanisms of cyanide degradation).....	100
5.3.4	Critical factors	104
5.3.5	Potential Environmental Impacts of TMFs containing cyanides.....	105

5.3.6	Prevention of cyanide incidents	109
5.3.7	Cyanide treatment.....	111
5.3.8	Cyanide Recovery or Recycling	114
5.3.9	Monitoring cyanide in the environment.....	115
5.4	TAILINGS CONTAINING ALUMINA SUBSTANCES	117
5.4.1	Origin and contaminants of the impounded material	117
5.4.2	Red mud disposal and risks	120
5.4.3	Migration pathways of red mud.....	122
5.4.4	Wind erosion	124
5.4.5	Environmental Impacts of TMFs containing red mud.....	124
5.5	TAILINGS CONTAINING RADIOACTIVE SUBSTANCES	128
5.5.1	Characteristics of Uranium mill tailings	128
5.5.2	Overview of contaminants and their impact on humans and nature	133
5.5.3	Mineral resources which after processing could produce tailings with radioactive contaminant content.....	138
5.5.4	Characteristics of main types of radio-nuclides resulted from the fine bodies originated during processing	145
5.5.5	Pathways for radio-nuclides migration into environment factors.	148
5.6	RADIOACTIVE TAILINGS FACILITIES INTERACTING WITH THEIR ENVIRONMENT – CASE STUDIES	151
6.	CONCLUSIONS.....	157
7.	FAILURE OF TAILINGS IMPOUNDMENTS – CASE STUDY	159
8.	REFERENCE	168

Table of Figures

Figure 1:	Simplified event tree for a dam failure due to seismic impact	8
Figure 2:	Erosion process	15
Figure 3:	Arrangement of further ring dykes in the catchment area of a tailings dam to control the water inflow	17
Figure 4:	Sludge disposal (Gyöngyösoroszi, Hungary)	18
Figure 5:	Removal of contaminating tailings material	18
Figure 6:	Damage of heap leaching pile, Hungary 1978	19
Figure 7:	Erosions cracks on the side of heap leaching piles	20
Figure 8:	Abandoned pile used for road constructions (Hungary)	20
Figure 9:	Geometrical Quantities influencing the potential of dam failure	31
Figure 10:	Behaviour of soil characteristic quantities in dependency of the spilling location	35
Figure 11:	Water and air content in a soil	36
Figure 12:	Void ratio relationship	37
Figure 13:	Typically weights, void ratios and densities for different tailings	37
Figure 14:	Grading curves for tailings	38
Figure 15:	Typical state parameters for tailings	39
Figure 16:	Equations for the prediction of the coefficient of permeability	40
Figure 17:	The location of phreatic surface within a stream embankment	41
Figure 18:	Influence of beach width on the phreatic surface	41
Figure 19:	Conceptual model of permeability variation within a tailings deposit	42
Figure 20:	Variation in permeability and anisotropy for well segregated tailings beaches according to distance from discharge	43
Figure 21:	Variation in average permeability as a function of void ratio	44
Figure 22:	Stress-Strain characteristics of fine copper tailings	45
Figure 23:	Shear strength parameters for tailings	46
Figure 24:	Strength envelope curvature at low stress levels	48
Figure 25:	Variation in effective friction angle with stress level	49
Figure 26:	Minimum and Maximum Densities of Sand Tailings	50
Figure 27:	Average in-place relative density of sand tailings	50
Figure 28:	Typical values of coefficient of consolidation	51
Figure 29:	Increase in average in-place density with depth	53
Figure 30:	Effect of sundrying on void ration and consolidation characteristics of tailings	53
Figure 31:	Plasticity chart	54
Figure 32:	The consistency states of soil	55

Figure 33: Soil profile as shear profil	57
Figure 34: Dependence of the cycle numbers and the earthquake magnitude	58
Figure 35: Dependency between c_r und relative density	59
Figure 36: Risk of liquefaction regarding the sieve line of in-situ soil	59
Figure 37: r_D in dependence of depth h	60
Figure 38: Limited curve for risk of liquefaction for fine sand with 2 groundwater layers	61
Figure 39: Regions of stress cycles for estimate the risk of liquefaction (1ft = 0.3048 m)	62
Figure 40: Approximations for the factor of safety against shallow failure within tailings dams	68
Figure 41: Filter criteria and grain size distribution	70
Figure 42: Construction methods of tailings dams	71
Figure 43: Upstream tailings dam	72
Figure 44: World distribution of gold or gold/silver mines that utilise cyanidation in the year 2000	96
Figure 45: General overview of the gold extraction operations	97
Figure 46: Cyanide cycle within gold extraction process	98
Figure 47: Gold heap leaching	99
Figure 48: Gold vat leaching	99
Figure 49: Simplified diagram of cyanide degradation processes in TMFs [125]: 1. Waste Rock Embankment. 2. Decant pond. 3. Natural ground.	101
Figure 50: Classifications of cyanides	102
Figure 51: Cyanide chemical loss pathways in the environment	104
Figure 52: Dam failure and leakage	110
Figure 53: Alumina tailings production	118
Figure 54: Percentage proportions of various disposal techniques used for the storage of red mud	120
Figure 55: Radioactive decay series for uranium and thorium	130
Figure 56: Uranium mill tailings activity	131
Figure 57: Uranium mill tailings hazards	132
Figure 58: Heavy metals and other components in tailings	133
Figure 59: U - Ra corespondency diagram	139
Figure 60: Cetatua tailings pond (Romania) – grid system for gamma dose measurement (left) and gamma dose debit distribution	140
Figure 61: Particulates of different sources	151
Figure 62: Dependence of the emanation coefficient E from moisture content w_t	152
Figure 63: Radon diffusion coefficient in relation to moisture saturation	153
Figure 64: Radon concentration in the vicinity of tailings ponds (inactive sites)	154
Figure 65: Radon concentration decreasing with distance from the source	155

Figure 66: Radiological parameters measured in Pellerd (Hungary, March-July 2002) 156

List of Tables

Table 1:	Parameters of static Acid Base Accounting procedures	89
Table 2:	Acid-Base Accounting Screening Criteria	90
Table 3:	Kinetic tests for acid rock drainage prediction	91
Table 4:	Natural degradation of cyanide in tailings impoundments	103
Table 5:	Response of humans and animals to HCN in air	106
Table 6:	Preliminary selection guide for cyanide treatment processes	115
Table 7:	Chemical composition of alumina red mud from some European refineries as compared with the range found in residues from worldwide production	119
Table 8:	Mine contaminant pathways	122
Table 9:	Average annual effective dose (mSv/year) for the whole population – natural irradiation	135
Table 10:	Average annual effective dose in Romania	136
Table 11:	Distribution of U and Ra and granulometry class	138
Table 12:	Phosphate deposits of high uranium content	141
Table 13:	Uranium and radium content in the NP fertilizer production	143
Table 14:	Contents of radionuclides in Oradea alumina plant (Romania)	143
Table 15:	Properties of the natural uranium isotopes	146
Table 16:	Isotopic composition of natural uranium	146
Table 17:	Natural radium radioisotopes	146
Table 18:	Natural radon radioisotopes	146
Table 19:	The invariable deposit of radon	147

1. Introduction

1.1 *General remarks on failure, risk and reliability*

When are Tailings Facilities interacting normally or as intended with their environment? When all the goals have been met with adequate confidence, and when there are no adverse effects on humans or on the environment! However, nature and in particular our technical world remain unknown to a certain degree. This is an old question in our culture as well as a field in philosophical science discussed from St. Augustine through Ludwig Wittgenstein to Karl Popper [28]. Daily we define ourselves and our culture as we confront the limits of knowledge in defining risks and we embrace uncertainties while deciding how to act on the stage we are given. Therefore in discussing whether a tailings facility poses any risk to humans or to the environment, the analytic philosophy begins with the questions: What is risk? How can we analyse risk? How can we distinguish the idea of risk from more fundamental concepts such as danger, threat, failure, possibility or severity?

Risk has several broad meanings that refer to different aspects of our experience [28]. In scientific and economical fields we try to use an objective concept. The subjective one is preferred for environmental and personal decisions, but there is also a psychological meaning of risk.

The objective meaning of risk refers to the measurable properties of the world and of its technical aspects such as frequency, severity, probability and variability. In this view risk is defined as a combination of the frequency with which an adverse effect or a failure occurs and the severity of that outcome. Risk increases as the frequency, severity or both increases. Mitigation or improvement decreases one of these components. Assessing risk requires analysing both the frequency of a possible adverse effect and the severity of the outcome. Therefore this quantitative description of objective risk requires that both are measurable properties.

The subjective understanding of risk agrees that risk is influenced by frequency, severity and variability of an effect. But in this school, risk exists inside our mind as a response to the unknown effects and not as an objective calculable property. In addition to frequency and severity we have to consider uncertainty and subjective qualities such as vulnerability, fear, dread, hope, trust or belief. Risk in this view can be defined as the individual mind characterized by uncertainty about the outcome of a process, a sense that the future might hold uncontrolled or undesired effects. An estimation of frequency and severity will influence the sensitivity but it is not possible to absolutely quantify risk. So subjective risk considerations have to do with opinion and belief, not with physics.

Beside these both fundamental approaches, a more psychologicistic school takes pieces of both and melds them. This approach accepts that risk deals with measurable objective properties such as frequency, probability of failure and severity of the outcome. But it defines that risk is

in some sense the human response to those characteristics. Therefore risk is a set of all adverse outcomes which a person might believe to be possible, when knowing the frequency and severity. Thus this view draws more attention to our knowledge about the processes and to the personal judgement instead of combining numerical values.

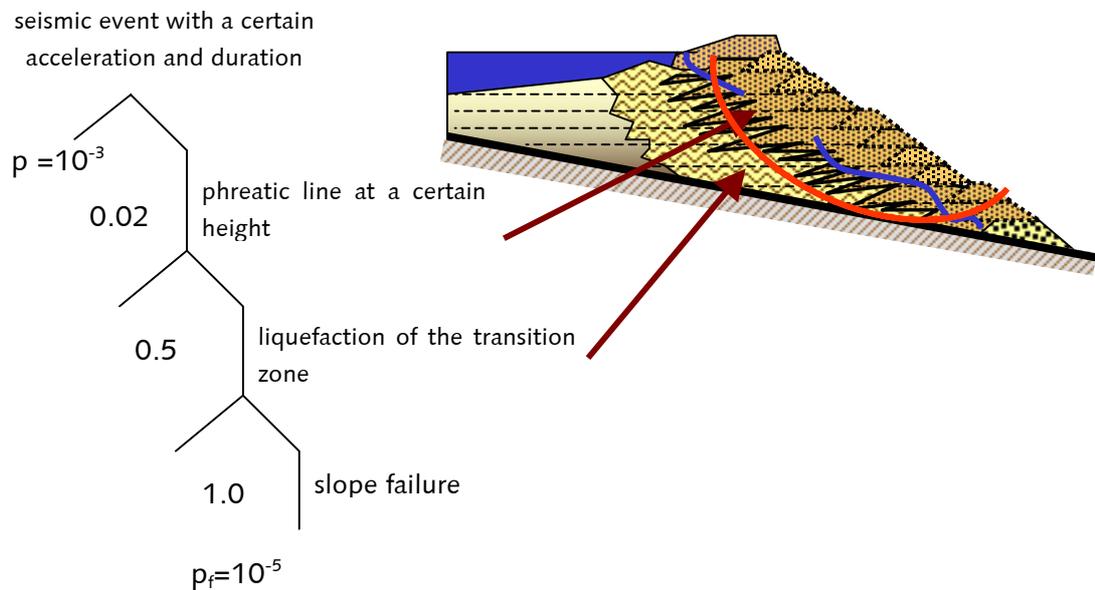


Figure 1: Simplified event tree for a dam failure due to seismic impact

To illustrate the meanings of those different definitions we consider certain environmental impacts of a tailings facility. Figure 1 shows a simplified event tree for an upstream built tailings dam. The analysis is a graphical device for laying out chains of events that could lead from an initiating event to adverse performance. That is what we call failure modes and effect analysis. For each event in the tree we have to assess the conditional probability of occurrence. The total probability for the particular scenario is found by multiplying the probabilities of the branches. The result corresponds to the frequency in an objective risk analysis. As severity we can consider the population that might be affected by a dam failure but also the environmental or economical consequences of damage. If we assess different located tailings impoundments with different geometry, different seismic impact and different downstream conditions, we will get a strongly varying objective risk for the population of each location. If the model fits the reality and we use the "true" data for this analysis, we get the "true" probabilities and therefore the objective "true" risk of each location. Nevertheless, from the view of a particular person, which might be affected in the case of damage, the subjective risk has an individual amount independent of the location. Risk as an individual subjective response of a person living downstream of the dam will be related to the frequency and the imagination of a possible adverse outcome in the case of damage but will also be tempered by many more factors such as experience, dread, trust, age and understanding of the processes etc.

Imagine that there are two studies which determine the probability of dam failure under the same conditions with an event tree. Both studies estimate the probabilities of the branches. The final result differs from $p_f = 5 \cdot 10^{-4}$ to $p_f = 10^{-5}$. Both are reliable under some circumstances but it is not possible to decide, which study is the true one. If asked to give their best estimate of the true failure probability, the objective school will line up the different branch probabilities, will give reasons for the greatest rational support and will declare one of the results to the most likely true failure probability. With this objective view uncertainty is a part of the ability to determine the true risk. The psychologistic school will line up the two estimates of failure probability, but will assign each with a certain level of confidence (perhaps 30% and 70%). This view considers the estimated range when combining with the severity. Therefore this approach treats uncertainty as a part of the risk itself and not as a part of the knowledge. The aim is not to find the best estimate for the true risk but an estimation of its possible range. In engineering science and especially for geotechnical applications, this is the best performance of risk assessment.

Traditionally safety considerations are far from this. They balance favourable and ultimate conditions and express this balance primarily in terms of a factor of safety. However, this traditional approach believes in a real but unknown level of safety, while working with conservative assumptions that should cover all the unknown uncertainties. The advantage of considering these uncertainties in a risk analysis is decomposing a system into smaller pieces that are easier to handle. Recombining the different sequences in a logically coherent way gives a better understanding of the range of risk. This is worthwhile and also sufficient, because even if we might determine a realistic estimate of the true failure probability or risk, we run in a more subjective or social claim when we have to decide, which level of risk we are willing to agree in life. That is what we call risk management, the process whereby decisions are made to accept a known or estimated risk and the implementation of actions to reduce the frequency or the severity. The frequency can be understood as a probability of occurrence of an event or as a failure probability, the severity as the consequence of a failure.

The inverse term, the probability of a system performing its required function adequately for a specific period of time under stated conditions, is called reliability. Therefore a reliable tailings facility is one that has a high probability of processing without any adverse effect. Concerning a tailings dam, reliability means a low probability of structural damage, collapse, erosion, overtopping or extensive deformation. Concerning a tailings pond, reliability stands for low probability of harmfully polluting exposed humans, surface water, groundwater or air. And when we consider and assess processes for far future periods we meet the sense of sustainability.

These fundamental explanations and definitions of risk and reliability might help to understand the way to reach the goals of a sustainable improvement of the safety of Tailings Facilities. However, this final report of WP 2 does not deal with the more or less philosophical considerations about risk, nor with the specific methodologies of analysing risk and reliability.

This report will point out and analyse the several components of risk with a tailings facility, the impact of nature to the impoundment on the one hand and as the response, the impact of a tailings facility to the environment on the other. Therefore the following chapters deal with the several points that might affect risk, to deliver data and ideas for the branches of a possible event tree, to build up chains of performance of the system and to identify interactions and correlations.

1.2 Usual features of Tailings Facilities

Tailings Facilities are restraint systems, used for the deposition of fluids in suspended matter. The residues shall be stored for long term. The different types of the slurry are well founded by the different types of technical processes, like extraction of raw materials or ashes.

Tailings Facilities are different to normal retaining facilities. They are built in time dependence of the production of residues and the quality of them. Normally the whole construction is never ending over decades. Disused Tailings Facilities have to store the inside residues over geological periods. Therefore the reliability of the whole construction and a compatible impact on the environment is to be guaranteed during rehabilitation and long term.

The economic developments and the given boundary conditions are normally changed during operation. Hence the construction methods can be different step by step. The phases can be recognized in the structure of the soil models in any location of the tailings dam or in the pond.

Depending on the morphological reasons, the useful time span of Tailings Facilities is limited. Continued slurries from production make it necessary to realise additional deposition possibilities. For this reason the Tailings Facilities will be enlarged through enlarging the dam with Tailings Facilities near the old one, so that the residues are deposited in bounded layers. The old pond can be the base for the newer pond and so on. The deposited material can be used for enlargement because the use of external ground substance is expensive. Therefore structure of the dam and the lagoon will be extremely heterogeneous. The detailed determination of the deformation or the bearing capacity is impossible.

The single time steps for Tailings Facilities as opposed to engineered constructed retaining dams are known. So it is possible to plan the construction period, time of operation, rehabilitation and long term use.

1.3 *Embankment dams versus tailings dam*

Conservatively designed earth and rockfill dams and other geotechnical structures do not present major risk to the public. The care with which dams are designed, operated, and maintained manages risk to a level such that society accepts their presence [56]. The reliability of dams and their operation must be very high because the possible risk for residual things must be managed as long as there are people. Often the guideline intended for conventional dams is used for evaluating the safety of tailings dam.

Tailings dams engineering and design is strongly influenced by the methods of conventional dam engineering. This is conditional since the basic engineering methods, dam stability, seepage or foundation analysis, deformability are similar, but not the same. The most differences between tailings and conventional dams result from the following characteristics of tailings dams [122]:

- tailings dams are usually constructed during the entire mine production phase,
- large changes in dam operating regime typically occur during the production phase,
- post-production (often more than one) dam operating phases exist, including the long closure phase during which tailings dam must remain as safe as during the production phase (long after profit making ceases),
- design intervals (ie., time periods during which neither the consequences of hypothetical dam failure nor dam operating conditions are subject to significant changes) may vary widely, say, from 1 year (time between consecutive stage-raises of tailings dam) to more than 500 years (closure phase),
- containment/control of actual or potential contamination.

Based on this the tailings dam engineering is developed to a separate discipline. Here the mentioned characteristics relate to tailings dam safety concerning to risk and reliability. Apart from these tailings dams can be subdivided in low permeability, pervious and highly pervious dams. Only low permeability tailings dams work in principle similar to conventional embankment dams. Traditionally tailings dams store solid materials cover with more or less contaminated water in the pond in opposite to conventional embankment dams storing water. Nevertheless, stored tailings may represent a key factor when analysing the consequences of hypothetical dam failure, examining potential for contaminant generation, determining seepage losses or loads on the upstream face of dam [122]. Both tailings and conventional dam engineering are partly based on the empirical methods. These and the individual experiences in risk and reliability are involved in design methods and safety factors. To underline cost effective design the theoretical dimension methods are combined with monitoring results and inspection programs.

The failure rate for large embankment dams built before 1950 is approximately 2.2 percent and approximately 0.5 percent for dams built after 1950. Most of the failed dams were less than 30 meters in height, but the failure of dams doesn't exclusively depend on height. Approximately 70 percent of the failures have occurred within the first 10 years following construction, and many of those failures were within the first year following commissioning. The most common cause of failure in earth and rockfill dams is overtopping, followed closely by internal erosion of the dam body or its foundations. Concrete dam failures have often been associated with shear failure or internal erosion in the foundations. There are many other significant modes that have resulted in failures, including those related to appurtenant works and conduits through embankments. The failure rates for dams built in China after 1930 is approximately 1 percent from internal erosion and 2 percent from floods during operation. These percentages apply to a population of some 80,000 of dams most of which are 10 meters to 30 meters in height. These failures indicate a level of risk that may not be evident when looking at data for dams in the more industrialized countries [56].

The failure rate of tailings dams depend directly on the application of engineering methods in design as well as monitoring and inspection programs in the other life phases. Worldwide, the mining industry has experienced several significant impoundment failures per year over the past 30 years. It is estimated that there are more than 3500 mentionable tailings dams worldwide [30]. Major tailings dam failures are summarised in [136]:

- 1994 USCOLD database of tailings dam failure incidents,
- 1996 UNEP database on mine waste incidents,
- 1997 USEPA summary of relatively recent tailings dam incidents largely focusing on non-compliant events and limited to certain jurisdictions of the United States, and
- WISE Internet site.

Based on this summarised database, it can be concluded that for the past 30 years, there have been approximately 2 to 5 "major" tailings dam failure incidents per year. During no year were there less than two events (1970-2001, inclusive). If one assumes a worldwide inventory of 3500 tailings dams, then 2 to 5 failures per year equates to an annual probability somewhere between 1 in 700 to 1 in 1750. This rate of failure does not offer a favourable comparison with the less than 1 in 10,000 that appears representative for conventional dams. Furthermore, these failure statistics are for physical failures alone. Tailings impoundments can have environmental "failure" while maintaining sufficient structural integrity (e.g. impacts to surface and ground waters) [29].

Therefore, risk reduction is an important part of responsible ownership and operation of Tailings Facilities. Risk reduction should be considered for the full range of safety for the whole life of an impoundment, based on the observation of a range of failure modes in the historical data. When plausible threats are examined together with a broad view of possible ways to detect problems or warn the downstream population, then risk reduction may be accomplished with a holistic, balanced and therefore cost-effective approach. This view should include conditions that may not be directly associated with the dam, all kinds of environmental risks are to consider.

The aim of this report is not to deliver a scheme for carrying out risk assessment but to figure out the relevant impacts and their effect on safety and reliability of Tailings Facilities.

2. Operational or disused Tailings Facilities interacting 'normally' or 'as intended' with their environment

Today, the release of pollutants is restricted. Different governmental regulations define limits for possible concentration of physical, chemical and radioactive contaminants. As result of ore processing or chemical processing, tailings will be discharged as slurry in Tailings Facilities. During the transport, the sedimentation, the deposition and the remediation the tailings interact with their environment. An import role in this consideration is the water cycle. In mineral processing plants today, the trend is clearly toward maximum recycle of process water. The water balance around a processing plant is well defined. Water enters with the ore and though difficult to regulate. Water may also enter as groundwater flows into the tailings pond but this can be almost eliminated by the use of diversion channels around the tailings area.

The migration pathways of contaminants and related parameter are summarised in Chapter 5.1 of this paper. The following enumeration is an overview in addition with case studies to interaction pathways.

2.1 *Transport loss*

In the interaction between Slurry, Tailings, Tailings Facility and the environment a number of potential mechanism for release of pollutants exist. Besides the erosion by water and wind, pollutants can be released by spilling during the transport of tailings to the impoundment, erosion of cover or embankments, structural failure of embankments, controlled/ uncontrolled release of contaminated water and infiltration into the groundwater, especially for radioactive pollutants where the radon emanation can pose a risk.

Slurried tailings may contain between 40-70% of water. These will be transported to the impoundment area through piping systems which can be many kilometres long. Even dry tailings may contain 15-30% water. They will be transported by truck ore conveyor. Regardless of the means of transport, tailings spill may cause contamination of nearby areas. The probability of spilling can be reduced by constructing the mill and impoundment as close together as possible, but it can not entirely be avoided.

2.2 Erosion and siltation

Another important way for contaminant release is erosion by wind and water. Potential erosion processes, which could lead to the failure of the cover or dam and the release of pollutants are illustrated in Figure 2. Once the impoundment cover has been stabilised, wind erosion tends to be less of a problem than water erosion. In arid areas are both mechanisms are important.

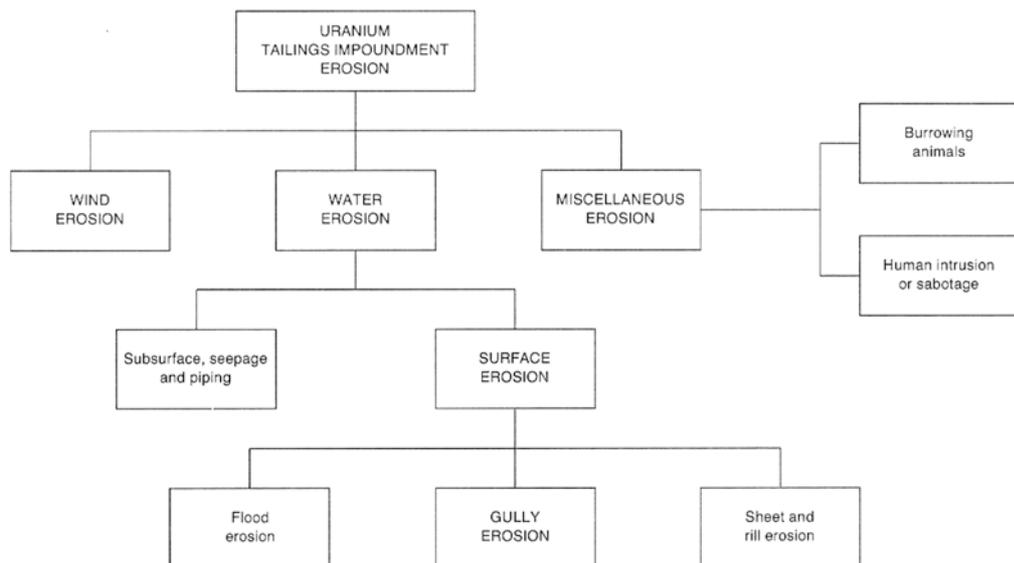


Figure 2: Erosion process

Surface water erosion is one of the most likely long term mechanisms for the disruption of tailings impoundment. To minimise this, erosion diversions and spillway structures have to be constructed. Wind erosion of dry tailings in arid regions, e.g. in south west part of the United States and parts of Australia, can be a serious problem and result in the suspension of radioactive particles in the environment. However, with proper design of cover and control structures, wind erosion effects can be reduced in long term. The amount of tailings dispersed by the wind is a function of the physical characteristics of the tailings (particle size, moisture content, surface texture, etc.) and of the meteorological conditions. Estimation of wind erosion of tailings into the atmosphere is similar to that of estimating re-suspension of substances from soil. Air dispersion, plume depletion, and radon daughter in-growths model have been developed for use in determining radiological effects from this source [90]. Wind erosion and re-suspension can be estimated by using a soil loss equation as developed by a number of researches [139]. Using these developed methods, the accuracy of the calculated values needs to be confirmed experimentally.

Site specific conditions will define the most appropriate strategies to stabilise a pile against the erosion. In some cases well-protected diversion channels, aprons and other construction features may be needed to further improve durability. Although vegetation as a cover system is a generally accepted method for minimising surface erosion. More research is needed to understand the possible remobilization by vegetation of certain contaminants such as radium and radon. There are certain areas where it is not practical to maintain vegetation cover to stabilise the tailings impoundment facility in the long term, e.g. in arid areas.

2.3 Water cycle in Tailings Facilities

Controlled release of contaminated water

For mill sites where the evaporation exceeds the precipitation, mill circuits may be designed for zero discharge (Africa, parts of Australia etc.). Here it is possible to rely on natural evaporation to remove excess water. In areas with low evaporation, the natural water balance may be realized by a controlled release into the environment, discharge to deep geological media or artificial evaporation of decant solution from the tailings impoundment. If radiological components are in the tailings, then the discharged water has to be treated to reduce radionuclide components such as radium-226, polonium-210 and uranium. Furthermore non-radionuclide pollutants like heavy metals, magnesium, etc are also to be reduced. The technical water treatment is expensive. Therefore the water volume should be minimized.

Uncontrolled release of contaminated water

Overtopping of the tailings dam after excessive water inflow could result in erosion and breaching of the embankment. Such a breach could release a large amount of tailings and contaminated water to downstream areas. Depending on the catchment area, the inflow volume from natural precipitation is very high. Ideally, the inflow volume should be minimised by careful selection of the site and use of diversion channels to remove the water from the impoundment. The design should provide the requirements, so that heavy rainfalls can not cause erosion.



Figure 4: Sludge disposal (Gyöngyösoroszi, Hungary)

2.4 Unauthorised removal and use of tailings for building or landfill

The coarse fraction of tailings looks like clean and high quality sand. In the past there have been cases where tailings were removed from piles and used as a material for road construction or house building [129]. The US Department of Energy estimates that in app. 4400 individual locations in the US, tailings has been used for construction work. Similar problems have occurred in Australia and in Ukraine. To reduce such unauthorised removal of tailings the repository system should make public access to the tailings difficult. Unauthorised removal of tailings contaminated by heavy metals is shown in Figure 5 (Hungary, Tailings ponds for disposal of wastes from lead-zinc ore processing). Heap leaching tailings (originated from processing of ore graded 130 g/t) has been used for road construction in small quantities (app. 40 thousand t) from an abandoned heap.



Figure 5: Removal of contaminating tailings material

2.5 *Structural failure of tailings embankment*

Structural failure of a tailings dam could result in the release of large quantities of tailings solution and/or solids from the impoundment. Where tailings are saturated, large discharges of solids may occur after this type of breach. Such an incident occurred in Church Rock, New Mexico, in 1979. Approximately 357 thousand m³ of tailings liquid and an estimated 990 t of tailings solids were released into an adjacent stream. The liquid eventually emptied into the nearby Rio Puerco [134]. The dam height is an important factor for the safety of tailings dam.

Consequences of dam failure can be severe with saturated, low-density tailings. These characteristics increase the potential for the tailings to flow as a liquid, which would result in a large discharge that might travel in a considerable distance. The potential for a dam failure by liquefaction during an earthquake is reduced if the tailings are dry or dense.

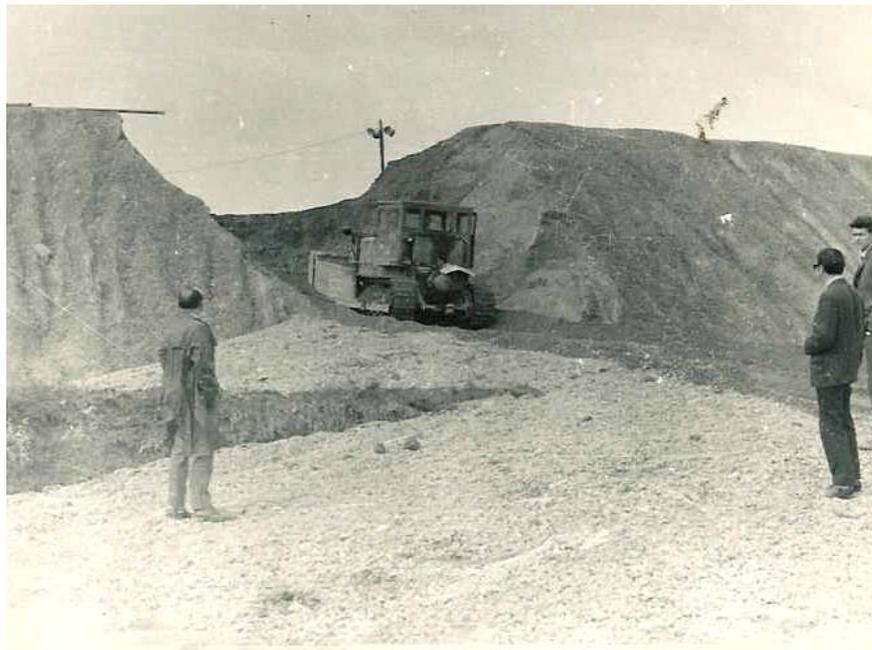


Figure 6: Damage of heap leaching pile, Hungary 1978



Figure 7: Erosions cracks on the side of heap leaching piles

A structural failure of a leaching pile heap is illustrated in Figure 6 (Hungary, 1978). Approximately 3 thousand tons of tailings and 1 thousand m³ of process water were released into the nearby forest. Erosion tracks on heap piles can be seen in the figure above. To prevent the dam failure the water table on the piles should be divided into some smaller lakes as shown in Figure 8.

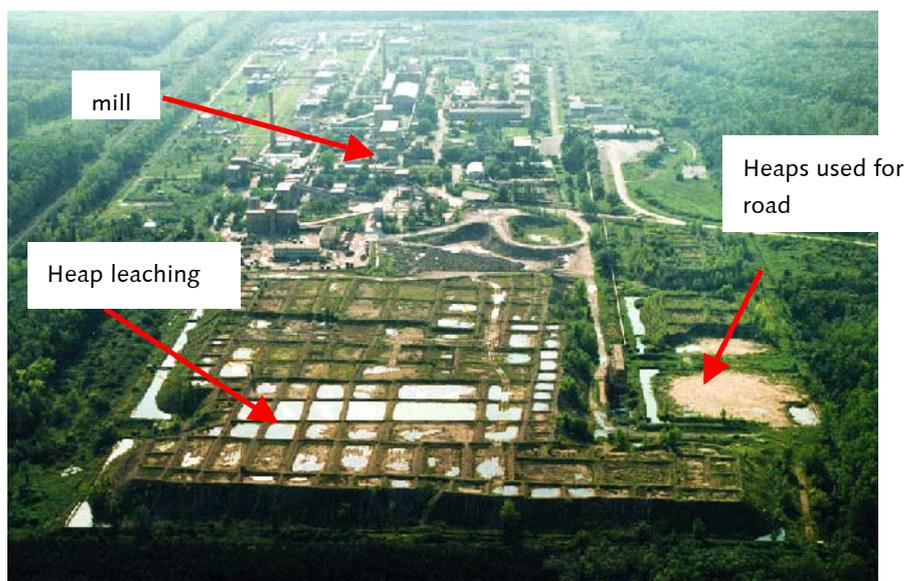


Figure 8: Abandoned pile used for road constructions (Hungary)

3. Operational or disused Tailings Facilities impaired by adverse factors, natural or human

3.1 *General remarks on dangerous impacts and failure modes*

Possible dangerous impacts of Tailings Facilities consist firstly of the accumulation of water and secondly of the soil parameter of the sediments or deposited tailings. The dependency of the ore type and the milling process on the interaction between water and the stored pollutants inside the tailings can be dangerous, if there is any pathway of pollution. In the designing and operation of a tailings facility from the beginning the incorporation of long term environmental considerations is very important. The tailings facility can be considered in the phases of sitting and planning, design, construction, operating, rehabilitation and long-term. Subsequently, only the impaired natural or human factors in the operating and disused phases will be examined. Potential threats for Tailings Facilities can be considered by [9]:

- water tributary and water precipitation
- frost action
- construction and other human processes
- chemical reactions
- actions from flora and fauna
- mass deformation and sliding
- seismic events
- terrorism.

Tailings may be produced in either a dry or slurry form. Dry tailings are from coal fire stations, coarse rejects from some coal, metal or mineral recovery processes. Subject to the economics of transportation and placement, the tailings can be classified into one of seven broad subcategories [2]:

- Base metal tailings that are generally fine grained, and may have significant clay content. These tailings commonly also have a high acid generating potential.
- Gold tailings that have been subjected to a cyanidation process, and may be acid generating.
- Uranium tailings that are radioactive and contain a high percentage of chemical precipitates (gypsum and metal hydroxides).
- Coal tailings that usually have two components: a coarse reject and a washery fines, which may be disposed of separately or may be combined. They are sometimes both acid generating.

- Nickel Tailings that have been subjected to high temperature and pressure leaching, followed by neutralisation, that influence the material properties.
- Alumina Red Mud tailings that have a high alkalinity.
- All other types, including potash tailings, phosphate tailings, clay tailings (or "slimes").

A sufficient level of reliability has to be independent of the current usage given to a tailings dam. This level should be determined by worst case scenario. This is difficult in particular for shut down dams, because no or only some empirical values are available for assessment the reliability. The change of the strength of the influences and the different influence weighting join under each other. Based on current knowledge, the following three classes of failure mechanisms should be considered, for the long term phase of tailings dams [54]:

- slope failures in the foundation or in the dam proper
- extreme events such as floods, earthquakes and high winds
- slow deterioration actions, such as water and wind erosion, frost and ice forces, weathering of fill materials and intrusion by vegetation and animals.

Necessary changes of the dam are caused by the influence of water, wind, natural hazards and human actions. Therefore, the so called environmental influences such as physical and chemical interactions are included in the dam just like topography, flora, fauna and weathering appearances of the settlement caused by climatic fluctuations.

3.2 Environmental impacts / adverse natural impacts

Slow deteriorative processes or progressive degradation may lead to overall failure and may have severe adverse long-term environmental consequences. Sustainability of remediate measures and improvement must thus be given careful consideration in the planning, design, operation and closure of tailings deposits. Such slow deteriorative processes are:

- internal erosion by water (piping)
- external erosion and siltation by water
- external erosion and siltation by wind
- frost and ice action
- weathering
- anthropogenic damage
- climatic variability and substrata deterioration

It is usually the combined effect of one or more of these long-term deteriorative processes which results in failure or environmental damage. Climate, tailings properties and geometry will however influence which of these processes is likely to be the most prominent.

3.2.1 Ageing and internal erosion

Internal erosion or piping may be caused by perennial or transient flow of water through erodible tailings material. The erosion can cause migration of tailings into the environment and could open up pathways for further erosion by stormwater. Internal erosion could also cause local steepening and thereby induce slope stability and further loss of integrity. Internal erosion can only occur where a hydraulic gradient exists in erodible tailings material and where a flow path, such as cracks or voids from eroded material, along which dislodged material can migrate, is present. In special cases the very fine fraction of tailings could migrate through a coarser matrix without the presence of a crack or void (suffusion), but such cases are extremely rare [54]. In practice piping usually occurs along cracks created by desiccation, shrinkage distortion of the outer walls of tailings impoundments or along dislocations due to settlement and deformation. Finer grained tailings which dry out slowly, may only develop cracks of sufficient depth or lateral extent long after closure. This may represent a significant environmental hazard unless ongoing maintenance is provided.

The most common forms of long-term instability brought about by piping are the following:

- in berms or benches where shrinkage associated with desiccation provides a pathway for water which accumulates on the berm to erode away the retaining structure and thereby create a route for more severe storm water erosion to follow
- along abandoned penstock, berm drainage pipes and structures which eventually corrode and collapse thereby creating pathways for piping stormwater erosion and possibly overall instability
- in outer perimeter dykes constructed by hydraulic filling, which may shrink and crack after placement
- in instances where the rate of rise is high, freshly placed tailings may not dry sufficiently before the next layer is deposited. Deep shrinkage cracking may then follow in dry periods and water, or fresh tailings, can then pipe through these cracks
- in outer perimeter dykes freezing may inhibit adequate desiccation and consolidation of the tailings. Subsequent thawing could result in cracking, through desiccation and distortion.

Deteriorative action may be prevented by adequate planning, suitable design and control of operation and at closures. Measures required will depend on climatic conditions, properties, geometry and construction techniques.

Internal erosion (piping) in the foundation has caused a number of earth and rockfill dams to fail. The foundation failure of concrete dams has generally occurred in the first 3-5 years [100], but there are exceptions. Failures caused by internal erosion in the body of fill dams or to embedded pipes are not uncommon.

The progress in reliability is correlated to the progress in design and construction practices, but possibly even more to improvements in maintenance and monitoring, and in particular to proper visual inspection and careful follow-up of increases in leakage. In dam engineering, monitoring has prevented many failures and has reduced the consequences of others. If we consider the long term behaviour and the sustainability of the safety of Tailings Facilities the same effects should be demanded.

3.2.2 Influence of water / external erosion and siltation by water

Abandoned or reclaimed tailings dams are more vulnerable to overtopping and to erosion than active managed impoundments. This is primarily due to the water management of an active tailings dam being controlled by using flood prevention techniques, and return process water usage. All tailings dams require an assessment of return period flooding and risk analysis of overtopping and erosion failures. Flooding that passes the toe of an embankment can cause

erosion, undercutting, and failure of the exterior face. The velocity of the water is a key parameter in determining the scale of erosion and risk of failure. One solution is to provide diversion ditches and mitigation techniques used to prevent inflow of water into the impoundment. Ideally the location of an impoundment should be away from low-lying coastal areas, narrow valleys and areas of particularly soft rock. Flooding around the embankment toe and overtopping of the impoundment causes an increase in the phreatic surface of the embankment. This is due to the natural drainage characteristics of the impoundment being obstructed and further saturation from heavy rains and flood waters. The *in-situ* stresses of the embankment can be too great if the phreatic surface rises too rapidly and so failure can occur.

Operational Tailings Facilities are able to pollute the surface and the groundwater. Effects on streams and watercourses can arise from a number of factors of operational Tailings Facilities. Accidental discharge can of course result in contamination of receiving waters with process solutions such as cyanide or with high loads of sediment. Acidification of sulphidic tailings and subsequent seepage can result in trace metal contamination. Failures of ancillary equipment such as pipelines can also result in the escape of tailings and reagents. The resulting impacts can vary depending on the type of tailings and reagents involved. For example, cyanide solutions discharged to a stream may have a very serious initial impact but will rapidly degrade without long term effect. A discharge containing high levels of toxic metals would however, be much more likely to have permanent or very long term impact.

The erosion rate is a function of a variety of significant factors. These are summarised in the universal soil equation [54].

$$A = R \cdot K \cdot L \cdot S \cdot P \cdot C \quad (1)$$

Here the soil loss by erosion is A, precipitation factor R, soil factor K, the slope length factor L, slope gradient factor S, erosion control factor P and crop management factor C.

For a given type of tailings and a given climatic region only the slope inclination and length can be varied to control erosion unless artificial stabilisers or vegetation is introduced. Vegetation usually represents the most cost effective means of controlling erosion of tailings surfaces. The relative effectiveness however also depends on climatic factors and the level of maintenance provided or required. The greater the water deficit, the more difficult it becomes to maintain vegetation on tailings dams and in arid climates it may not be viable to do so. Sustainable methods using vegetation can only be established by trial and error on a site specific basis. No simple formula exists for all situations but where experience has been built up notable success has been achieved. Specific developments of chemical or mineralogical improvement of the surface soil stabilizer in order to give more fixations against erosion can solve the problem under certain circumstances.

The main problem of the hydrologic planning is the planning over the operating lifetime and over the much longer period after reclamation. The longer the impoundment exists, the greater the probability of a failure. As in natural processes of land forming, erosion can also be initiated by hazardous events of stormwater and by overtopping of impoundments due to flood. The shape of tailings dams must thus be planned to control erosion during the operational and post closure phase under consideration of a probable maximum event. In this regard special attention should be given to the design of the following components of a typical impoundment:

- benches or berms and perimeter dykes
- penstocks and spillways to discharge excess water
- catchment paddocks which arrest the flow of storm water and facilitate settling out of suspended solids
- water dams which impound contaminated runoff and facilitate settling out of suspended solids
- trenches, berms and structures which divert storm water arising upslope of the impoundment away and around the tailings dam.

The choice of design recurrence interval for each of these elements will depend on the circumstances. Risk and consequences of failure severity must be weighted in making the choice. The acceptable probability of post closure failures may be set much lower than during the operational phase since the consequences may be more severe and because the operator will no longer be present to undertake the necessary remedial work. In areas of heavy rainfall, some form of protection against erosion is usually required. Tailings embankments may be susceptible to erosion failure in two major locations, embankment abutments and the embankment face. Erosion along the contact line between the embankment and the abutment may result from stormwater flow that concentrates there. Typically, this type of failure is preventable with proper stormwater diversion methods and so results from faulty design or maintenance. Erosion of embankment faces may result from rupture in tailings lines installed on the embankment crest. Again, maintenance (and alternate siting of tailings lines) may prevent this type of failure [20]. Rainfalls results in different flood events during construction, operation and after closure of an impoundment. In each period of the Tailings Facilities, these flood events have different consequences. Nevertheless every event of transient overtopping results in a breaching or structural dam failure.

Large floods not endangering the impoundment but inundating large areas may also be handled more effectively as a result of emergency planning and early warning possibilities. In future the global human risk associated with these floods may well be higher than the failure risk. This risk is often overlooked, as the risk is considered to be beyond the responsibility of the owner and the operator of a tailings facility.

In rehabilitation and long term the erosion by water and by wind is to be prevented as mentioned above. Summarised, the basic philosophy of erosion control in the rehabilitation of tailings deposits is to protect the surface against the action of water and wind forces either by rock-cladding or by biological means (vegetation cover). At present the establishment of a good vegetation cover is considered the preferred solution [124]. Three different types of surface protection exist, (i) physical and chemical methods, (ii) biological methods and (iii) slope flattening.

The following methods of rehabilitation and amelioration exist especially for gold mines:

- leaching
- leaching with horizontal lime trenches
- elementary amelioration
- minimum cultivation with topsoil
- optimum amelioration
- optimal amelioration with drip irrigation
- optimal amelioration with impact sprinkler, and dry land

As opposed to an operating plant closed plants have to be fixed on the specified Probable Maximum Flood. In face of the fact of increasing fluctuations, which leads to a more extreme precipitation event, the resulting long-term safety for the dam resulting from the PMF is questionable. For all types of impoundment – cross valley, sidehill, valley-bottom and completely enclosed ring dikes – accumulation of water may be prevented by capping the impoundment with material that is peaked near the centre of the impoundment and sloped to drain toward its perimeter.

Another aspect of erosion and siltation in connection with the pollution or in interaction with the environment is given by infiltration and evaporation. Infiltration rates are generally low because of the small particle size and low permeability's in the tailings. Infiltration rates are a function of a soil's moisture content, capillary pressure, unsaturated hydraulic conductivity, and the distance below the surface. There is no runoff or ponding when the infiltration rate is less than the saturated hydraulic conductivity. Runoff or ponding occurs when the infiltration rate is larger than the infiltration capacity and the saturated hydraulic conductivity [125].

Evaporation is a function of wind velocity, atmospheric pressure, temperature, and real extent of surface water. In general, it is proportional to the surface area of the free-water pond. Impoundments in arid areas are designed to conserve and recycle water for mining processes during the mine's active life.

Evaporation data for certain areas are available from NOAA (National Oceanic and Atmospheric Administration). In example the pan evaporation tests can be used to determine evaporation rates if the site is not located in a basin monitored by NOAA. In essence, the pan evaporation test monitors daily water loss in a Class A pan (four feet in diameter and ten inches deep) which is mounted one foot above the ground. A pan coefficient (0.64 to 0.81) is used to adjust pan evaporation rates because they will be higher than normal lake evaporation rates. When the evaporation rates for a basin are known, the designer can determine if surface area dimensions will provide the required evaporation rates. Because net evaporation, like precipitation, is not constant from year to year, it may be beneficial to reduce the calculated evaporation rate by a safety factor to account for annual variability [26].

3.2.3 Influence of wind / external erosion and siltation by wind

Similar to sands, tailings can show the phenomenon of migrating dunes which clarifies the need for protection against wind erosion. Wind erosion mainly is a factor on flat, unbroken surfaces. Wind erosion is a function of climate, erodability of tailings (cohesion Properties), vegetation, length and steepness of slopes. Factors controlling it are in the Wind Soil Loss Equation. Wind erosion can be controlled in much the same way as water erosion, hence measures against water erosion will generally also serve the purpose of controlling wind erosion. Possibilities to check or even to prevent erosion of tailings deposits are laying out a permanent water reservoir and/or a vegetation zone. In the two cases as a consequence of the intervention wind, which means the necessary target of erosion gets taken away.

High wind from tornados, cyclones and low pressure systems will have a negative influence on the dam stability. Risk of failure is associated with wave actions and overtopping of the dam crest. Dams surrounding ponds with large water surfaces must have a high freeboard and the upstream slope must be protected against erosion and be stable against wave actions.

3.2.4 Influence of frost

The effects of frost and ice on tailings and their containment structures can be large and have an influence on the long-term stability. Frost can cause problems for any structure built on saturated soils. As the pore water spaces consist of ice, then once thawing occurs a release of a greater volume of water than can be normally accommodated by the soil pore space occurs. The once frozen water saturates the soil with excess water and the strength of the soil mass is reduced. Under these circumstances slope failures are likely to occur even on gentle slopes [5]. Normally the effects can be divided into two groups, the ice accumulation and the seasonal frost action. In areas of severe winter where cold continuous or discontinuous permafrost develops, ice accumulation should be taken into consideration. During placement of tailings onto beaches, layers of frozen material may be formed. The thickness of each layer depends on the climate and the rate of the use of the impoundment. The effect of freezing tailings is to prevent consolidation and drainage for as long as the tailings are frozen. In northern Canada

large accumulations of frozen tailings have occurred under the beach areas of annually layered tailings impoundments. The zones under the pond tend not to freeze. On thawing the low density frozen tailings will consolidate resulting in large surface settlements and pore water pressure. Such settlements would disrupt surface drainage and capping layers.

Cycles of freezing and thawing have two effects. On freezing the water is drawn from the tailings pores and forms ice crystals and lenses which separate the tailings solids. Thus the tailings solids are consolidated between ice filled channels. On thawing, these channels permit the rapid drainage of fluids resulting in an increase in the tailings mass permeability. Some of these channels close as the thawing tailings slump under the overburden loads. In [70] it is demonstrated that for gypsum rich tailings, the frost induced consolidation may exceed the evaporation induced consolidation, while overall tailings permeability remains considerably higher than for equivalent unfrozen load stress consolidated tailings.

Frost effects on surficial layers of tailings (and cover soils) results in an increase of the surface permeability and consequently an increased infiltration rate. This has considerable significance if surface infiltration is to be minimized to reduce leaching and contaminated seepage.

Another effect of frost can result in a damage of dewatering of process water such a reason can lead to overtopping with the well known problem of dam failure. An impressive example is the failure of Baia Mare, Romania

Frost heave is one of the biggest problems associated with slope stability. As the water in the pore spaces freezes an increase in volume of approximately 9% occurs. For a saturated soil the volume of the soil mass above the level of freezing will increase by the same amount, representing an increase in the soil mass of 2% to 5%, depending on the void ratio [27]. Frost creep occurs as a consequence of continuous freezing, heaving and thawing which results in the downslope movement of soil and sediment. Other forms of instability as a result of frost creep are solifluction, and gelifluction. Solifluction can occur on shallow gradients and is the slow downslope flow of soil and sediment that is saturated with water. Gelifluction is a form of solifluction where the moving soil mass shears over a permafrost layer.

Skinflows (also known as active layer glides) are rapid failures involving the detachment of a thin covering of vegetation and soil that flows downslope over frozen subsoil. Skinflows result of a rapid change in climate that heats the active layer causing it to slip from frozen subsoil. Some examples of frost induced failures are:

3.2.5 Weathering

Weathering is brought about by physical and chemical processes. Physical processes include such phenomena as slaking, cracking and spalling. They are used by freezing-thawing, successive wetting and drying and by thermal variations. Degradation brought about by vegetation and root penetration can also be classified as physical weathering. Chemical weathering depends on mineralogy of the materials, availability of oxygen, organic acids and

water. Common chemical processes are dissolution and acidic reactions, oxidation-reduction, hydration and hydrolysis. The management of Tailings Facilities also imposes some level of danger on their surroundings, depending on the location and standard of operation. The goal is the reduction of the responsible operators, so that risk would be acceptable.

Natural materials such as soils and rocks are often more resistant to weathering than artificial materials. The finely ground rock particles that comprise tailings may depend on the type of rock to be susceptible to rapid weathering. Artificial materials such as concrete, asphalt, plastic, etc., have relatively short lifetimes (mostly less than 100 years) and should not be used where long-term [54].

3.2.6 Anthropogenic damage

Damage and loss of integrity may be brought about by a variety of agents. Use of tailings dam surfaces for recreational and other purposes which may damage structures especially created to provide for long-term stability, should be controlled. Post closure land use must thus be controlled if long-term stability is to be maintained. Vegetation, which plays a vital role in stabilising of slopes and horizontal surfaces, can readily be destroyed, or its efficacy can be impaired by inappropriate land use. Generally, farming and game conservation on rehabilitated tailings impoundments must be managed more carefully than usual to prevent overgrazing, grass fires, etc., which could result in denudation and degradation [54].

3.2.7 Climatic variability and substrate deterioration

Vegetation which is established on the surfaces of tailings impoundments in order to control erosion is often more sensitive and less resistant to extreme climate variation than usual, owing to the relatively harsh substrata on which the vegetation is established. Vegetation may thus only be sustainable for normal conditions and may require special maintenance after extreme droughts, exceptionally cold or hot weather and fires. Coarse or fine grain size, salinity and toxicity are obstacles to establishment of vigorous vegetation cover in many instances. Long term deterioration of substrata may occur where seepage and/or capillary action draws salts up to the surface and causes a build-up which eventually kills off vegetation. These conditions are usually associated with finer grained tailings but it may also be prevalent where the substrate breaks down or weathers chemically.

3.2.8 Topography

The potential of possible threats brought out from Tailings Facilities are especially defined through the dimensions, construction type and environmental arrangement. The potential for dam failure in dependence on the geomorphologic aspects is shown in Figure 9.

Characteristic	Impact to danger
Dam	
Height h	Potential increases with h
Length l	Potential increases with l

Pond	
Volume Sediments V_{sed}	Potential increases with V_{sed}
Volume Water V_{water}	Potential increases with V_{water}

Type of Impoundment	
cross valley	high potential
sidehill	high potential
valley bottom	medium potential
open pits	low potential

Figure 9: Geometrical Quantities influencing the potential of dam failure

The geoenvironmental influences are not concerned. Furthermore the hydraulic conditions are important for the arrangement of the impoundment. The location of the Tailings Facilities in relation to surrounding populated areas defines the total risk and therefore the necessary reliability of the facility. Thus the sensitivity and the use of downstream land, the population, and the buildings of the infrastructure are considered.

With a view to the geological conditions of the site, the stability and the deformation potential have to be recognized with respect to the loads forming the tailings facility in the different time spans of use. For assessing the location it is necessary to proof the geology of the subsoil as well as the properties of the accompanying units as well as bedding, layers, fractures, joints, glide horizons, water tables and karst.

In mountainous regions also the properties of the slopes around are a concern. Depending on this the following failure potentials exist:

- Deformation of the subground in case of weak and compressible layers
- Glide potential at predominated horizons with low shear resistance
- Subground erosion due to changes in groundwater flow and chemistry
- Landslides and other mass movement
- Stability problems of cave networks like karst passages, rooms or chambers
- Pore water pressure effects and seepage due to changes in groundwater flow confined aquifers, perched water tables and seasonal sources

3.2.9 Material risk potentials

In respect of the ore type and the milling process, a wide range of discharged materials exist in the impoundment. The different elements are more or less in solution. The kind of the material compositions and accumulation determines the interaction between the environment and the impoundment, so that various threats can arise. The threats formed by the slurry can be subdivided into the following sections [9]:

- emaciation and stability
- probability for explosions and for fire
- toxically and radioactive threat

Therefore not only the residues should be considered, but also the slurry and the infiltration by precipitation and tributary. The chemical and physical interactions are mentioned above.

3.2.10 Natural hazards

The most hazardous impact in areas with seismic activity is the earthquake. Concerning embankment dams, the average annual failure probability is lower than 10^{-6} in most cases, but it may be in the range of 10^{-3} for some dams in seismic areas.

The consequences of earthquakes may be [100]:

- sudden failure due to liquefaction of fine non-cohesive materials,
- structural failure of buttresses, or
- delayed failures due to cracks, which may continue to extend hours or days later.

Seismic risk assessment is less precise than for floods, and emergency planning is less effective in the case of sudden failures. More expensive structural measures may be necessary. Seismic stability considerations include:

- the magnitude and distance of design earthquake and the design ground motions appropriate for the tailings storage site
- the geotechnical properties of the tailings dam foundation and dam fill materials
- the design and construction of the dam
- the potential inundation of and/or environmental damages to downstream areas in an event of dam incident or failure, and appropriate seismic design criteria
- the selection of an appropriate method of seismic stability analysis; and
- the range of remedial measures available for enhancing the seismic resistance of an existing tailings facility and/or for mitigating potentially detrimental downstream effects that could result from its unsatisfactory performance.

Because seismic failures usually occur without pre-warning, critical observations during the failure process are practically non-existent. Seismic behaviour of tailings dams includes settlement, horizontal movement, cracking (longitudinal and/or transverse), pore pressure build up, slope slumping, slope failure, internal erosion, seepage increase and impoundment breaching.

Crest movement of embankment dams during earthquakes reflects dam deformation associated with compression, lateral spreading and slope movements induced by earthquakes. Reviewed in [19] the seismic performance of earth and rockfill dams includes the crest movement (settlement and horizontal movement).

The empirical data [24] relating tailings dam behaviour to magnitude and epicentral distance of the earthquake reflects the behaviour of older conventional "upstream construction" type. Tailings dams offer a valuable practical tool for preliminary stability assessments of these structures. Tailings dams constructed using downstream and centreline methods would be expected to exhibit better performance under earthquake loading. For seismic stability assessment of tailings dams involving medium dense sands in either the embankment or the foundation, both the selection of methods of seismic stability analysis and the interpretation of results obtained from these analyses require special attention. This is often carried out in a staged approach, which involves starting with a simpler analysis and progressing to more complex analyses as required by the specific case.

In ascending order of cost and complexity, these are:

- Static Limit-equilibrium Stability Analysis using Steady-State Strength
- Simplified Seismic Stability Analysis
- Finite-Element Seismic Stability Analysis

The different calculation models are explained in ICOLD 98. In case of a disused impoundment similar to an operational, liquefaction of loose deposited tailings in high seismic areas may raise the spectre of massive flow-type slides during earthquakes. However, the expected lack of saturation after abandonment and reclamation ordinarily precludes liquefaction, even under major seismic shock. To be able to check the repletion of the tailings deposit, the installation of permanent measuring stations of the ground-water level and the phreatic line in the dam is meaningful.

Dynamic forces due to earthquakes may result in liquefaction of the tailings. Liquefied tailings in the pond will place additional transient forces on the dam, which rarely occurs for conventional water dams, as they are seldom subjected to heavy silt deposit loading. Consequently, determination of the stability under earthquake loading is an important aspect for the stability of the tailings dam.

3.2.11 Extraordinary human factors

Dam constructions and dykes not only fail by exceeding structural stability or through hazardous natural impacts. The breaching of dams and dykes is also a widely used military and terrorism option.

Such attacks can be:

- the destruction of dams and dykes through military weapons
- the destruction of dams and dykes through terrorist actions
- cyber attacks on dam constructions and dyke control systems

Breaching of river dykes or dams in China has been a military tool for more than 1000 years. In 1938 it caused the loss of more than 500,000 lives. The number of years when dams higher than 30 m have been involved in wars is small. The corresponding yearly failure probability is between 10^{-2} and 10^{-3} . The dams at risk may well be those the failure of which would have the worst consequences [100].

In current time the impact of terrorism can act like military forces in war. The aim to influence a great volume of people can be reached by contaminating water reservoir, deactivating energy systems or destroying of retaining systems.

A cyber attack is particularly important to consider that in the larger context of economic activity, water system failures, power outages, air traffic disruptions and other terror scenarios are routine events that is important but do not affect in general the national security. Cyber-terrorists would need to attack multiple targets simultaneously for long periods of time to create terror, achieve strategic goals or to have any noticeable effect.

Although hazardous analysis can hardly determine probabilities of wartime failure, it may be very useful for identifying the potential failure modes from acts of war or terrorism and ways to protect against them, for estimating possible consequences and for preparing emergency planning and early-warning systems.

4.2 3-Phase System of soil

Equivalent to natural soil, the fine body of tailings is a combination of solid material, water and air. A soil mass consists of solid particles that are separated by spaces or voids. These voids can either be filled with air, water or a mixture of both. If the voids are filled with air the soil mass is dry, whereas if only water is present in the voids the body is said to be saturated. If a mixture of air and water is present then the body is partially saturated. Figure 11 shows the three degrees of saturation in a unique volume. The indices stand for V_A - air volume, V_w - water volume, V_s - solid volume.

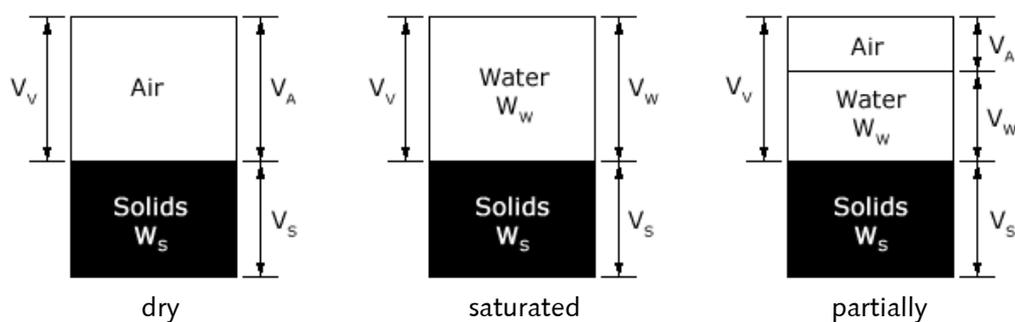


Figure 11: Water and air content in a soil [118]

The following equation is used to calculate the degree of saturation, usually expressed as a percentage.

$$S_r = \frac{\text{Volume of water}}{\text{Volume of voids}} = \frac{V_w}{V_v} \quad (2)$$

This 3-phase system of the tailings can be characterised by their chemical and their mineralogical components. For the valuation of the soil properties both criteria are important.

The considerable components are:

- non-soluble mineralogical solids
- chemical soluble components
- radioactive and toxic components.
- In some cases organic content

For further information it is necessary to validate the composition by a detailed chemical and mineralogical analysis. Especially the potential for emission of environmental contaminants is to be examined, following the source-pathway-target framework.

4.3 Soil characteristic properties

4.3.1 Void Ratio

The void ratio e is simply the ratio between the volume of voids to the volume of solids. This is the most important index to evaluate density, deformation and shear resistance.

$$e = \frac{V_v}{V_s} \quad (3)$$

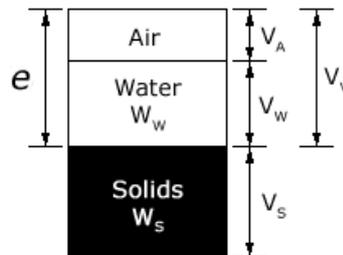


Figure 12: Void ratio relationship

Typical void ratios for different residues are mentioned in the next figure. The varying of the void ratio is a good measure for characterisation of dry density and the consolidation process.

Type of residue	Solid unit weight	Void ratio	Dry density
Fine coal			
East of USA	1,5-1,8	0,8-1,1	45-55
West of USA	1,4-1,6	0,6-1,0	45-70
Great Britain	1,6-2,1	0,5-1,0	55-85
Sand containing oil			
Sands	-	0,9	87
Slimes	-	6-10	-
Lead-Zinc			
Sands	2,9-3,0	0,6-1,0	93-113
Slimes	2,6-2,9	0,8-1,1	80-103
Gold-Silver			
Slimes	-	1,1-1,2	-
Molybdenum			
Sands	2,7-2,8	0,7-0,9	92-99
Copper			
Sands	2,6-2,8	0,6-0,8	93-110
Slimes	2,6-2,8	0,9-1,4	70-90
Taconite			
Sands	3,0	0,7	110
Slimes	3,1	1,1	92
Slimes	3,1-3,3	0,9-1,2	97-105
Phosphates			
Slimes	2,5-2,8	11	14
Gypsum			
	2,4	0,7-1,5	60-90
Bauxites			
Slimes	2,8-3,3	8,0	20
Trona			
Sands	2,3-2,4	0,7	92
Slimes	2,4-2,5	1,2	68

Figure 13: Typically weights, void ratios and densities for different tailings

4.3.2 Grainsize distribution

The grain size distribution of the solid phase of tailings is determined by the ore type and the milling process. It is beyond the control of the impoundment design engineer. The grain size distribution is the fundamental property of tailings that controls the permeability of the body. The variation of the grain size distribution causes spatial variation in permeability. The type of variation is a result of sedimentation and segregation during tailings placement.

Variations in permeability between the sand and slime zones of a tailings impoundment may be as much as three orders of magnitude [101]. Thus the sand zone from a cyclone split tailings, or at the top of a well segregated spigot discharge placed beach, may be one to two orders of magnitude greater than the average (un-segregated) tailings.

Layering of beach deposited tailings is highly anisotropic with permeability parallel to the layering being one to two orders of magnitude greater than normal to the layering. Such layering is adversely oriented for drainage purposes and is therefore undesirable, because the phreatic surface cannot be defined as expected. Where ice layers are included in the beaches (as occurs in northern Canada) they are also barriers to drainage. In vertical direction the slime zones with a greater fine material content are composed in layers with a couple of centimetres with varying thickness up to 20 percent. If the spilling locations are far away from each other then changing of the fine grain content up to 50 percent is possible. These extreme compositions can for example arise through a periodically increasing pond water surface. The typical particle size distribution, influenced by the ore type, varies in a wide range (Figure 14).

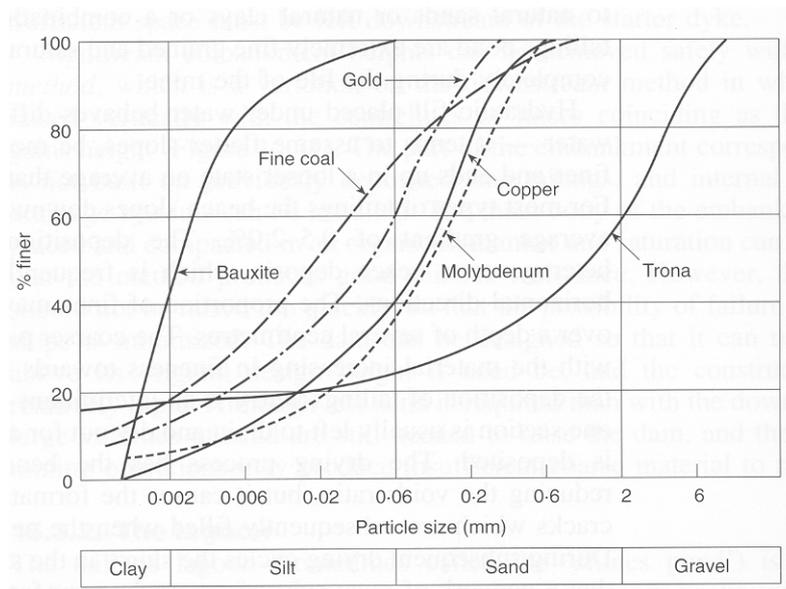


Figure 14: Grading curves for tailings [108]

The so named soft-rock tailings are derived from shale ore. They consist of sand materials. Despite the sandy material, the natural clay content influences the behaviour of the whole

body. Fine tailings have only a little or no sand content. They are included phosphatic clays, bauxite red muds, fine taconite tailings and slimes from tar sands.

Hard-rock tailings are dominated by the sand friction. Slimes are derived from the crashed host rock. Coarse tailings are determined by the sizeable coarse sand friction.

The general characteristics of tailings from different ores are summarized in [108]. Resulting of the different grain size distribution in Figure 15 ranges of characteristic values are given. In comparison to Figure 13 some quantities are different from each other.

Type of residue	Specific gravity	Voids ration	Dry density [Mg/m ³]	Bulk density [Mg/m ³]
Coal				
Slimes	1,4-2,1	0,5-1,1	0,7-1,4	0,8-1,7
Lead-Zinc				
Slimes	2,6-3,0	0,6-1,1	1,3-1,8	1,7-2,2
Molybdenum				
Sands	2,7-2,8	0,7-0,9	1,5-1,6	1,6-1,8
Copper				
Sands	2,6-2,8	0,6-0,8	1,5-1,8	1,8-1,9
Slimes	2,6-2,8	0,9-1,4	1,1-1,4	1,5-1,9
Taconite				
Sands	3,0	0,7	1,8	1,9
Slimes	3,1-3,3	0,9-1,2	1,5-1,7	1,9-2,2
Phosphates				
Slimes	2,5-2,8	11,0	0,22	1,1
Bauxites				
Slimes	2,8-3,3	8,0	0,4	1,2
Trona				
Sands	2,3-2,4	0,7	1,5	1,7
Slimes	2,4-2,5	1,2	1,1	1,6

Figure 15: Typical state parameters for tailings [108]

4.3.3 Permeability

On the base of the Darcy's simple equation for the laminar flow of water to a porous medium with the velocity v and the gradient i the permeability k is defined as:

$$k = \frac{v}{i} \tag{4}$$

$$i = \frac{\Delta h}{l}$$

The gradient itself is defined by the ration between the height difference and the flowing path length. The permeability k depends in general on the grain size distribution, especially D_{10} – the diameter corresponding to percents finer than 10%. Average permeability spans five or more orders of magnitude form 10^{-4} m/s for clean, coarse sand tailings to as low as 10^{-9} m/s for well consolidated tailings [131]. The permeability varies as a function of grain size, plasticity,

saturation, fluid viscosity, depositional method and the depth within the deposit. For prediction of the coefficient of permeability various functional assumptions exist (Figure 16).

Soil type	Relationship
Clean sands (Hazen, 1892)	$k = C(d_{10})^2 \text{ mm/s}$ d_{10} , the 10% particle size (in mm); C , a factor varying between 10 (dense) and 15 (loose state) approximately
Sandy soils (Carmen, 1956 — Korzeny-Carmen equation)	$k = C_1 \left(\frac{e^3}{1+e} \right) \text{ mm/s}$ C_1 , factor of the order of 1 (fine sand) to 100 (coarse sand); e , voids ratio
Soils (general) (Taylor, 1966)	$k = d^2 \left(\frac{\gamma_w}{\mu} \right) \left(\frac{e^3}{1+e} \right) C$ d , equivalent particle size; μ , viscosity of the permeant; C , factor accounting for the shape of the cross-section through which flow occurs; γ_w , unit weight of water
Medium to fine sand (Shahabi <i>et al.</i> , 1984)	$k = 1.2 C^{0.735} d_{10}^{0.88} \left(\frac{e^3}{1+e} \right)$ C , shape factor; d_{10} , finest 10% of the particles; e , voids ratio
Remoulded soft clay (Carrier & Beckman, 1984)	$k = \left(\frac{1}{1+e} \right) \left(\frac{LI + 0.242}{95.21} \right)^{4.29} \text{ m/s}$ LI , liquidity index = $(w - w_p)/(w_L - w_p)$
Clay soils for low-permeability barriers in waste disposal sites (Sarsby & Williams, 1995)	$k_{opt} = \frac{25(1+e)100^{\log d_{10}}}{CU} \left(\frac{d_0}{d_{10}} \right)^2 \text{ mm/s}$ CU , coefficient of uniformity; d_0 , particle size for zero percentage on the grading curve; d_{10} , the 10% particle size; e , voids ratio

Figure 16: Equations for the prediction of the coefficient of permeability [83]

For a preliminary estimate it has been demonstrated [83] that the average vertical permeability for sand tailings is predicted reasonably by the 'loose sand'-form of Hazen's equation, with $C = 0.01$:

$$k = 0,01 \cdot d_{10}^2 \tag{5}$$

More important is the grain size distribution for the permeability of the tailings near the embankment during the precipitation of the fine material of the slurry. The difference in permeability of embankment material and tailings, especially for starter dam, has a high influence on location of phreatic surface (Figure 17).

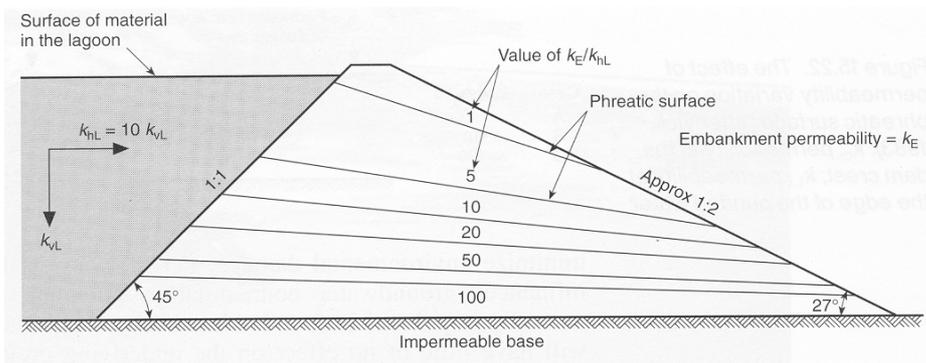


Figure 17: The location of phreatic surface within a stream embankment [108]

The sediment can be subdivided into three groups, the fine material, the transmission zone and the coarse material. The sandy coarse material and the fine material can differ by more than 3 orders of magnitude. The influence of the beach permeability variation for non-homogeneous embankments is shown in Figure 18. In [78] it is shown that Hazen's formula can be extended in application to nonplastic slimes tailings. It is possible to use this together with similar formulas to cycloned sand. Estimates of average permeability on the base of grain size distribution, cannot account for several factors that control the permeability of tailings deposit as a whole. Therefore it is necessary to consider the isotropy behaviour, the distance from discharge and the affects of void ratio.

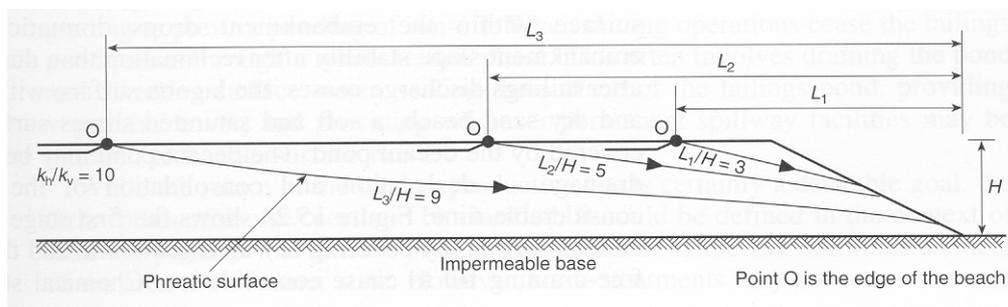


Figure 18: Influence of beach width on the phreatic surface [108]

Due to their genesis, tailings exhibit considerable variations in permeability between the horizontal and vertical direction. The differences in measurements of horizontal to vertical permeability are in the range of 2^{-10} for reasonably uniform beach sand deposits and for underwater-deposited slime zones. The transmission beach zone between the clean coarse sands and slimes shows higher anisotropy factors due to the interlayering of finer and coarse particles. For tailings deposits with not so well controlled deposition, the anisotropy ratio can be greater than 100 [131]. Typically tailings permeability ranges are shown in the following table [131]:

Type	Average Permeability [m/s]
Clean, coarse, or cycloned sands with less than 15% fines	$10^{-4} - 10^{-5}$
Peripheral –discharged beach sands with up to 30% fines	$10^{-5} - 5 \cdot 10^{-6}$
Nonplastic or low- plasticity slimes	$10^{-7} - 5 \cdot 10^{-9}$
High-plasticity slimes	$10^{-6} - 10^{-10}$

Permeability values for tailings as reported in the literature:

Material	Source	k_v [m/s]	k_h [m/s]
Sands	Mittal & Morgenstern (1975) [83]	$2 \cdot 10^{-4} - 9 \cdot 10^{-6}$	-
	Genevois & Tecca (1993) [41]	$4 \cdot 10^{-5} - 5 \cdot 10^{-6}$	-
	Vick (1983) [130]	$10^{-4} - 10^{-5}$	-
Slimes	Genevois & Tecca (1993) [41]	$10^{-8} - 5 \cdot 10^{-9}$	-
	Routh (1984) – China clay [104]	$5 \cdot 10^{-6} - 5 \cdot 10^{-7}$	$2 \cdot 10^{-5} - 5 \cdot 10^{-5}$
	Routh (1984) – Tungsten [104]	$2 \cdot 10^{-7}$	10^{-6}
	Blight (1994) [14]	$10^{-7} - 10^{-9}$	-

The autocorrelation between the permeability and the distance from the discharge location describes the variation of the permeability. The classical model [66] subdivided the impoundment in three zones (Figure 19).

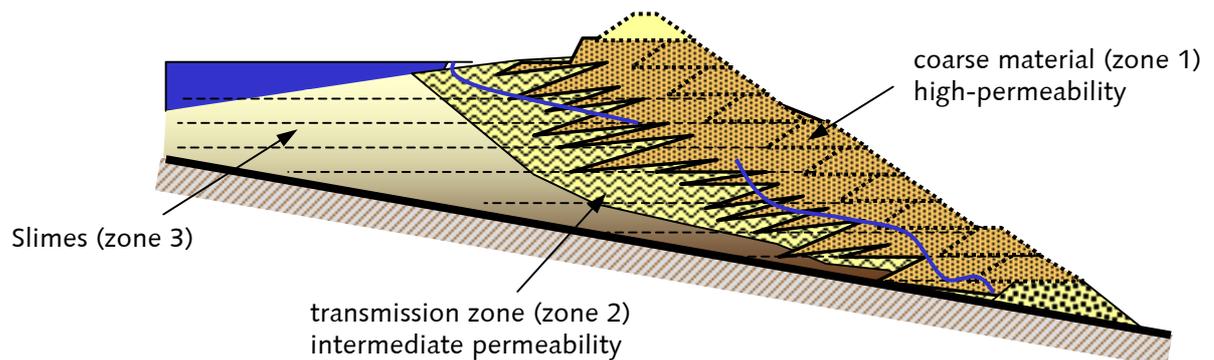


Figure 19: Conceptual model of permeability variation within a tailings deposit

An example can be shown detailing exactly the decreasing permeability depending on the distance from the discharge point of two Tailings Facilities (Figure 20).

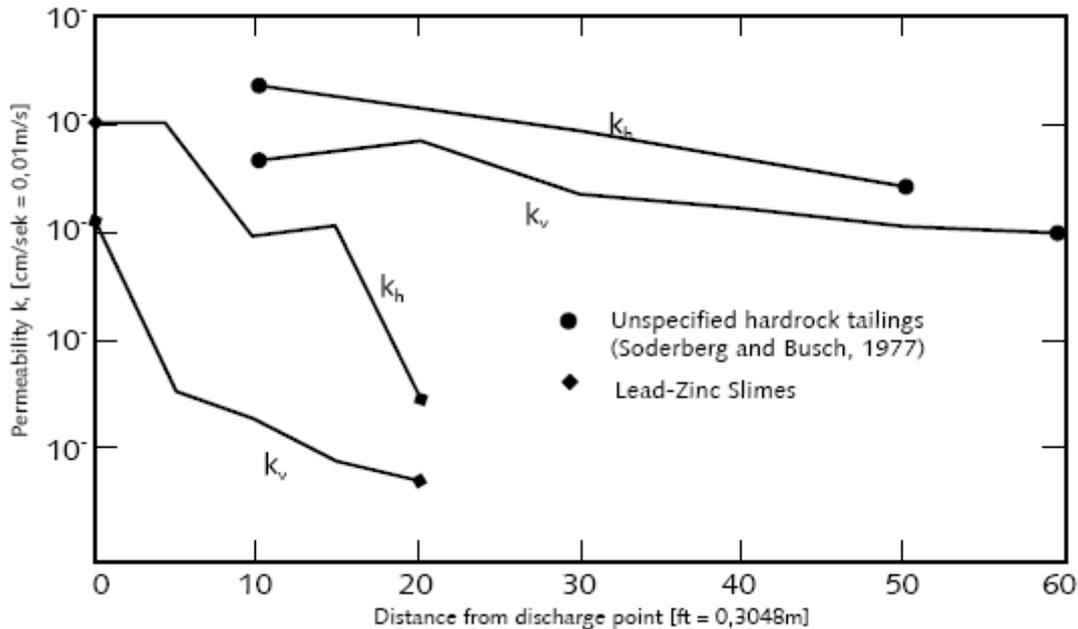


Figure 20: Variation in permeability and anisotropy for well segregated tailings beaches according to distance from discharge [130]

The variation in permeability increases with the range of particle size in the slurry, where the discharge is at a low pulp density and where the discharge points or spigots are sufficiently closely spaced to minimize the deposition of slime layers on the beach. For mechanical separation, e.g. by cyclones, is it necessary to determine the permeability variation by testing. A direct estimation without testing is difficult. The influence of the third effect, the void ratio, has been studied in laboratory tests. Although the permeability differs reasonably, the dependency between void ratio and permeability is nearly linear and is consistent for most tailings sand and low plasticity slimes (Figure 21). The range of void ration encountered over the depth in most tailings deposits. Sands may show a permeability decrease by around a factor of 5. The permeability of slimes may decrease by roughly a factor of 10, because of their higher compressibility. As a result of the greater permeability decrease show slimes layers, which generally control vertical permeability, the anisotropy ratio k_h / k_v may tend to increase with depth in a deposit of interlayered sands and slimes. Phosphate slimes and oil sands sludge constitute exceptions to generalization about void ratio effects. In spite of the high plasticity and clay content, the permeability of these materials can be as great as 10^{-6} m/s at the extreme void ratio that follows sedimentation. By high degrees of consolidation, the permeability may be 10^{-10} m/s or lower.

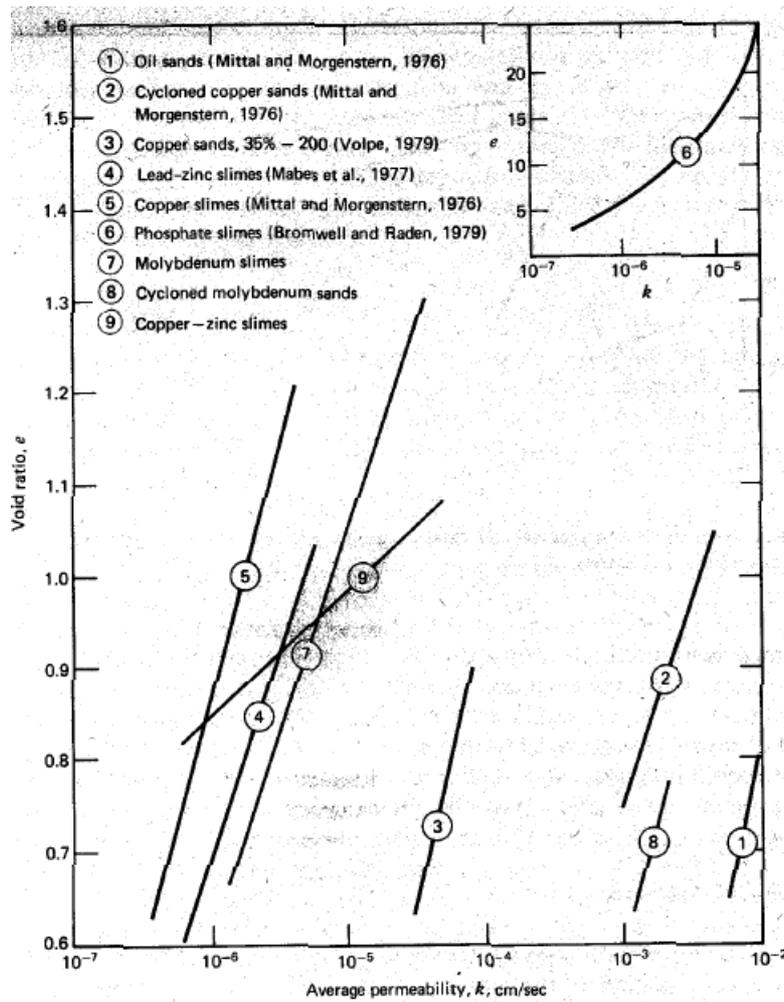


Figure 21: Variation in average permeability as a function of void ratio [130]

For numerical simulations in finite element programs, it is necessary to restrict the differences in permeability between the single soil layers on 10^{-5} m/s otherwise unrealistic flow matrices can appear. Furthermore for numerical calculations the permeability can be expressed as function of void ratio in dependence of the density.

$$\log\left(\frac{k}{k_0}\right) = \frac{\Delta e}{c_k} \quad (6)$$

Here is Δe the change in void ratio, k the permeability during consolidation and k_0 the start value of permeability for beginning consolidation. The parameter c_k and the compressibility index have in general the same range of values [11], [76]. In consolidation calculations the pore water pressures are determined by permeability and not by the material characteristics.

4.3.4 Shear strength

The shear strength of tailings is generally of importance only where tailings are used for embankment construction, or for trafficability on the surface of the tailings [101].

When tailings material is drained, then the shear properties can be determined by conventional soil tests. Here the direct shear box or the triaxial test can be named. Against the usually unexceptional behaviour of drained tailings material, the undrained situation is more complicated. Generally the stress-strain curves in triaxial shear similar to those of loose to medium-dense natural soils of similar grading. Normally there are high strains to failure and small or no reduction in post failure strength at large strains. Dilatency seldom occurs. Typical examples of stress-strain curves for fine tailings, in undrained copper tailings, are shown in Figure 22.

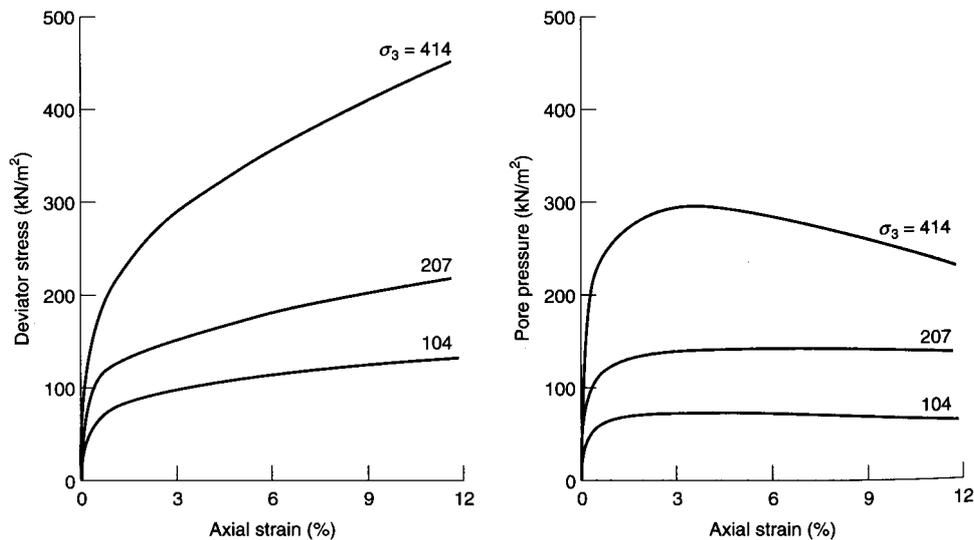


Figure 22: Stress-Strain characteristics of fine copper tailings [108]

The stress-strain curves on the left side normally increase without any peak. For undrained loading of slimes the pore pressure rises with strain to reach a maximum and then remains constant or decreases slightly. The peak point is nearly 5-10 percent of axial strain.

Source	S_u/σ'_v	c'	ϕ'	Comments
Street (1987)	—	0	25–33	China clay sands, relative density 50–60%
Genevois & Tecca (1993)	—	0	42.3	Sands — fluorite tailings
	—	0	35–38	Slimes — fluorite tailings
	0.15–0.22	—	—	Slimes — fluorite tailings
Routh (1984)	—	0	28–38	Sands — tungsten
	—	0	32	Fines — tungsten
	—	0	25–36	Fines — china clay
Wroth & Hughes (1973)	—	0	34	China clay slimes
Vick (1983)	—	0	31–32	Gypsum slimes
Jennings (1979)	—	0	36	Gold slimes
Mittal & Morgenstern (1975)	—	0	33.5	Sands — copper
Charles (1986)	0.2–0.4	—	—	Hydraulic fills — slimes
Volpe (1979)	—	0	33–37	Sands — copper

Figure 23: Shear strength parameters for tailings [108]

Tailings are characterized by low cohesion and high frictional strength parameters, Figure 23. Generally loose depositional notwithstanding, tailings have high drained shear strength owing primarily to their high degree of particle angularity. Therefore tailings show at most an effective friction angle which is 3–5° higher than that of similar natural soils with the same density and stress level. To develop frictional strength requires the dissipation of excess pore pressures resulting from imposed loads.

The effect of void ratio on the effective stress of tailings is small. The friction angle varies between 3–5° with regard to different densities. Similarly, overconsolidation has a relatively small effect on the effective friction angle of slimes tailings. The most important property, influencing the shear strength, is the present stress level. Relatively low stress levels supply very high contact stresses of angular grains. Resulting in the curvature of the strength envelope, especially at low applied stress. The combined effect of particle crushing and dilatancy are mentionable at stresses up to 300 kN/m². At higher stresses the effective friction angle becomes essentially constant [108].

After [108] in situ slime tailings slimes exhibit the same undrained strength profiles as normally consolidated deposits. They can be characterized by the ratio S_u/σ'_v , where S_u is the undrained shear strength and σ'_v is the effective vertical stress. This strength ratio depends on the overconsolidation ratio. Slimes are at most normally consolidated. The strength ratio is the interval of 0.15–0.40.

Sand sized tailings are usually sufficiently rapidly draining so that they may be used for embankment construction by any of the construction methods. Silt sized tailings are generally so slow draining that they cannot be used for embankment construction by any of the conventional methods, except the paddock system in which dewatering (drainage) is achieved

by evaporative drying. Some silt sized tailings can be mechanically dewatered to a sufficient extent to permit "dry" placement. Significantly or predominantly clay size tailings can not usually be used for embankment construction, except by the paddock system. Where silt and clay sized tailings have been dewatered it is necessary that they remain so through effective drainage, if additional embankment construction is to proceed over them. To achieve trafficable conditions on silt and clay sized tailings, they must be dewatered to a partially saturated state to a depth of 0.4 to 0.6 m below the surface.

4.3.5 Effective stress

Void ratios decrease in a soil sample when a load is applied so consequently changing the mechanical properties of the soil mass. This is known as the effective pressure. In a saturated soil, neutral stresses exist within the water and solids in all directions and are equivalent to the pore water pressure. The other stress is a function of the inter-granular pressure between soil particles within a soil mass and is known as the effective pressure. This is represented as:

$$\text{Effective stress} = \text{total stress} - \text{neutral stress}$$

Remembering that for a saturated soil mass, the neutral stress is equal to the pore water pressure. Applying loads to saturated soils causes grain to grain stresses, and increased pore water pressures (as a result of the increased load being partially carried by the water in the voids). The effective stress of a saturated soil mass is demonstrated by the soil structure which is under a load, consolidating with time. The individual grains will compress, but the water content is incompressible. If drainage is impeded then the load is entirely resisted by the pore water pressure. If drainage is sufficient as to allow the water to drain allowing the soil mass to compress then the pore water pressure is dissipated increasing the effective stress and rate of consolidation. Effective stress increases causing consolidation can result from [101]:

- Additional layers of incumbent tailings
- Drawdown of water table due to underdrainage
- Pore suction due to evaporative drying
- Frost action.

For clay soils the drainage rate is usually slow due to the intrinsically low permeability of the soil mass. The effect of initial density or void ratio on the effective stress strength of tailings is small. Over the range of densities commonly encountered in tailings deposits, the friction angles varies no more than 3-5° for sand tailings. If overconsolidation is present, it has relatively little effect on the friction angle for slimes tailings.

An important relation exists between the friction angle and the stress range. Even at relatively low levels of applied stress, stresses at the point to point contacts of the angular grains are very high. When the stress level increases, it produces a partial crushing. The result is the typical curvature of the strength envelope (Figure 24).

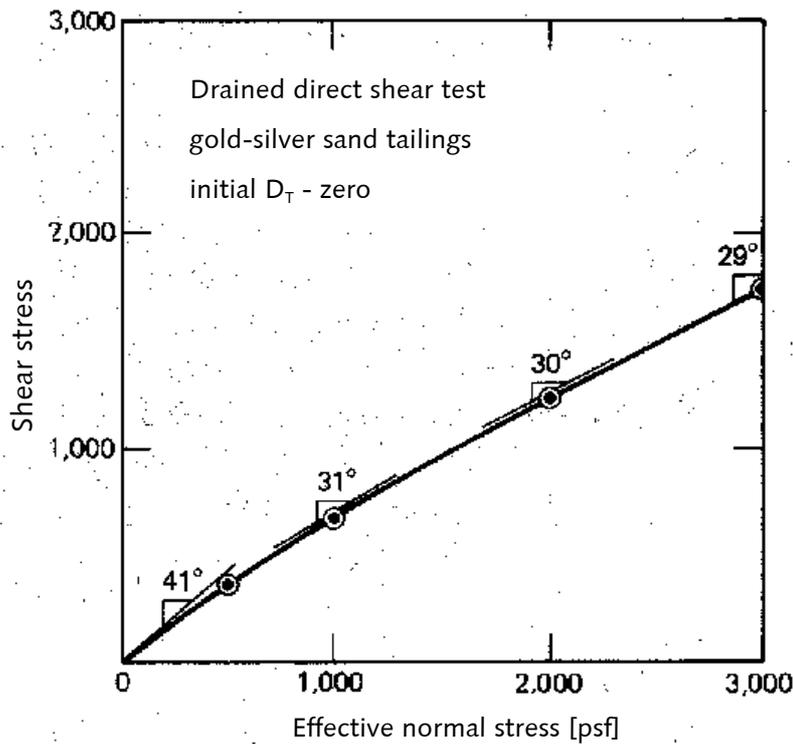


Figure 24: Strength envelope curvature at low stress levels [83]

Data for higher stress levels and denser sand tailings are shown in Figure 25. For these materials, the combined effects of particle crushing and dilatency are most pronounced at stresses up to about 40°. At higher levels the effective friction angle is nearly constant.

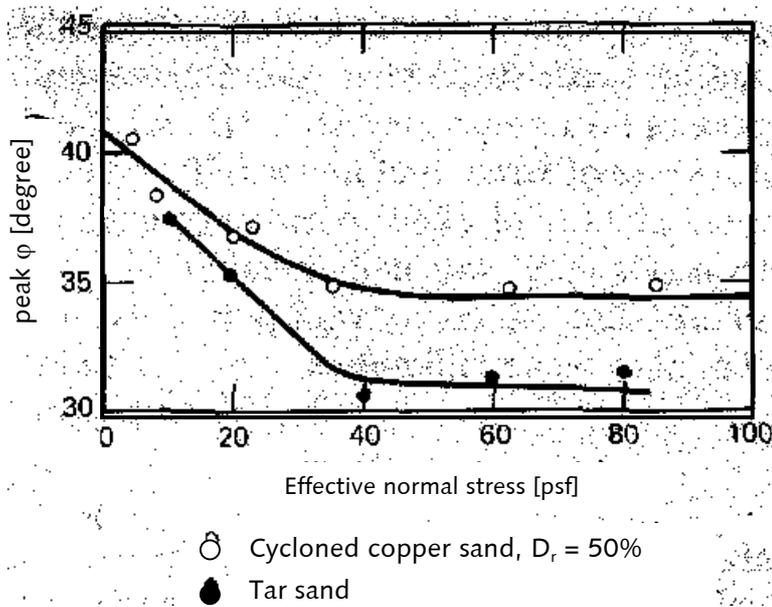


Figure 25: Variation in effective friction angle with stress level [83]

4.3.6 Density

The dry density of deposited material is a function of the type of residue, unit weight and clay content. Hence all parameters that can vary the dry density can be strewn over a wide range. Phosphate slimes with clay proportion have a dry density of 224.3 kg/m^3 while lead-zinc tailings, clay type copper slimes can exceed dry densities of 1762 kg/m^3 . The in-place density can be expressed either in terms of dry density γ_d or in terms of void ratio e . Both quantities are in relation and depend on the depth of the deposit. A low void ratio n results in a high dry density ρ_d with the grain density ρ_s and visa verse.

$$\rho_d = (1-n) \cdot \rho_s \quad (7)$$

Conversely, highest void ratios or lowest dry densities are usually situated near the surface of tailings shortly after deposition. The in-place density depends on the specific gravity, type of tailings, clay content and on the load i.e. the overburden pressure. Because the dry density spans a wide range, the void ratio will be used for further explanations. The grain size distribution and clay content control the void ratio. For instance, the in-place void ratios of sand are in the range of 0.6 to 0.9. Slimes from these tailings types of general low to moderate plasticity show a higher in-place void ratio in range of 0.7 to 1.3 [131]. Slimes of higher plasticity or unusual composition (phosphatic clays, bauxite, oil sands tailings slimes) have a very high void ratio between 5.0 and 10.0. Large impoundment volumes with these materials are often resulting in significant disposal problems. An overview about typical densities and void ratios is given in [131] and Figure 13, Figure 15.

Relative density

Relative density D_r is a measure of the in-place density with respect to the densest and loosest states that the material can attain in laboratory test.

$$D_r = \frac{e_{max} - e}{e_{max} - e_{min}} \quad (8)$$

The slurried tailings have at first a relative density nearly 35 to 60 percent, depending on the transport water [99]. The relative density of sandy residues above the water level from spigots tailings is situated between 30-50%. Other literature sources provide 50-65 percent and 30-40 percent for sand above the water level and 10 to 50 percent, mainly between 25 and 40 percent, for saturated sand [99], [40]. The results from [83] and [96] are presented in Figure 26.

Reference	$\gamma_{d,min}$ [t/m ³]	$\gamma_{d,max}$ [t/m ³]	e_{max}	e_{min}
Mittal and Morgenstern [83]	1,201 – 1,538	1,586 – 1,794	0,72 – 1,23	0,51 – 0,68
Pettibone and Kealy [96]	1,362 – 1,586	1,682 – 2,067	0,99 – 1,32	0,51 – 0,67

Figure 26: Minimum and Maximum Densities of Sand Tailings

For hydraulic deposition of sand tailings the ranges given by various authors are shown in Figure 27. In addition to the spigotted tailings, the data includes cycloned tailings sand. The deposition is done without technical devices and mechanical compaction. Relative densities in this range can be achieved only by relatively clean sands.

Reference	Type	D_r (%)
Mittal and Hardy [82]	Tar Sands	30 – 50
Nelson et al. [87]	Molybdenum Sands	31 – 55
Klohn and Maartman [69]	Cycloned Copper Sands	33 – 54
Mittal and Morgenstern [85]	Cycloned Copper Sands	45 – 68
Brawner [17]	Cycloned Copper Sands	10 – 55
Sandic [106]	Cycloned Lead-Zinc Sands	30
Unpublished	Lead-Zinc Sands	17 – 43
Unpublished	Copper Sands	37 – 60

Figure 27: Average in-place relative density of sand tailings

The relative density of hydraulically deposited beach sands has important influences on dynamic strength behaviour. The derived liquefaction potential is a common cause of impoundment failure and a crucial factor regarding the susceptibility of a soil is the relative density.

4.3.7 Consolidation characteristics

The time rate of consolidation for materials conforming to THERZAGI's theory can be subdivided in primary and secondary phase. Primary consolidation defines the rate of pore water pressure dissipation under constant stress level resulting in a decrease of void ratio and increase of density. This consolidation phase is important for different stability and phreatic line problems but also for all kind of deformation. For example the primary consolidation for sand layers occurs so rapidly, that it is nearly impossible to measure it in laboratory tests. The consolidation coefficient can be expressed as a quotient of the permeability k , the water unit weight γ_w and the coefficient of volume change $m_v = d\varepsilon/d\sigma$ (stress-strain relationship in one-dimensional compression):

$$c_v = \frac{k}{\gamma_w \cdot m_v} = \frac{k \cdot d\sigma}{\gamma_w \cdot d\varepsilon} \quad (9)$$

The small amount of available data suggests that the coefficient of consolidation c_v varies from $5 \cdot 10^{-3}$ to 1 m/s for beach sand deposits [131]. For slimes tailings the consolidation coefficient is in the range 10^{-4} to 10^{-6} m/s, like typical natural clays (Figure 28).

Reference	Type	c_v [cm ² /s]
Volpe [132]	Copper beach sands	$3,7 \cdot 10^{-1}$
Volpe [132]	Copper slimes	$1,5 \cdot 10^{-1}$
Mittal and Morgenstern [84]	Copper slimes	$10^{-3} - 10^{-1}$
Nelson et al. [87]	Molybdenum beach sands	10^2
Blight and Steffen [12]	Gold slimes	$6,3 \cdot 10^{-2}$
Kealy et al. [67]	Lead-Zinc slimes	$10^{-4} - 10^{-2}$
Wimpey [135]	Fine coal refuse	$3 \cdot 10^{-3} - 10^{-2}$
Somogyi and Gray [121]	Bauxite slimes	$10^{-3} - 5 \cdot 10^{-2}$
Bromwell and Raden [18]	Phosphate slimes	$2 \cdot 10^{-4}$

Figure 28: Typical values of coefficient of consolidation

During the consolidation, the effluent in the pore spaces is squeezed out. The result is a closer spacing of the grains. The consolidation under an increase of stress results in a decrease of the void ratio and an increase in the dry density of the tailings. Each tailings material has its own void ratio-effective stress relationship. The density and the stress increase with the increased depth in tailings impoundment. Generally the density increase is higher, when the void ratio is higher at the beginning (Figure 29). Slimes tend to consolidate more than sands for the same stress action. The full consolidation occurs only, when all excess pore water pressure has dissipated. Excess pore pressure dissipation is time dependent. This depends on the stress state, permeability and drainage path length. Contrary to the classical consolidation theory from TERZAGHI the best way to explain the consolidation is by the finite strain theory as by Schiffmann 1980 [109], Carrier and Bromwell 1983 [21], Geocon 1986 [42] which provide more realistic model [101]. It is possible under these aspects to considerably increase the estimates of drainage and consolidation rates. The drawdown water table increases the effective stress in the tailings. This event cause important settlements, particularly in the region of slime ponds. The low moisture content results in consolidation and cracking of surface tailings layers. The high pore suction results from evaporation drying. This method of dewatering and consolidation is exploited to advantage for the creation of the consolidated drained zone of the outer portion of the retaining embankment in the paddock construction system. The increases in effective stress and the reduction of void ratio have been documented by [15] (Figure 30).

Evaporation compaction in the main pond especially results in an increase in tailings density and in the capacity of the tailings facility. Thus dependent on the climatic conditions, it can be necessary to minimize the tailings facility pond size and maximize the beach area.

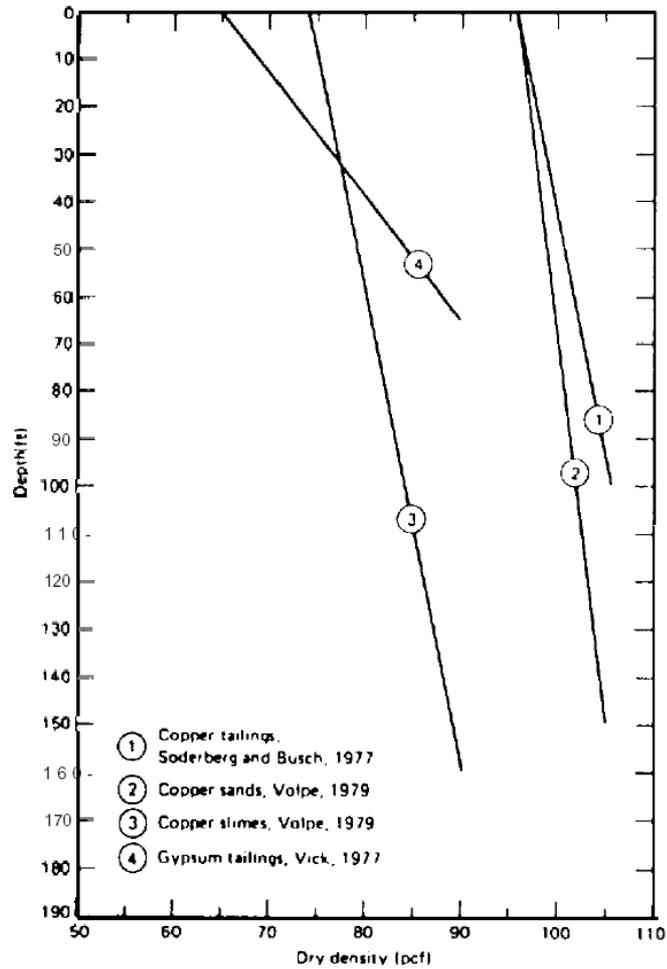


Figure 29: Increase in average in-place density with depth [101]

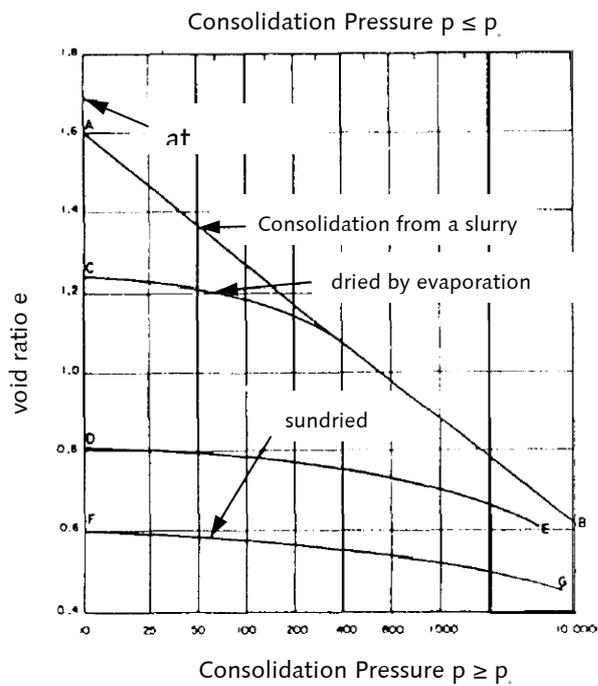


Figure 30: Effect of sun-drying on void ration and consolidation characteristics of tailings [101]

4.3.8 Plasticity of fine grain soils

Plasticity describes the ability of a soil mass to undergo unrecoverable deformation at a constant volume without cracking or crumbling. It is a very important characteristic of fine grained soil, indicating that a significant content of clay and/or organic material is present within a soil mass.

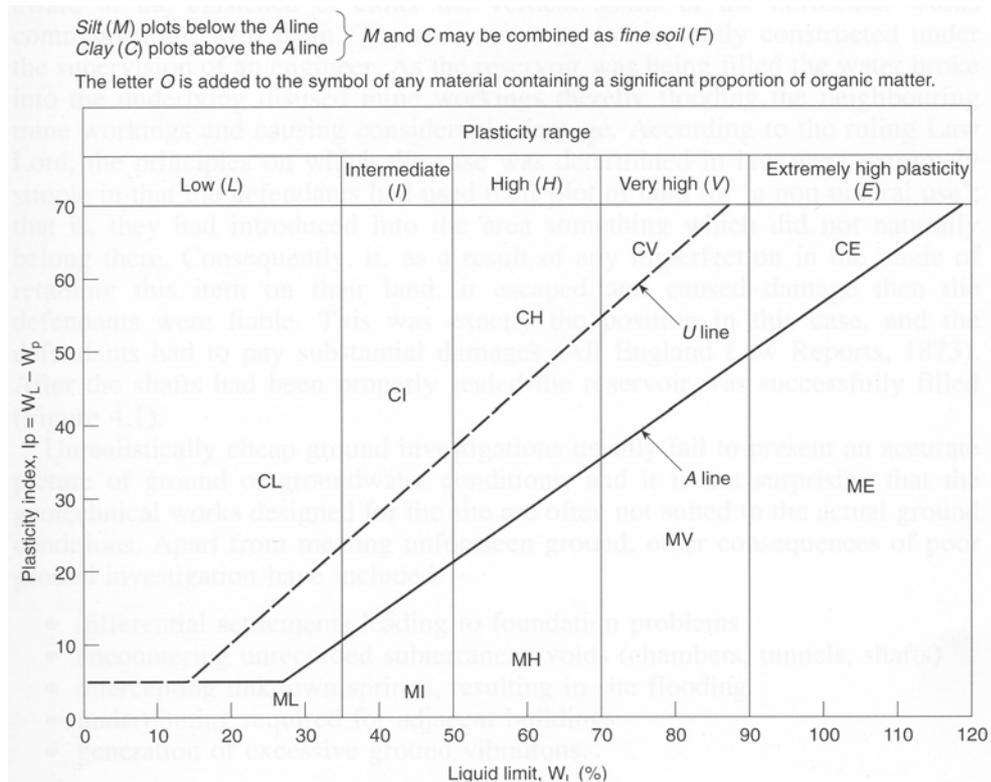


Figure 31: Plasticity chart [27]

Depending on the saturation of a soil mass, a soil may exist in liquid, plastic, semi-solid and solid states. Generally most fine grained soils exist naturally in the plastic state [27]. If there is a decrease in the saturation of a plastic soil, then the capillary layer decreases resulting in the increase of the net attractive forces between particles in the soil mass. The plastic characteristics of a soil occur when the net attractive forces between particles is such that they are free to slide relative to each other, whilst cohesion between them is maintained.

For a soil to exhibit plastic behaviour it must have a level of saturation that lies between the liquid limit w_L and the plastic limit w_p . This saturation zone is known as the plasticity index I_p , and is represented as:

$$I_p = w_L - w_p \quad (10)$$

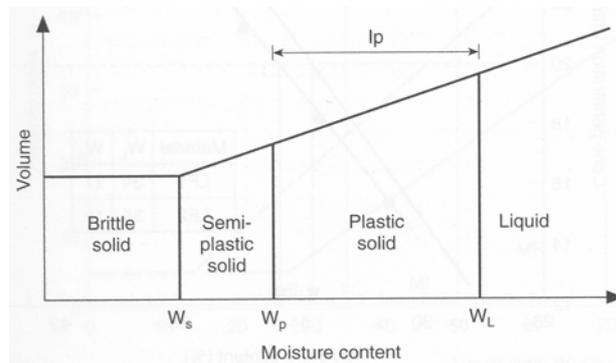


Figure 32: The consistency states of soil [27]

The degree of plasticity of the clay fraction (known as the activity of a soil mass) is expressed by the ratio of the plasticity index to the percentage of clay size particles.

4.3.9 Spatial variability

Within a sand/soil mass there are large spatial variabilities even within homogeneous zones. It is very difficult to determine this spatial variability and so probability methods are preferred which are cheaper and not subjected to uncertainties as a result of measurement errors. Field tests are the only way to understand spatial variability consequences for particular sites and materials. The Monte Carlo simulation technique is one method used to determine soil properties for specific soil conditions.

Soil liquefaction adds further complexity, as the empty pore spaces become liquefied during seismic shaking, which can only be accounted for by considering the stochastic nature of spatial variability of soil properties [97].

4.3.10 Water content

The characteristic of fines is sensitive against the water content. It can be subdivided in both saturated and unsaturated soils.

4.3.11 Chemical properties

Tailings are chemically similar to the parent ore. The presence of process reagents, weathering after deposition and evaporation of water may significantly change their properties. All tailings have been subjected to a physical and/or chemical separation processes. Such processes can be flotation, cyanidation and acid leaching. Constituents of environmental concern may therefore be present in a tailings facility, particularly in the pore fluid, including a variety of metals, chemicals, salts and radioactive materials. In arid regions the process water may also be saline or hypersaline, posing an additional threat to the environment.

The chemical and physical properties for tailings materials are explained in detail in chapter 5.3, 5.4 for the slurry and fine materials.

4.4 *Flow deformation behaviour and liquefaction potential*

Liquefaction of a soil is a temporary state in which the structure of the soil is disturbed, causing the particles to lose contact. If the soil is deposited in a loose saturated condition, so fine sand and silt particles are susceptible to rapid and large reduction in strength due to very minor disturbances.

The different dam construction types are more or less susceptible to liquefaction under seismic (dynamic) events, for example, this motion may result from earthquakes, from nearby mine blasting, or even from nearby motion of heavy equipment. Especially noted in this context can be upstream dam construction or cross-valley dams [131], because tailings deposits typically comprise unconsolidated, saturated deposits of similarly-sized grains, they are susceptible to temporary suspension in water. Liquefied tailings may behave like a viscous fluid and as such they may pass through narrow openings and flow considerable distances. Accordingly, even small dam failures may result in substantial releases of impounded materials if those materials become suspended. Factors affecting liquefaction [125] potential include:

- **Soil type** - Uniform grain size materials, mostly in the fine sand sizes (the typical grading of a tailings material) are the most susceptible to liquefaction.
- **Relative density or compactness** - For a given material, the more compact or dense it is the more resistant it will be to liquefaction.
- **Initial confining pressure at the time subjected to dynamic stress** - This offers an opportunity in certain areas to prevent liquefaction by applying overloads to loose deposits.
- **Intensity and duration of the ground shaking** - Liquefaction may occur due to an intensive earthquake, or due to prolonged earth movement.
- **Location of the water table** - A high water table is detrimental. Consequently, a tailings deposit constructed on a pervious foundation or a dam with a phreatic line kept low by providing adequate internal drainage features may have a greatly reduced potential for liquefaction.

Hydraulically placed tailings sands and silts are usually in a state of low relative density (less than 60%) and if saturated are susceptible to liquefaction under dynamic (seismic) loadings. In areas subject to seismic risk, it is usually necessary to design accounting for the potential of liquefaction. From three dimensional FE simulations and in-situ observations it is known that the soil layers with the highest potential of liquefaction are situated in between 15 m and 20 m deep under the surface [45]. This means the avoidance of upstream construction methods (where embankment sands cannot be compacted) or, if used, the installation of sufficient under drainage and a sufficiently secure placement method to prevent the build up of saturated uncompacted zones in the embankment. Cycloned tailings sands and borrow

materials can be compacted to relative densities above the approximately 65% value required for dynamic stability.

By incorporating drainage facilities, maintaining a low pond surface and compacting the fill materials during construction, the density, saturation, and confining pressures can be controlled to reduce the likelihood of liquefaction. If the tailings embankment is constructed of fine sands, compaction of these sands will increase their density and reduce their susceptibility to liquefaction. Compaction to obtain relative densities of 60% or greater provides reasonable protection by an acceleration of less than 0.1 g. Sandy soils with greater densities, such as are used in the constructed dam crest, are not dangerous. Also partially saturated soil above the phreatic surface is uncritical to liquefaction. However the densification is likely to reduce the effective stress to such an extent, that large deformation occur. Therefore, provided embankment materials possess a relative density of 60% or greater or provided the phreatic surface is maintained at a position well below the embankment surface, the embankment can have a sufficient factor of safety against liquefaction failure. Design calculations generally are needed to verify this for each individual dam.

The risk of liquefaction depends on the grain size distribution and the ration between the horizontal acceleration and gravity acceleration. An important influence for the liquefaction risk of soils is given through the field test methods and laboratory test methods, especially through the device parameters and the boundary conditions of the tests [112]. Dependent on the content of silt and sand the corn size distribution of the soil sample can refer to a standard sieve line district. The quotient of the deduced shear stress and the effective stress in the soil with regard to a defined curve explains the measure of liquefaction risk.

In [112] a simplifying model is mentioned to assess the liquefaction risk with a small range of dynamic parameters. The first step is to determine the shear stress in a soil layer.

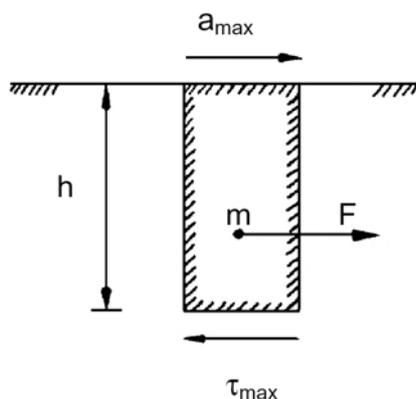


Figure 33: Soil profile as shear profil [74]

$$\tau_{\max} = \sigma_o \cdot \frac{a_{\max}}{g} \cdot r_D = \gamma \cdot h \cdot \frac{a_{\max}}{g} \cdot r_D \quad (11)$$

- σ_o total vertical stress in the soil in depth h
- σ_o' effective vertical stress in the soil in depth h
- r_D correction factor as a function of h , I_D , soil material

The soil isn't a rigid material. Therefore the correction factor was inserted. It describes the non-rigid behaviour of the soil as a function of the depth h , regarding compactness I_D and in-situ soil (Figure 37). The maximal acceleration of the assumed earthquake will be reduced by a factor of 0.65, so that shear stress cycles have constant amplitude:

$$\tau_{av} = 0,65 \cdot \tau_{\max} \quad (12)$$

- τ_{av} shear stress under the assumption of constant amplitude
- τ_{\max} maximum shear stress

The loads are set in cycles with constant boundary conditions on the sample soil. The number of repetitions depends on the earthquake magnitude.

Magnitude	N _{eq} [Zyklen]	Zeit [s]
8	30	60
7,5	20	40
7	10	20

Figure 34: Dependence of the cycle numbers and the earthquake magnitude [74]

The results of the laboratory tests are be scaled to an in-situ situation using the correction factor above.

$$\tau_{av} = \tau_{10} = \sigma_o' \cdot \left(\frac{\sigma_{dc}}{2\sigma_a} \right)_{50} \cdot \frac{I_D}{50} \cdot c_r \quad (13)$$

- τ_{av} shear stress under the assumption of constant amplitude
- σ_o' effective vertical stress in the soil in depth h
- $\frac{\sigma_{dc}}{2\sigma_a}$ coefficient
- c_r coefficient dependent on the relative density

The last two coefficients are a measure for liquefaction under the acceptance of d_{50} for $I_D = 50\%$. This equation is able to determine an $a_{max, in situ}$. Should the maximal acceleration a_{max} of the earthquake be less than $a_{max, in situ}$ the liquefaction of the soil is improbable.

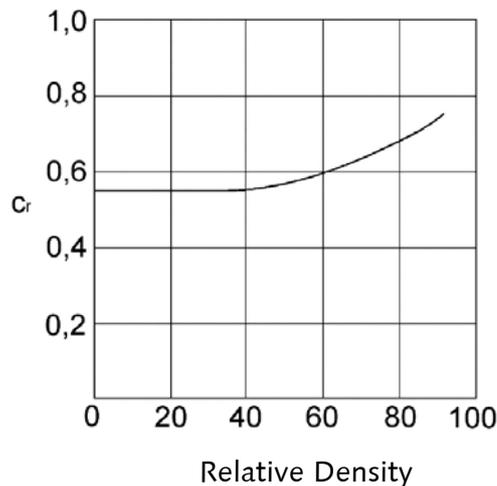


Figure 35: Dependency between c_r und relative density [45]

Another simpler method is explained in the KTA 2201.2 [72] explained. The sieve line of the in-situ soil is marked in the diagram below.

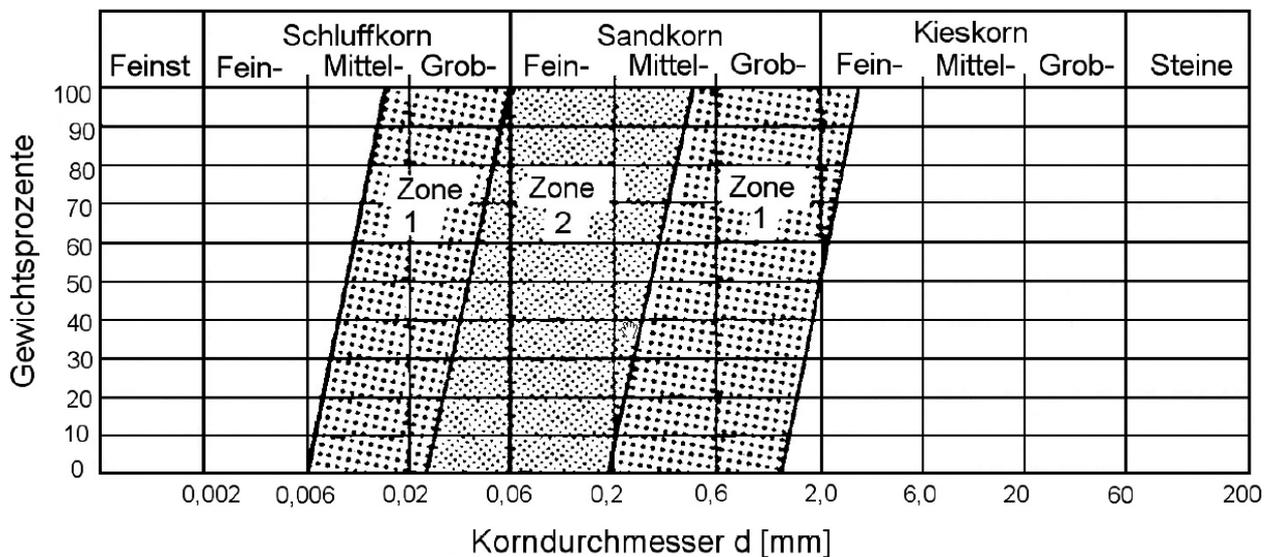


Figure 36: Risk of liquefaction regarding the sieve line of in-situ soil [45]

The major part of the registered sieve line determines the "Zone". Following is the dynamic shear stress ratio calculation τ_{max} / σ'_o .

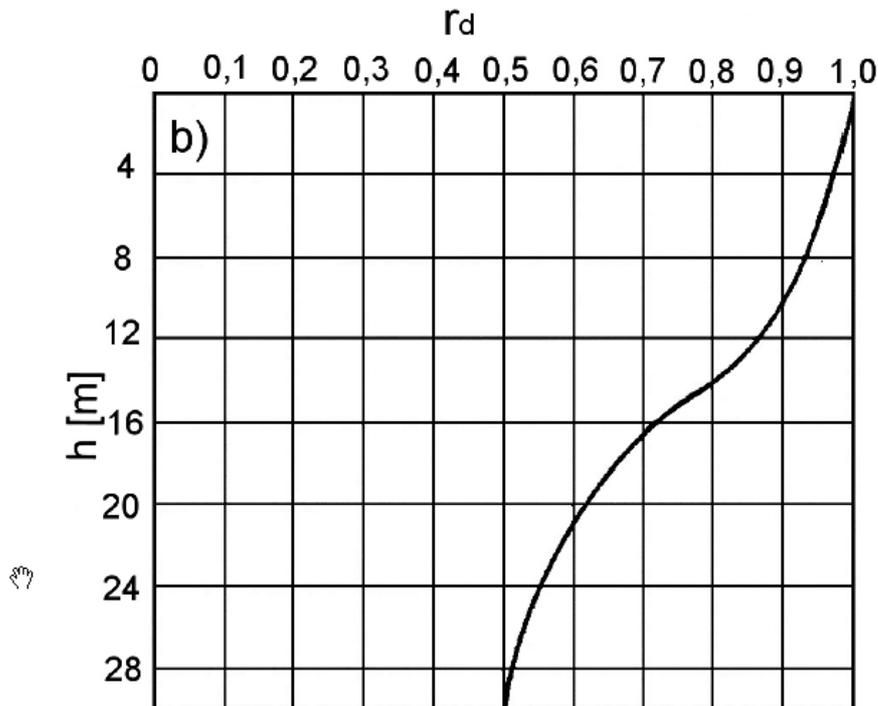


Figure 37: r_d in dependence of depth h [45]

As previously mentioned, the soil layers between 15 m to 20 m deep are especially endangered against liquefaction and flow deformation. In [112] an estimation method is shown based on stress cycles of the SPT-test (drop sounding test) is shown. This method defines two extreme situations:

- | | | | |
|----|--------------|------------------|---------------|
| a) | favourable | $d_{50} = 0.075$ | $N_{eq} = 20$ |
| b) | unfavourable | $d_{50} = 0.200$ | $N_{eq} = 10$ |

For a given data set under the condition of a) a risk of liquefaction exists when the point is below the limited state line. For the condition b) a risk of liquefaction exists when the datapoint is situated above the limited state line.

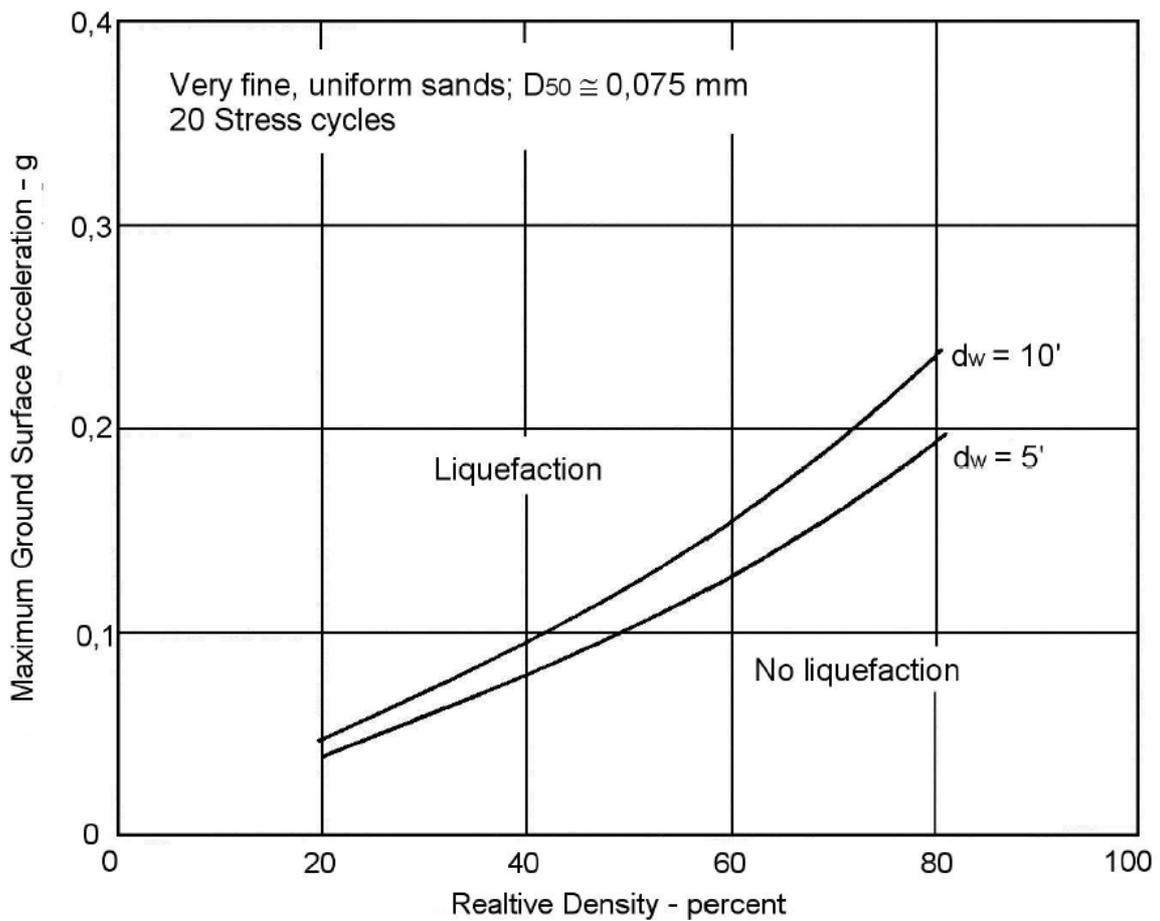


Figure 38: Limited curve for risk of liquefaction for fine sand with 2 groundwater layers [45]

Dependency on the present stress cycles and the existing maximum acceleration with respect to the both conditions a) and b) the risk to liquefaction can be subdivided into 3 parts:

- improbable risk to liquefaction
- possible liquefaction with respect to the soil type and dynamic magnitude
- very probable risk to liquefaction.

For ranges, where it is impossible to make exact specifications, detailed field investigations are necessary.

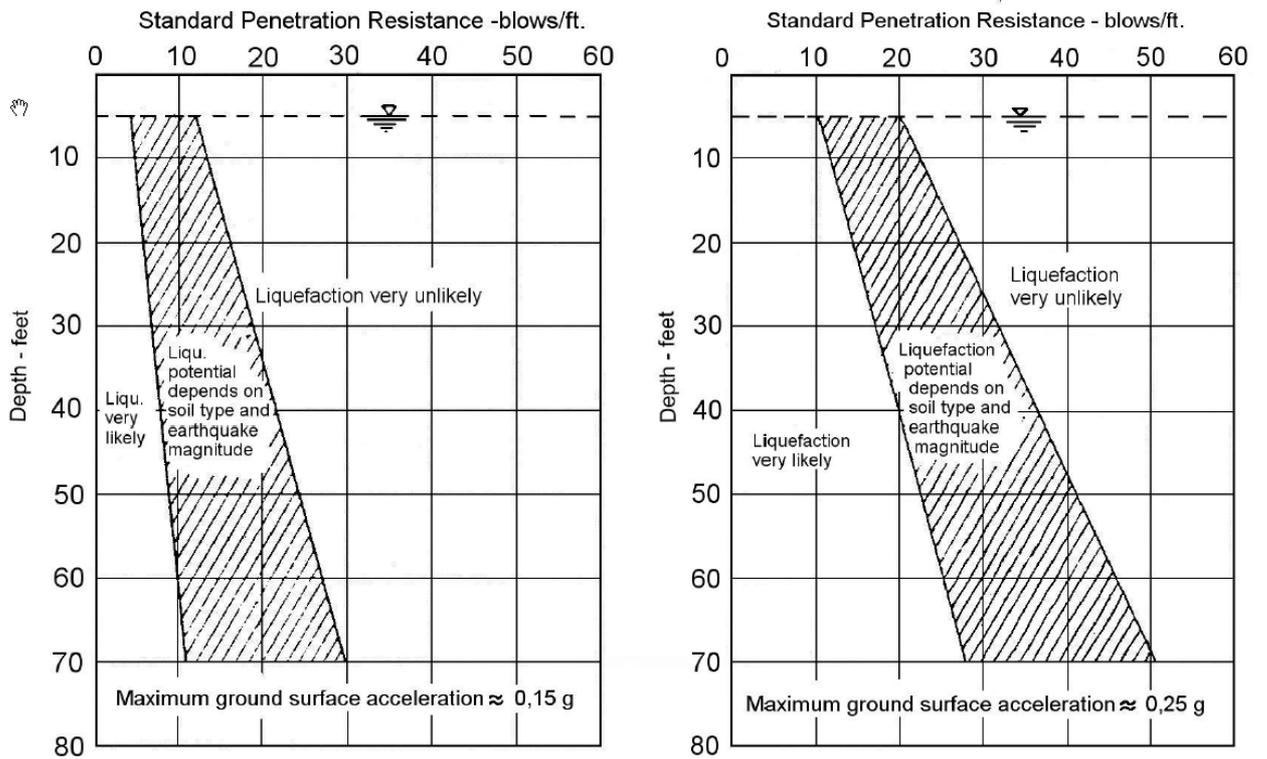


Figure 39: Regions of stress cycles for estimate the risk of liquefaction (1ft = 0.3048 m) [45]

4.5 Variations of the properties of fine material by dynamic events and different natural hazards

The term "dynamic soil properties" relates to the mechanical properties of soils under dynamic loading, these are shear wave velocity v_s , shear modulus G , damping ratio D , and Poisson ratio ν . The primary cause of slope failure with tailings dams constructed of saturated cohesion-less soils is the build up of pore water pressure and the consequent reduction in shear strength. Hence stability analysis is commonly performed using pseudo static or deformation approach. When a soil mass is subjected to wave propagation, low levels of strain are induced. However, when soils are subjected to dynamic loading, stability is a concern as large strains are experienced. For tailings impoundments seismic activity is the primary hazard that induces dynamic loading across the entire area of the impoundment site. The embankment, impoundment and tailings will all experience variations in the seismic energy, but general dynamic soil parameters will apply.

Natural hazards

Natural hazards pose a threat to disused and operating tailings impoundments. The main hazards that cause a potential risk of failure are seismic events, and flooding caused by storms or heavy rains. There have been in the past many failures that have occurred due to these two types of natural hazards. Another hazard is frost causing freezing of the water and air within the pore spaces of the soil mass. These three hazards are discussed and examples of specific dam failures are stated.

Seismic events

Seismic loading is the biggest threat to a tailings impoundment and creates a great challenge for geotechnical engineers. There are many parameters involved in producing stresses on soils and material used in the construction of an impoundment. Establishing ground stability during earthquakes is a complex procedure involving fields of geophysics, geology, seismology, and geotechnics. Earthquakes can cause both rupture and stability in the ground by horizontal and vertical displacement, caused by tectonic plate convergence and divergence.

Soft soils can increase the amplitude of seismic waves within soil layers close to the ground surface. The spatial variations and the saturation level of the soil can cause reflection and refraction of certain waveforms, which can alter the ground motion characteristics on the surface [133].

Settlement and liquefaction

The settlement and packing of a soil mass induced by seismic shaking, plays an important role. For granular soils in various states of saturation the amount of settlement can be assessed during cyclic stresses by monitoring the variation in the void ratios [5]. The variation in the relative density as a result of seismic activity can be evaluated by the stratification and the initial relative density of the deposit.

Most tailings dam embankments contain some saturated soils that are known by assessing the depth of the phreatic surface. Where drainage is restricted or even impeded, seismic shaking can cause an increase in pore pressure reducing the strength of the embankment. Shear stresses build up in the soil mass within the embankment and when the pore water pressure equals the external load the soil begins to act like a fluid. When this occurs, normally under extreme seismic conditions, liquefaction has occurred. If very strong shaking occurs then high consolidated soil and clays are subjected to liquefaction events. Failures can occur along fracture planes, fissures or along variations in the properties of a soil mass. The risks of liquefaction events occurring can be reduced by understanding the contributing parameters of a soil mass.

Relative density and confining pressures

For tailings embankments the relative density should be a minimum of 0.7 (70%) to avoid liquefaction events from occurring. This is for an impoundment that has low confining pressures and is surrounded by material that cannot liquefy. For impoundments situated in areas of high seismic activity, for example Turkey, and embankment drainage is restricted in any way, then relative density should increase to 0.8 (80%). For embankments being raised, a high relative density can be obtained by using clean cycloned sand materials, or by using the hydraulic fill placement and compaction technique. This is also cost affective compared to using mechanical compaction techniques. The varying grain size distribution and shape variation of grains can result under seismic loads in a deviation of the cyclic parameter in the liquefaction calculation.

If an embankment has a low relative density material and high confining pressure then the risk of liquefaction is greatly increased. With the added complexity of tailings characteristics for individual sites, the cost associated with measuring the dynamic behaviour of an embankment material is complex, assuming tailings are used for the raising of the embankment. It is therefore best practice to predict the risk of liquefaction by means of synthetic criteria rather than *in-situ* or field testing. One such field test for determining the relative density is the standard penetration test (SPT). By far one of the best methods to use to predict the risk of liquefaction is to concentrate on the seismic data collected for a particular site, rather than establishing and predicting worst case scenarios.

Plastic (cohesive) soils with little or no consistency behave differently to cohesion-less soils. The main difference being, that once a cohesive soil undergoes rapid loading the soil mass

gains strength and alternating the load reduces it. If a cohesive soil (particle size < 0.005mm) contains clay greater than 15%, then generally soil liquefaction doesn't occur [113].

Liquefaction can occur for embankment slopes with an angle as low as 20°. Once liquefaction has begun the slope can flow and subside to an angle of repose as low as 4°. For a tailings embankment that has undergone liquefaction the slope will only flow and settle over a short distance. The major impact to the environment occurs when the tailings are released as a consequence of embankment failure. There have been hundreds of tailings embankment failures all over the world as a result of liquefaction.

4.6 *Structural stability with respect to the soil parameter*

4.6.1 Common failure modes

Potential causes of impacts on the environment from Tailings Facilities fall into the following broad categories [2]:

- **Structural failures** – These are cases generally involving the collapse, subsidence or slippage of a part of a containment structure such as a dam wall. Such failures can give rise to the discharge of large quantities of tailings and treatment solutions and lead to very severe environmental damage as a result. In extreme cases such events can threaten human life due to inundation or damage inflicted on structures.
- **Operational failures** – In these cases the root cause of the incident is the failure to operate or control the operation of a facility adequately. For example, failure to monitor the water level of a TSF could result in an overflow. Lack of adequate operational control can ultimately result in a structural failure. For example, if water levels were allowed to rise to the point of overflow, the dam wall could then be eroded and fail.
- **Equipment failures** – Wherever mechanical equipment such as pipelines, pumps, valves etc used for tailings management activities there is potential for mechanical failure. Burst pipelines, coupling breakage and pump failure are common causes of accidental discharge. Failures of this kind are not usually as serious as those due to operational or structural causes but can nevertheless cause significant harm to the environment.
- **Unforeseen consequences** – Some instances of damage to the environment are the result of simple oversight in the design, operational procedures or closure of a TSF. For example, the long term consequences of slow seepage from the base of a TSF may not have been adequately considered at the design stage and may not become apparent for some years.

In the following, the structural failures shall be identified and examined. The operational, equipment failures and unforeseen consequences are themes for the work packages intervention and prevention.

4.6.2 Failure modes for embankments

For the determination of the reliability or detailed safety factor, the analysis of the potential failure surface of the embankment is necessary. There are a number of common failure modes to which embankments may be vulnerable. These include slope failure from rotational slide, overtopping, foundation failure, erosion, piping, and liquefaction. Each failure mode may result in partial or complete embankment failure. The single modes are described as follows [125].

Rotational sliding, so named because the failure surface appears as a segment of a horizontal cylinder, may result in slope failures ranging from local sloughing of tailings at random areas along the face of an embankment to massive circular arc slides extending over the entire structure. In general, for a stable slope, the shear strength resisting movement along a potential failure surface exceeds the shear stress tending to induce movement. Instability occurs when the shear stress on the failure surface equals the shear strength [131]. Limitations of use for natural soils can be derived. The arrangement of the different soil structures in the dam have to guarantee that the phreatic line doesn't exceed the maximum high, like in the dam stability analysis is mentioned. Therefore, especially for using natural soils, is between the various internal zones and the core material a sufficient difference in permeability to control the phreatic line and to prevent breakout of seepage on the embankment face necessary. The permeability can be influence like mentioned above. Another limitation is the moisture content of natural soils. When available natural soils are at or near saturation because of high groundwater levels is the material handling difficult and compaction to densities required for adequate strength may be impossible. Shall the deposit or natural soil use then is drying before necessary. In contrast to natural soil deposited tailings material can be used for increasing the embankment construction. For embankment raised less than 5-10 m each year excess pore pressure may be assumed to dissipate as rapidly as the load is applied [130]. For raising the embankment more than 10 m each year, so the embankment has to be modelled as a series of discrete raises. In dependency on the distance to the discharge location are exist different grain size structures. Any use of tailings material in embankment constructions requires careful attention in filter design and moisture content. The different heights of lifts of embankments make the control of compaction difficult. Equivalent to natural soils are the risks of increasing phreatic line or liquefaction by increasing the effective stress to minimize. Specifically, causes of rotational failure may include changes in the water table, changes in the permeability of the foundation materials, disturbances to the embankment caused by vibration or impact loading, settlement of the foundation materials.

Slope stability analysis for tailings embankments concentrate on initial rotational-type, slides incorporating the rigid-body assumption of limiting equilibrium analysis. These analysis types

can only represent the possible failure mode, but a prediction for behaviour after loose the stability are not possible.

Case	Relationship
Dry slope	$\text{FOS} = \frac{\tan \phi'}{\tan i}$
Seepage parallel to the slope, phreatic surface at ground level	$\text{FOS} = \left(\frac{\gamma - \gamma_w}{\gamma} \right) \frac{\tan \phi'}{\tan i} = \left(1 - \frac{\gamma_w}{\gamma} \right) \frac{\tan \phi'}{\tan i}$
Horizontal seepage, fully saturated soil	$\text{FOS} = \left(\frac{\gamma - \gamma_w \sec^2 i}{\gamma} \right) \frac{\tan \phi'}{\tan i}$

Figure 40: Approximations for the factor of safety against shallow failure within tailings dams [106]

The use of numerical methods is advantageous in contrast to analytical calculation methods. Here the different lifts can be separately described with different constitutive models. The deformations of different stage constructions and applied load steps can be added, so that the previous pre-deformation calculation step can be related to the next one.

The exact determination of the safety factor requires an extensive study of the physical properties. The density functions of each calculation parameter or their bounded ranges can be integrated in computer model sampling routines. The results of these models are again density function or ranges. The result isn't a determined value, but a probability measure for failure of construction.

Foundation failures are not uncommon among earthfill structures. Where a weak layer of soil or rock exists at shallow depth in the foundation below the structure, movement along a failure plane will occur if the earthfill loading produces stresses in excess of the shear strength of the soil in the weak layer. Classic design measures for weak foundations include flattening embankment slopes to an inclination consistent with the initial undrained shear strength of the foundation material. When the tailings embankment is raised to its ultimate height, which usually needs many years, then significant pore pressure dissipation and the corresponding increase in undrained shear strength will be occur. An analysis for a tectonic tailings embankment on a weal peat foundation is described in [123]. The enlargement steps are to fast, so that previous soil layers can dissipate the pore pressure. The problem was solved by modifying the impoundment layout. A larger impoundment area can store the same volume of tailings but with a lower ultimate embankment rise in a slower time.

One of the most common causes of failure is **overtopping** by flood waters. Overtopping typically results when the volume of run-on entering an impoundment, from improper diversion of surface water flows or excessive storm water flow, exceeds the capacity of the impoundment. Because tailings embankments are constructed of highly erodible materials, the friction caused by rapid flow over an unprotected embankment crest may quickly erode a gully in the fill material, allowing sustained release to occur. Additionally, a rapid increase in pore pressure associated with large storm water inflow may result in the liquefaction of unconsolidated impounded sands and slimes. Sustained high flow over the crest of an embankment can thus result in a major failure of the overall impoundment within minutes.

In areas of heavy rainfall some form of protection against **erosion** is usually required. Tailings embankments may be susceptible to erosion failure in two major areas, embankment abutments and the embankment face. Erosion along the contact line between the embankment and the abutment may result from storm water flow that concentrates there. Typically, this type of failure is preventable with proper storm water diversion methods and therefore results from faulty design or maintenance. Erosion of embankment faces may result from rupture in tailings lines installed on the embankment crest. Again, maintenance (and alternate siting of tailings lines) may prevent this type of failure.

Piping refers to subsurface erosion along a seepage pathway within or beneath an embankment which results in the formation of a low-pressure conduit allowing concentrated flow. Piping may result from the phreatic line exiting the face of an embankment with sufficient velocity to erode the embankment face. The resulting void space promotes progressive erosion extending upstream toward the source of the seepage. In the worst case, the seepage may result in the creation of a direct channel from the tailings pond to the dam face. Excessive piping may result in local or general failure of the embankment or the embankment foundation. The filter requirements have a great influence on piping. They are well established from the conventional water dam design. A comprehensive list of filter requirements is listed in [138] and a short summary is shown in Figure 41.

Autor (Jahr)	Kriterium	Grenzzustand	Grundlage
PRINZ (1923)	$d_F/d_B \leq 4,42$	Eindringen von Basis- körnern in Filter	Geometrische Betrachtungen unter Kugelannahme
BERTRAM (1940)	$d_{15F}/d_{15B} \leq 9$ $d_{15F}/d_{85B} \leq 6$	Reduzierung der Filter- durchlässigkeit	Experimente mit eng- gestuften Sanden
TERZAGHI und PECK (1948)	$d_{15F}/d_{85B} \leq 4$	beinhaltet nicht quanti- fizierte Sicherheit	nicht näher beschriebene Experimente
SICHARDT (1952)	$d_F/d_B \leq 2,42$ $\leq 6,46$	Eindringen v. Basisk. in locker/dicht gelagerten Filter	Geometrische Betrachtungen unter Kugelannahme
U.S. CORPS OF ENGINEERS (1953)	$d_{15F}/d_{85B} \leq 5$ $d_{50F}/d_{50B} \leq 25$	Filter hält Basis nicht zu- rück. Beinhaltet Sicherheit	Experimente mit enggest. Sanden und versch. Filtern
LEATHERWOOD u. PETERSON (1954)	$d_{15F}/d_{85B} \leq 4,1$ $d_{50F}/d_{50B} \leq 5,3$	Reduzierung des Druckver- lustes an Kontaktflächen	Experimente mit engge- stuften Sanden
KARPOFF (1955) (U.S. BUREAU OF RECLAMATION)	$d_{50F}/d_{50B} < 5:10$ (enggest. Filter) $d_{15F}/d_{15B} < 12:40$ $d_{50F}/d_{50B} < 12:58$ (B.+ F. weitgest.)	Bereich des maximalen Durchflusses bei konstanter Druckhöhe	Experimente mit jeweils einem Basiserdstoff und variablem Filter $C_{U,B} < 30$; $C_{U,F} < 28$

Figure 41: Filter criteria and grain size distribution [138]

Although obtaining proper grades of filter materials is often difficult, they may be essential to the proper functioning of the whole structure. When it isn't possible to obtain graded material from the deposit for the different construction elements of the dam, then the use of synthetic filters is to be preferred. Considerable care with regard to the filters is required during impoundment operation.

4.7 Typical residues and abnormalities

Tailings are the residue of the milling process that is used to extract metals of interest from mined ores. During this process, ores are first milled and finely ground, and then treated in a hydrometallurgical plant. Since the extracted metal represents only a small percentage of the whole ore mass, the vast majority of the material mined ends up as fine slurry.

The tailings contain all other constituents of the ore but the extracted metal, among them heavy metals and other toxic substances. Moreover, the tailings also contain the chemicals added during the milling process. In addition, as a result of the milling process, all these contaminants now are more easily available for dispersion into the environment than when in the original ore. Also, the mechanical stability of the tailings mass is very poor, due to its small grain size and the usually high water contents.

Considering the landscape morphology, a tailings pond could be located on a valley (cross valley impoundment type), at the base of a slope (side valley impoundment type), on a plane land (ringed impoundment type). Disposal of granular materials presents certain particularities depending on the specific type of tailings pond.

Valley type tailings ponds

Depending on their construction mode, the tailings ponds have a different internal structure (Figure 42):

- upstream method of tailings dam construction (arising towards interior)
- downstream method of tailings dam construction (arising towards exterior)
- centreline method of tailings dam construction (central vertical arising)

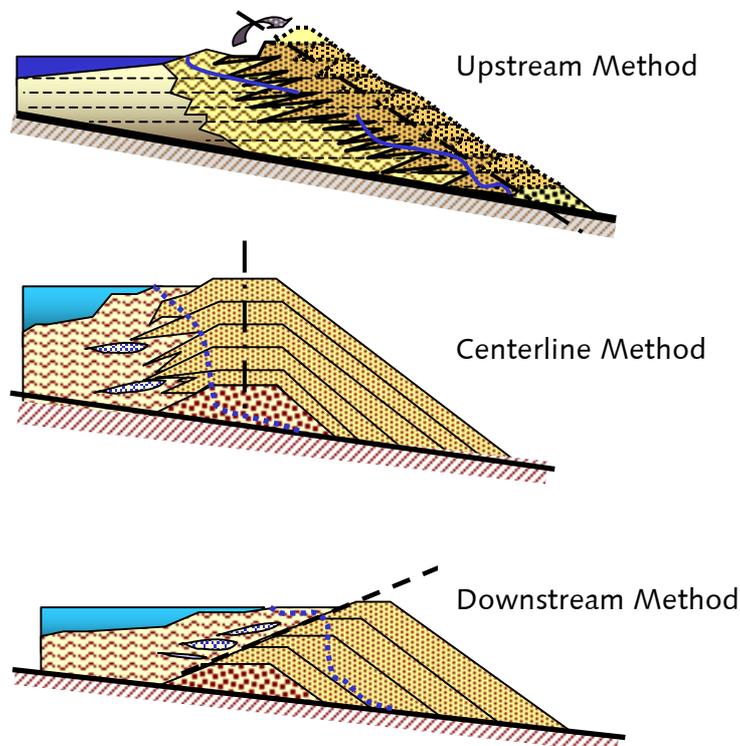


Figure 42: Construction methods of tailings dams

In the tailings pond after the pulp has overflowed onto the beach, the deposition of the material from suspension starts according to the grain sizes: first of all sand, then dust and at the end, the clayish fraction. The limit between those three fractions is not a distinct limit but a ragged one. The reason is that the flow speed of the pulp is not uniform over all its extent.

The maximum speed is in the centre and decreases towards extremities so that a part of dusty or clayish fraction is deposited in the sandy fraction body. That also happens to the overflowing interruption when as flow speed decreases, the fine fractions deposit near to the

dam. The layers of fine materials have a thickness of few centimetres, varying 10-20 percent according to spill technology. The variance increases with the distance of the spill locations. The normal deposition manner of the three grain fractions in the valley type pond with upstream, downstream and vertical method of tailings dam construction is schematically presented in Figure 42. We notice that in the three situations, the slope of the grain particle layers conforms to the dam height.

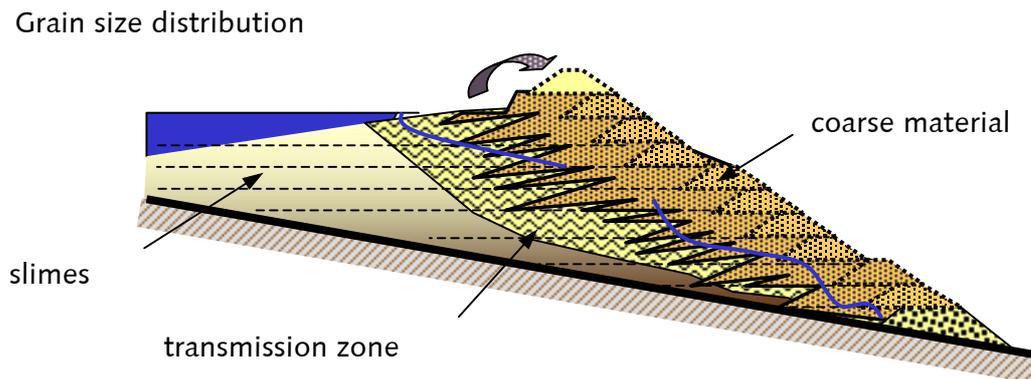


Figure 43: Upstream tailings dam

This method, while available at low cost, implies a number of specific hazards for dam stability. These hazards require a thorough assessment, continuous monitoring and control during siting, construction and operation of the dam. Experience shows that these conditions are not often maintained.

4.7.1 Coastal and field type tailings ponds

Coastal and field type tailings ponds have three and four sides respectively which do not benefit by any natural support. A profile through any side without natural support presents a pond internal structure similar to the one shown at the valley type pond with upstream method of tailings dam construction. This is the reason the internal structure of those ponds follows the same manner of the grain particles deposition. Starting from the dam towards inside the order of deposition is: the sandy fraction, the dusty fraction and then the argillaceous fraction.

4.7.2 Abnormal situations in the internal structure of the tailings pond

As stated above, the normal order of material deposition in a tailings pond from the dam towards the beach is as follows: the sandy fraction, the dusty fraction and then the argillaceous one. In the building and operation process of a tailings pond critical situations can appear such as: pulp overflow interruption as a result of flotation process stoppage, the overflow of a high clayey content pulp, damages to the hydrocyclone system or of the pure or pluvial water discharge system and special natural phenomena (earthquakes), all of which are able to produce a disturbance of the typical sediment deposition process in the pond. Thus, in the pond body pockets of clayey material with high humidity (mud pockets), subsidence funnels or simply the liquefaction of the whole material from the pond can appear. The fine residues are saturated and can possibly never finish the consolidation.

4.7.3 Mud pockets

Clayey material pockets can appear in the condition of the pulp overflow stopping when a deposition of the clayey material takes place close by the dam in the area where the clayey material usually deposits or when the pulp has a high content of clayey fraction. Thus, within the sandy zone an unsubstantial clayey material pocket with high water content (mud pocket) can be formed which endangers the pond stability and sometimes stands out due to the leakage excess which can appear even in the drought periods, i.e. the Valea Selistei pond in Romania where an important leakage appears after a period of abundant precipitations. This pond started as a valley type pond with downstream method of tailings dam construction. After a time, it became obvious that the hydrocyclone method is not an adequate method for the production process and it was abandoned, the pond was re-built with upstream method of tailings dam construction. Thus, the natural manner of sediments deposition in the pond was altered and a clayey material pocket was formed which worked as a collector of precipitations waters.

4.7.4 Subsidence funnels

The main causes of subsidence funnels are the suffusion processes or damage to the pure water discharge system. Suffusion is the carrying process of the finest particles by the ground water as it runs through sandy rocks under a critical hydraulic gradient. Sandy sediments from the tailings pond body are easily affected by suffusion as the ground water from the pond has a high hydraulic gradient whilst sand compaction is low. If there is a suffusion void in the leakage zone from the slope of a tailings pond towards it, the flow lines of the ground flow will focus so increasing the carrying away capacity of the sandy particles. As a result, the void develops inside the pond on the direction of the flow lines. Because of the suffusion void subsidence there is a subsidence funnel which comes out at the slope surface or on the pond beach. We mention a few tailings ponds from Romania where subsidence funnels have

appeared due to the suffusion process: Valea Devei, Rovina – Gura Barza, Tarnicioasa and Paraul Cailor. Subsidence funnels can also appear as a result of a deflection of the pure water discharge system. After the subsidence of the pure water discharge system, the soft part of the pond is carried away by water leading to the formation of a subsidence funnel which usually appears on the damage point vertical. Such subsidence funnels were at Valea Podului – Teliuc and Tarnicioasa ponds as a result of impairment of the pure water discharge pipes situated on the pond bed.

4.7.5 Clay content

Normally residues contain no considerable amounts of clay. The residues show no plastic behavior [131]. But residues from diamond mines, kimberlite, as well as residues from phosphate- and bauxite mines have relatively high clay content [13].

4.7.6 Liquefaction of the pond material

The liquefaction process takes place in sands situated under ground water level because of vibrations. Most liquefaction processes are generated by vibrations produced by earthquakes but can take place also as a result of some detonations connected with industrial purpose (in the big pits). Such phenomena happened in Chile, during the earthquake from 1965 when 11 tailings ponds were destroyed by liquefaction.

5. Bodies of such materials containing toxic/or hazardous substances, with particular attention to the release of contaminants

In almost all the processes applied for the extraction of metal values, the mined ores are first milled and finely ground, then treated in beneficiation and hydrometallurgical plants. Since the extracted metal represents only a small percentage of the entire ore mass, the vast majority of the material mined usually ends up in the form of fine slurries. The tailings contain almost all the constituents of the ore, among them heavy metals and other toxic substances. As a result of the milling process, all the contaminants contained in tailings may become easily available for dispersion into the environment. Moreover, the tailings contain the chemicals added during the beneficiation and hydrometallurgical treatment process.

5.1 Migration pathways of contaminants and related parameters

As precipitation percolates through the mine tailings, heavy metals and metalloids as well as toxic or non-toxic substances present in tailings may be dissolved, impairing the quality of water as it seeps downward. In the specific case of sulphidic wastes the infiltrating water becomes more acidic, dissolves the arsenic and heavy metal compounds in the tailings and carries these contaminants downward. The run-off water from the tailings dam site may also dissolve and transport contaminants into nearby water resources. Furthermore, removal of water from the tailings during periodic dry periods and on mine closure, or deposition of tailings in dry form, results in wind erosion and transportation of tailings material to the surroundings.

5.1.1 Seepage

Contaminated water may be formed from downward migration of impoundment constituents or ground water movement through tailings. Most contaminant transport in ground water systems is from the advection (fluid movement and mixing) of contaminants. Factors affecting the rate of advection include ground water/leachate velocity, chelation, pH and partition coefficient values. The geochemistry of the aquifer, physicochemical properties of the tailings and seepage will determine the buffering capacity of the soil, types of chemical reactions (precipitation or neutralization) and the rate of adsorption and ion exchange.

Neutralisation, oxidation/reduction, precipitation, adsorption, ion exchange and biological reactions play a major role in the chemical composition of the tailings seepage. There are two basic options for controlling contaminated water in impoundments: keeping it in the impoundment or capturing it after it exits the impoundment. Seepage controls are typically evaluated in the early phases of impoundment design. The objectives are to maintain embankment stability, decrease water losses and maintain water-quality at the site. Options for seepage control include preparation of the TFM base by installation of liners beneath the entire impoundment (to contain water and to exclude ground water), constructing drains for

seepage collection, constructing seepage collection and pump-back (or treatment) systems, sometimes in conjunction with low permeability barriers, construction of low permeability embankments and embankment barriers (i.e. cores and lines), dewatering of tailings prior to deposition and decreasing hydraulic head by locating the free-water pond away from the embankment. Some of these techniques are described in more detail below (Ritcey 1989).

5.1.2 Liners

Liners have not been incorporated into tailings impoundment designs until last decade or two. Even now, due to their high cost, mining companies tend to avoid the use of liners under an impoundment. Although liners may be used to seal the upstream face of the tailings dam, most tailings impoundments in use today do not contain lining system. The two major types of liners used to control flow through tailings dams are synthetic materials, which are very expensive and constructed liners made of local clays or other readily available materials. Slimes are also sometimes used as low permeability barriers.

A real coverage needed for the impoundments is a major cost consideration, especially for cross-valley dams. Thickness' vary depending on the liner type but most thickness' can be decreased if they are overlain with a drainage system to collect fluids, which reduces the hydraulic head on the liner. Liners have to be resistant to constituents in the tailings and seepage (such as acids or caustic substances), weathering if exposed to ultraviolet radiation, deformation from loading stresses and seismicity.

Clays and synthetic liners can be combined to form double and triple liners. To prevent large settlements, clay and synthetic liners are not placed over loose or easily compressed material. Designs usually incorporate covers to mitigate the effects of sunlight, wave and wind exposure on clay and synthetic liners and drying on clay liners. The effects of frost action and drying are incorporated as needed into a liner design, especially for construction, stress loading, exposure to ultraviolet radiation, or improper planning and construction of the sub-grade. Short term maintenance plans are generally implemented because many problems often occur within the first six months of operation.

Clay liners

Clay can be an inexpensive option for liners, especially in areas with a natural abundance of this material. The density of a clay liner depends on the particle size distribution, mineralogy and the method and degree of compaction. Clay can be compacted to a prescribed moisture content and density to provide a permeability of 10^{-6} to 10^{-7} cm/sec or lower. Grain-size distribution curves may be used to determine the amount of fine-plasticity clay because of its lower permeability, but construction and the climate of the site may have an effect on the decision. Chemical tests should be undertaken on the clay material to determine if it is resistant to the seepage produced by the tailings dam.

Clay liners may be supplemented with other liners (e.g. synthetic) to further reduce potential seepage. Clay liners can fail when the permeability increases considerably above the design value. The three major causes of failure are differential settlement of the foundation, causing

- localized cracking of the clay liner
- drying of the clay liner (desiccation) leading to the development of microcracks (that can occur in areas lined with clay too long in advance of the time when wet tailings will cover the liner or if tailings dry after deposition)
- alteration of the liner permeability, due to geochemical reactions between the liner and leach solution.

Synthetic liners

Synthetic liners are a relatively new development in the control of seepage in tailings impoundments. Of the rigid liners, concrete (rarely used) and gunite may be susceptible to acid and/or sulphate attack and asphaltic concrete may have questionable weathering and sun-aging characteristics. Sprayed membranes have demonstrated installation problems which may need to be resolved before being considered as a possible option. Synthetic rubber membranes may be too costly for tailings impoundments. However, these thermoplastic membranes are the most common liners considered for tailings impoundments.

Non-rigid liners are often grouped into a category called geomembrane. Geomembranes are often used in conjunction with clay liners to form a double or triple liner combination. Seepage losses through geomembranes are estimated on the basis of flow through a hole in the geomembrane. Most synthetic liners are generally 10^{-9} to 10^{-14} cm/sec with average thickness of 40 to 60 mils. As noted previously, both the cost and technical feasibility are major factors in selecting synthetic liners, given the large size and uneven terrain usually encountered.

Slimes

Tailings slimes are easy and inexpensive to install as low permeability layers to slow but not stop seepage. To be cost-effective, the slimes must constitute a majority of the whole mill tailings and the coarse and fine sands must be cycloned out of the slimes. In addition, there should be a system in place to guarantee even distribution of the slimes in the tailings dams (or lifts). Although slimes may offer a low-cost alternative to other materials, they have several disadvantages. In addition it is difficult to determine the long-term permeability of the slimes.

Embankment barriers

Embankment barriers are installed below the impoundment and include cut-off trenches, slurry walls and grout curtains. An impervious layer is generally required between them and the tailings. Barriers are installed underneath the upstream portion of a downstream embankment and the central portion of centre-line embankments; they are not compatible with upstream embankments. A good water-quality monitoring program is needed when using embankment barriers to ensure that they are completely effective in intercepting flows and also that seepage is not moving downward and contaminating the ground water.

Cut-off trenches, usually 5 to 20 feet in depth, are the most widely used type of embankment barrier for tailings dams especially in areas with large volumes of natural clays. Dewatering may be necessary during the installation of cut-off trenches when they are installed below the ground water table.

Slurry walls are narrow trenches that are best suited to sites with a level topography and containing saturated or fine-grained soils. They are not compatible with fractured bedrock systems. The slurry walls are installed by excavating a trench to a zone of low permeability material and filling the trench with soil/bentonite slurry which is then allowed to set to a consistency of clay. Depths average 40 feet and permeabilities obtained can be low as 10^{-7} cm/sec.

Grout curtains use cement, silicate materials, or acrylic resins as a barrier to seepage movement. They are limited to sites with coarse-grained material (medium sands to gravel or fractured rock with continuous open joints) and can extend to depths of more than 100 feet. Permeabilities obtained can be as low as 10^{-8} cm/sec. However, leaks can occur through curtain joints or by subsequent corrosion of the curtain.

Rather than simply intercepting and containing seepage flows, barriers may have gravel (or other pervious material, appropriately filtered) drains immediately upgradient to allow seepage to be removed or directed to embankment underdrains. Barriers and seepage collection systems may also be used downgradient of embankments to prevent further environmental releases.

Pump-back systems

Pump-back systems consist of seepage ponds and/or seepage collection wells installed downgradient of the impoundment that are outfitted with pumps that send seepage back to the impoundment or for use as process water. Current practices include the use of toe ponds or seepage ponds to collect seepage. In some cases, underdrains or toe drains are designed to flow into the seepage pond. In other cases however, these systems are installed after construction of the impoundment as a remedial action to collect unanticipated seepage. These units may be used in conjunction with slurry walls, cut-off trenches or grout curtains to

minimize downgradient seepage. Depending on effluent quality, the operation of the pumpback system may continue indefinitely.

Associated pathways

Human exposure to seepage contaminants may result from the use of contaminated surface and groundwater for domestic, industrial, and agricultural purposes. Local residents usually rely on groundwater as a portable water supply, and on surface water as irrigation water supply. The likelihood of human exposure to groundwater contaminants is usually minimized by the rural nature of the site and the lack of supply wells for portable water downgradient from the site; however, without monitoring results from nearby private wells, this pathway of human exposure can not be ignored. The potential exists for completing this pathway of human exposure in the future if groundwater wells are installed on-site or downgradient from the site.

5.1.3 Surface run-off water

Control of surface water should be one of the major factors involved in design and operation of a tailings impoundment. A mass balance approach to water management can be used, with variables categorized into outflows and inflows. Outflows from a tailings impoundment include overflows, evaporation, recycle and re-use and seepage. Overflows are dependent on the dam's storage capacity and the runoff volume of a storm event in the basin. Evaporation rates are a function of the climate and the surface area of the free-water pond and saturated tailings. Recycling and re-use volumes depend on the operation's capacities and needs. Seepage can exit the dam as ground water or seepage through or under the embankment. Both surface and subsurface components interact in a dynamic fashion and must be considered together in any analysis [125].

Estimation of surface water inflows and outflows using a mass balance approach includes both natural and man-made components. Variables include precipitation (including storm events), evaporation, run-on (including flood events), the liquid component of the tailings as it is discharged to the impoundment, water returned to the impoundment from any downstream seepage return systems, evaporation, infiltration, decanting and recycling tailings water any direct discharge (overflow) and also any discharge to the free water pond resulting from tailings consolidation.

5.1.4 Wind erosion

A typical mine tailings site contains many millions of tons of tailings deposited during the operating life of the mine. The tailings impoundment usually covers a large area, sometimes hundreds of hectares, and is comprised of the embankment made of earth fill or tailings, a sandy beach grading to fine tailings adjacent to the tailings pond. Much of the tailings fines

may be under water. Removal of water from the tailings, during periodic dry periods and on mine closure, results in the exposure of beaches which can dry out. Wind erosion can cause dusting from the embankment, sandy beaches or dried out fine slimes pond zones. In many instances this dust can be an annoyance or hazardous to the environment and needs to be controlled.

A wide range of control methodologies is available including the development of good vegetation cover, the placement of granular wind resistant materials and the spraying of bonding agents to form an erosion resistant crust. The practicality of use is dependent on the access that is required onto the tailings in order to place the dust control measure. Access onto embankments and sandy beaches is normally such that none of the above methods are precluded. Since the very soft nature of the tailings slimes often prevents access by heavy vehicles, the agent must be placed using extremely light equipment, possibly hand held sprays. This limits the agents that can be used to those agents that can be sprayed onto the tailings.

Without proper erosion control installed at tailings disposal sites, erosion of the dumped waste material can take place, leading to elevated sediment loading in runoff during rainfall. Given that tailings usually contain metals or other contaminants which can dissolve, they can lead to even greater contamination of the environment. It is desirable to plan the waste tailings rehabilitation to occur as early as possible in the life of the mine. Establishing the final faces of waste dumps early and re-vegetating these surfaces will significantly reduce wind erosion.

5.2 *Tailings containing sulphide*

The oxidation of sulphidic mine tailings and waste rock containing sulphides is a major environmental problem for the mining industry. The financial liability of sulphidic wastes is estimated to be several billion Euros. Without proper management and treatment, the presence of sulphidic mine tailings will generate Acid Mine Drainage (AMD) for years. Acid Mine Drainage (AMD), or Acid Rock Drainage (ARD) are the terms used to describe the drainage that occurs as a result of the natural oxidation of sulphide minerals contained in rocks when exposed to the combined action of water and atmospheric oxygen. It is common in polymetallic sulphide mining operations as well as in coal and lignite mining. The phenomenon is not necessarily confined to mining, but can occur when sulphide-bearing rocks are exposed to air and water; some natural springs lying in the vicinity of sulphide mineralization outcrops are acidic.

The common sources of AMD are underground or open-pit mining works, mining waste rock or overburden dumps, processing tailings, and temporary or permanent stockpiles of sulphidic concentrates, notably pyrites, of no commercial value under the current economic conditions. These sources may remain active for decades or even centuries after mine closure.

AMD is associated with coal and polymetallic sulphide mining. Polymetallic sulphide deposits constitute the major resource for a number of metals, such as copper, zinc, lead, and important resource for others, as gold and nickel. Mining of these deposits inevitably results in the generation of large quantities of waste rock and processing tailings containing residual sulphides. Improper waste management practised in the past, but to a certain degree in current mining operations as well, resulted in the production of AMD that has contaminated soil and groundwater over extensive areas with sulphuric acid and toxic metals. It is particularly catastrophic in areas with redundant mines, where the old wastes and mining works are now active AMD sources and pollute the environment continuously. The AMD from coal and lignite mining also contains sulphate, iron and aluminium in solution, however it is generally characterised by lower concentration of toxic metals [71].

The geochemistry of AMD has been the subject of numerous investigations. AMD results from the interactions of certain sulphide minerals with oxygen, water, and bacteria. The iron disulfide minerals pyrite (FeS_2) and, less commonly marcasite (FeS_2), are the principal sulfur-bearing minerals. Pyrrhotite (FeS), arsenopyrite (FeAsS), chalcopyrite (CuFeS_2) and other sulfide minerals containing Fe, Cu, As, Sb, Bi, Se and Mo also can produce acidic solutions upon oxidation.

The sources of AMD contain acid-forming sulphides and other minerals, some of which may be potentially acid-consuming. AMD is therefore the result of a complex series of chemical reactions involving: (a) Generation of sulphuric acid due to the oxidation of the sulphides by the combined action of oxygen and water. These reactions are auto-catalytic and their rate is

further accelerated by bacterial activity. (b) Consumption of the generated acid by the acid-consuming components; these reactions result in the precipitation of gypsum and metal hydroxides, oxy-hydroxides and other complex compounds. If the acid consumption capacity is higher than the acid-generation, it is possible that the resulting drainage will be neutral or even alkaline rather than acidic. However acid generation may still be taking place locally at "hot spots" with high sulphide concentration.

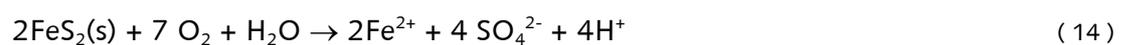
AMD solutions usually have a characteristic brown-red colour attributed to the ferric iron; however, a blue-green colour may prevail if the dissolved iron is in the ferrous state, which will turn to the usual brown-red upon exposure to the atmosphere. In addition, AMD contains in suspension products of the precipitation reactions that may impart their colour, as reddish (ferric oxy-hydroxides), white (aluminium hydroxide), etc.

In the following sections, the reactions leading to the formation of AMD will be discussed. These include acid generation and acid neutralisation reactions.

5.2.1 Oxidation–acid generation reactions

Oxidation of the sulphide minerals takes place through a complex series of reactions involving direct, indirect and microbially-assisted mechanisms; some oxidation reactions result in acid generation, while others result in the dissolution and mobilisation of heavy metals. Pyrite (FeS_2) is the main mineral responsible for acid generation; pyrite oxidation and its control has been reviewed in detail in [39]. Normally, pyrite is a stable, insoluble mineral as long as it does not come into contact with air and water. However, as a result of mining, it becomes exposed and is partially solubilised.

Pyrite can be oxidized directly or indirectly. The direct oxidation of pyrite is described by the following reactions:



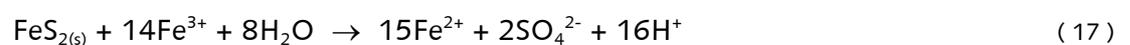
This reaction produces proton acidity; if the oxidation potential is maintained, oxidation of Fe^{2+} to Fe^{3+} by oxygen will take place, consuming part of the proton acidity produced:



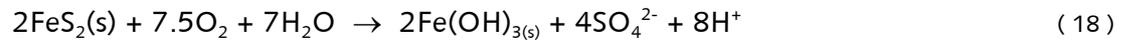
If the pH of the resulting solution is higher than 3 (as in the initial stages of the reaction), ferric iron will hydrolyse, precipitate as hydroxide and generate acidity:



Further, the Fe^{3+} generated by reaction (15) will oxidise FeS_2 by the indirect reaction:

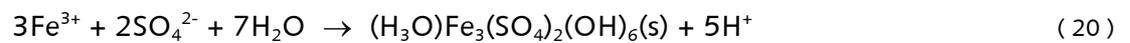
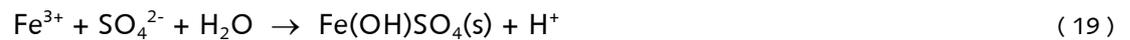


The resulting Fe^{2+} will be oxidised to Fe^{3+} by reaction (15) and will again become available to oxidise more pyrite (autocatalysis). The overall stoichiometric reaction describing the oxidation of pyrite to result $\text{Fe}(\text{OH})_3$ is commonly given as:



The heat energy produced in this reaction for complete conversion of one mole of pyrite to ferric hydroxide amounts to about 1490 kJ at 25°C.

At low pH, other precipitation reactions, mostly involving ferric iron, may take place yielding basic sulphates or jarosites:



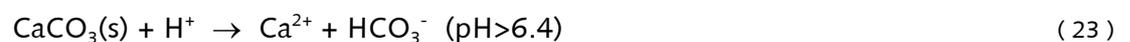
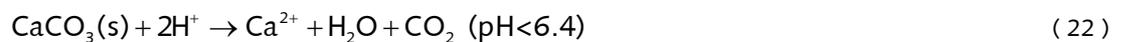
Other products, such as $\text{FeSO}_4 \cdot x\text{H}_2\text{O}$, have also been identified.

5.2.2 Acid neutralisation reactions

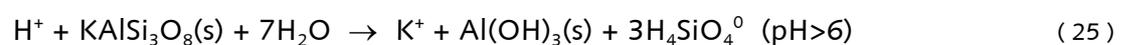
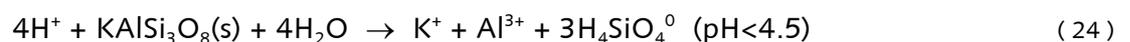
Acid consuming minerals, such as carbonates (calcite, aragonite, siderite, magnesite, aluminosilicates olivine, pyroxenes, feldspar, micas, etc.) usually coexist with the sulphides; these will react with the acid neutralising it. Calcite will react according to the following typical reaction:



The weak and unstable carbonic acid produced by this reaction decomposes, giving $\text{CO}_2(\text{aq})$, HCO_3^- and CO_3^{2-} ; the distribution of these species is a function of pH; for pH values between 7-10, HCO_3^- is the dominant species, while at pH below 6, CO_2 predominates. By taking into account the predominant carbonaceous species the net neutralisation reactions with CaCO_3 are:



These reactions show that twice as much calcite is needed to neutralise acid to neutral ($\text{pH} > 6.4$) than to slightly acidic levels ($\text{pH} < 6.4$). In the same manner, aluminosilicates, as for example the K-feldspar, react according to the reactions:



During the evolution of the reaction sequence, and as the pH rises, heavy metals will precipitate as hydroxides, producing acidity. Gypsum will precipitate once its solubility product is exceeded:



The solubility product predicts a solubility of 0.86 g/L; including complexes and activity corrections, the solubility of $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$ is calculated as 3.15 g/L at 25°C [7].

5.2.3 The role of bacteria

Bacterially assisted mechanisms of pyrite oxidation are very important, especially at pH less than 4. The bacteria that are important in pyrite oxidation belong to the genera *Thiobacillus* and *Leptospirillum*. The genus *Thiobacillus* includes the species *Thiobacillus ferrooxidans* (Th. f.), which is the most common microorganism involved in sulphide oxidation. It is an obligate chemoautotrophic acidophilic bacterium able to oxidise Fe^{2+} , S^0 and metal sulphides, as well as other reduced inorganic sulphur compounds to sulphate. The species *Thiobacillus thiooxidans* (Th. th.) has also been isolated; it can oxidise both S^0 and sulphide-S to sulphate, but, unlike Th. f., it cannot oxidise Fe^{2+} . The species *Leptospirillum ferrooxidans* participates in mixed cultures with Th. f. The above genera are active at pH 1.4-4 and temperatures 5-35°C, the optimum conditions being pH around 2 and temperature 30-35°C.

The mechanisms of pyrite oxidation by bacteria involve direct and indirect metabolic reactions. Direct metabolic reactions require physical contact between bacteria and pyrite particles. Indirect do not require physical contact, and therefore are mainly responsible for the aqueous phase oxidation of Fe^{2+} to Fe^{3+} , which then becomes available for indirect sulphide oxidation. In [91] it is reported that the predominant, and possibly the only, active mechanism is the indirect bacterial oxidation of Fe^{2+} to Fe^{3+} , which in turn oxidises pyrite to generate additional Fe^{2+} and acidity in a cyclic process.

5.2.4 Stages in AMD generation

It is obvious from the above that AMD generation involves a complex series of interrelated reactions that take place either chemically or through microbially assisted mechanisms, and depend upon the activity of bacteria and solution pH and Eh [68]. Three stages can be distinguished:

The *first stage* starts at circum-neutral pH and ends at pH of about 4.5; direct and indirect pyrite oxidation as well as ferrous iron oxidation take place mainly abiotically. At this pH range, the solubility of Fe^{3+} is limited, and consequently precipitation of ferric hydroxide will take place; therefore the overall pyrite oxidation-acid generation reaction at this stage can be expressed by reaction (18).

Because of the limited availability of Fe^{3+} , the indirect oxidation mechanisms are not important at this stage. The acid produced will be consumed by the neutralisation reactions as long as acid-consuming minerals are available. Once the alkalinity is consumed, pH will drop, although the rate of decrease is moderated because of the pronounced decrease in the rate of reaction of oxidation of Fe^{2+} ions.

The *second stage* of the process starts at pH values around 4.5 and finishes at about 2.5. As abiotic oxidation of ferrous iron slows down at the end of stage 1, bacterial oxidation through the action of iron oxidising bacteria, the most common being *Thiobacillus ferrooxidans*, becomes dominant, allowing reactions of iron oxidation and ferric iron precipitation to proceed at a fast rate. Direct pyrite oxidation takes place both abiotically and via direct bacterial action. At pH less than 3, the increased solubility of Fe^{3+} results in a decreased rate of ferric hydroxide precipitation and therefore, in an increase in the Fe^{3+} activity in solution; this accelerates the rate of the indirect oxidation reactions.

The *third stage* of the process starts as the activity of Fe^{3+} becomes significant at a pH of approximately 2.5, where essentially precipitation stops. Fe^{3+} is now the main oxidant that promotes pyrite oxidation. Reaction of iron oxidation proceeds at a rate totally determined by the activity of *Th. f.* and this reaction now is the rate-determining step in pyrite oxidation. In addition, Fe^{3+} promotes oxidation of the other sulphide minerals, as galena, chalcocite, chalcopyrite, and sphalerite. Although these minerals do not produce acid, they release heavy metals, intensifying the adverse environmental impact of AMD. The net result at stage 3 is the production of highly acidic water with pH <2.5, with high sulphate, acidity, total iron, $[\text{Fe}^{3+}]/[\text{Fe}^{2+}]$ ratio, and dissolved heavy metals concentration.

Bacterial oxidation is very important for stages 2 and 3; it has been reported that *Th. f.* can accelerate the rate of Fe(II) oxidation by a factor of 10^6 [115], although in real systems acceleration factors up to only 5 have been reported. The microbial involvement in AMD makes the overall kinetics extremely complicated and difficult to reproduce in the laboratory.

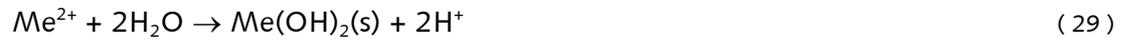
During the third stage, both chemical and bacterial mechanisms responsible for sulphide oxidation/acid generation reactions become autocatalytic; therefore, once AMD develops, the problem tends to worsen and is extremely difficult to halt.

5.2.5 Dissolution of other contaminants

Usually pyrite is the predominant sulphide mineral, and is responsible for the bulk of the acid generated; If metallic minerals (such as galena, chalcopyrite, sphalerite) in addition to pyrite and pyrrhotite are present (as is usually the case in the natural oxidation of a mineral deposit and the oxidation of products from the mining of a mineral deposit) then there may be a secondary effect of the oxidation of the iron-sulphur minerals to sulphuric acid and ferric iron.. Bivalent metal sulphides, as MeS , where $\text{Me} = \text{Fe}, \text{Zn}, \text{Cd}, \text{Pb}, \text{Cu}, \text{Ni}$, will oxidise directly or indirectly by the action of Fe^{3+} to yield the corresponding sulphates (Kontopoulos 1998):



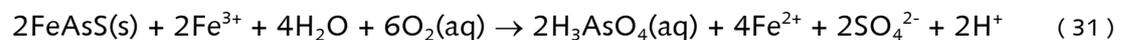
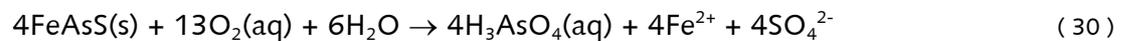
These reactions do not generate proton acidity, unless the Me^{2+} ion hydrolyses:



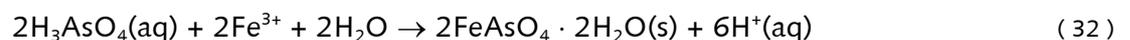
The solubility of Cd^{2+} , Zn^{2+} and Ni^{2+} is high in the neutral to 4.5 pH range, therefore these ions will not hydrolyse to any appreciable degree to produce acidity; however they will contribute to the loading of the resulting drainage with heavy metals. Cu^{2+} will precipitate as Cu(OH)_2 at pH 5-6, producing acid. Pb will form the insoluble PbSO_4 .

It is by this process that significant amounts of heavy metals may be solubilised by AMD. In addition, many metallic elements are often present at trace levels within the minerals pyrite and pyrrhotite. Oxidation of these minerals can therefore release and mobilize these trace elements. Untreated (not neutralized) AMD creates two quite distinct environmental problems - the acidity from sulphuric acid (which is invariably a product by definition) and the heavy metal solubilization. It is important that these two effects be recognized as separate, since their consequences to ecosystems are distinct, and because AMD generation and heavy metal transport are separate processes.

Arsenopyrite, if present, will oxidise directly or indirectly:



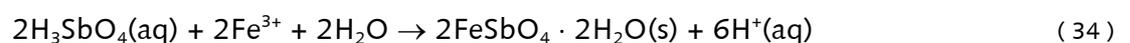
In all of the above reactions, dissolved As is depicted in the penta-valent state as non-dissociated $\text{H}_3\text{AsO}_4(\text{aq})$; however, it may be found partially dissociated as $\text{H}_2\text{AsO}_4^-(\text{aq})$. It must also be stressed that direct and indirect oxidation of arsenopyrite may yield trivalent rather than penta-valent As in solution as $\text{HAsO}_2(\text{aq})$ or $\text{AsO}_2^-(\text{aq})$. Precipitation of ferric arsenate will occur at $\text{pH} > 3$ yielding ferric arsenate, $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$, or basic ferric arsenates, $\text{FeAsO}_4 \cdot x\text{Fe(OH)}_3$:



Precipitation of arsenate with lime is also possible:



Antimonate ions will precipitate with similar reactions:



5.2.6 Factors affecting the AMD generation

AMD generation requires the combined action of bacteria, water and air. Equation (2) is catalysed by the presence of site-specific strains of *Thiobacillus ferrooxidans* that utilize the sulphur present as their source of energy. They are autotrophic, obtaining their nutritional needs from the atmosphere (nitrogen, oxygen, carbon dioxide and water) and from minerals (sulphur and phosphorus). While these bacteria are not catalysts by true definition, they do act as accelerating agents if their habitat conditions are at or close to optimal and they are a most important factor in the generation of ARD.

It must be stated that in addition to pyrite, the presence of both *oxygen* and *water* is required for process progression. This means that the removal of the oxygen source (e.g. by total submersion under water) or the water source (e.g. conditions of aridity) will halt AMD production. AMD production would also be considerably slowed or halted by the termination of *T. ferrooxidans* reproduction by a bactericidal agent. The end products are sulphuric acid and ferric sulphate. Also, sulphuric acid is an important intermediate product. From the onset of pyrite oxidation, pH falls (acidity increases) quickly and then stabilizes, typically at values around pH 2.5 to 3.0. The pH of stabilization is normally determined by the optimal habitat requirement of the site-specific strain of bacteria.

If pyrite and/or pyrrhotite are the only sulphide minerals open to atmospheric oxidation then the products of the oxidation process are those described above. Depending upon the availability of water and oxygen, reactions may not always approach completion as indicated by equations 1 to 5, and in such cases intermediate phases of chemical compounds or minerals may remain at the oxidation site.

5.2.7 AMD prediction methods

To predict the metal leaching/AMD potential, static and kinetic tests are usually applied. The static tests, aim at determining the balance between acid-producing and acid-consuming mineral components in the individual samples. These are easy and quick tests; specific screening criteria will then be applied to evaluate the results and classify the material according to its acid generation potential. The material that will be characterised as potentially acid generating, based on the specified screening criteria, will then undergo a series of kinetic tests that will determine the actual rate of acid generation-neutralisation and will therefore define the actual potential for acid generation over time and the expected drainage quality. Kinetic tests require long testing periods, ranging from weeks to months or even years.

Static tests - acid base accounting

Static Acid Base Accounting tests are short term (usually measured in hours or days) and relatively low cost. tests developed to provide an estimate of a mine waste's capacity to produce acid and its capacity to neutralize acid. These tests do not consider parameters like availability of acid-producing and acid-neutralizing minerals and differences between the respective dissolution rates of acid-producing and acid-neutralizing minerals. Thus, these tests are commonly used as a screening tool, and their implications are subject to further verification. The most common of such procedures include:

- Sobek Acid Base Accounting (ABA) procedure
- BC Research Inc. Initial Test procedure
- Net Acid Production (NAP) test
- Net Acid Generation (NAG) test
- Modified Acid Base Accounting (ABA) procedure
- Lapakko Neutralization Potential Test procedure
- Peroxide Siderite Correction for Sobek ABA method
- Despite individual procedural differences, all these methods involve:
- Determination of the Acid Potential (AP) based on the total sulphur or sulphide-S content
- Determination of the Neutralization Potential (NP) including:
 - the reaction of a sample with a mineral acid of measured quantity
 - the determination of the base equivalency of the acid consumed
 - the conversion of measured quantities to a Neutralizing Potential in g/kg or kg/tonne or tonne/1000 tonnes calcium carbonate (CaCO₃).

Initially the most commonly-used static test was the standard ABA [120]. Variations of ABA commonly applied nowadays include the modified ABA [73], NAG test and the B.C. Research Initial Test [1].

The static tests quantify the acid potential (AP) using either total sulphur or sulphide-sulphur content. The total sulphur content (Standard ABA) though, overestimates the actual AP of samples containing substantial non acid-producing sulphate minerals (e.g. barite or gypsum). On the other hand, the sulphide-sulphur measurement (modified ABA) will underestimate the actual AP of samples containing substantial acid-producing sulphate minerals (e.g. melanterite or jarosite). Knowledge of the mine waste sulphate mineralogy will indicate whether the sulphate minerals present, if any, are acid producing and allow selection of the more appropriate AP quantification. However at present it is accepted that the AP is calculated based on sulphide sulphur.

Different static test methods can produce markedly different neutralization potential values (NP) for the same sample. Protocol variables which may contribute to these differences include mine-waste particle size (tailings are typically run "as received"); the amount of acid added (i.e. digestion pH), and the endpoint pH of the "back titration", if a back titration is used. The extent to which protocol variables will affect the measured NP is dependent on the sample mineralogy. The conditions and minerals reported to dissolve by various ABA procedures are summarized in Table 1. It is noted that carbonates are considered the most reactive acid neutralizing minerals whereas minerals including plagioclase feldspars, K-feldspar, muscovite and quartz are slow weathering to inert minerals.

The Net Acid Production (NAP) and Net Acid Generation (NAG) tests are based on the principle that hydrogen peroxide accelerates the oxidation of iron sulphide minerals. The acid consequently produced dissolves neutralizing minerals present, and the net result of the acid production and neutralization can be measured directly. This test does not require sulphur determinations and is, therefore, more readily conducted in a field laboratory than other static tests. Based on previous studies, the application of NAP to wastes with sulphur content higher than 10% may underestimate the acid generation potential due to incomplete oxidation [1].

Table 1: Parameters of static Acid Base Accounting procedures

Procedure	Acid	Amount of acid added	End pH of acid addition	Test duration	Test temperature	Minerals dissolved
Sobek	Hydrochloric	Determined by Fizz Test	0.8-2.5	Until gas evolution ceases (~3 h)	Elevated (90 °C)	Mineral carbonates Ca-feldspar, pyroxene, olivine (forsterite-fayallite) Some feldspars anorthoclase>orthoclase >albite ferromagnesians – pyroxene hornblende, augite, biotite
BCRI Initial	Sulphuric	To reach pH 3.5	3.5	16-24 h	Ambient	Ca + Mg carbonates. Possibly chlorite, limonite
Modified ABA	Hydrochloric	Determined by Fizz Test	2.0-2.5	24 h	Ambient	Ca + Mg carbonates Some Fe carbonate, Biotite, chlorite, amphibole olivine (forsterite - fayallite)
Lapakko	Sulphuric	To reach pH 6.0	6.0	Up to 1 week	Ambient	Ca + Mg carbonates
Sobek – Siderite Correction	Procedure as for Sobek, but with peroxide correction for siderite					Ca + Mg carbonates, excludes Fe+Mn carbonates. Otherwise as per Sobek.

* (Mills, www.infomine.com/technology/enviromine/ard)

Critical parameters

Two parameters are calculated to classify materials in terms of acid drainage generation potential. These are:

- The Net Neutralisation Potential (NNP) which is the difference in value between the neutralisation potential (NP) and the Acid Potential (AP), expressed in kg CaCO₃ / t of material and
- The neutralisation potential ratio (NPR) which is the ratio of NP value to AP value.

The former is the preference for Appalachian coal mines, and the latter for Western Canadian metalliferous mines. Materials with sulphide minerals whose net neutralising potential is negative are likely to be an acid drainage source. Exceptions are possible if the sulphide content of material is very low and/or there are slow dissolving, non-carbonate sources of alkalinity. Based on the NPR values, the Acid-Base Accounting screening criteria recommended by the British Columbia Ministry of Employment and Investment of Canada are given in Table 2 [98].

The above guidelines define a "grey zone" for NPR ranging between 1 and 4. The acid drainage potential of materials that fall in the grey zone is considered uncertain and kinetic test work has to be conducted to characterise them with regard to acid generation potential. It is noted that the British Columbia guidelines recommend that the neutralisation potential is determined based on the expanded version of the Sobek method and acid potential is determined based on the sulphide sulphur content of the samples.

Table 2: Acid-Base Accounting Screening Criteria [98]

Potential for ARD	NPR	Comments
Likely	< 1:1	Likely AMD generating.
Possibly	1:1 – 2:1	Possibly AMD if NP is insufficiently reactive or is depleted at a faster rate than sulphides.
Low	2:1 – 4:1	Not potentially AMD generating unless significant preferential exposure of sulphides or extremely reactive sulphides in combination with insufficiently reactive NP.
None	> 4:1	No further testing is required unless material is going to be used as a source for alkalinity.

It is recommended to use Modified ABA [73] together with the mineralogy of the sample as the basis of a reliable AMD screening programme. Modified ABA has a lower risk of misclassification of samples into the wrong category and is a cost-effective screening test.

Paste pH is a common and simple field test used to assess the presence of soluble acid salts on mine wastes. Most methods use a 1:1 weight ratio of distilled water to air dried solids, with pH measurements of the mixture. Sample mass and equilibration time of the water-solids mixture prior to pH measurement vary among methods. The procedure determines pH of a mixture of 10 g sample (-60 mesh) and at least 5 ml distilled water (water addition is adequate to saturate, but not cover, the sample). The Acid Concentration Present test is slightly more complicated but supplies an estimate of acidity present rather than simply pH. A mixture of 20 g sample (-200 mesh) and 50 ml deionised water is agitated, the initial pH is recorded, and the mixture is titrated to pH 7 with NaOH.

Kinetic tests

Kinetic tests are performed for sulphide wastes that according to static test results are characterised as potentially acid generating or fall in the zone of uncertainty. Kinetic tests can also be used to determine the metal leachability of trace elements of environmental concern. Kinetic testing is required to determine the acid generation rate and quality of drainage of these materials, critical information for the environmental management of wastes. A number of laboratory kinetic tests have been developed with humidity cells, columns and lysimeters (Table 3), being the three most commonly used laboratory methods of determining kinetic acid drainage characteristics of drill-core samples, waste rock and tailings. All kinetic testwork procedures involve two main stages, i.e. subsection of sample to periodic leaching and collection of drainage for analysis.

Table 3: Kinetic tests for acid rock drainage prediction

a/a	Type	Procedure	Comments
1	Humidity cells (ASTM D5744-96)	Sample mass: 1 kg Oxidative wet/ dry cycles Test duration: 20 weeks minimum	Standard procedure Determination of acid generation / neutralization rates Real conditions may not be simulated.
2	Column test	Site or material specific operation Simulation of oxidizing, reducing environment.	Flexible allowing simulation of field conditions Long duration
3	Lysimeters test	Simulation of field conditions	No standardized practice. Long duration

The humidity cell is a standard kinetic test (ASTM D5744-96) recommended by the government of B.C, Canada for the prediction of the geochemical behaviour of mining wastes. It is usually referred as an accelerated weathering procedure because it is designed to accelerate the natural weathering rate of potentially acid generating samples and reduce the length of time for which testwork must be run. A cell 203 mm in height by 102 mm diameter is specified for material 100% passing 6.3 mm (crushed core or waste rock and coarse tailings) and a cell 102 mm in height by 203 mm diameter is specified for material passing 150 µm (fine tailings). The humidity cell operational procedure is a cyclic one during which the sample is subjected to three days of dry air permeation, three days of humid (water saturated) air permeation and one day of water washing with a fixed volume of water, i.e. 500 ml for 1 kg of sample. The sample mass used is about 1 kg and minimum test duration of 20 weeks is recommended.

Column testwork may be undertaken to determine the geochemical behaviour of waste rock and tailings disposed on the surface and exposed to atmospheric weathering (sub-aerial disposal) or disposed under water cover (sub-aqueous disposal). Unlike humidity cell procedure, there is little, if any, standardisation of column testwork procedure allowing considerable flexibility. This flexibility permits column operation to be highly site or material specific with regard to material particle size, sample mass, wet/dry cycles, volume of water washing etc. Columns for sub-aerial and sub-aqueous testwork are typically 76, 102 or 152 mm in diameter and from about 1 m to more than 3 m in height.

Lysimeters may be also used to determine the acid generation/ neutralisation rates of sulphidic wastes and assess the drainage quality. Like column kinetic test, lysimeter test allows the simulation of the conditions encountered in the waste disposal area. Lysimeters usually have larger diameter and smaller height as compared to columns, (e.g. 30 or 70 cm in diameter and height 30 to less than 100 cm).

It is noted that a humidity cell will usually determine if a given sample will produce acidity but neither when the sample will go acid nor the on site drainage quality. On the other hand, column and/ or lysimeter operation may simulate field conditions and as a result, it can be used to give indications of drainage quality, i.e. allow the determination of lower and higher bound. Monitoring parameters in a kinetic test include mass/ volume of leachates, pH, conductivity, redox potential (mV), acidity/ alkalinity, sulphate and dissolved metals.

Critical parameters (levels and sensitivity)

Acidic drainage from sulphidic tailings is a complex of elements that interact to cause a variety of effects on aquatic life that are difficult to separate into individual components. Toxicity is dependent on discharge volume, pH, total acidity, and concentration of dissolved metals. pH is the most critical component, since the lower the pH, the more severe the potential effects of mine drainage on aquatic life. The overall effect of acidic drainage is also dependent on the

flow (dilution rate), pH, and alkalinity or buffering capacity of the receiving stream. The higher the concentration of bicarbonate and carbonate ions in the receiving stream, the higher the buffering capacity and the greater the protection of aquatic life from adverse effects of acid drainage. Secondary effects such as increased carbon dioxide tensions, oxygen reduction by the oxidation of metals, increased osmotic pressure from high concentrations of mineral salts, and synergistic effects of metal ions also contribute to toxicity. In addition to chemical effects of mine drainage, physical effects such as increased turbidity from soil erosion, accumulation of tailings fines, and smothering of the stream substrate from precipitated metal compounds may also occur. In areas where surface and groundwater have been contaminated by mine drainage, treatment of water supplies becomes more difficult, more time consuming, and more expensive. Listed below are parameter constituents which are typically elevated in drainage emanating from sulphidic tailings or in groundwater recharged by acidic drainage and their properties which can render a municipal or domestic water supply unusable without treatment, unpalatable, or aesthetically offensive.

pH - The hydrogen ion concentration can affect the taste of water. At a low pH water tastes sour. The bactericidal effect of chlorine is weakened as pH increases and it is advantageous to keep the pH close to 7. Water with a pH below 7.0 is corrosive to plumbing and can result in constituents such as copper, zinc, cadmium, and lead being dissolved in drinking water.

Metals - Heavy metals can increase the toxicity of mine drainage and also act as metabolic poisons. Iron, aluminium, and manganese are the most common heavy metals which can compound the adverse effects of mine drainage. Heavy metals are generally less toxic at circumneutral pH. Trace metals such as zinc, cadmium, and copper, which may also be present in mine drainage, are toxic at extremely low concentrations and may act synergistically to suppress algal growth and affect fish and benthos.

Iron - The taste threshold of iron in water has been given as 0.1 and 0.2 mg/L of iron from ferrous sulfate and ferrous chloride respectively. It has been reported that ferrous iron imparts a taste at 0.1 mg/L and ferric iron at 0.2 mg/L. Staining of plumbing fixtures occurs at 0.3 mg/L. Certain animals are sensitive to minor changes in iron concentration. Cows will not drink enough water (taste threshold 0.3 mg/L) if it is high in iron, and consequently, milk production is affected.

Iron precipitates at a pH greater than 3.5 and does not re-enter solution at higher pH. Because iron can form precipitates at a lower pH than aluminium and can be present in streams with pH less than 4.5, separating the effect of iron from the effect of low pH is difficult. Precipitation of ferric hydroxide may result in a complete blanketing of the stream bottom,

adversely affecting both macroinvertebrates and fish. The severity is dependent on stream pH and the thickness of the precipitate.

Aluminium - Aluminium rarely occurs naturally in water at concentrations greater than a few tenths of a milligram per litre; however, higher concentrations can occur as a result of drainage from coal mines, acid precipitation, and breakdown of clays. The chemistry of aluminium compounds in water is complex. Aluminium combines with organic and inorganic ions and can be present in several forms. Aluminium is least soluble at a pH between 5.7 and 6.2; above and below this range, aluminium tends to be in solution. Of the three major metals present in mine drainage, aluminium has the most severe adverse effects on stream aquatic life. The addition of aluminium ions compounds the effect of low pH by interacting with hydrogen ions, further decreasing sodium uptake, and increasing sodium loss in blood and tissues. High calcium concentrations generally reduce mortality and sublethal effects of low pH and elevated aluminium by reducing the rate of influx of hydrogen ions into the blood. Streams most susceptible to degradation from elevated aluminium, however, normally have low concentrations of calcium.

Manganese - Elevated manganese causes several specific problems when encountered in drinking water, such as unpleasant tastes, deposits on food, laundry staining, reduction in water main capacity, and discoloration of porcelain fixtures. Staining may occur at concentrations above 0.5 mg/L.

Sulphates - High sulfate levels in water may have laxative effects and cause taste and odour problems.

Total Dissolved Solids (TDS) - Excessive TDS in drinking water is objectionable because of possible physiological effects and unpalatable mineral tastes. Physiological effects related to TDS include laxative effects, effects on the cardiovascular system, and toxemia associated with pregnancy.

5.3 *Tailings containing cyanides*

Tailings containing cyanides refer to tailings containing residual amounts of cyanide generated as wastes by activities associated with cyanidation operations. Typically these wastes are treated to neutralize or destroy cyanide prior to final disposal or closure. This treatment though identifies issues that are still outstanding.

5.3.1 **Origin and amounts**

Cyanide is a general term for a group of chemicals containing carbon and nitrogen. The main forms of cyanide that are manufactured by humans are gaseous hydrogen cyanide and solid sodium and potassium cyanide. Cyanide has unique properties that make it appropriate for a numerous applications, including the mining industry.

This industrial activity uses not less than the 20% of the total cyanide production. Approximately 20% of the total cyanide production is used to manufacture sodium cyanide, being a form of cyanide that is relatively easy and safe to handle. Of this approximately 90% (totalling to the ~18% of the total production) is used in mining, mostly in the extraction of the precious metals (gold, silver) from ore [61], [62], [63]. In addition it is used in low concentrations as an agent in the flotation process (as a depressant for pyrite FeS₂) that is used in the processing of the lead, copper and zinc metals (sulphide ores).

The usage of cyanide in precious metals recovery seems to be critical due to the resistance of these metals, especially gold, to attack by most chemicals resulting to its inefficient extraction. It is one of only a few chemical reagents that will dissolve gold in water, being available at a reasonably low cost.

Gold is a very valuable natural resource. Therefore it's worth mining even if it originates from low-grade (grams/tonne range) ores, utilising very dilute solutions of sodium cyanide typically at a range of 0.01% to 0.05% cyanide. This results in the production of large amounts of tailings relative to the amount of gold produced. 1 tonne of gold (assuming recovery of 100%) requires approximately 200,000 tonnes of ore (at a gold grade of 5 g/t) to be mined (EC BAT, 2003).

In 2000, there were about 875 gold or gold/silver mining operations in the world (without including relative contribution from base metal mines, where gold is recovered as by-product). 460 of these mines, representing a percent of 52%, utilised cyanide, 15% of them were heap leaches and 37% used cyanidation in tank leaching. The remaining 48% used a variety of processes that did not use alternative chemical agents or lixiviants but instead used processes like gravity separation and flotation to form a concentrate that was finally sent to a smelter for final processing. The world distribution of gold or gold/silver mines that utilise cyanidation in the year 2000 can be viewed in Figure 44.

In order to solve the problems that arise from the unique usage of cyanide to recover gold especially from finer gold particles, there is ongoing research that is focused on the replacing of cyanidation with less hazardous techniques, and research on the destruction or recycling of the cyanide in the tailings.

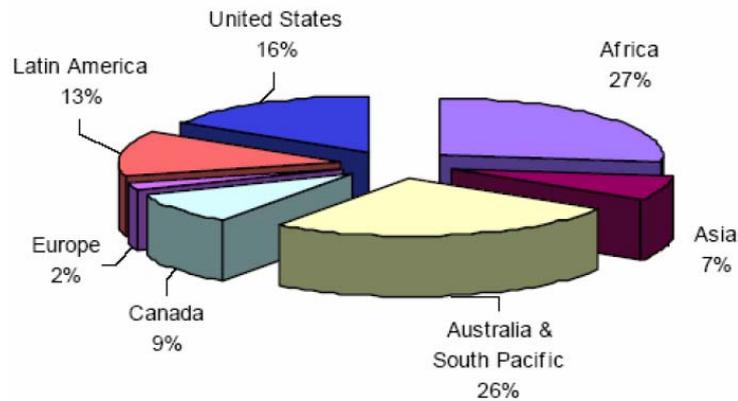


Figure 44: World distribution of gold or gold/silver mines that utilise cyanidation in the year 2000

5.3.2 Cyanidation process

Gold typically occurs at very low concentrations in ores (less than 10g/ton) [141]. At such concentrations the use of aqueous chemical extraction processes is the only economically viable method of extracting gold from ore. Typical hydrometallurgical gold recovery includes a leaching step during which the gold is dissolved in an aqueous medium, followed by separation of the gold bearing solution from the residues or adsorption of the gold onto activated carbon. After elution from the activated carbon, the gold is further concentrated by precipitation or electrodeposition.

Cyanidation uses solutions of sodium or potassium cyanide as lixivants to extract precious metals from ore. Cyanidation techniques used in the gold industry today include heap or valley leaching; agitation leaching followed by carbon-in-pulp (CIP) and agitated carbon-in-leach (CIL). Cyanidation is best suited in fine-grain gold in disseminated deposits. Heap or valley fill leaching is generally used to beneficiate ores containing less than 0.124 grams/ton. CIP and CIL techniques commonly referred to as tank or vat methods are generally used to beneficiate ores containing more than 0.124grams/ton. These cut-off values though are dependent on many factors, including the price of gold and an operator's ability to recover the precious metal. Figure 45 comprises a general overview of gold extraction and beneficiation, where cyanidation operations are clearly presented.

In order to ensure economic and optimal recovery of the gold prior to the leaching step, a reduction of the particle size by crushing and grinding of the ore is necessary for the liberation of gold. If the ore contains large gold particles, a gravity recovery process may be utilised in order to avoid long dissolution times during cyanidation leaching.

Ores that contain gold associated with sulphide minerals or even carbonaceous minerals may require additional pre-treatment. Usually this comprises of a flotation process in the case of sulphide minerals, since otherwise during leaching, cyanide preferentially leaches the sulphide minerals rather than gold, consuming cyanide in the formation of thiocyanate. In case of the carbonaceous minerals, they adsorb gold once it is made soluble, thus an oxidation step (pressure or biooxidation) of the ore prior leaching may be included to prevent this phenomenon.

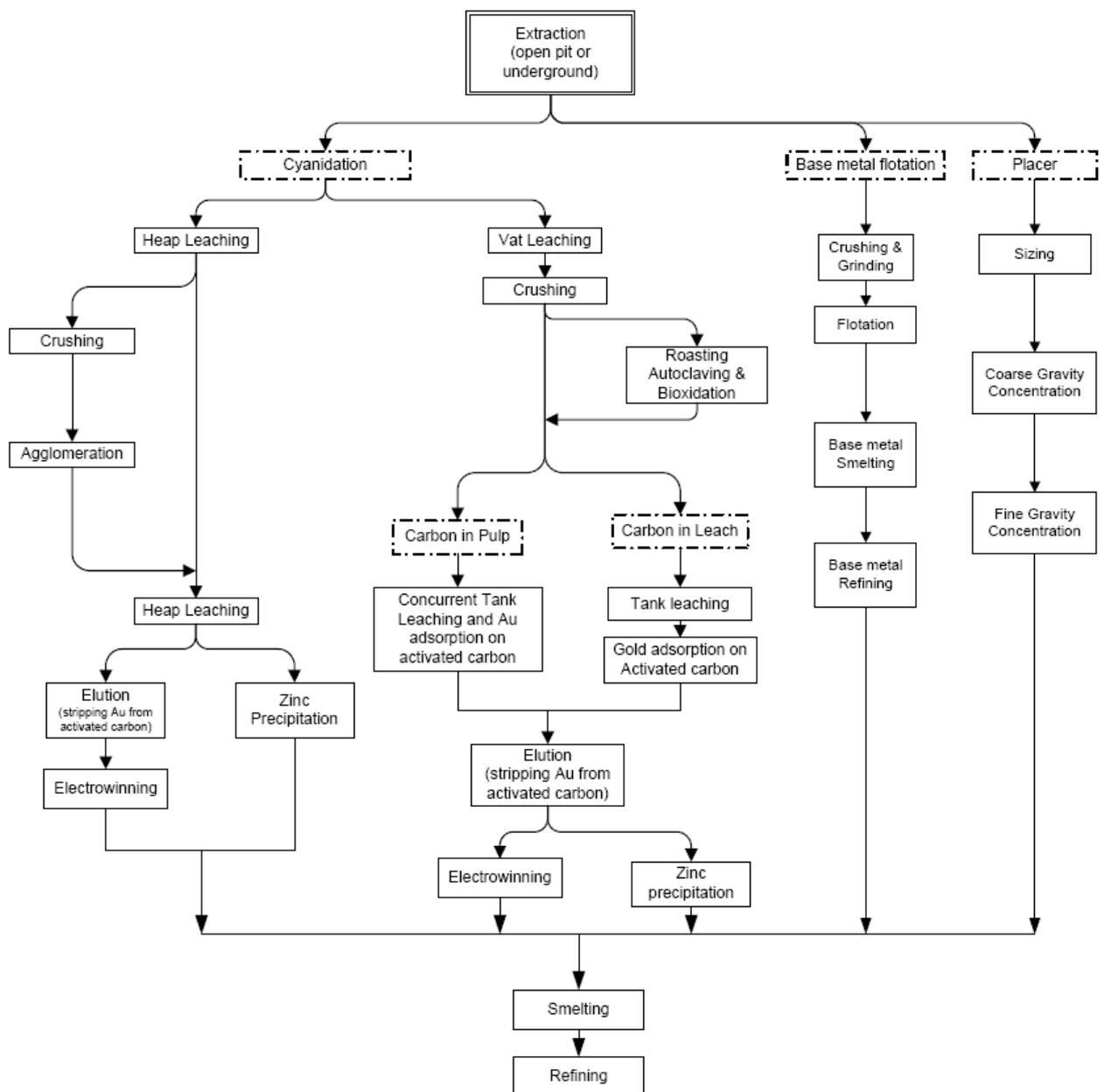


Figure 45: General overview of the gold extraction operations [35], [36]

A generalised scheme describing the overall cyanide cycle within the gold extraction process is given below (Figure 46).

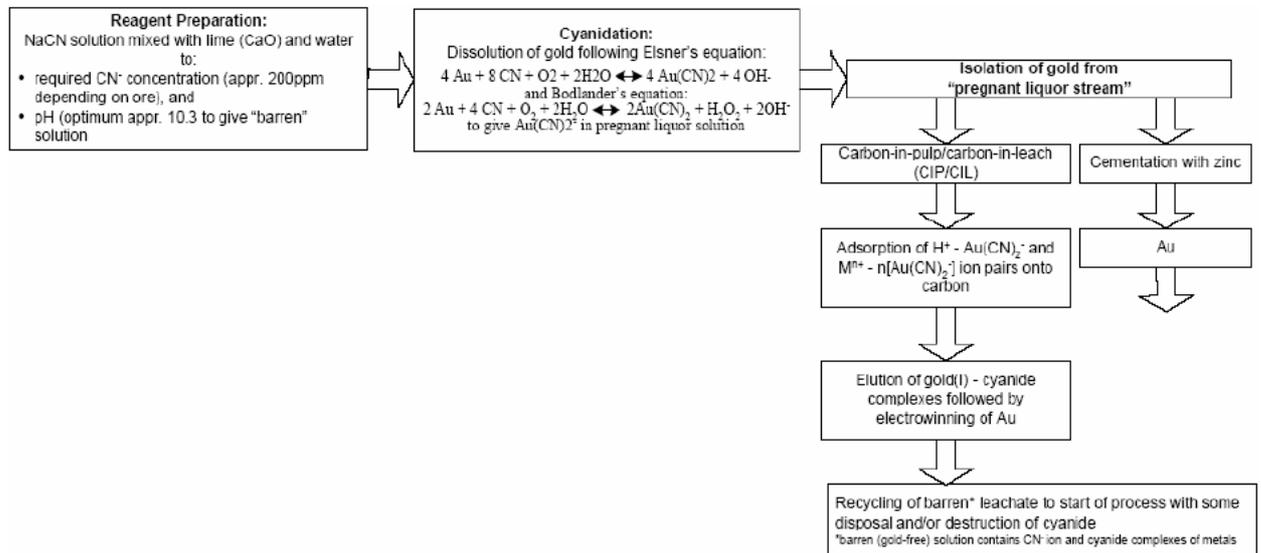


Figure 46: Cyanide cycle within gold extraction process

Practically, gold is leached in aqueous cyanide by oxidising it with an oxidant such as dissolved oxygen and complexing it with cyanide to form a stable gold cyanide complex. Cyanide concentration though fluctuates, since it depends on the ore body and whether metals like copper, iron, zinc are present, as they are also cyanide consumers (they form stable complexes).

With heap leaching, the ore is stacked in heaps on a pad liner with an impermeable membrane. Cyanide solution is then introduced with sprinklers or a drip irrigation system; the cyanide leachate percolates through the heap, leaching gold from the ore and is collected by pipes to ditches around the pile. Pregnant solution is collected on the impermeable membrane and channelled to storage facilities (lined pond or tank) for further processing. Irrigation of the heap generally stops when the pregnant solution falls below about 0.016 g of gold per ton of solution. Recovery rates range from 60-80%. Heap leaching is attractive due to the low capital cost involved, while its disadvantage lies on the slow process resulting to relatively low gold extraction efficiency. An "average/normal" leach cycle takes approximately three months.

When no further recovery of gold occurs, thus leaching has been completed, spent ore and the remaining cyanide solution become wastes. Typically the heap is rinsed with water until the cyanide concentration in the effluent is below a specific standard.

Agitated or tank leaching requires the ore to be of a powder consistency. Leaching of the milled ore is taking place to a series of leach tanks, by means of agitation either mechanical or air injection, in order to enhance the efficiency of the leach process. pH is settled to 10-11 using lime in order to keep cyanide in the solution, without generation of hydrogen cyanide gas which escapes from the solution. Oxide ores require a 4 to 24 hour period at a pulp density of 50%, while sulphide ores a 10 to 72 hour period at a pulp density of 40% solids. Tank operations present significant recoveries from 85-98% of the gold contained in the ore.



Figure 47: Gold heap leaching



Figure 48: Gold vat leaching

The recovery of the dissolved gold is taking place either by the Merrill-Crowe process (cementation with zinc powder) either with the adsorption of gold onto activated carbon, elution and /or cementation with zinc and electrowinning [35], [36].

Zinc cementation requires a clear solution which implies the use of a solid liquid separation process by means of filtration or counter current decantating. Due to the fact that these are capital intensive operations, the adsorption onto activated carbon becomes more attractive. The adsorption is performed by introducing highly activated carbon either directly into the leach tanks (carbon-in-leach, CIL) or into separate tanks following leaching (carbon-in-pulp, CIP). This method can recover more than 99.5% of gold within 8 to 24 hours. Loaded carbon is separated from the pulp using screens or hydrodynamically swept.

The material from which the gold has been removed by adsorption or liquid solid separation, or otherwise known as barren ore, form the tailings. These are either dewatered to separate the cyanide containing solution for destruction or recycling or for disposal in the tailings management facility (TMFs).

Gold is then recovered from the carbon by means of elution, which uses a caustic aqueous cyanide solution. Gold is recovered from the eluate either by zinc cementation or electrowinning. Carbon is regenerated and returned to the adsorption circuit. Calcination of the gold concentrate follows or it is directly sent for smelting into gold bullion that typically contains about 70-90% gold. Further refining of gold leads to 99.99% fineness, using chlorination, smelting and electrorefining and nowadays, solvent extraction. In gold operations tailings generally are not stored underground, if possible; it is prudent to dispose of tailings on the surface in order to be easily available when higher gold prices and/or more efficient extraction processes exist.

Cyanide in tailings may be treated for cyanide removal (most European sites) or left as is for removal by natural degradation in the tailings pond (international standard). Disposal in rivers and lakes is frowned upon for environmental reasons. Generally tailings are dewatered (as previously mentioned in the gold operations above) prior pumping to the tailings dump, either by counter current decantation or by filtration.

Seepage collection trenches of the dam may recycle the collected water to the tailings or to the mill, depending on the water balance of the plant, or dispose of it elsewhere after treatment. The same goes for the excess solution from the dam (return to the mill), if the extent of the dissolved base metal impurities allows such recycling. In the case where it has to be rejected, environmental regulations apply to the so-called bleed stream. It may have to be discharged through activated carbon, ion exchange columns or to be submitted to a process for the destruction of the residual cyanide.

5.3.3 Fate of cyanides (mechanisms of cyanide degradation)

The processes that decrease the concentration of cyanide in solution, whether in the natural environment, or in engineered facilities, are called "attenuation" [60].

Cyanide is generally considered as a "transient pollutant", since its toxic properties decrease rapidly over time, without the need to destroy it with chemical agents. This refers to the natural degradation of cyanide and which is caused by volatilization, photodecomposition, biodecomposition and conversion to thiocyanate. The simplified diagram of Figure 49 presents this route that cyanide undergoes naturally within a tailings management facility.

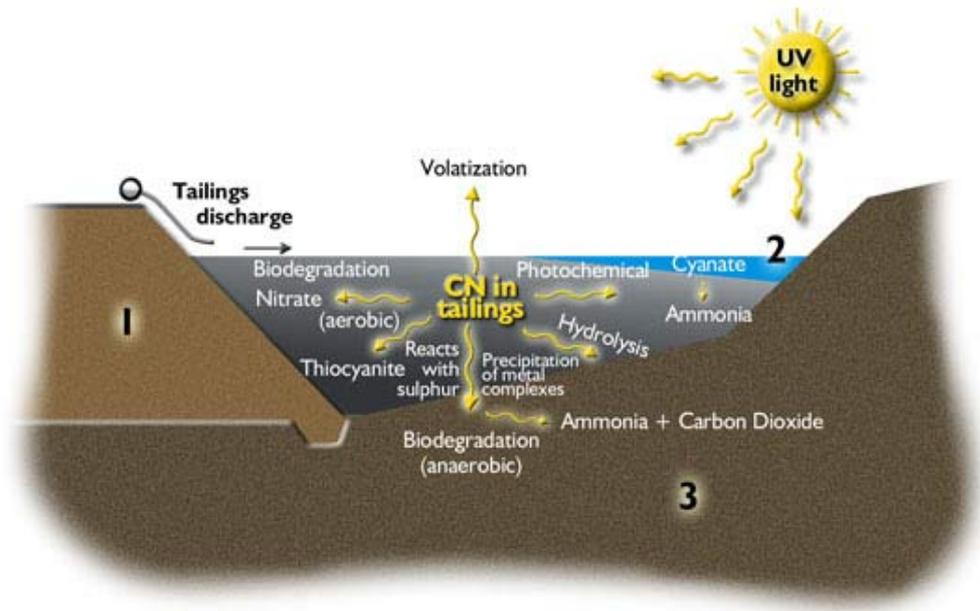


Figure 49: Simplified diagram of cyanide degradation processes in TMFs [117]: 1. Waste Rock Embankment. 2. Decant pond. 3. Natural ground.

Cyanide decay and its transformation processes can be very effective in reducing cyanide concentrations both within the decant pond water and within the tailings itself.

Study and research has been conducted (i.e. [4]) in order to understand the environmental fate of cyanide in and around tailings dams. Drilling has also been carried out to investigate the cyanide levels at a depth within the tailings storage, along with lateral sampling, which determines the variation of cyanide concentrations from the point of deposition. These variations can result either from the depth and/or the "age" of the tailings within the tailings management facility.

This investigation certainly concluded the fact that the cyanide concentration decreases significantly with respect to the depth due to decay and transformation processes. A knowledge of cyanide chemistry helps in understanding and predicting its behaviour in such an environment. For this reason some general information on the chemistry of cyanide is included here as well.

The term cyanide as it is used by the mining industry, regulators and laboratories, refers to one of three classifications of cyanide and which are: total cyanide (CN_T); weak dissociable cyanide (CN_{WAD} or WAD); and free cyanide (CN_f), as they are presented in Figure 50 [62], [63].

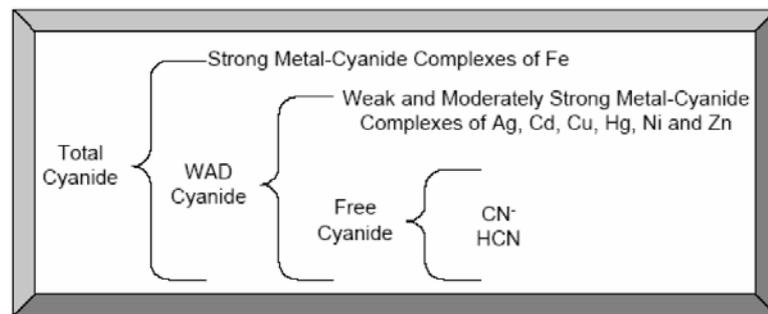


Figure 50: Classifications of cyanides [16]

Cyanide in chemistry, is a singularly charged anion consisting of one carbon atom and one nitrogen atom joined in a triple bond. The most toxic form of cyanide is free cyanide, comprising of the anion itself and the hydrogen cyanide in a gaseous or aqueous state. Diagrams describing the CN^-/HCN equilibrium with pH, show that at a pH of ~ 9.4 under normal conditions of temperature and pressure, their concentrations are in equilibrium. Nearly all free cyanide is present as HCN at a pH of 8 or less (making it able to volatilize and disperse into air), while at a pH greater than 10.5 nearly all free cyanide is present as CN^- [111].

Cyanide is also very reactive forming simple salts with alkali earth cations and complexes of varying strengths with numerous metal cations; its stability depends on the cation and the pH value. Salts of sodium, potassium and calcium cyanide (KCN , $\text{NaCN} \cdot 2\text{H}_2\text{O}$, $\text{Ca}(\text{CN})_2$) are quite toxic, as they are very soluble in water, resulting in the formation of free cyanide. Weak or moderately stable metal complexes such as those of cadmium, copper and zinc ($\text{Cd}(\text{CN})_4^{2-}$, $\text{Cu}(\text{CN})_2^-$, $\text{Cu}(\text{CN})_3^{2-}$, $\text{Cu}(\text{CN})_4^{3-}$, $\text{Zn}(\text{CN})_4^{2-}$) are known as weak-acid dissociable (WAD). Their solubility is described in their name and although they are less toxic than free cyanide, in case of dissociation they can release toxicity by the free cyanide produced and also by the potential toxicity of the metal cation. WAD metal cyanide complexes can dissociate even in neutral pH range of surface waters, presenting potential danger for the environment.

Cyanide forms complexes with gold and iron, ($\text{Au}(\text{CN})_2^-$, $\text{Fe}(\text{CN})_6^{4-}$, $\text{Fe}(\text{CN})_6^{3-}$) which are stable under mildly acidic conditions. Both ferro- and ferricyanides however decompose to release free cyanide when exposed to direct ultraviolet light in aqueous solutions, a process that is reversed in the dark. Metal cyanide complexes can also form salt type compounds with alkali or heavy metal cations, such as potassium ferrocyanide ($\text{K}_4\text{Fe}(\text{CN})_6$) or copper ferrocyanide ($\text{Cu}_2[\text{Fe}(\text{CN})_6]$). Nearly all alkali salts of iron cyanides are very soluble, liberating free cyanide upon dissolution. Heavy metals of iron cyanides though form insoluble precipitates at certain pH values.

Thiocyanate, SCN^- , is formed when cyanide combines with sulfur, which dissolves in weak acidic conditions. It is not included in the WAD cyanides since it presents similar complexity with cyanide. Moreover it is approximately seven times less toxic than hydrogen cyanide,

causing though lung irrigation since it oxidizes chemically and biologically into carbonate, sulfate and ammonia.

With oxidation of cyanide either in a natural or in a treatment process, cyanate, CNO^- , can be produced, which is less toxic than hydrogen cyanide and it is transformed into ammonia and carbon dioxide through hydrolysis [61], [62], [63].

With respect to the chemistry data presented for cyanides, drilling investigation presented that in the upper four to six metres of an active tailings pond, free cyanide rapidly decays. Below that depth the remaining WAD cyanide is present mainly as copper cyanide complexes, which through transformation processes is converted to the non-toxic and very stable iron cyanide and insoluble $CuCN$. These processes have been observed in other similar facilities and prevent long term leaching of cyanide from the tailings. This phenomenon is reflected in the cyanide concentration in the tailings underdrainage water.

Table 4 shows data from natural degradation systems at a number of gold mines around the world, demonstrating the ability of this degradation to reduce the cyanide concentrations in solutions.

Table 4: Natural degradation of cyanide in tailings impoundments [110]

Mine	CN entering the tailings system (ppm)	CN discharged from the tailings system (ppm)
Lupin, NWT, Canada	184	0.17
Ridgeway, S.Carolina, USA	480	0.09
Golden Cross, N. Zealand	6.8 (WAD)	0.33 (WAD)
Cannon, Washington, USA	284	<0.05

A more detailed diagram that focuses on natural degradation of cyanide within the boundaries of a tailings dam, mainly through hydrolysis, photodegradation, chemical and bacteriological oxidation, precipitation as metal complexes and the important processes of dissociation of metal cyanide complexes and volatilization of cyanide as hydrogen cyanide, is presented below, Figure 51. Along with these, the main factors affecting the degradation, which are considered to be pH, temperature, ultraviolet irradiation and aeration can be observed.

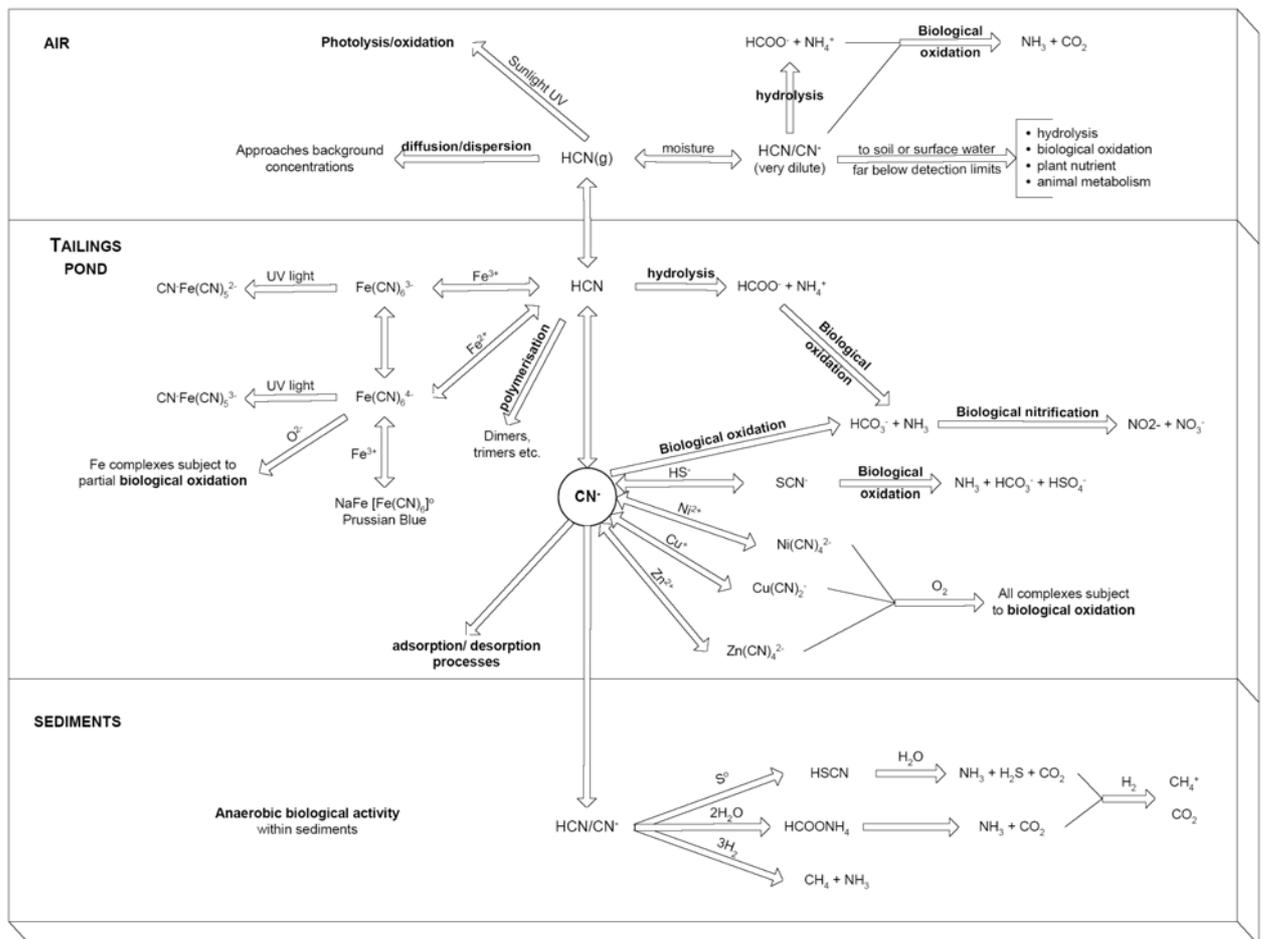


Figure 51: Cyanide chemical loss pathways in the environment [117]

5.3.4 Critical factors

This section attempts to gather the critical factors or parameters of a tailings management facility that associate with the cyanide behaviour within its boundaries, or the parameters, better referred to as characteristics, of the tailings containing cyanides associated with the operation of a tailings management facility.

In terms of the behaviour of tailings containing cyanides in a tailings management facility, the previous section that described the mechanisms of cyanide fate and migration, presented critical factors that associate with the processes of natural degradation. These include pH, temperature, ultraviolet irradiation, aeration, that govern the ways through which cyanide is lost within a tailings dam, remaining in the long term as its less toxic and strongly complex forms. Therefore the preservation and maintenance of the tailings dam in order to provide such appropriate conditions can enhance the cyanide tailings degradation process, with minimum hazard potential to the environment. Furthermore it has been noted that large surface areas can increase the contact with CO_2 in air, thus lowering pH values and enhancing the volatilisation pathway of cyanide. Tailings dams are generally ideally designed to provide long term storage of materials containing such complexes that result from natural cyanide

degradation, as well as security against escape of more toxic forms (i.e. WAD or free cyanide) through seepage, overtopping, breaching, pipe/channel failure, etc.

On the other hand cyanidation tailings exhibit unique characteristics besides the elevated levels of cyanide, complexed metals, cyanate and thiocyanate and the general characteristics of tailings such as, grain size distribution, solid to liquid ratio, ARD-characteristics, mineralogy and trace element content. These tailings-specific factors determine the leaching characteristics of the material and must be used in the design of the tailings dam. This is in order to evaluate the potential impact on its structural stability, its seepage characteristics along with the potential chemical aspects of seepage or other discharges from the dam. Cyanide presence dictates the importance of the seepage water quality, which depends on the operating conditions. Initially when tailings deposition commences, it is apparent that the flows and the cyanide concentrations in the tailings underdrains will be high. This is due to the fact that the water can short-circuit the tailings and find its way directly into the underdrainage system. Over time, the tailings begin to consolidate over the underdrains and the flow of water begins to decrease. The same behaviour follows the concentrations of cyanide. Thus after the high values of the concentrations present with the tailings deposition, the cyanide level can drop to a point where its concentration is so low that the underdrainage water may not even require treatment at a water treatment plant. Moreover, cyanide settles within the tailings as ferrocyanide which is non hazardous and emission of ferrocyanide in seepage from a slowly leaking tailings structure is generally limited in amount. If exposed to sunlight, the complex could break down into cyanide but it is likely to be very diluted.

Besides cyanide, cyanidation tailings as a product of gold operations usually associated with sulphide or carbonaceous minerals, may contain an array of metal contaminants such as arsenic, copper, lead, manganese etc. and/or exhibit an acid drainage generation from the oxidation of the present sulphides. The possibility of acid production in tailings dams (especially after decommissioning) constitutes the real threat to the environment. Acid mine drainage could create conditions which not only dissolve ferrocyanide within the tailings, releasing the cyanide but may also free many heavy metals.

5.3.5 Potential Environmental Impacts of TMFs containing cyanides

The potential environmental impact due to the operation of TMFs containing cyanides are presented in detail in the following paragraphs.

Humans

Cyanide is a very fast acting poison that is capable of killing a person within minutes if he or she is exposed to a sufficiently high dose. Humans may be exposed to cyanide by inhalation, ingestion or absorption through the skin. Cyanide prevents oxygen from being used by the cells, causing tissue hypoxia and "cyanosis" (a bluish discoloration of skin). The respiratory system fails to nourish the cells with oxygen, a condition, which, if untreated, causes rapid, deep breathing followed by convulsions, loss of consciousness and suffocation.

Although there are many everyday sources of exposure to cyanide (automobile exhaust, tobacco smoke, fires), cyanide does not accumulate in tissues because the body transforms such small amounts into a less toxic compound called thiocyanate, which is then excreted. Cyanide is not known to cause cancer or birth defects or adversely affect reproduction. The most toxic form of cyanide is HCN gas. The American Conference of Governmental Industrial Hygienists (ACGIH) lists the ceiling threshold limit at 4.7 ppm. At concentrations of 20 to 40 ppm of HCN in air, some respiratory distress may be observed after several hours. Death occurs in minutes at HCN concentrations above approximately 250 ppm in air.

For free cyanide, the lethal dosage to humans by ingestion or inhalation ranges from 50 to 200 mg (1 to 3 mg of free cyanide per kg of body mass). The lethal dosage for dermal adsorption is considerably higher at about 100 mg/kg of body weight. Table 5 gathers information on typical doses versus response data of humans and various animal species to concentrations of HCN in air.

Table 5: Response of humans and animals to HCN in air

Species	Concentration (ppm)	Response
Human	270	Immediately fatal
	181	Fatal after 10 minutes
	135	Fatal after 30 minutes
	110-135	Fatal after 30 to 60 minutes or longer
	45-55	Tolerated for 30 to 60 minutes without immediate or subsequent effects
	18-36	Slight symptoms after several hours
Mouse	1300	Fatal after 1 to 2 minutes
	110	Fatal after 45 minutes exposure
	45	Fatal after 2.5 to 4 hours exposure
Cat	315	Quickly fatal
	180	Fatal
	125	Markedly toxic in 6 to 7 minutes
Dog	315	Quickly fatal
	115	Fatal
	90	May be tolerated for hour; death after exposure
	35-65	Vomiting, recovery, convulsions; may be fatal
	30	May be tolerated
Rabbit	315	Fatal
	120	No marked symptoms

Fauna and Flora

Fauna

Cyanide affects not only humans, but also ecological receptors. For mining environments three groups of ecological or environmental receptors are of concern.

- Mammals, reptiles and amphibians
- Birds (especially migratory wildfowl)
- Fish and other aquatic life

Table 5 provides the cyanide intoxicification data according to different mammalian species. Birds are most at risk from tailings dams containing cyanide. Other species are also at risk (e.g. bats, lizards and small mammals), although the impacts on these species are rarely monitored. If tailings ponds contain WAD cyanide at levels less than 50 mg/L and access to the ponded area and releases of water to the environment are avoided, the impact on wildlife is low.

In addition to minimising cyanide use and impact through the effective application of process chemistry and risk assessment procedures, further practical measures can minimise the effect of cyanide in tailings dams and ponds on wildlife and domestic animals as a second line of protection. These should supplement actions taken to reduce risks to wildlife through careful process chemistry design, operation and monitoring, and should never be seen as an adequate wildlife protection measure in themselves. These measures are generally either to prevent access, or to scare away, and include:

- fencing,
- floating balls,
- netting (though this is not effective for large impoundments and is now being replaced by floating balls), and
- a variety of hazing (fireworks, lights, music, sound guns).

Surrounding the tailings facility with water drinking troughs and having decoy wetlands can also be effective in minimising the risk of poisoning wildlife. Proximity to human settlements may influence the choice of technique, e.g. possible community impacts from noise and light need to be taken into consideration.

While netting and hazing can help prevent bird losses, correct tailings dam design is also important in this respect. If dams have extensive surface areas they will attract more birds, particularly waders and swimmers.

To reduce the need for methods such as hazing, management should first ensure that:

- the cyanide levels in the tailings are as low as practicable; and
- ponds have a small surface area preferably covered with netting and fenced.

Cyanide is one of the most toxic chemicals to which fish are likely to be exposed. Fish are approximately one thousand times more sensitive to cyanide than humans. Dose levels as low as 0.03 mg/L HCN can be ultimately fatal to sensitive species, while 0.2 mg/L is lethal to most species. In each case, less than lethal levels do provoke physiological and pathological responses that reduce swimming ability, interfere with reproductive capacity and can lead to seriously deformed offspring, and also leave fish more vulnerable to predators. The ANZECC Water Quality Guidelines provide trigger values for cyanide of 4, 7, 11 and 18 $\mu\text{g/L}^{-1}$ in order to ensure confidence of species protection at the 99, 95, 90 and 80% levels respectively in ecologically significant waters. Mine design must incorporate features to avoid the release of water containing cyanide that may approach these trigger values for any ecologically significant aquatic systems in the vicinity.

Cyanide is not a persistent toxic chemical in the environment and under normal conditions will not permanently destroy a fish habitat. Due to this acute sensitivity, fish make excellent biological markers for the presence of cyanide in water. If fish are living after exposure, then no other form of life will be harmed.

In addition to the total cyanide level in the water, a number of other factors associated with water chemistry exert a modifying effect on acute toxicity; these include: dissolved oxygen concentration, temperature, pH, salinity, and other dissolved constituents (e.g., zinc, ammonia). Cyanide toxicity in fish increases 3-fold with a 12° C decrease in temperature. Furthermore, seventeen parts per thousand chloride ion (full strength sea water) decreases the survival time.

Flora

Limited data exists on the effect of cyanide on plant growth. Given that cyanide is not stable and decomposes when exposed to atmospheric conditions, no severe impact is expected on plant growth over tailings containing cyanides. Plant growth trials over oxidised tailings, involving the establishment of both grass and native species in both fertilised tailings and tailings with compost added, have shown that the survival rate of native plants in tailings was 100%. The addition of compost caused significantly higher growth rates in the first six months. Beyond six months no significant differences were observed.

Surface, ground water - Cyanide in tailings sites mainly seeps slowly out of the tailings, potentially contaminating surface and ground waters. The environmental impacts of these spills have nearly all been related to the toxic effects on aquatic life resulting from cyanide entering surface water, such as a river or the alluvial aquifer. In some instances, the impacts

can be more severe than anticipated due to the type of response taken by the gold mining operation.

Soil - Soils can be contaminated with tailings through wind erosion phenomena. However cyanide contamination is considered to be limited due to the natural degradation of cyanide.

Air - Given the decomposition of cyanide compounds when exposed to atmospheric conditions no environmental impact is expected in the air quality surrounding the tailings management facility.

Natural degradation of cyanide within the tailings dam boundaries, as presented above, implements a cyanide pathway of migration in air via its transformation to hydrogen cyanide whilst a plant is operating. Moreover it is generally assumed that it is the main way cyanide passes into the environment, over the fairly short term, once tailings slurry enters the tailings structure. Based on these assumptions, the effect of the HCN on the environment is insignificant because this is a relatively small amount compared to the HCN that is pumped into the atmosphere by forest fires. Also, HCN is very soluble in water and so, it is likely to get degraded quite quickly by contact with any moisture in the air.

5.3.6 Prevention of cyanide incidents

The occurrence even nowadays of significant environmental events involving cyanide indicates that knowledge and systems related to cyanide management in mining demand improvement. In particular engineering standards in the design, construction and maintenance of tailings dams, along with the water management or water balance, and tailings management require more attention.

Generally the failure modes considered are instability, overtopping of dams and internal erosion. In long-term safety and failure modes, seepage, dust and long-term erosion are also considered, with even more efficient measures in case of the retention of hazard potential for a long time, like cyanide. The accidents though, implement a combination of reasons for failure of tailings dam. A chain of events for such accidents includes the use of an inappropriate design, the acceptance of that design by the permitting authorities and the inadequate monitoring and dam construction, operation and maintenance.

The most severe cyanide spills have resulted from the breaching or overtopping of tailings dams during high precipitation or runoff events, or the rupture of pipelines without adequate secondary containment and collections systems. In conjunction with the spills, improper emergency treatment exacerbated the impacts on aquatic life and the environment due to toxicity of the cyanide antidote chemicals employed, such as chlorine, sodium or calcium hypochlorite. In order to minimize the number and severity of these spills, there is a need for development and implementation of both spill prevention, plans and procedures.



Figure 52: Dam failure and leakage

Generally the transport, storage, use and disposal of cyanide at a mine site can be hazardous both to human health and the environment. Best practice environmental management requires minimising the risk of environmental damage from accidental cyanide release. The main goals should be to:

- use the minimum effective amounts of cyanide required to recover metals;
- dispose of cyanide in a way that eliminates or minimises environmental impacts; and
- monitor all operations, discharges and the environment to detect any escape of cyanide and subsequent impacts of such a release - and conduct regular cyanide audits.

Other important actions include:

- allocating clear responsibility for cyanide management;
- understanding the actual and potential hazards and environmental impacts in transporting, storing, using and disposing of cyanide;
- maximising the recovery of cyanide where economically feasible or ensuring its destruction;
- constructing physical containment measures and implementing procedural measures to deal with spills or leaks of cyanide;
- conducting regular training in the above measures;
- to dispose of cyanide in a way that eliminates or minimises environmental impacts;
- having emergency response plans in place to ensure immediate action to minimise environmental effects should an accidental or unplanned release occur; and

- keeping adequate records with regular assessment so that future environmental problems can be anticipated.

A review of cyanide related spills over the last quarter century has shown the causes to include:

- the lack of a dynamic site water balance and comprehensive water management plan;
- the lack of or implementation of improper water treatment capabilities; and
- the lack of integrity and secondary containment within the solution conveyance system (i.e. storage tanks, mixing systems, pipelines or drains along which cyanide-contaminated waters may flow).

The three critical components for developing and implementing a cyanide spill prevention plan are therefore water management, water treatment and water quality, both generally throughout the mine site, and specifically in relation to the tailings storage facility.

5.3.7 Cyanide treatment

Cyanide treatment in a mining operation should most times be simultaneously considered as water management. Cyanide treatment refers to WAD cyanide since this includes the toxicologically important forms of cyanide, including free cyanide and weakly complexed metal cyanides. Total cyanide as already mentioned includes the relatively non-toxic iron cyanide complexes.

Cyanide treatment processes are classified as either a destruction-based process or a recovery-based process. Destruction processes use either chemical or biological reactions to convert cyanide to a less toxic form, while recovery processes focus on a recycling approach in which cyanide is removed from solution or slurry in order to be reused in the metallurgical circuit.

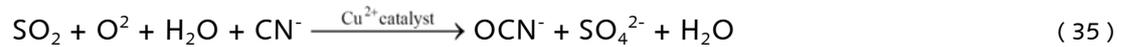
Cyanide treatment is also the natural attenuation of cyanide or natural degradation processes. These have been reviewed earlier in this report since they describe the fate of cyanide within a tailings impoundment.

Cyanide destruction processes

The main principle of the cyanide destruction processes is the conversion of cyanide to a less toxic form through an oxidation reaction.

INCO sulphur dioxide/air

This process was developed by INCO Ltd in the 80's and is used at over thirty sites worldwide. The Inco sulfur dioxide (SO₂)/air process is a simple method that requires little supervision and does not interrupt gold recovery. SO₂ in liquid or gaseous form acts with air to oxidise WAD cyanide to cyanate and sulfuric acid while releasing metals into solution:



Its applicability can take place either in tailings (slurry) or solutions. The process is performed in agitated tanks where SO₂ (as liquid or as sodium metabisulfite), oxygen (atmospheric air), lime for pH maintenance and copper (as a solution of copper sulphate) as a catalyst, are introduced. Their quantities are: approximately 3.5-4.5 g SO₂/g CN⁻ oxidised, lime appropriate for pH 8-9, and copper in order to provide copper concentration of about 10-50 ppm.

Hydrogen peroxide

An increasingly popular method for detoxifying residual cyanide is the Degussa peroxide process. Put simply, hydrogen peroxide oxidises free and WAD cyanides to cyanate, which is further hydrolysed to biodegradable ammonia and carbonate:



Metals such as copper, zinc and cadmium complexed with cyanide are precipitated as hydroxides and iron cyanide complexes. These are then removed by a further treatment step, which precipitates the iron cyanide complex by combining it with copper ions. It applies mainly in solutions other than slurries due to the high consumption of hydrogen peroxide in slurry applications. H₂O₂ used ranges from 2-8 g/g CN⁻ oxidised. The reaction takes place at a pH of 9-9.5 for optimal removal of cyanide and metals such as copper, nickel and zinc. Copper is required as a soluble catalyst and it is added as copper sulphate to provide a copper concentration of 10-20% of the initial WAD cyanide concentration.

Caro's acid

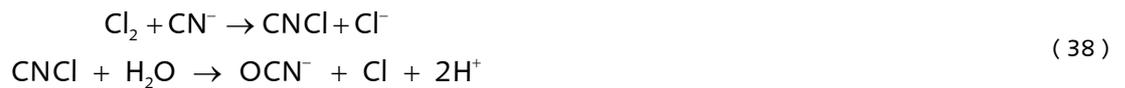
The most cost-effective way to partially detoxify tailings slurries is to use peroxymonosulfuric acid (Caro's acid), when a copper catalyst is not desirable. It can be generated safely on-site from hydrogen peroxide and sulfuric acid since it decomposes rather quickly. Because it is a stronger oxidising agent than peroxide, it is a more cost-effective use of these chemicals.



the quantity used is approximately 5-15 g H₂SO₅/g cyanide oxidised, while lime can be used to neutralise acid produced in the reaction.

Alkaline chlorination

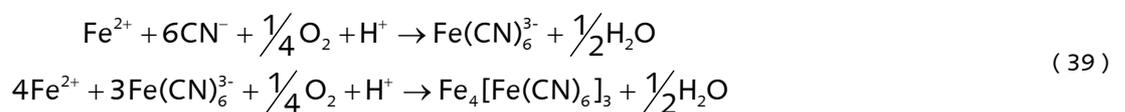
Alkaline chlorination is effective at treating cyanide to low levels but it can be relatively expensive to operate due to the high reagent consumptions. Its primary application is with solutions rather than slurries due to the high consumptions of chlorine in slurries. It is a two step reaction where cyanide converts to cyanogen chloride and cyanogens chloride to cyanate. Metals previously complexed with cyanide, such as copper, nickel and zinc, precipitate as metal-hydroxide compounds.



The reaction takes place at a pH of greater than 10 to ensure completely conversion to cyanate. Cl₂ used in the process can be provided as a liquid Cl₂ or as a 12.5% solution of sodium hypochlorite (NaOCl).

Iron-cyanide precipitation

This process is applicable in situations where the precipitation reactions can be controlled and the precipitated solids can be separated and disposed. Free, WAD and total cyanides react with ferrous iron, provided as ferrous sulphate, to yield a variety of soluble and insoluble compounds, primarily hexacyanoferrate (III) (Fe(CN)₆³⁻), Prussian blue (Fe₄[Fe(CN)₆]₃) and other insoluble metal iron cyanide compounds such as with copper and zinc.



Activated carbon polishing

Activated carbon has a relatively high affinity for many metal cyanide compounds, including soluble cyanide compounds of copper, iron, nickel and zinc. Activated carbon is suitable for use as a polishing treatment process to remove cyanide to relatively low levels. This is a simple and effective process, convenient for installation at sites where activated carbon is used in metallurgical processes for precious metals recovery.

Biological treatment

At present, considerable effort is being expended internationally to identify and isolate bacteria that can be used to facilitate degradation of cyanide in waste streams, tailings storage facilities and contaminated environments. A bio-degradation process has been used successfully since 1984 at the Homestake Mining Company in South Dakota, USA and at Homestake's Nickel Plate Mine in Canada. The first site employs an aerobic growth biological

treatment which is used to remove cyanide, thiocyanate, cyanate, ammonia and metals from tailings impoundments decant solution prior to discharge into a trout fishery. The second site treats tailings impoundment seepage and it is a suspended sludge system with both aerobic and anaerobic treatment sections.

The applicability of biological processes for treatment of cyanide solutions in the mining industry is somewhat limited, although it presents itself to be the optimal selection in certain situations. Overall, biotechnological approaches potentially have a significant role to play in detoxifying cyanide-containing waste waters but are not a universal panacea. Each mine site waste stream is unique; some possess the correct nutrient balance and availability for bacterial growth, others may require significant input of, say, phosphate.

5.3.8 Cyanide Recovery or Recycling

Stripping and recovery

When residual cyanide occurs as free and WAD cyanide, it may be recovered using various non-oxidative processes. Two such processes rely on reducing pH to release HCN. One is AVR (acidification-volatilisation-absorption) which uses shallow aeration basins and high pressure air blowers to recover free cyanide and some metal-cyanide complexes. The other method which is more efficient and cost-effective is CYANISORB[®], which was developed in New Zealand in 1989 and recovers about 90% of cyanide from tailings. HCN is removed when the tailings contact high volumes of turbulent air in stripping towers and is captured by hydrated lime slurries in absorption towers. Recovered cyanide is recycled to leaching operations as calcium cyanide. The three main reactions involved with the cyanide recovery process are as follows:

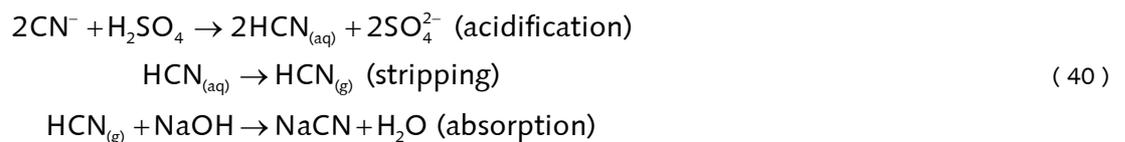


Table 6 summarises the suitability of treatment processes for removal of cyanide and its related compounds cyanate, thiocyanate, ammonia and nitrate. This simplified summary can be used as a conceptual screening tool when tabular evaluating cyanide treatment processes.

Water Recycling

Cyanide can be effectively recovered and reused by recycling cyanide containing solutions within a metallurgical circuit. This takes place by means of thickeners which separate solution from tailings solids and recycling the solution usually to the grinding and/or leaching circuit.

In the following table the advantages and disadvantages of cyanide treatment processes appear as a preliminary selection guide, which can also be used as a screening tool when evaluating these processes.

Table 6: Preliminary selection guide for cyanide treatment processes [16]

Treatment Process	Iron cyanide removal	WAD cyanide removal	Slurry application	Solution application
SO ₂ /air	√	√	√	√
Hydrogen peroxide	√	√		√
Caro's acid		√	√	
Alkaline chlorination	√	√		√
Iron precipitation	√	√	√	√
Activated carbon	√	√		√
Biological	√	√		√
Cyanide recovery		√	√	√
Natural attenuation	√	√	√	√

The key to successful implementation of cyanide treatment processes is to consider the following:

- site water and cyanide balances under both average and extreme climatic conditions;
- the range of cyanide treatment processes available and their ability to be used individually or in combination to achieve treatment objectives; and
- proper testing, design, construction, maintenance and monitoring of both water management and cyanide management facilities.

By carefully considering these aspects of water and cyanide management before, during and after mine operation, operators can reduce the potential for environmental impacts associated with the use of cyanide.

Another aspect of cyanide treatment to be considered is the potential environmental impact of the cyanide related compounds cyanate, thiocyanate, ammonia and nitrate. These compounds may be present in mining solutions to varying extents and may require treatment if water is to be discharged. Each of these cyanide related compounds is affected differently in the treatment processes discussed and this should be considered when evaluating cyanide treatment alternatives for a given site.

5.3.9 Monitoring cyanide in the environment

Monitoring levels of cyanide in the environment is an essential part of best practice management. It is generally accepted that only the analysis of total and WAD cyanide (by distillation) can be considered reasonably reliable. The first one is used to determine total

cyanide in solutions, including free cyanide and metal bound complexes, such as the more stable, non-toxic iron cyanides. WAD cyanide determination is used for free and complexed forms of cyanide, except iron cyanide. An older, but still used as alternative method for WAD determination is the one called "cyanide amenable to chlorination". In interpreting results of a monitoring program, the possible sources of cyanide reported in surface waters of treated effluents needs to be taken into account. The first is analytical error, the second naturally produced cyanide excreted by plants, micro-organisms and insects, and the third manufactured cyanide.

Cyanide analysis is needed for operational control, regulatory compliance and toxicity evaluation, as well as for public information. Monitoring results from the gold recovery operation can be used for the evaluation of good practice and for the protection of health and the environment. In order to achieve that it presents as good practice the data collection from periodical monitoring, which will refer to WAD cyanide in tailings water leaving the detoxication unit and at the water intake from the tailings dam, heavy metals such as As, Sb, Cd, Hg, Pb, Zn, Cr in the tailings water, water quality analysis including WAD cyanide downgradient of the tailings dam and hydrogen cyanide measurements at various locations and in the tailings dam area.

The environmental monitoring program, which should form part of the environmental management plan, should include:

- specification - of sampling sites, frequency of sampling, method of analysis, parameters to measure, use of certified reference materials, required action on detection of outliers or on non-compliance;
- baseline information - existing water quality (surface- and ground water);
- monitoring during and after operations -
- water levels and quality of surface- and ground water, process ponds, drinking water, tailings dams
 - dust generation and deposition
 - fauna
 - rehabilitation.

The cyanide-monitoring program must consider all these factors. Generation of acid tailings from waste rock dumps, for example, may change the chemical form of the cyanide and cause HCN gas to be released. Warning of such potential problems can be gained from knowledge of environmental exposure pathways linked to monitoring of pH, WAD cyanide and free cyanide.

5.4 Tailings containing alumina substances

Worldwide, the major problem the alumina refineries are facing nowadays is the disposal of the bauxite residue. The large amount of the bauxite residue produced by the Bayer Process (known as "red mud"), globally exceeds the 70 million dry metric tonnes per year, thus rendering compulsory the need of a systematic and liable waste management, giving emphasis on the reduction of the alumina tailings.

The fundamental issues that render the alumina tailings disposal a key subject of the alumina mining industry are of economic but mostly, of environmental nature. Although red mud is now safely disposed and it is therefore considered as a non-hazardous material according to the European Commission Decision 2001/118/EC regarding the list of wastes, there are still a number of environmental issues which call upon a systematic and reliable confrontation of the alumina tailings management facilities. Moreover, there is no guarantee that the designation and safety of the alumina tailings management facilities will remain unchanged in the future, or that there will not be a reclassification of bauxite residue as a hazardous waste at some future time [3].

By now, national and domestic environmental regulations worldwide, underline the necessity of a thorough environmental impact assessment by the alumina industries along with their operational and productivity framework. This includes studies on the potential environmental impacts affecting the natural habitat around the waste disposal sites, the land degradation, potential drainage, pollutants leached from tailings, the feasibility of rehabilitation etc. [119]. This is particularly obligatory for the alumina plants located in areas of considerable fauna and flora.

Concluding from the above, it is important to investigate the way in which alumina tailings affect the environment and also to detect all those critical parameters that should be considered during the designation and the operation of an alumina tailings management facility.

5.4.1 Origin and contaminants of the impounded material

Practically, all the world's alumina is made by the Bayer process, with variations to account for differences in the types and quality of bauxite used. Consequently, alumina tailings arise by the five-step process as shown in Figure 53.

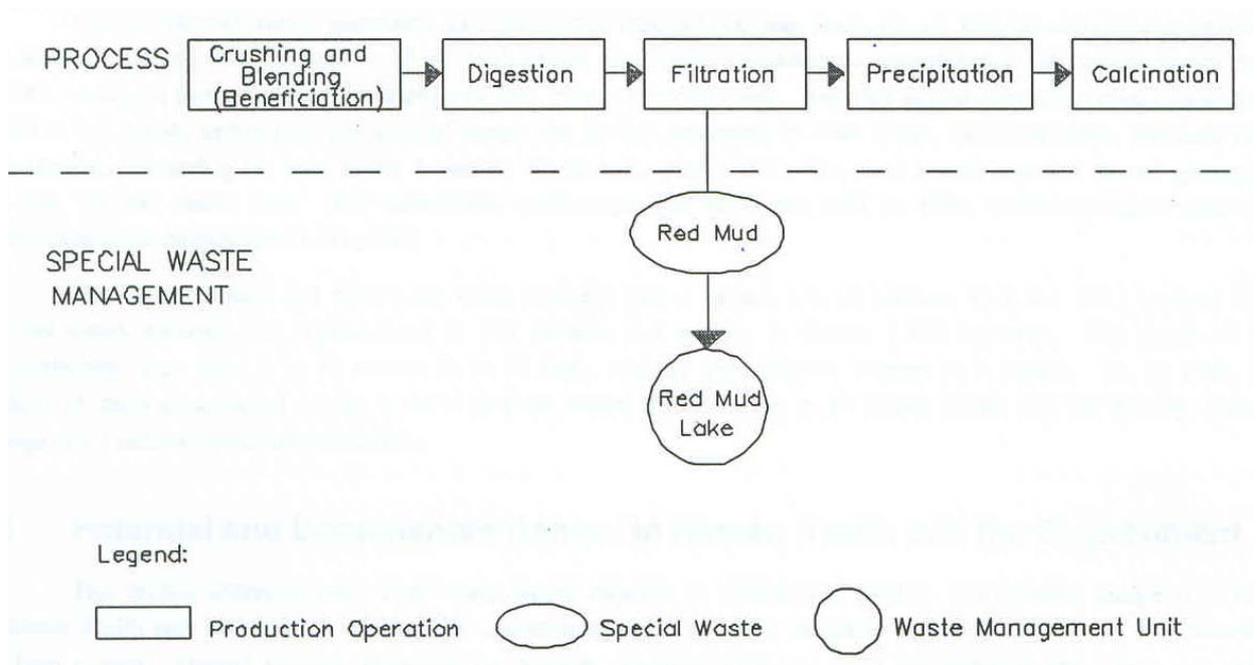


Figure 53: Alumina tailings production [34]

According to the Bayer process, in general, the bauxite ore is crushed, screened and mixed with a caustic alkaline solution (NaOH). The slurried ore is then routed to digesters, where the aluminium is heated and solubilized as sodium aluminate. In the third step, the solution is cooled and purified. Sand is removed in a settling tank or cyclone and sent to disposal. Iron oxide, silica and other undigested portions of the ore comprise the special waste, known collectively as red mud, which is also removed in settling, thickening and filtration units and sent to treatment and disposal units [34].

Thereby, alumina tailings consist of two major parts. The fine fraction, which accounts for 80-95% of total and is the red mud and a coarser fraction, commonly referred to as "process sand".

These two portions represent the 97-100% of the total tailings. In some cases the remaining 3% consists of salt cake, from a salting out liquor purification process and sludge (principally aluminium hydroxide) from the underflow of the clarifier. However, red mud is the main by-product of the Bayer process and any reference to alumina tailings management facility has to do with red mud.

Depending on the quality of the bauxite and the technical layout of the Bayer process, between 1.1 and 6.2 tons of red mud are generated per 1 ton of alumina produced [46]. However, a typical mass flow from bauxite to aluminium for European refineries gives 0.75t tailings for every 1 tonne of alumina produced [38].

Based on the mineralogical composition of bauxites and the Bayer process applied to extract alumina (reagents, flocculants etc.), red mud consists partly of minerals contained in the

bauxite that do not dissolve during the caustic treatment (e.g. iron and titanium oxide), calcium carbonate and calcium aluminates from lime addition, and also the desilication product, which contains silica and unrecoverable alumina and soda. The residue may also contain trace amounts of elements like chromium, nickel, magnesium, manganese, potassium, sulphur, zinc, zirconium, vanadium [38].

The chemical composition of red mud among some European refineries compared with the range found in residues from worldwide production is shown in Table 7.

Table 7: Chemical composition of alumina red mud from some European refineries as compared with the range found in residues from worldwide production

Component (%)	Fe ₂ O ₃	Al ₂ O ₃	SiO ₂	Na ₂ O	CaO	TiO ₂	LOI
Range ¹	30-60	10-20	3-50	2-10	2-8	Trace-10	NR
Ajka, Hungary ²	40	18	15	8	7	4	7
Aluminium de Greece ³	45	16	6	3	11	5	13
Aughinish, Ireland ²	47	17	7	5	8	12	3
Eurallumina, Sardinia ²	18	26	20	12	8	6	9
Gardanne, France ⁴	42	14	6	2	14	11	NR

NR: Not reported, LOI: Loss of ignition

¹<http://www.world-aluminium.org/environment/challenges/residue.html>

²European Commission (2003)

³Pontikis et al, 2003

⁴www.altech.pechiney.com/Gardanne

As can be seen, iron minerals constitute the principal components of red mud, followed by aluminium, calcium and silica compounds. Using available data on the composition of red muds, EPA evaluated whether this waste exhibits any of the four hazardous waste characteristics that is corrosiveness, reactivity, ignitability and extraction procedure toxicity. Based on available information and professional judgement, EPA came to the conclusion that red mud does not exhibit any of the characteristics of hazardous waste [34]. However, the same organisation identified intrinsic hazards in chemical constituents of red muds that may pose a risk to the environment.

Despite repeated washing during the clarification stage of the Bayer process, the solution entrained within the red mud still contains small amounts of caustic. Generally, red mud exits the process stream as a highly alkaline slurry (pH: 10-12.5) at ~15-40% solids content, while the water content is about 60%.

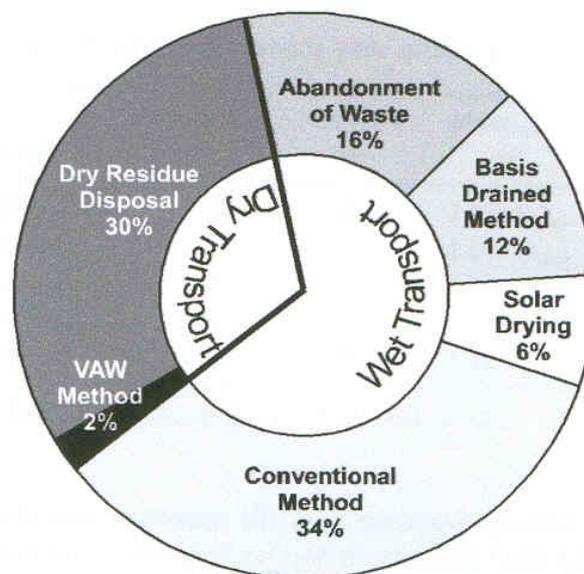
Research from EPA's Office of Water has also shown that the supernatant removed from red mud impoundments has a pH of roughly 11.6. Most of the caustic converts to sodium carbonate and sodium bicarbonate on the tailings disposal area. The residual alkali content of

the muds that are left in the impoundments makes it difficult to use these impoundment areas for agricultural production [6]. As well as the high alkalinity, red muds have some other physical characteristics that limit the possibility of revegetation, such as high salinity, low porosity and complete absence of nutrients or organic constituents [44].

Another parameter of environmental significance is the cation exchange capacity of red mud that is a measure of the potential leaching of sodium. Studies have shown that the amount of exchangeable sodium is higher for the red mud containing sodalite $[(\text{NaAlSiO}_4)_6 \cdot 2\text{NaCl} \cdot 0.6\text{H}_2\text{O}]$, i.e. a desilication product [75]. Thus the control of Bayer operating conditions to form alternate desilication product like hydrogarnet and cancrinite $[(\text{NaAlSiO}_4)_6 \cdot 2\text{CaCO}_3]$ might be beneficial. These compounds have lower $\text{Na}_2\text{O}/\text{SiO}_2$ values as compared with sodalite, resulting in lower amounts of soda loss in the red mud. A higher dose of synthetic flocculant added to the clarification stage of the Bayer process has also been seen to increase the exchangeable sodium of the red mud, however, further study is required to quantify the effect of this parameter [75].

5.4.2 Red mud disposal and risks

Tailings storage by its very nature poses significant risk to the environment. Therefore, all aspects of the design of a tailings management facility must carefully consider potential environmental impacts and must attempt to minimise these potential impacts. Typically, there are two red mud disposal techniques, those based on "wet" and "dry" transport conditions of the waste material, with subsequent techniques as shown in Figure 54. It is important to note that about 16% of red mud is not stored in proper disposal sites.



**The VAW method necessitates the disposal site to be constructed level by level (dry method).*

Figure 54: Percentage proportions of various disposal techniques used for the storage of red mud [46]

The mainstream of red mud, after separation from the pregnant solution and washing to recover the caustic soda, is commonly pumped to disposal areas. At some plants (e.g. Aughinish, Ireland), the red mud is neutralised by mixing with sea water. The amount of seawater required is 12-18 times the volume of red mud. Conventional disposal methods have revolved around the construction of clay-lined dams or dykes, into which the red mud slurry is simply pumped and allowed to dry naturally. The operation of conventional disposal areas was simple and inexpensive, however the potential impact on the surrounding environment, and difficulties associated with surface rehabilitation, forced significant changes in disposal practices [25]. Doubly sealed impoundments, incorporating a composite clay/geomembrane lining, and drained lakes, having a drainage network incorporated in the lining material, have subsequently seen widespread use. Drained disposal systems reduce the potential for seepage and increase the consolidation of the residue, improving the storage efficiency of the disposal area [37].

The wet disposal practice of pumping large tonnages of the residue to pondage in the form of dilute mud-sand slurries, which accounts for 70% of the red mud deposition, poses many technical, economical and environmental problems.

There are great volumes of material to handle and transport, a high energy requirement for pumping and consequential wear in pipeline and pumps. Other problems include slow rate of sedimentation and consolidation of fine mud leading to reduced life of lakes and slow rehabilitation, and potential leakage of caustic and alkaline water into underground water [22], [87].

Over the past 20 years, new techniques have been developed for the conditioning of red mud, which allow the residue to be thickened to 50-65% solids. These techniques mostly use vacuum or high pressure filters (e.g. Aughinish) or superthickeners (e.g. Alcoa). Deep or super thickening technology was developed by Alcoa to be applied for the thickening of Jamaican red mud. This mud is known to have the slowest settling behaviour of all muds, possibly due to the goethite therein and thus filtering was not feasible. Thickened red mud, from all alumina plants regardless of the bauxite source, behaves as a Bingham plastic material i.e. flow occurs for values of pressure exceeding the yield stress and is kept laminar for a specific range. The material is pumped to the disposal area and allowed to dry in the sun, resulting in a dry density of around 70% solids.

Dry stacking is primarily applied in arid climates, as solar radiation to concentrate the mud is needed. It is thought to be the best method of red mud disposal now available since it has reduced pumping costs and allows an effective use of land, minimising the environmental impacts [26]. Additionally, a thickened tailings disposal system, even when only requiring a small dam to be constructed, will only be a local problem if failure occurs and not an ecological disaster. This is because the material contained behind the dam, being well consolidated and without free liquid, can slump but cannot flow [102]. However, it requires large drying areas thus raising the possibility of dust generation [10].

An alternative method of red mud disposal is ocean dumping and in particular deep sea disposal. However, alumina operators using this method are limited. Pechiney, an operator in Marseilles, France has applied the deep sea storage method since 1967. Having a disposal rate of 4000 tonne/day, a submarine canyon at the depth of 1200m has been generated [140].

Environmental risks concerning the deep sea disposal of red mud include the smothering of benthic fauna at the ultimate deposition and along the route of the tailings flow which occurs when the deposition rate is greater than the ability of slow moving organisms to move away. At depth, the extent of this impact is difficult to predict because little is known about deep sea benthic organisms and deep sea ecosystems. Possible reactions and dependence between benthic fauna and surface fauna would imply risks to marine species throughout different ocean depths. A potential breakage of pipelines is, also, a subject of consideration at the deep sea storage facility of red muds.

5.4.3 Migration pathways of red mud

In general, the predominant migration pathways of tailings contaminants include air-borne and water-borne, as shown on Table 8. Accordingly, the potential migration of the alkaline and corrosive contaminants of the alumina tailings may involve aerial or aquatic pathways.

Table 8: Mine contaminant pathways

Pathway	During mining	After mining
Air-borne	Potentially very important and the major issue in most surface mine applications.	After revegetation, usually not a problem.
Water-borne	Not usually a problem, as heavily controlled by effluent treatment consent standards.	Often a problem after flooding of voids complete, or where tailings not capped.

(Source: 'Proceedings of a seminar on proposed EU directives on mining waste', Burlington House, London, 2001).

The implication arising from the above is that consideration, related to the migration pathways and affects of tailings contaminants, must be given not only during the mining process but mostly during the use and closure of the tailings storage facilities, where long-term pollution indications may be observed.

Regarding the alumina tailings contaminants, studies have shown that the most common environmental aspects that must be considered during the planning of a tailings management facility is the potential of alkaline seepage and, also, the potential of surface water overflows of the red mud ponds. In cases where the dry stacking method of red mud disposal is used, as well during the drying phase of red mud ponds, the potential of wind erosion is likely to

occur, especially in areas where the climate favours strong winds. These environmental aspects concerning potential migration pathways of red mud contaminants are discussed below.

Alkaline seepage

It is generally recognised and accepted universally that seepage from a tailings storage facility is an inevitable consequence of the deposition of tailings slurries. However, it does not imply that seepage is acceptable and this is because uncontrolled seepage from a tailings storage facility may become a significant environmental issue as it may affect the surrounding environment.

The common problem of a tailings management facility handling the alumina residue is the, so-called, alkaline seepage. Alkaline seepage occurs, mostly, in wet disposal methods and results by the percolation of the caustic residue (sodium), which is characterised by pH values up to 9. If the base of an impoundment extends into the saturated zone then shallow ground water contamination is possible. Potential migration of the alkaline liquor of red muds, under hydraulic pressure, may, therefore, have serious impacts in groundwater and in aquifers [34]. Alkaline seepage impacts include damage to vegetation as a result of lateral seepage through the starter impoundment or through the intermediate raise embankments. Vertical seepage may contaminate the soils and groundwater through chemical alteration due to the alkaline nature of red muds.

Unlike the wet disposal method of alumina residue, dry stacking of red muds does not pose any serious risk of alkaline seepage. At Ewarton Works in St. Catherine, Jamaica where the dry stacking method for the disposal of red mud is used, seepage of contaminated water is virtually eliminated. Standing pondwater, which exerts the most hydraulic pressure, is confined to small areas such as the toe areas of the facility and the holding pond [92].

Surface run-off water

Constituent of potential concern in the red mud disposal facilities comprises the surface run-off water. Surface water releases from tailings dams are, usually, caused by storm water of heavy rainfalls and the impacts of such an incident are closely related to the high alkalinity of the muds, as leaching of alkaline water is possible to occur. If the receiving water into the impoundment is not well-buffered, its pH could exceed levels that are protective of aquatic life. Alkaline water may, also, have low resource value due to its corrosive properties [34]. Any migration of mud contaminated surface water is likely to pose an ecological threat regarding the distance between the tailings facility and the drinking water intakes. Therefore, the consideration of nearby water bodies is of importance.

Unlike the ground water discharges, where low level chronic loads of alkaline contamination are expected, pH excursions in surface waters are more likely to be caused by periodic direct discharges. A damage case regarding surface water release was reported at Ormet

Corporation's Aluminium plant in May 1983. The plant is located south of Rouge in Burnside and the facility is situated near the Mississippi River. During heavy rainfall events, discharge of excess water has resulted in some cases of high pH excursions. For example, excess water was discharged to the Panama Canal. Due to improper operation of the neutralisation station, combined with communication problems, high pH excursions were not detected until after the discharge event. The excessive pH levels ranged from 9.4 to 9.8 [94]. The emergency discharges to the Panama Canal imparted a red colour to the canal water, resulting in complaints from local residents [77].

5.4.4 Wind erosion

Red mud contaminants of the tailings storage facilities can, also, be released to the environment in the form of windblown dust. During the operational phase of the impoundments, the potential for dusting from the muds exists mostly in arid areas, where the mud is less likely to remain moist and particularly in the case of the dry stacking method of disposal, where the contaminants can easily be transferred and overlaid on vegetation or be inhaled by human and animals.

The muds dry to a very fine particle size which sometimes may be less than 1 micrometer in diameter [34]. This characteristic, together with the caustic soda presence, makes red muds highly susceptible to wind erosion. In Jamaica it has been observed that a considerable amount of alumina is spilled on clouds of dust being carried downwind from loading equipment [107]. Although it has been argued that the dust is chemically inert, it does adversely affect the respiratory system, pollute the residential cisterns and deface property. Therefore, regular water spray of stockpiles and re-vegetation of tailings surfaces, as soon as they are no longer in use and their material has the ability to support vegetation, must be applied for preventing wind erosion.

5.4.5 Environmental Impacts of TMFs containing red mud

Impacts on surface and ground water

As described in previous sections alumina red mud tailings are not of a toxic nature. However, they are potentially significant sources of pollution due to the high sodium residue that they contain and consequently the characteristic high values of alkalinity. The eventual discharge of surface run off, produced by rainfall, is one mechanism by which pollutants are released into surface waters. Other mechanisms of contaminant releases include seepage from the impounded areas and indirectly releases of pollutants via ground water that has a hydrological connection to surface water [81]. According to this mechanism, environmental impacts of contaminants on surface and ground water may be closely related to each other.

Impacts on surface water involve high pH levels that contaminate the drinking water supplies and accordingly affect human and animal health. Indeed, studies conducted in Jamaica bauxite

mining have shown that the Jamaican population is particularly subjected to hypertension, which can be aggravated by high levels of sodium [64].

It should be noted that seepage assists in drying the tailings in the impoundment, particularly after the abandonment. In the dry climate of Western Australia, seepage has been recorded as having caused the groundwater table to rise, bringing saline water closer to the surface and damaging vegetation [126].

Generally, ground water impacts in mining are not as widespread as surface water impacts because of the much slower velocity of ground water movement, the more limited extent of many affected aquifers and the lack of available oxygen to continue the oxidation process [81]. Nevertheless, ground water contamination once it occurs makes it a serious concern. Red mud releases of contaminants may cause disturbance in the ground water flow regime affecting the quantities of water available for local use. Contaminated water may, also, recharge surface water down gradient of the tailings storage facility through contributions to base flow in a stream channel of springs.

Impacts on soils

Human health and environmental risks from soils generally fall into two categories; contaminated soil resulting from windblown dust and soils contaminated from chemical spills and residues. In the case of alumina tailings management facilities, the most significant parameter that affects soils is the high alkalinity of contaminants that characterizes the soils as 'alkaline'.

Alkaline soils are soils with a high pH (>9), a poor soil structure and a low infiltration capacity [93]. Soil alkalinity around red mud tailings disposal areas is associated with the presence of sodium carbonates generated by the Bayer process. The sodium carbonate when dissolved in water dissociates into 2Na^+ and CO_3^{2-} . The sodium carbonate can react with water to produce carbon-dioxide (CO_2), escaping as a gas, and sodium-hydroxide ($\text{Na}^+ \text{OH}^-$), which is alkaline and gives high pH values.

Problems associated with the high alkalinity of soils include the difficulty to be taken into agricultural production. In addition, rain water stagnates on the soil easily and in dry periods irrigation is hardly possible. Agriculture is limited to crops tolerant to surface water logging (e.g. rice, grasses) and the productivity is low. Particular awareness must be given to the ability of restoration of red mud tailings due to soil contamination. In 1991, the International Aluminium Institute has carried out the first Bauxite Mine Rehabilitation Survey which was intended to provide for the first time a collection of definitive data on a wide range of key topics relating to the environmental impacts of bauxite mines and their rehabilitation. The survey described the impact of programmes in place at 18 mining locations, representing 65% of total world bauxite production. The survey has since been repeated (in 1998) and the latest

report, published in August 2000, covers 27 mining locations, representing 72% of the total world bauxite production.

The results of both these surveys indicate that the bauxite mining industry is broadly in harmony with the approach set out in the United National Environment Programme (UNEP) "Guidelines for the Environmental Management of Alumina Production". Of the land disturbed each year by bauxite mining, 76% is forested, 19% agricultural and pasture and 2% scrubland. Post-mining land use shows 70% being returned to native forest, 3% to commercial forest, 17% to agriculture and pasture and 7% used for urban and industrial development, housing and recreational purposes. Rehabilitation plans are in place at 90% of the surveyed mines (compared to 55% in 1991) and the current rehabilitation rate is 1256 hectares per year [59].

Impacts on air

Wind entrains dust from red mud tailings, either dry as disposed or the dry portions of the impoundments. Thereby, tailings can be a potent source of fine particles that may consequently be inhaled by humans and animals, be ingested by surface water and affect soils and vegetation. Health risks posed by inhaled red mud dust particles are influenced by both the penetration and deposition of particles in the various regions of the respiratory tract and the biological responses to these deposited materials. When inhaled, these particles are deposited in the trachea and bronchia section of the lung. Silica dust, often suspended near the red mud storing facilities, has been a widespread problem in the past for workers in mining industries. Silica is non-toxic and safe except where specific crystalline forms of silica occur in dust. Long term inhalation of silica dust may lead to the formation of scar tissue in the lungs and can result in 'silicosis', a serious and life threatening lung disease [33].

Damage to vegetation and agriculture, due to the alkalinity of dust, is possible through mechanisms such as the blocking of leaf stomata, the inhibition of gas exchange and the reduced photosynthesis due to smothered surfaces. Contaminated dust may, also, be deposited on surface water and migrate through it onto agricultural areas, harming the vegetation or rendering the soil desolate.

Impacts on fauna and flora

Further impacts associated with red mud tailings during the operational phase of the tailings involve the effects on fauna and flora. By its very nature, it is known that mining causes disturbances to the natural habitat that is to the aquatic resources, wildlife, vegetation and wetlands. By the time the operations of tailings management facilities affect the surface and ground water and cause serious damage to soils and air, it is inevitable that they affect fauna and flora as well. Accordingly, red mud contaminants when existing in excessive amounts may

destruct the ecosystem. This fact requires the necessity of rehabilitation planning in order to restore the natural habitat as effectively as possible.

Alcoa of Australia Limited has mined and restored the Jarrah forest of Southwest Australia since the late 1960s. The goal of rehabilitation is to re-create the forest ecosystem that existed before mining occurred. Long-term monitoring of flora and fauna has shown that mine sites retain a majority of the qualities of the surrounding forest, which can be an indicator of restoration success. Bauxite mining is an intensive process that destroys all aspects of the ecosystem. Efforts by Alcoa of Australia Limited show that quality ecosystems can be reproduced after a severe human disturbance [114]. Another study of Alcoa of Australia Ltd has shown that bird communities in rehabilitated sites in Australia varied in their successive development according to the techniques used after mining [8].

However, other studies conducted at the reforestation area at a bauxite-mined site at Porto Trobetas in western Para' State in Brazil have shown that several species living in the forest before the mining operations, are now rare since reforestation [95]. The same study suggests that while the reforestation program has been successful in creating a favourable environment for regeneration of a native primary forest species, further management interventions such as enrichment plantings, may be required to accelerate regeneration of large-seeded primary forest species.

Regarding the ocean dumping of red muds it has been detected that this method of disposal is potent to harm the benthic fauna, although little is known about deep sea organisms. Indeed, biological studies have revealed that the damage caused to benthic invertebrate faunas by alkaline, ferruginous discharges (such as red muds) is generally as severe as that caused by acidic mine water discharges [65]. This is because smothering of the benthos with iron hydroxide precipitates prevents photosynthesis and therefore locally removes the foundations of the trophic web, i.e. the 'food chain'.

5.5 Tailings containing radioactive substances

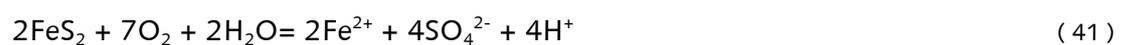
5.5.1 Characteristics of uranium mill tailings

Uranium mill tailings are normally dumped as sludge in special ponds or piles, where they are abandoned. The amount of sludge produced is nearly the same as that of the ore processing. At a grade of 0.1% uranium, 99.9% of the material is left over. Apart from the portion of the uranium removed, the sludge contains all the constituents of the ore. As long lived decay products such as thorium-230 and radium-226 are not removed, the sludge contains 85% of the initial radioactivity of the ore. Due to technical limitations, all of the uranium present in the ore can not be extracted. Therefore, the sludge also contains 5% to 10% of the uranium initially present in the ore.

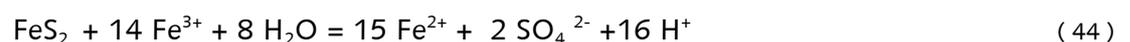
In addition, the sludge contains heavy metals and other contaminants such as arsenic, as well as chemical reagents used during the ore processing. The main radiological, chemical, and mineralogical characteristics of tailings which need to be assessed when considering the environmental effect of the particular tailings are the acid generating potential, chemical precipitate containing in the tailings and chemical form of existing compounds, uranium ratio of the original ore for determining the radioactivity of tailings. Some of them are described in detail in descriptions before. Here is a short summary.

Uranium ores very often contain sulphide minerals especially pyrite and have therefore an acid generation potential. These minerals are regarded as reactive minerals and are susceptible to bacterial oxidation in the presence of moisture and oxygen, forming sulphuric acid according to the equations [48]:

Direct oxidation:



Indirect oxidation:



The production of sulphuric acid can result in elevated concentration of toxic heavy metals. Radionuclides appear sometimes in discharge waters, including seepage because the pyrite usually contains these elements or there is a reduction in the pH-value of adjacent water systems and disruptive effects on the ecology. If acid leachate is treated, large quantities of sludge may be formed which therefore produces an additional waste disposal problem.

This problem is significant and can be reduced if removal of pyrite during processing takes place. For prediction of the possible acid generation, the neutralisation potential (mainly carbonate content) also has to be considered. This is especially so in the case of alkaline leaching, where the residues can contain some amount of carbonate. For example, in dry areas, salts containing contaminants can migrate to the surface of the pile, where they are subject to erosion. If the ore contains the mineral pyrite (FeS_2), then sulphuric acid forms inside the deposit which can be accessed by precipitation and oxygen. This acid causes a continuous automatic leaching of contaminants.

Uranium grade ore in composition with other radiological components that has been processed in past decades ranged from less than 0.1% to several percent and the average mill tailings radioactivity reflects these original uranium concentrations. As during the ore processing essentially only uranium is removed, the tailings usually contain all decay products from the mother elements of uranium. The decay series for uranium and thorium (which usually also present in the ores to some degree) is illustrated in Figure 55. For each 0.1% of uranium present, the virgin ore has about 12.4 Bq of each member of the ^{238}U series per gram of ore or about 174 Bq of total activity per gram.

Among the decay products radium-226 is considered to be the most important radiotoxic element. The radiotoxicity is connected with the element itself as well with the production from it of radon-222, a radioactive inert gas whose decay products can cause lung cancer. Radium itself tends to deposit in bone [49]. Radon-222 gas emanates from tailings piles and has a half life of 3.8 days. This may seem short, but due to the continuous production of radon from the decay of radium-226, which has a half life of 1600 years, radon presents a long term hazard. Further, because the parent product of radium-226, thorium-230 (with a half life of 80,000 years) is also present, there is continuous production of radium-226.

²³⁸ U series		²³⁵ U series		²³² Th series	
Nuclide	Half-life	Nuclide	Half-life	Nuclide	Half-life
²³⁸ U ₉₂	4.51 × 10 ⁹ a	²³⁵ U ₉₂	7.1 × 10 ⁸ a	²³² Th ₉₀	1.41 × 10 ¹⁰ a
↓		↓		↓	
²³⁴ Th ₉₀	24.1 d	²³¹ Th ₉₀	25.6 h	²²⁸ Ra ₈₈	5.8 a
↓		↓		↓	
²³⁴ Pa ^m ₉₁	1.17 min	²³¹ Pa ₉₁	3.4 × 10 ⁴ a	²²⁸ Ac ₈₉	6.13 h
↓		↓		↓	
<div style="display: flex; justify-content: space-between;"> 99.87% 0.13% </div> <div style="border: 1px solid black; width: 100%; height: 20px; margin: 5px 0;"></div> <div style="display: flex; justify-content: space-between;"> ²³⁴Pa₉₁ ²³⁴Pa₉₁ </div>	6.75 h	²²⁷ Ac ₈₉	21.6 a	²²⁸ Th ₉₀	1.910 a
↓		↓		↓	
²³⁴ U ₉₂	2.47 × 10 ⁵ a	²²⁷ Th ₉₀	18.6 d	²²⁴ Ra ₈₈	3.64 d
↓		↓		↓	
²³⁰ Th ₉₀	8.0 × 10 ⁴ a	²²³ Ra ₈₈	11.7 d	²²⁰ Rn ₈₆	55 s
↓		↓		↓	
²²⁶ Ra ₈₈	1602 a	²¹⁹ Rn ₈₆	3.9 s	²¹⁶ Po ₈₄	0.15 s
↓		↓		↓	
²²² Rn ₈₆	3.823 d	²¹⁵ Po ₈₄	1.8 × 10 ⁻³ s	²¹² Pb ₈₂	10.64 h
↓		↓		↓	
²¹⁸ Po ₈₄	3.05 min	²¹¹ Pb ₈₂	36 min	²¹² Bi ₈₃	60.6 min
↓		↓		↓	
<div style="display: flex; justify-content: space-between;"> 99.98% 0.02% </div> <div style="border: 1px solid black; width: 100%; height: 20px; margin: 5px 0;"></div> <div style="display: flex; justify-content: space-between;"> ²¹⁴Pb₈₂ ²¹⁸At₈₅ </div>	26.8 min	²¹¹ Bi ₈₃	2.2 min	<div style="display: flex; justify-content: space-between;"> 64.0% 36.0% </div> <div style="border: 1px solid black; width: 100%; height: 20px; margin: 5px 0;"></div> <div style="display: flex; justify-content: space-between;"> ²¹²Po₈₄ ²⁰⁸Tl₈₁ </div>	304 ns
↓		↓		↓	
²¹⁴ Bi ₈₃	19.7 min	²¹¹ Po ₈₄	0.5 s	²⁰⁸ Pb ₈₂	Stable
↓		↓			
<div style="display: flex; justify-content: space-between;"> 99.98% 0.02% </div> <div style="border: 1px solid black; width: 100%; height: 20px; margin: 5px 0;"></div> <div style="display: flex; justify-content: space-between;"> ²¹⁴Po₈₄ ²¹⁰Tl₈₁ </div>	164 μs	²⁰⁷ Pb ₈₂	Stable		
↓					
²¹⁰ Pb ₈₂	13.8 min				
↓					
²¹⁰ Bi ₈₃	5.01 d				
↓					
<div style="display: flex; justify-content: space-between;"> ~100% 0.00013% </div> <div style="border: 1px solid black; width: 100%; height: 20px; margin: 5px 0;"></div> <div style="display: flex; justify-content: space-between;"> ²¹⁰Po₈₄ ²⁰⁶Tl₈₁ </div>	4.19 min				
↓					
²⁰⁶ Pb ₈₂	Stable				

Figure 55: Radioactive decay series for uranium and thorium [51]

After about 1 million years, the radioactivity of the tailings and thus its radon emanation will have decreased so that it is only limited by the residual uranium contents, which continuously produces new thorium-230 (Figure 56).

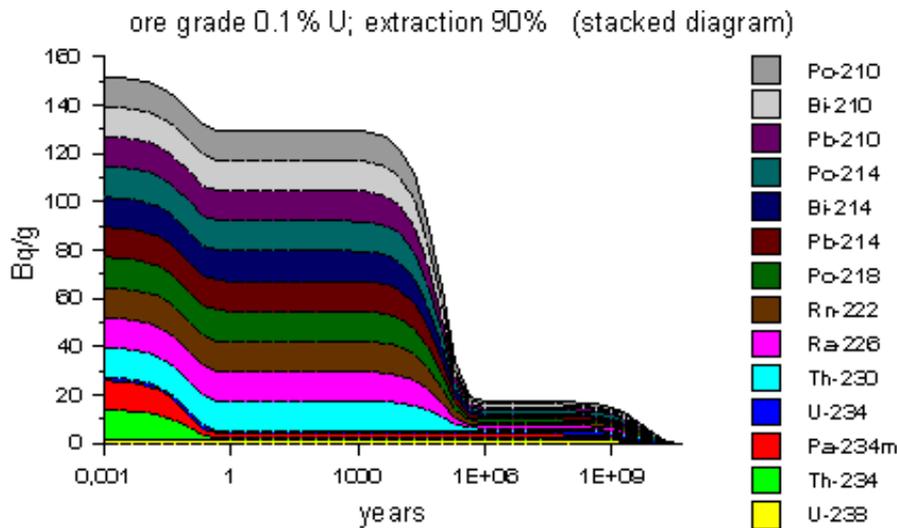


Figure 56: Uranium mill tailings activity [31]

Radon releases are a major hazard that continues after uranium mines are shut down. The U.S. Environmental Protection Agency (EPA) estimates the lifetime excess lung cancer risk of residents living nearby a bare tailings pile of 80 hectares at two cases per hundred. Since radon spreads quickly with the wind, many people receive small additional radiation doses. Although the excess risk for the individual is small, it cannot be neglected due to the large number of people concerned. EPA estimates that the uranium tailings deposits existing in the United States in 1983 would cause 500 lung cancer deaths per century, if no countermeasures are taken [31].

210 Pb lead also an important decay product, both because of its radiotoxicity and because of the mobility of its subsequent daughter element 210 Po. Therefore the contribution to the dose of these two radionuclides should not be overlooked. Uranium recovery from the ores leads to the removing app. 90-95% of uranium and at the same time 15% of total radioactivity. Therefore the tailings initially contain about 85% of radioactivity originally present in the equilibrium ore but this activity rapidly decreases to 70% as short lived daughters such as 234Th decay. The radium-226 concentration in the process solution before treatment may vary from about 10 to 200 Bq/l.

In acid solution more than 50% of thorium-230 may be dissolved and could be of equal radiological importance because radium-226 is a decay product. Neutralisation of tailings solution decreases the thorium-230 concentration significantly. Although the radioactivity of tailings are relatively low in most tailings, some radiological hazard will last virtually forever because of the long half-lives of the radionuclides involved (thorium-230 84 thousand years, remained uranium 4.5 billion years, etc.).

If, for example, 90% of the uranium contained in an ore with 0.1% grade was extracted during the milling process, the radiation of the tailings stabilizes after 1 million years at a level 33

times that of uncontaminated material. Due to the 4.5 billion year half-life of uranium-238, there is only a minuscule further decrease. Potential uranium mill tailings hazards are presented in Figure 57.

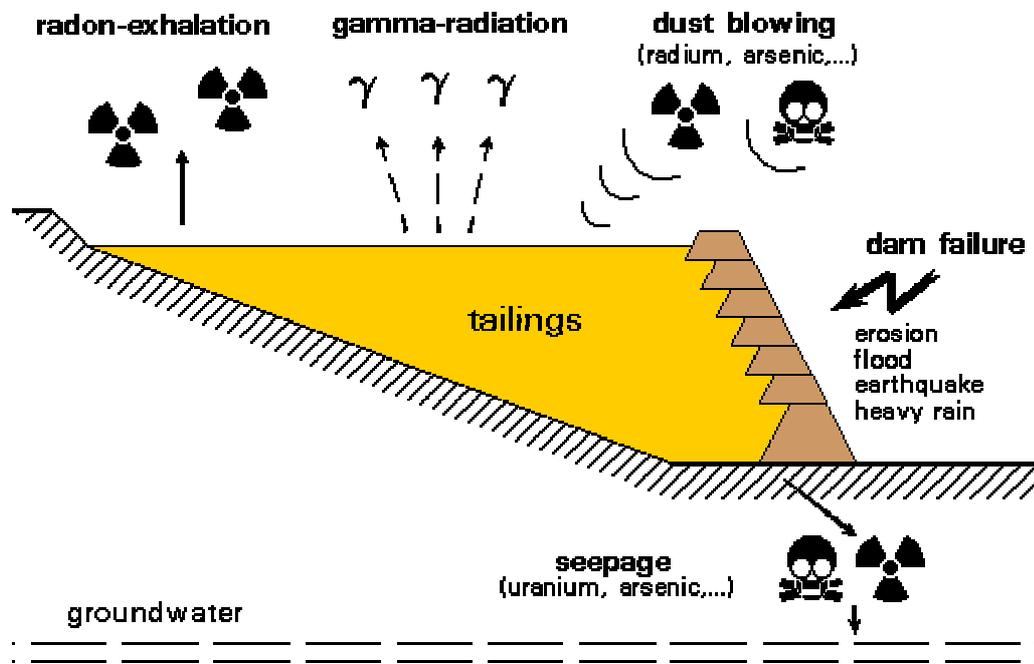


Figure 57: Uranium mill tailings hazards [31]

Radionuclides contained in uranium tailings emit 20 to 100 times as much gamma-radiation as natural background levels on deposit surfaces. Gamma radiation levels decrease rapidly with distance from the pile.

Tailings are contaminated with different non-radioactive constituents such as arsenic and other heavy metals as listed in Figure 58. Also a large number of non-radioelements can be present in the ores, which can also cause environmental problems. These elements are listed in Figure 58. Some pollutants originate from the process itself. These elements and compounds can leave the impoundment so therefore they have to be controlled in water samples. In the process of ore treatment, different precipitates can be formed. Perhaps the most important is the gypsum which is formed during leaching and partly during the neutralisation of tailings and tailings water treatment. The gypsum concentrates in the fines, therefore it can be found in the slime core if hydrocyclon discharge technology is used. It is important to underline that during the neutralisation of tailings, different hydroxides (iron, aluminium, magnesium etc.) are formed, which can increase the volume of tailings and hinder their consolidation process.

Arsenic	Barium	Boron	Cadmium
Chromium	Copper	Iron	Lead
Manganese	Mercury	Molybdenum	Nickel
Nitrate	Radium	Selenium	Silver
Thorium	Uranium	Vanadium	Zinc
Ammonia	Carbonates	Chlorides	Cyanide
Di(2-ethylhexyl) phosphoric acid		Sulphyric acid	Isodecanol
Kerosene	Pyrite	Pyrrhotite	Tertiary amines
Tributyl phosphate			

Figure 58: Heavy metals and other components in tailings

Mining and processing removes these hazardous constituents in the ore from their relatively safe underground location and converts them to a fine sand and then sludge, whereby the hazardous materials become more susceptible to dispersion in the environment. Moreover, the constituents inside the tailings pile are in a geochemical disequilibrium that results in various reactions causing additional hazards to the environment.

5.5.2 Overview of contaminants and their impact on humans and nature

Tailings deposits are subject to many kinds of erosion. Due to the long half-lives of the radioactive constituents involved, depositional safety has to be guaranteed for very long periods of time.

After rainfall, erosion gullies can appear, floods can destroy the whole deposit, plants and burrowing animals can penetrate into the deposit and thus disperse the material, enhance the radon emanation and make the deposit more susceptible to climatic erosion. When the surface of the pile dries out, the fine sands are blown by the wind over adjacent areas. Villages located in the immediate vicinity of uranium mill tailings piles have been threatened by radioactive dust storms.

Due to wind distribution, these dust particles represent a physical pollution. Particulate matter is monitored in the United Kingdom as PM_{10} : particles generally less than 10 microns in diameter. A large number of epidemiological studies have shown that day to day variations in concentrations of particles are associated with adverse effects on health. Because of the continuous relationship between concentrations of particles and effects on health, a different approach to devising bands of air quality has been advised by Committee on the Medical Effects of Air Pollutants (COMEAP).

The following break points between bands were agreed [23]:

- less than 50 µg/m³ (running 24 hour average): "low" levels of air pollution.
- 50-75 µg/m³ (running 24 hour average) : "moderate" levels of air pollution.
- 75-100 µg/m³ (running 24 hour average): "high" levels of air pollution
- more than 100 µg/m³: "very high" levels of air pollution.

Seepage from tailings piles is another major hazard. Seepage poses a risk of contamination to ground and surface water. Residents are also threatened by radium-226 and other hazardous substances like arsenic in their drinking water supplies and in locally caught fish. The seepage problem is very important with acidic tailings, as the radionuclides involved are more mobile under acidic conditions. In tailings containing pyrite, acidic conditions automatically develop due to the inherent production of Sulfuric acid, which increases migration of contaminants to the environment.

Tailings dams are often not of stable construction. In most cases, they were made from sedimentation of the coarse fraction of the tailings sludge. Some, including those of Culmützsch and Trünzig in Thuringia, were built on geological faults and are therefore subject to earthquake risk. As the Thuringian tailings deposits are located in the centre of an area of earthquake risk in the former GDR, they suffer a risk of dam failure. Moreover, strong rain or snow storms can also cause dam failures [31].

Occasionally, because of their fine sandy texture, dried tailings have been used for construction of homes, Figure 5, or for landfills. In homes built on or from such material, high levels of gamma radiation and radon were found. The U.S. Environmental Protection Agency (EPA) estimates the lifetime excess lung cancer risk of residents of such homes at 4 cases per 100.

As a consequence of the ionizing radiation impact on human body, a long series of phenomena are released. This begins with ionization and excitement of the atoms and molecules from the interaction medium. These physical perturbations stimulate physio-chemical reactions followed by chemical reactions, ending with biological effects. Roughly, these biological effects can be classified in non-stochastic effects and stochastic effects [79], [80].

Non-stochastic effects are immediate (precocious) effects with thresh hold reversible and imperfect additive effects. With a view to their avoidance, the divided administration of the radiation doses has great importance. Among these effects are: malignant lesions of the skin, cataract, medullary hipolesia which stimulates a hematological deficiency and damages gonad cells with repercussions on fertility [79]. There are always late somatic effects without thresh hold, irreversible and additive effects. Their appearance follows a probabilistic dose-effect relation, apparently at random, whence the name of stochastic.

Among them there are the somatic effects which appear later as malignant diseases (cancer) and genetic effects (hereditary) which appear in descendants. In practice, the time between irradiation and the appearance of malignant disease can be years or even tens of years. The doses administered on long term being additive, it is not important if the dose produces the effect in a second or in a month. The biological effect produced is additive and at present there are not probability reduction methods for the appearance of the stochastic effects due to some doses already received.

The main objective, in the case of industrial activities of non-nuclear (but including associated radioactive elements) raw material processing, consists of the assessment of supplementary doses received due to internal irradiation and to the external ones which could be accidental or permanent. Because people live in a geographic area characterized by a well defined geological substratum and by different altitude, these facts lead to an individual effective annual dose which varies from one zone to another due to the natural background.

Both international and Romanian national (issued in August 2000) radio-protection norms allow that for an individual person, an additional dose "caused" by all the natural sources via all the pathways from the region in which he lives, 1 mSv/year, which has to be added to the effective dose due to the natural background of the investigated area.

Table 9: Average annual effective dose (mSv/year) for the whole population – natural irradiation [128]

Source	External Dose	Internal Dose	Total
Cosmic Radiation – direct ionizing	0.30	-	0.30
Cosmic Radiation - neutrons	0.055	-	0.055
Cosmogenic radionuclides	-	0.015	0.015
Potassium – 40	0.15	0.18	0.33
Uranium – radium (total) series	0.1	1.24	1.34
From which – radon and its daughters	-	(1.1)	-
Thorium series (total)	0.16	0.18	0.34
From which – thoron and its daughters	-	(0.16)	-
TOTAL	0.8	1.6	2.4

Table 10: Average annual effective dose in Romania [105]

Pathway	Source	External Dose	Internal Dose	Total
Terrestrial	Cosmic radiation – direct ionizing	0,25	-	0,25
	Cosmic radiation - neutrons	0,03	-	0,03
Aquatic	Uranium, radium and thorium series	0,46	1,2	1,66
Air	From which – radon and thoron with their daughters	-	(1,1)	-
	Other radionuclides	-	0,23	0,23
	TOTAL	0,74	1,43	2,17

The estimated value of the annual effective dose (now named EDE = Equivalent of Effective Dose), mediated on the whole planet due to the natural background is 2.4 mSv/year [128]. From this value, 1.1 mSv/year is due to radon (Rn222) and its daughters with short half life time, 0.16 mSv/year is due to thoron (Rn²²⁰) and its daughters, 0.35 mSv/year is due to the cosmic radiation and 0.33 mSv/year is due to K⁴⁰.

The methodology of total effective dose followed the recommendations of the international organisations, such as: International Atomic Energy Agency (IAEA) [50]; International Commission for Radioprotection (ICRP); UN Scientific Committee for Effects of Atomic Radiations (UNSCEAR) and Manual for Radioactivity Measurement (WISMUT – PHARE Project).

The total effective dose has two components:

- The external effective dose due to gamma external irradiation – it is experimentally determined by directly measuring the dose debit in the place where it is or it could be the individual for whom the investigation is performed;
- The internal effective dose due to the radionuclides inhalation and ingestion – it is indirectly determined if the radioactive concentration is known for U, Ra and Rn, and the incorporated quantity of the "substances" entering the organism: air, water, food (milk, meat, fish, etc.)

The annual effective dose E_T is the sum of all the doses obtained (received by a person) via the following pathways transfer from the source to the "target":

- Terrestrial pathway
- external gamma irradiation – external effective dose E
- internal irradiation by ingestion of contaminated "materials" (soil, mud, fines from the waste dump, etc.) – internal effective dose E_{ci}
- Aquatic pathway
- internal irradiation by ingestion:
 - directly: water
 - indirectly: milk, butter, meat, fish, vegetation (rutty plants, herbaceous)
- internal effective dose E_{ai}
- Aerial pathway
- internal irradiation by inhalation of radioactive dust with long life daughters (U, Th, Ra, Po) – internal effective dose E_{hi}
- internal irradiation by inhalation of radon and short life daughters – internal effective dose E_{hRn}

Total annual effective dose E_T (named until now EDE = effective dose equivalent) is composed of:

$$E_T = E + E_{ci}^c + E_{ci}^a + E_{hi} + E_{hRn} \quad (45)$$

where:

- E = external effective dose
- $E_{ci}^c + E_{ci}^a$ = effective dose by ingestion
- $E_{hi} + E_{hRn}$ = effective dose by inhalation

Effective dose by ingestion + Effective dose by inhalation = Internal effective dose. From the total annual effective dose E_T subtracting the effective dose due to the natural background $E_{T_{fond}}$ the total supplementary effective dose obtained is $E_{T_{sup}}$ received by a person via all the pathway transfers from source to the target.

$$E_{T_{sup}} = E_T - E_{T_{fond}} \quad (46)$$

Natural irradiation – supplementary products are, like any other irradiation, a risk to peoples health by biological effects which can be divided into:

- determined effects which appear from a certain value of the dose;
- stochastic effects from which the effect could possibly be determined by the appearance.

The main task of the risk studies is to identify the supplementary irradiation sources from the investigated area and to assess the irradiation of organisms due to the supplementary component which is not allowed to exceed an effective dose of 1 mSv/year.

5.5.3 Mineral resources which after processing could produce tailings with radioactive contaminant content

Uranium mineral resources waste resultant from uranium ore processing depends on these ore characteristics:

- ore with/without organic matter content
- ore with carbonated components
- ore with argillaceous components.

In order to determine the characteristics of the tailings, samples were collected from the tailings pond surface, resulting a U and Ra content specific for each grain size class (Table 11).

Table 11: Distribution of U and Ra and granulometry class [57]

Class	Percentage (g,%)	U%	Ra (Bq/g)	Class Matrix
+ 500 µm	0.07	1.0	120	37% bitumen, 60% resin, 2% wood slivers, 1% quartz
+120 µm	7.0	0.13	10	31% bitumen, 2% resin, 32% mica, 3% quartz, 1% wood silvers, 24% conglomerate.
+ 63 µm	30.13	0.0190	4.2	2% bitumen, 5% mica, 75% quartz + feldspar, 18% carbonate.
+ 45 µm	20.10	0.0215	20	Lack: bitumen, resin, wood silvers, prevail: argillaceous conglomerates with minerals: kaolinite 22%, illite 18%, montmorillonite 12%, feldspar, quartz, carbonate.
- 45 µm	42.7	0.0130	8	
Total	100 %	0.0250	9.4	

The results obtained are:

SiO ₂ = 12.8%	Al ₂ O ₃ = 14.3%	Fe ₂ O ₃ = 3.45%	CaO = 6.2%	MgO = 0.9%
K ₂ O = 2.3%	Na ₂ O = 5.7%	Mn = 0.2%	Pb = 0.010%	Cu = 0.030%
Zn = 0.028%	Co = 0.003%	Sn = 0.0005%	Ni = 0.004%	V = 0.0120%
Mo = 0.0008%	S = 0.34 %	org.matter=0.7%	U = 0.025%	Ra = 9.4 Bq/g
Humidity = 22 %				

In a statistical interpretation in U - Ra coordinates (Figure 59) of the results obtained both for the tailings and for the granulometry fractions, a lack of equilibrium is to be noted towards Ra in tailings and granulometry fractions in which, depending on their composition, discrepancies are high.

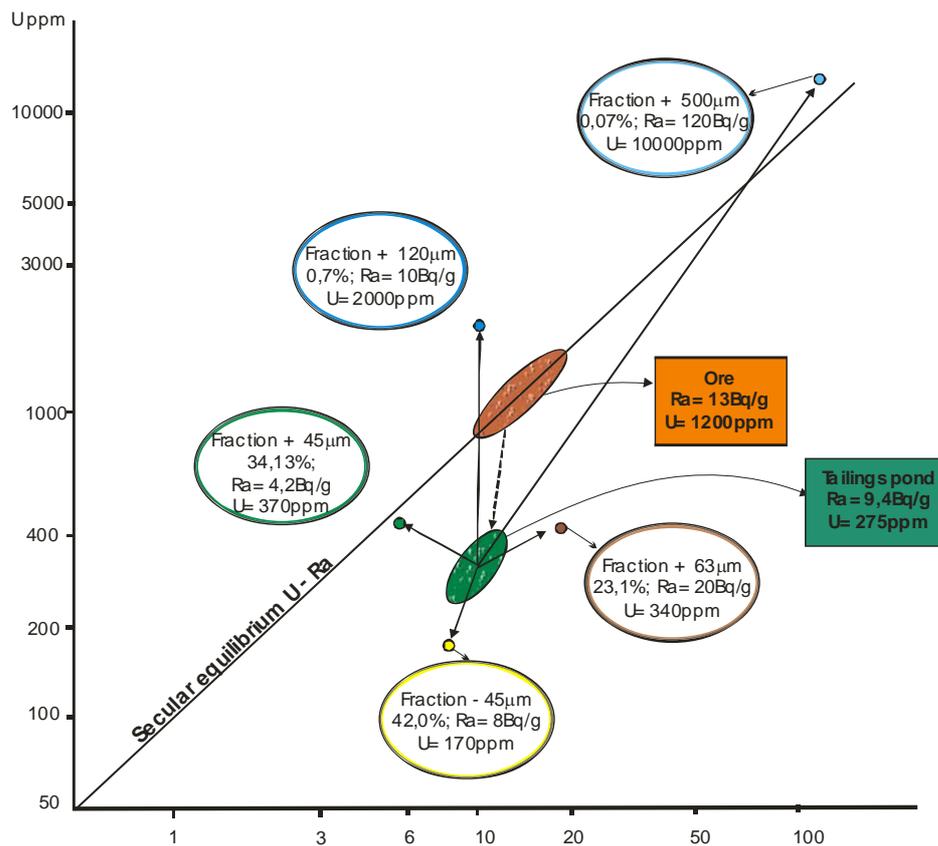


Figure 59: U - Ra correspondency diagram [57]

The ratio Ra/U is a very important metrological factor for irradiation potential assessment and for correct interpretation of dosimetric measurements. Because of the Ra content, the Rn release into the environment lasts for a long time.

Due to their radioactive element content, their fine texture and due to the lack of equilibrium towards Ra, the uranium ore processing tailings ponds present the highest risk from the whole uranium mining industry. The global activity of the materials from the tailings ponds is about 1100 GBq.

For radiometric characterization of the tailings pond surface, measurements must be performed in a defined grid system. These Rn-measurements are to be made on the tailings pond surface and also in holes dug some 20-40 cm deep on the beach (Figure 60). A median gamma dose debit of 1.2 $\mu\text{Sv/h}$ is measured in the water at a depth of 5 cm. At a depth of 25 cm, this decreases to 0.4 $\mu\text{Sv/h}$. The highest Ra content is in the fine fractions representing $\sim 92\%$ of the tailings.

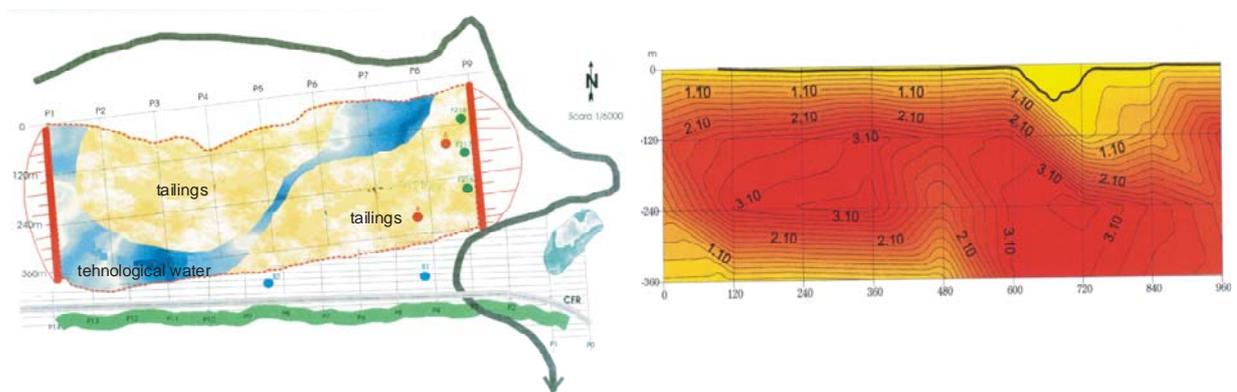


Figure 60: Cetatua tailings pond (Romania) – grid system for gamma dose measurement (left) and gamma dose debit distribution

Furthermore phosphatic ore, aluminium ore and coal processing have the potential to produce tailings with radioactive contaminants. Many of the above categories haven't been considered such in industries with a potential radiological impact on people or environment.

The *natural phosphates* from sedimentogene genesis are radioactive as U is in equilibrium (there could be exceptions) with Ra^{226} , while magmatic nature phosphates have low U and Ra contents, thorium predominating. The phosphatic rock world production is about 130 million tones, the main producers being: China, Morocco, Russia, USA and Tunisia. In Romania, 7 fertilizer plants based on H_2SO_4 and HNO_3 attack, were built and commissioned in the period 1964-1978.

Using H₂SO₄ – attack, two products resulted:

- phosphoric acid (liquid phase) where more than 90% of uranium from the rock is found;
- gypsum usually named phosphogypsum is the solid phase which may also carry 80% of the Ra-226 initial existent in the phosphate rock. Phosphogypsum may also carry 5-20% of uranium from the rock.

Phosphoric acid, the intermediate product, is separated from phosphogypsum and is used in fertilizer manufacture. Such a fertilizer includes all uranium of phosphoric acid and that existent in the phosphate rock used in the neutralization process as well as the whole ²²⁶Ra in the rock. Therefore the TSP fertilizer from sedimentary rock has an average 80-140 mg/kgU and 300-700 Bq/kg Ra²²⁶. If Kola phosphate rock has been used in the phosphoric acid neutralization then no radium was carried by TSP.

The alternative is that phosphoric acid is neutralised with ammonia, the fertilizer usually produced is DAP (diammoniumphosphate). The alternative of DAP is that the radioactivity involved is from phosphoric acid and no radium is present. If uranium is eliminated and recovered by solvent extraction, then TSP uranium content is minimized and in the case of DAP, no radioactivity is present. However there are also natural phosphates with much higher radioactivity resulting from uranium phosphates content which may be 20-30 times higher than in commercial phosphates.

In this case, radium follows a similar end and its high radioactivity in phosphogypsum is a great problem. Some literature data on uranium content in the phosphate rock is given in Table 12.

Table 12: Phosphate deposits of high uranium content

Phosphate rock used	Uranium mg/kg
Brazil	1200
Central African Republic	1600-5600
Siberia (depressions)	up to 4000
Utah (USA)	up to 3000
India	up to 800

The high uranium content of phosphate rock is a health hazard and the radioactivity must be removed from phosphoric acid before producing a fertilizer. In this alternative, the TSP production must be avoided and replaced by DAP. The dihydrate wet process as a further method consists of the attack of the phosphatic ground rocks at an optimum granulation (below 100 mesh) with concentrated sulfuric acid (94-98%) in a recirculated phosphoric acid medium (16-18% P_2O_5 concentration). Phosphoric acid obtained (50-52% P_2O_5) can be combined with ammonia to produce ammonium phosphate or can be combined with phosphorite in triple ratio to produce superphosphates.

From the phosphoric acid facilities results the following sub-production wastes:

- Solid waste – phosphogypsum
- Liquid waste – acid waters
- Gaseous waste – fluorine

For contamination prevention in the environment, the factors of water, air, soil, foundation by one of the above-mentioned wastes, using specific technological measures.

For phosphogypsum, solid waste resulting from the filtration stage can be evacuated from plant:

- hydraulically, when it is collected together with chemically impure waters from plant in a phosphogypsum recipient where it is neutralized at the same time with the waters by $Ca(OH)_2$ 10%, it is then deposited by pumping into an impermeable settling tank, the decanted waters being recirculated in the plant.
- mechanically, when it is evacuated from the plant, by means of a conveyer belt, onto a waterproofed waste heap with ditches.

The chemically impure waters can be neutralized at the same time with phosphogypsum, evacuated in neutralization recipients, decanted and then evacuated in sewage. In exemplary method for deposition and remediation is explained in [58].

In nitric acid attack of phosphate rock, the rock is completely dissolved including uranium and radium which are 100% found in the liquid Phosphonitric (PN) solution. The advantage of the process is that nothing like phosphogypsum has resulted. In the process of fertilizer (complex) production of this type, the calcium nitrate $Ca(NO_3)_2 \cdot 4H_2O$ is usually separated by strong cooling as a solid crystallized product which is later converted to NH_4NO_2 and $CaCO_3$ by $(NH_4)_2CO_3$. The results have determined that no radioactivity was present in $Ca(NO_3)_2 \cdot 4H_2O$ separated and therefore no radioactivity was carried by solid $CaCO_3$. Thus the whole uranium and radium of initial rock are present in liquid PN solution after separation of $Ca(NO_3)_2 \cdot 4H_2O$.

In the process of complex fertilizer production of NP(K) type the PN solution is neutralized by ammonia. Therefore the end product includes the whole uranium and radium. This behaviour is shown in the following Table 13.

Table 13: Uranium and radium content in the NP fertilizer production

Phosphate rock used	Uranium mg/kg	²²⁶ Ra Bq/kg
Florida	100-120	1000-1200
Marocco	100-130	1200-1300
Algeria	90-100	1000-1100
Israel	90-100	900-1100
Jordan	80-90	800-900

The manufacture sub waste is calcium carbonate at the calcium nitrate conversion stage. The product is easily filterable, very reactive in the case of when used as agricultural amendment. The granulation is comprised between 40÷100 microns, the medium one being between 60÷80 microns. The humidity is of max. 15% free water when it is sent to the waste heap. For special demands, the nitro-phosphate plant is equipped with calcium carbonate drying equipment in a pneumatic column with adequate storage silos and specialized equipment for dispatch to the consumers.

The waste calcium carbonate quantity depends strictly on CaO content in the phosphatic rocks utilized in nitro phosphate producing. At a capacity of about 300 t P₂O₅ 100% / day in complex fertilizers, for the calculus medium rock about 700 t dry CaCO₃ / day, respectively 231000 t CaCO₃ / year. A tailings pond with this sub-product is Targu-Mures. The properties of the deposited material and the deposition strategy are explained in [57].

The main waste of Aluminium processing is "red sludge". This residue will deposited in Tailings Facilities. In Romania the Al₂O₃ alumina is produced at Oradea from bauxite exploration in Padurea Craiului Mountains and at Tulcea from imported ore. An example for the above mentioned processing plant in Oradea the radioactive contaminants are summarised in [58].

Table 14: Contents of radionuclides in Oradea alumina plant (Romania)

	ppm U	Bq/gr Ra	ppm Th
Material of red colour	11	0,21	61
Material of dark colour	<10	0,137	70
Material of light colour	127	1,675	11

- In air at the pond surface
- Rn222=21 Bq/m³
- Thoron=1220 Bq/m³
- In water from the red sludge pond
- Rn222=101 pCi/l
- Thoron=0
- Ra226=0,023 Bq/l
- pH=12,05

The link between high uranium concentrations and the organic matter from sedimentary rocks explains the concentration of radionuclides in coal processing. Uranium accumulation in coal storages hasn't a wide development but some ore bodies are known in which these accumulations reach an appreciable quantity and constitutes industrial exploitation object. The existent material analysis proves that in aged coals, represented mainly by hard coal, the uranium content is low. As a rule; its highest concentrations are linked to low metamorphosed brown coal or by Mesozoic or younger age lignite. There are known cases in which the uranium industrial concentrations are linked to actual peat exploitation.

If uranium is in coal and peat in high quantities, given "Clark" values situation and also an important number of rare and dispersible elements, in certain cases this can constitute auxiliary exploitation object.

For uranium exogenous ores forming in carboniferous formations, a complexity of favourable conditions is necessary, the most important among them being the mineral feeding sources closeness and possibly the migration and uranium concentration in sediments and rocks especially concerning both the composition and physic-mechanical conditions (porosity, permeability) and the geo-chemical particularities.

It seems that for none of the exogenous ores, the uranium source problem isn't solved with the same precision as in the case of uraniferous coal ores. Usually, there is a direct link either with the hydrothermal ores in degradation close by the sedimentation basin or with acid intrusive rock massifs and effusive coverings which present high uranium "Clarks" or with aged emerged sedimentary complexes, for example with bituminous schist of high uranium content. Pit coal type ore is processed in the plants: Coroiesti, Petrila and Lupeni, which are located in Petrosani mining area. The coal is washed and divided in granulometric sorts. Measurement performed on the liquid phase of the waste discharged in the tailings pond shows: $U = 0.026 \text{ Bq/l}$, $Ra = 0.006 \text{ Bq/l}$.

5.5.4 Characteristics of main types of radio-nuclides resulted from the fine bodies originated during processing

Even since forming, our planet has inherited a considerable dowry of terrestrial radioactivity from which only a part has been kept respectively high half-life radioactive isotopes and elements:

- ^{238}U , ^{235}U and ^{232}Th generating the three natural radioactive families and their daughters (Table 9, Fig. 13);
- ^{42}K radioisotope which accompanies natural potassium in all the combinations (0.118%);
- Tritium (^3H) and Carbon (^{14}C) which are formed by the neutronic component action of cosmic radiation with nitrogen nucleus;
- other natural radioisotopes of some elements such as: ^{87}Rb , ^{115}In , ^{130}Te , ^{138}La , ^{147}Sm , ^{171}Lu , ^{189}Re and other high half-life ones.

The different decay series and decay chains are well described in [47], [137]. Those radio elements are only some rests which speak to us about those days when almost all the nuclear species were radioactive. Among the most distributed ones in the earth's crust, uranium, thorium and radium are presented in a larger or smaller measure in the eruptive, metamorphic or sedimentary rock, in hydrosphere and troposphere. Under the action of atmospheric agents (wind, precipitations, temperature variations), the radioactive minerals are partially altered, solubilized and transported by waters, contributing to the environmental radioactive pollution. At the same time in atmosphere, also concentrated are a considerable amount of radon active emanation released by successive transformations of uranium which enters in equilibrium with radon from mine waters, released by solubilized uranyl salts.

The main radioactive elements which are found in the tailings ponds of radioactive ores are uranium, radium, radon and thorium (in the case of the tailings ponds from the processing of monazite content radioactive ores).

Natural uranium is a mixture of radioisotopes: ^{238}U 99.28% uranium family generator, ^{235}U (UAc) 0.720% actinium family generator and ^{234}U (UII) 0.0055% member of uranium family.

Among the recoverable minerals (found also in the tailings ponds in a small measure), the most important are: Uraninite, UO_2 , Pechblende, U_3O_8 , and Carnotite, $\text{K}_2(\text{UO}_2)_2(\text{VO}_2)_2 \cdot 2\text{H}_2\text{O}$. It is also spread in rivers, seas and oceans, in oil, coals and natural gases (sea water contains $0.4 \cdot 10^{-7}$ - $2.3 \cdot 10^{-6}$ g U/l under the form of soluble salts, clay schist and lignite contains 0.7-1% U_3O_8 (Marcu 1997).

Table 15: Properties of the natural uranium isotopes

	U-234	U-235	U-238
Half-life	244,500 years	$703.8 \cdot 10^6$ years	$4.468 \cdot 10^9$ years
Specific activity	231.3 MBq/g	80,011 Bq/g	12,445 Bq/g

Table 16: Isotopic composition of natural uranium

	U-234	U-235	U-238	Total
Atom %	0.0054%	0.72%	99.275%	100%
Weight %	0.0053%	0.711%	99.284%	100%
Activity %	48.9%	2.2%	48.9%	100%
Activity in 1 g U_{nat}	12,356 Bq	568 Bq	12,356 Bq	25,280

Radium. Although it hasn't its own minerals it is spread under the form of four radioisotopes: ^{226}Ra , $^{228}\text{Ra}(\text{MsTh}_1)$, $^{224}\text{Ra}(\text{ThX})$ and $^{223}\text{Ra}(\text{AcX})$, members of the three natural radioactive families. The main source of radium acquirement is Pechblende (U_3O_8) which contains about 200 mg Ra/t U_3O_8 . In the earth' s crust, it is spread $2,0 \cdot 10^{-10}$ %, while in the natural waters it is spread 10^{-13} g Ra/l.

Table 17: Natural radium radioisotopes

Radioisotope	λ (s^{-1})	T
^{226}Ra	$1.38 \cdot 10^{-11}$	1600 a
$^{228}\text{Ra}(\text{MsTh}_1)$	$3.29 \cdot 10^{-9}$	5.75 a
$^{224}\text{Ra}(\text{ThX})$	$2.20 \cdot 10^{-6}$	3.64 d
$^{223}\text{Ra}(\text{AcX})$	$5.50 \cdot 10^{-4}$	11.43 d

Radon. Is spread in nature as three radioisotopes: radon ^{222}Rn , thoron ^{220}Tn and actinon ^{219}An , which are members of the three natural radioactive families (U, Th, Ac), resulted by the disintegration of radium under the form of active emanation.

Table 18: Natural radon radioisotopes

Radioisotope	λ (s^{-1})	T
^{222}Rn	$2.07 \cdot 10^{-6}$	3.824 d
^{220}Rn	$1.27 \cdot 10^{-2}$	55.6 s
^{219}Rn	$1.77 \cdot 10^{-1}$	3.96 s

Generally, radon solubility in water is low but 2.5 hours later an equilibrium is established between radon from air and radon from water. In equilibrium state, depending on temperature, its solubility in water is of about 0.51% in comparison with its solubility in air at 0 °C. On the other hand, its solubility in fats is over 100 times higher than in water.

Disintegrating by α particle emission, emanations turn into Po, Bi, Pb and Tl radioisotopes, half-life active α and β radioisotopes which are generally small and form the so-called invariable deposit.

Table 19: The invariable deposit of radon

Radioisotope	T
$^{214}\text{Pb}(\text{RaB})$	26.8 min
$^{210}\text{Pb}(\text{RaD})$	22.3 a
$^{214}\text{Bi}(\text{RaC})$	19.8 min
$^{210}\text{Bi}(\text{RaE})$	$3.5 \cdot 10^6$ a
$^{218}\text{Po}(\text{RaA})$	3.05 min
$^{214}\text{Po}(\text{RaC})$	164 μs
^{210}Po	138.38 d
$^{210}\text{Tl}(\text{RaC})$	1.3 min

Penetrating the human body, radon solubilizes easily in blood where it produces the active deposit which is then transported to different organs of the body, being able to produce lesions. This is the reason radon joins the radionuclides with high radiotoxicity.

Permanently released by the deposits which contain radioactive ores, from those radioisotopes only radon has a half-life high enough to be drawn in atmosphere, in soil and in natural waters. Thoron and actinon are found only in the air layer close to the soil. On an average, radon concentration in the atmosphere is low, respectively 10^{-13} Ci/l \approx 37 mBq/l and grows above the areas rich in radioactive ores reaching 10^{-9} – 10^{-8} Ci/l.

5.5.5 Pathways for radio-nuclides migration into environment factors.

The existent radionuclides in the waste disposed in the tailings ponds spread through the air current (gone with the wind) as dust or migrate in solutions. The waste resulting from uraniferous ores processing represents a mixture of non-radioactive mineral dusts and radioactive dusts which contain all the elements of ^{238}U , ^{234}U , ^{230}Th and ^{226}Ra series and alpha-transmitter with long periods (whence the name radionuclides with long life-time).

Radionuclides migration by air due to wind action is realized in a physical form and depends to a large measure on the granulometry and humidity of the material deposited in the tailings pond, the weather conditions of the zone (wind speed, warmth interval cycle) and the vegetable layer coverage degree of the tailings ponds surface and surroundings. There are a few important aspects of the migration of solid natural state radionuclides in solutions (because of infiltration water or rainfall on the tailings pond):

- the transport of a certain radionuclide type to its destination has the implication of the generation of all the daughters which derive from its decay; for example, radium transported and concentrated is dangerous by itself and by the radon and its daughters which are formed;
- Due to Szillard-Chalmers effect, derived waters especially draw radio-elements comparatively with the initial element, which explains why in natural waters there are:
 - ^{228}Th is in excess compared with ^{232}Th ;
 - ^{234}Th is in excess compared with ^{230}Th , ratio 10/1;
 - ^{234}U is in excess compared with ^{238}U ;
 - Ra is in excess compared with ^{238}U ;
- Excepting radon which can migrate in a gaseous state and radon daughters which migrate at the same time with the free air, all the radionuclides migrate in solution, either dissolved or in suspensions absorbed or not on fine particles.

Radionuclides transport in solution is favoured by:

- the radionuclides concentration in the waste disposed in the tailings pond;
- the existence of some propitious hydrodynamic conditions;
- the existence of some migration routes (the sterile porosity, storm flow on the tailings pond, evacuated water);
- the existence of favourable physio-chemical conditions (all the processes of drawing in solutions take place in oxidizing conditions).

Radionuclides deposition from suspensions takes place in common conditions existent for such depositions while from the dissolved state they are dependent on the physio-chemical conditions changing (depositions being mainly favoured by the existence of a reducing medium: bitumen, earth oil, coal, peat, sulphurous solutions). Uranium migration from the tailings pond depends on factors such as:

- oxidizing or reducing conditions of the medium;
- climate particularity;
- liquid phase composition.

Uranium can migrate either in solution or in sorbed state on the colloidal mechanical suspensions or in association with the fine material. Fine material and suspension are deposited according to common physics laws; the phenomenon is emphasized especially when the transport speed decreases. Fine material doesn't generally migrate far from the tailings pond. Dissolved uranium can precipitate at the change of the physico-chemical characteristics as follows:

- From sulphates acid solution by: dilution of acid waters with meteoric waters; interaction between ammonium and uranyl ion; interaction between solutions and limestone; sorption on colloids; iron hydroxides;
- From carbonates solutions by: pH decrease; presence of Cu, Ca, Pb cations; reduction with organic matter; intense evaporation in medium with Cu and Mg;
- From soluble humus compound by: sorption on different sorbing agents (organic matters); U (VI) to U(IV) reduction by organic matter.

Radium is leached by the infiltration of meteoric waters from the tailings pond more easily than uranium due to both chemical properties and its good position in the crystalline network. Waters with Na, Ca, etc. chlorides are favourable for radium solubilization and its maintenance in solution, while the waters with carbonates and hydrocarbonates are less favourable for radium changing in solution. The waters with sulphides are also less favourable for radium maintenance in solution; from these kind of waters it precipitates with Ba and Ca sulphates. Radium is easily absorbed by a variety of different matter and can be found enriched in them. Due to those characteristics, radium can present a potential contamination risk even away from its source.

In the surface waters, migration is influenced by the following factors:

- radium in chemical or physical form (radium dissolved can exist as ion pairs: RaSO_4 and Ra^{2+});
- water composition;
- adsorption of dissolved radium or from solid suspensions on sediments;
- radium co-precipitation with solids existent in water;
- biological transport.

Thorium generally migrates with difficulty, existing only in solution:

- in cases where there are conditions to form soluble compounds with organic acids;
- in waters with high concentrations of carbonates ions.

The majority of thorium migrates as clastic material and in fine suspensions, colloiddally but in low concentrations and not away from its source being adsorbed from solution by the organic matter and by iron and manganese hydroxides.

When a radionuclide in solution migrates with the waters through the porous medium constituted from pond sediments, many interactions between the sterile medium and radionuclide take place; these interactions generally lead to radionuclide concentration changes as time goes on. There are the following phenomena which lead to radionuclide concentration changes: hydrodynamic dispersion, transport by convection and sorptive influence of natural ion exchangers.

5.6 Radioactive Tailings Facilities interacting with their environment – Case Studies

Radon-222, the gaseous decay daughter of radium-226, can emanate from tailings and become free to diffuse to the surface of the pile and escape to the atmosphere. The radon half-life is 3.8 days and many of the gaseous radon atoms decay to solid polonium-218 before reaching the surface. Because the radon is an inert gas that can readily migrate, it cannot be assumed that radon or its daughters are in secular equilibrium with radium-226. Because radon is free to move through a tailings pile to the atmosphere, the radon activities through the tailings and the release rate from the surface (or flux in Bq/m²/s) will depend among other things, on the extent to which this movement occurs. The background release depends on the type of soil but it is on the level of app. 50 mBq/m²/s. At the same time the releases from tailings piles can reach more than 10 Bq/m²/s value.

Emission source	Particulates (MBq·a ⁻¹)			Radon-222 (TBq·a ⁻¹) ^a
	U-238 U-234	Th-230	Ra-226 Pb-210 Po-210	
Ore hauling and storage pad ^b	24.8	24.8	24.8	} 2.5
Ore crushing and grinding ^b	33.3	33.3	33.3	
Yellowcake drying and packaging ^c	5550	27.0	5.6	Negligible
Tailings pile	322	4440	4440	163
Dispersed ore and tailings ^d				1.8

Figure 61: Particulates of different sources [86]

The radioactive emissions generated by a model mill [86] are illustrated in Figure 61. It can be seen that radon is generated to a great extent on tailings piles, while uranium is released from the yellow cake drying and packing system. The emanation of radon from the particles and its movement through the pile to the atmosphere is controlled by many factors [142], [52] including:

- emanation coefficient,
- self-confinement,
- adsorption,
- moisture,
- atmospheric pressure,
- multi-layering effects,
- surface vegetation, and surface covers.

Emanation coefficient

The emanation coefficient E is about 0.03 to 0.5 for most uranium tailings, depending on mineralogy, moisture and the size of the tailings particles. The alkaline extraction process will produce a higher value of E since the ore must be milled more finely. Dependence of E related to the moisture content is illustrated in Figure 62.

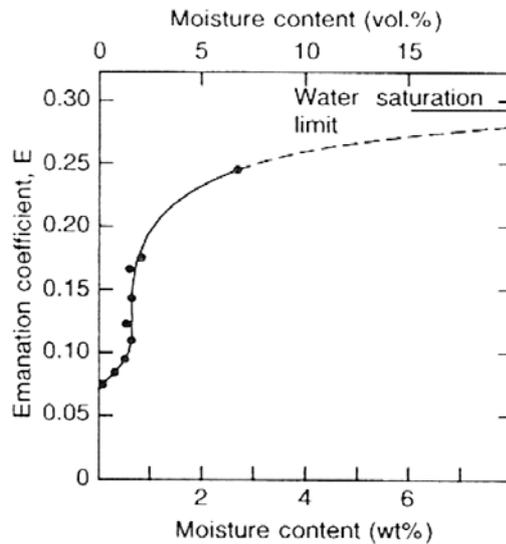


Figure 62: Dependence of the emanation coefficient E from moisture content wt [135]

Self-confinement factor of the tailings pile

The self-confinement factor ϵ is defined as the fraction of the radon atoms released from the ore particles that reaches the atmosphere without decaying within the tailings pile. This factor is a function of the tailings thickness, tailings density and the effective relaxation length for the porous medium. The effective relaxation length H_{eff} is a measure of the depth of tailings through which radon can diffuse before decaying in its daughter. The value of H_{eff} depends on the specific conditions in the layer of tailings and is a function of the diffusion coefficient of the radon-222 in air ($D_A = 1.03 \times 10^{-5} \text{ m}^2/\text{s}$ at 20 °C), the tortuosity factor, and λ , the decay constant of radon-222 ($\lambda = 2.06 \times 10^{-6} \text{ s}^{-1}$). The effective relaxation length is typically between 1 and 2 m. If the thickness of tailings pile is large compared with H_{eff} , then the number of atoms released by the pile is not dependent on the thickness of the pile. The reason for this is that only the radon portion formed in the top layers (1-2 m), can escape from the pile, independent of how thick the pile is.

Adsorption

The adsorption can have the effect of reducing the effective relaxation length and hence reducing the radon-222 release from the tailings pile.

Moisture effect

Moisture content of the tailings has a great effect on the radon release mainly but not exclusively because of the absorption and adsorption process of radon atoms between the liquid and the gas phases, this will result in a reduction in the effective diffusion coefficient. The diffusion coefficient of radon in water is $D_w = 1.4 \times 10^{-9} \text{ m}^2/\text{s}$ (at 20°C). The effective diffusion coefficient thus varies from 1.03×10^{-5} to $1.4 \times 10^{-9} \text{ m}^2/\text{s}$ when the water content of the medium changes from 0 to 100%. The dependence of the effective diffusion coefficient from the moisture is illustrated in Figure 63.

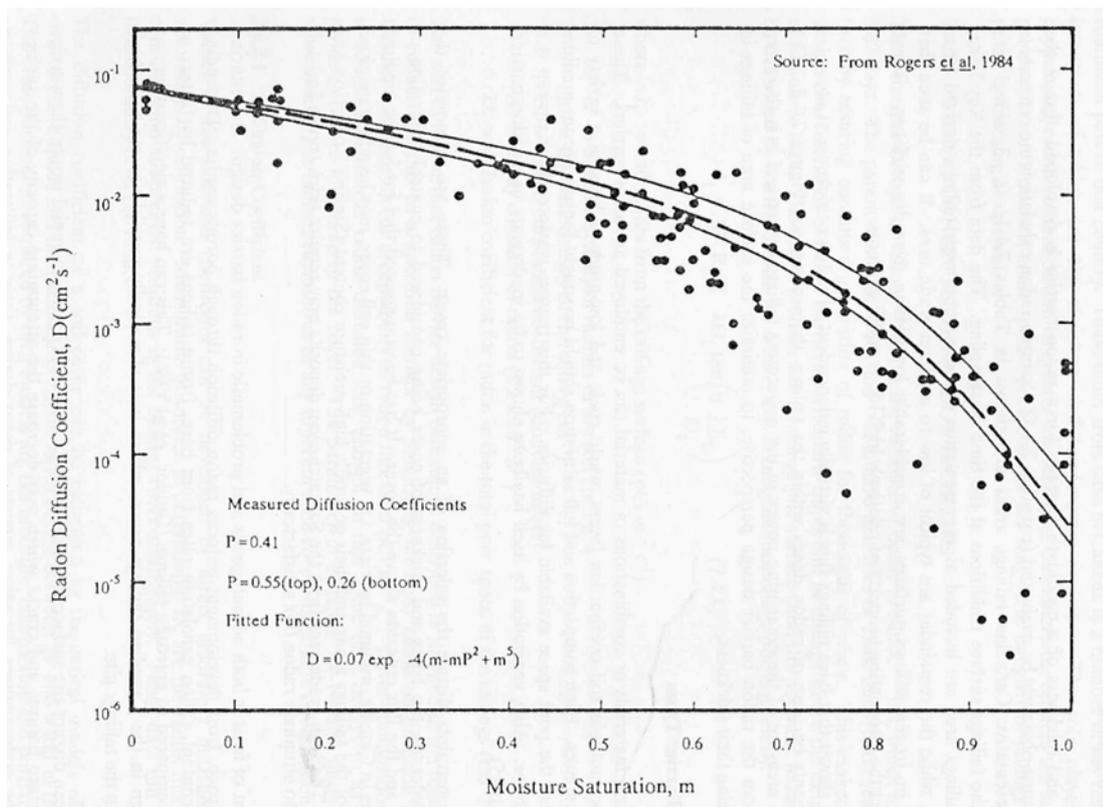


Figure 63: Radon diffusion coefficient in relation to moisture saturation [99]

Atmospheric pressure effect

Changes in atmosphere pressure can result in temporary change in radon release rate. Usually the duration of atmospheric pressure changes is much less than the half-life of radon (3.8 d). In this case the effect of cyclic change of atmospheric pressure of strength Δp is to release all the radon emanated to the pore space in the top fraction ($\Delta p/p$) of the tailings pile. There is a lot of data for radon flux from tailings piles. Some data is presented in Figure 64.

Location	Flux from tailings ($\text{Bq}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$)	Concentration over tailings ($\text{Bq}\cdot\text{m}^{-3}$)	Other concentrations ($\text{Bq}\cdot\text{m}^{-3}$)
<i>USA</i>			
Baggs	3.1		0.074 at 2 km
Belfield	1.7 (max.)		
Bowman	1.5 (max.)		
Durango	1.3–11.5		
Falls City ^a	0.1–2.9	70	Background reported as 37 measured at 0.5 km
Grand Junction ^b	0.9–24	2200 (max. reported)	96 at 0.5 km
Green River ^b	1.2–4.8		85 at 5.4 km, 33 at 0.2 km, background 55
Gunnison Site ^a		133 (max. reported)	37 at 1 to 8 km
Lakeview ^a	22.6–26.3		22 to 26, distances not reported
Lowman	1.8–5.5		Background averages 44
Maybell ^b	2.8–3.7	540	Background as high as 110, owing to large amount of ore in region
Mexican Hat	0.6–59	3220–10 400	7.4 at 0.6 km, background 22 at 2 km
Monument Valley	0.5–1.1	250	Background 22 at 0.6 km
Naturita ^b	28–94		Maximum 560 at 0.3 km distance, background of 74 at 5 km
New Rifle	2.6–52	1080	Background at 1.6 km
North Continent ^b	0.15–9		
Old Rifle ^b	7.8–48	5200	Background at 1.6 km
Phillips/United	1.5–11		1040 at 0.2 km, background reported at 110–180
Riverton ^a	1.9–3.0	244 (max.)	Background 41 at 1.6 km
Salt Lake City (Vitro Chemical)	3.7–33	7.4–500	Background 22 at 0.8 km
Shiprock ^a	2.0–5.8	474	37 at site boundary
Spook	7.0–106	630 (max.)	Background at 3 km
Tuba City	0.4–15	810	74 at 0.2 km, background 26 at 5 km
Union Carbide ^b	0.2–0.9		140 at 0.3 km
<i>Canada</i>			
Lacnor, Ontario	0.008–6.7	8–59	5–30 off edge
Gummar, Saskatchewan	0.04–5.6	700–1500	10–230 off site but tailings surface contamination present in some areas
Beaverlodge, Saskatchewan	0.5–13.3	18–1307	42 over waste rock pile
<i>Spain</i>			
Andujar	10	178–560	97 in local houses

^a Covered with material of various thicknesses.

^b Covered with up to 15 cm of material, various states of erosion.

Figure 64: Radon concentration in the vicinity of tailings ponds (inactive sites) [51]

It can be seen that the radon concentration above tailings piles varies through a wide range but usually is some hundred Bq/l. Variation of atmospheric radon concentration with distance at Shirley Basin mine/mill complex is illustrated in Figure 65.

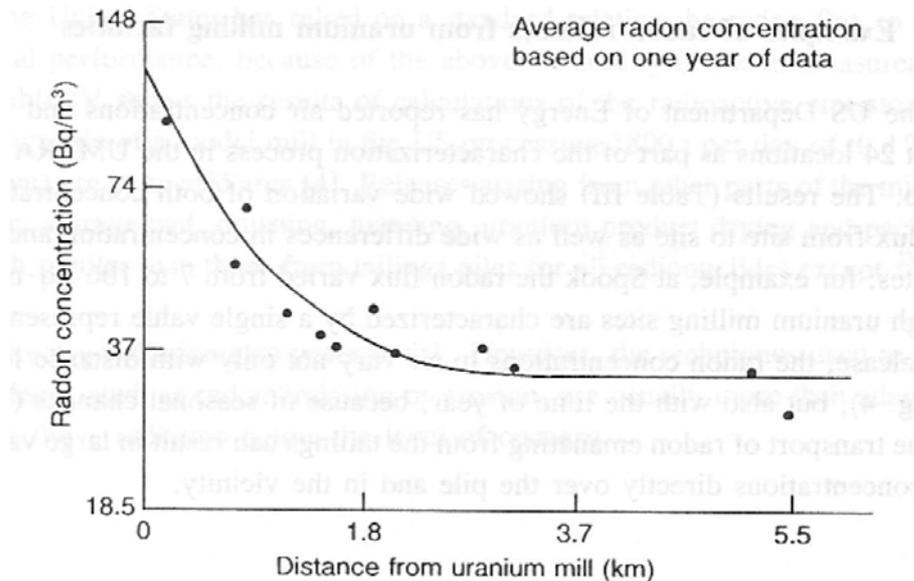


Figure 65: Radon concentration decreasing with distance from the source [86]

Atmospheric radon concentration varies in time. This is illustrated in Figure 66, where the long-term measurement results are presented for a village (Pellérd) 1 km away from tailings piles. It can be seen that the radon concentration reaches an extremely high value (some hundred Bq/l) for a short period, then drops to the average value.

Estimation of the radon release from tailings

Radon release into the environment involves two mechanisms for the liberation from the particle in which it is formed. This is characterised by radon emanation coefficient and transport thorough the bulk medium to the atmosphere, characterised by the diffusion coefficient in the bulk medium. In addition, the release to the environment will be affected by the presence of covering layers and meteorological conditions. On the bases of above mentioned principles and approaches some equations have been developed for calculation of radon release from tailings. The mathematical relations are discussed in detail in [52]. The flux from an uncovered tailings pile is directly related to the radium activity, emanation coefficient and bulk density. If any of these variables increase, the surface radon flux increases proportionally. The flux also increases as the diffusion coefficient increases. It is shown in [43] that the thickness has no effect beyond about 2 m.

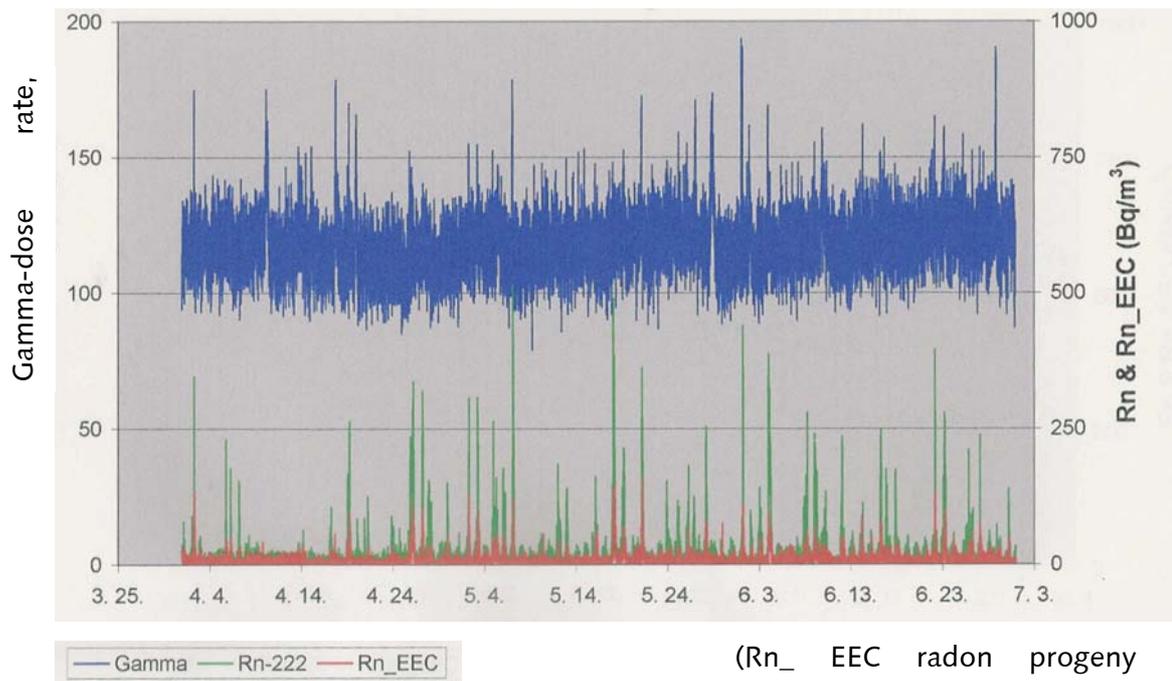


Figure 66: Radiological parameters measured in Pellerd (Hungary, March-July 2002)

6. Conclusions

The previous chapters of this workpackage explain the different processes and parameters which can produce adverse effects on tailings facilities. Summarizing this there are two general components relevant for safety considerations: the impacts to the site and the effects as their response.

Impacts can be gravity of the tailings, different forces of water and wind, earthquake accelerations or the content of contaminants within the impoundment. Effects are the interactive response of the parameters and of the environment due to several impacts. With a view to risk and reliability we can characterise two different groups of effects, decrease in structural safety and environmental damage. However, for both we have to define what failure means, how adverse the result should be to be called a failure. In engineering terms this means to define limit state conditions.

If the effect results in a decrease or in a loss of structural stability of the impoundment or the surface we are forced to make adequate measures of strengthening. For assessing this we need several geometrical data. Investigation and monitoring should deliver soil-mechanical parameters of the dam and the transition zone at one hand and an idea of the future maximum possible impacts at the other.

In all questions of environmental hazards we can use the well established source-pathway-target framework of contaminated land risk assessment. But there is one evident difference to common contaminated sites that is the amount of the contaminants within the impoundment and the duration of possible threats running out from this. Beside this often there is no specific legislation to demand remediation measures for mining waste and tailings facilities.

The final key question in this field might be: "What is the major risk, running out from all this impacts and parameters we analysed in this report?" There is no clear answer to this question. The most sensitive parameters and the most critical impacts, its frequency as well as the severity of an adverse event can change from site to site depending on the kind of tailings, the periodical change in dam construction, the pore water pressure distribution within the transition zone, the capacity of the subground to transport or adsorb contaminants, the surrounding geological conditions and the distance of the downstream residential and infrastructure. However, in most of the cases such an analysis will end with the result that risk will be coupled with any effect of water. Therefore water management and monitoring as well as the derivation of threshold values of pore water pressure and seepage level is one of the most important tasks. Anyway, if we want to identify the relevance of the risk posing parameters or the vulnerable points of a facility we are forced to analyse complex processes and scenarios and we must estimate the likelihood of interactive influence of impacts and effects. The steps we have to do in this task for any specific site are

- (i) HAZARD IDENTIFICATION That means data collection, identification of the site conditions, physical parameters of the impoundment and the tailings, topography, the geology and the hydrology. The question in this first step is: *What are the fundamentals of a possible problem?*
- (ii) HAZARD ASSESSMENT The aim of this step is to set up event trees or fault trees for imaginable scenarios. This includes to compound processes with different impacts and interactions. The question in this step is: *How big the problem might grow?*
- (iii) RISK ESTIMATION is the quantitative analysis of all those scenarios, combining the branches of the event trees with estimated probabilities and sum it up to the likelihood of the certain scenario. The question therefore is: *What will be the effect of a certain scenario and how strong is it?*
- (iv) RISK EVALUATION During this final step that summarises all the knowledge about the probable adverse effects and under consideration of the socio-economical and legal aspects the assessment of risk and reliability is reduced to the simple question: *What might happen and does it matter?*

Doing this steps we will get a sufficient and objective data base for better decisions concerning design, processing and after use of tailings facilities and that's what we call Risk assessment.

7. Failure of Tailings Impoundments – Case Study

Dam/Mine Name:	Barahona
Mine Location:	Chile
Ore/Tailings Type:	copper
Dam Height (m):	61
Dam Type:	US (Upstream)
Dam Fill Material:	CST (Cycloned Sand Tailings)
Impoundment Volume (cu. m):	20,000,000
Incident Information:	
Date:	10-01-1928
Incident Type:	1A (failure of active impoundment)
Cause:	Earthquake
Quantity of Tailings Released (cu. m):	2,800,000
Tailings Travel Distance (m):	

Incident Description:

The dam was constructed by cycloning sand tailings to form the outer shell. Embankment slopes were as steep as 1:1, and at the time of failure the last perimeter dike on the embankment crest had been constructed to a height of 55 feet. The dam failed by liquefaction during the M8.3 Talca earthquake of October 1, 1928. A tailings flowslide developed through a breach section approximately 1500 feet wide and flowed down the valley, killing 54 people.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name:	El Cobre Old Dam
Mine Location:	Chile
Ore/Tailings Type:	copper
Dam Height (m):	35
Dam Type: US Dam Fill Material:	T (Tailings)
Impoundment Volume (cu. m):	4,250,000
Incident Information:	
Date:	28-3-1965
Incident Type:	1A (failure of active impoundment)
Cause:	Earthquake
Quantity of Tailings Released (cu. m):	1,900,000
Tailings Travel Distance (m):	12,000

Incident Description:

The embankment failed catastrophically in the M7-7 1/4 La Ligua earthquake of March 28, 1965 by liquefaction. The dam had been constructed according to the upstream method by spigotting from flumes on the crest and was in use as an emergency impoundment at the time of the earthquake. Embankment slopes as steep as 1.2:1.0 and the presence of slimes layers near the face suggest that static stability may have been marginal even before the earthquake. The tailings flowslide destroyed the town of El Cobre, killing more than 200.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name:	Mochikoshi No.2
Mine Location:	Japan
Ore/Tailings Type:	gold
Dam Height (m):	19
Dam Type: US Dam Fill Material:	T (Tailings)
Impoundment Volume (cu. m):	480,000
Incident Information:	
Date:	01-15-1978
Incident Type:	1A (failure of active impoundment)
Cause:	Earthquake
Quantity of Tailings Released (cu. m):	3,000
Tailings Travel Distance (m):	150

Incident Description:

The embankment was constructed with rockfill starter dikes, and had slopes of 2.5:1 to 3:1. Liquefaction failure occurred the day after the January 14, 1978 M 7.0 Izu–Oshima– Kinkai earthquake, and about 5 hours after the two after shocks of M 5.4 and M 5.8.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name: Itogon-Suyoc

Mine Location: Baguio gold district, Luzon, Philippines.

Ore/Tailings Type: Gold

Dam Height (m):

Dam Type:

Dam Fill Material:

Impoundment Volume (cu. m):

Incident Information:

Date: 07-26-1993

Incident Type: IA

Cause: over topping

Quantity of Tailings Released (cu. m):

Tailings Travel Distance (m):

Incident Description:

Diversion tunnel taking the river around the impoundment, blocked, causing flood water to enter impoundment and overtop dam. Partial failure, i.e. collapse of part of the dam. Contributory cause; typhoon.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name:	Placer Bay, Tailings Pond No.5
Mine Location:	Surigao Del Norte, Philippines
Dam Height (m):	17
Dam Type:	WR (water retention)
Dam Fill Material:	E (earthfill)
Impoundment Volume (Cu. m):	
Incident Information:	
Date:	9:20 on 09-02-1995
Incident Type:	IB (failure, inactive impoundment)
Cause:	slope instability
Quantity of Tailings Released (Cu. m):	50,000
Tailings Travel Distance (m):	out to sea

Incident Description:

Inactive impoundment. Waste rock being placed on top of tailings for storage when dam failed into the sea forming 100m breach. Cause partly toe erosion and foundation on reclaimed land. 12 killed including safety inspector, and 14 pieces of heavy equipment were lost that had been working to place waste rock on the impoundment.

On 21st Dec 1986, Typhoon Ameng washed away a portion of the dam at the seafront. Another collapse occurred on 9th July 1987, both incidents releasing effluent with high levels of cyanide resulting in fishkill.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name:	Galena Mine
Mine Location:	Wallace, ID, USA
Ore/Tailings Type:	silver
Dam Height (m):	9
Dam Type:	US (upstream dam)
Dam Fill Material:	MW (mine waste)
Impoundment Volume (cu. m):	
Incident Information:	
Date:	01-15-1974
Incident Type:	1A
Cause:	OT (overtopping)
Quantity of Tailings Released (cu. m):	3,800
Tailings Travel Distance (m):	610

Incident Description:

Three tailings impoundments in a sidehill configuration adjoined each other within a narrow valley with a creek at their toe. During a rain-on-snow event, flooding on the creek reached estimated 100-yr. recurrence interval flows. A culvert in the creek upstream from the impoundments became blocked by debris, diverting a large portion of the streamflow into the uppermost impoundment. Lacking sufficient decant spillway capacity for these flows the uppermost embankment breached by overtopping, resulting in cascade failure of all three impoundments. Tailings released in the failure covered about 5 acres, including a short section of highway and railroad track. This incremental damage was insignificant in relation to general flood damages to public and private property.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name: N'yukka Creek
Mine Location: USSR
Ore/Tailings Type:
Dam Height (m): 12
Dam Type: WR (water retention)
Dam Fill Material: E (earthfill)
Impoundment Volume (cu. m):
Incident Information:
Date: 1965
Incident Type: 2A (accident, active impoundment)
Cause: foundation
Quantity of Tailings Released (cu. m):
Tailings Travel Distance (m):
Incident Description:

The dam was constructed as a starter dike for subsequent upstream raising, and operated initially to retain water. During first filling, sinkholes appeared in both abutments, and were initially treated by covering with tailings. When this proved ineffective, a concrete cutoff wall was constructed through the embankment and into the foundation. Sinkhole development was attributed to thawing of foundation permafrost that allowed ice-filled joints in foundation rock to transmit seepage and piping to occur.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name:	Aurul S.A. Mine
Mine Location:	Baia Mare, Romania
Ore/Tailings Type:	Gold
Dam Height (m):	A few m; future final height 20 m
Dam Type:	Initially DS (downstream), later US
Dam Fill Material:	T (cycloned tailings)
Impoundment Volume (cu. m):	Approx. 800,000
Incident Information:	
Date:	1-30-2000
Incident Type:	1A
Cause:	ST (structural)
Effluent released(cu. m):	100,000 (Estimate)
Tailings Travel Distance (m):	

Incident Description:

After extreme weather conditions (ice and snow on the tailings pond, high precipitation: 36L/m²), the tailings deposited on the inner embankment (starter dam) became saturated. Stability was affected, causing local displacement, and this subsequently developed into a breach of approximately 23 m in length. The effluent released through the breach filled the area between the starter dam and the outer perimeter dam, both surrounding the impoundment (93 hectares in area), and spilled over the outer embankment. Around 100,000 m³ of cyanide-rich (50-100 tonnes) effluent contaminated also with some heavy metals was released into the Somes and Tisza rivers and then into the Danube, finally reaching the Black Sea. Significant contamination occurred over a stretch of 150 to 180 m, then became more and more diluted. It caused significant fishkill and destruction of aquatic species in the river system.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

Dam/Mine Name: Kimberley
Mine Location: British Columbia, Canada
Ore/Tailings Type: iron
Dam Height (m):
Dam Type: US (upstream)
Dam Fill Material: T (tailings)
Impoundment Volume (cu. m):
Incident Information:
Date: 1948
Incident Type: 1A
Cause: SI (slope instability)
Quantity of Tailings Released (cu. m): 1,100,000
Tailings Travel Distance (m):

Incident Description:

The embankment was constructed by direct spigotting of tailings using upstream raising procedures. The foundation is believed to have consisted of low-permeability glacial till. The failure is attributed to freezing of the dam face during a period of high snowmelt and spring runoff that raised the phreatic surface and caused slope instability. A large tailings flowslide was triggered that moved toward, but apparently did not reach, the St. Mary River a few miles away. Frozen blocks of material were observed in the flow failure mass.

Source: (United Nations Environmental Programme (UNEP) Division of Technology 2001)

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