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# Disposal of waste materials at the bottom of pit lakes

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## Abstract

*Mine pits have been used as sites for disposal of wastes from mining, ore milling and refinery, oil sand processing, by-products of acid mine drainage (AMD) neutralisation, ashes of coal combustion in power plants or even industrial wastes. In several cases, pit lakes formed after disposal of the waste materials. In other cases, the disposal went on after formation of a pit lake or was even conducted in order to neutralise the pit lake. However, the deposition of waste in surface water is not allowed in many countries.*

*The purpose of the paper is to contribute to the discussion how to handle such existing waste deposits. In order to reach that goal, the paper gives a brief overview over processes relevant for the transport of substances from the waste into the main water body of pit lakes. Examples and experiences from Germany and from international literature are presented.*

*The presented examples and the literature show that there are advantages and disadvantages accompanying subaqueous disposal of waste. In general, the stability of the conditions inside the deposited waste and at its interface with its aqueous environment is a main prerequisite for successful long term storage of waste below a water cover. In this respect, meromixis is usually helpful. Risks such as long term change of conditions inside and around the waste deposits and the pit lakes, as groundwater contamination or as toxication of aquatic life have to be evaluated carefully and site specifically. However, there are no scientifically reasonable arguments for a general preclusion of the subaqueous disposal of waste in pit lakes.*

## 1 Introduction

Mines produce typical wastes. Besides overburden and waste rock, sludge from mine water treatment plants is another typical waste of operating mines. Further processing of the mined minerals close to the mine site produces additional wastes: tailings from ore milling and refining or from oil sand processing, slack from smelters, coal refuse from coal washing and ashes from power plants. For all such wastes, mine voids are potential disposal sites. If mining sites are closely associated with other industries like chemical industry, mine voids appear as potential disposal sites also for other industrial wastes.

Literature reports on several cases of subaqueous deposition of mine wastes with small and acceptable environmental impacts: on the sea floor (Garnett and Ellis, 1995; Jones and Ellis, 1995; Ellis and Robertson, 1999; Poling et al., 2002) and at the bottom of lakes (Pedersen et al., 1998; Ellis and Robertson, 1999). Wet covers were used as approach for remediation of tailings ponds (Amyot and Vezina, 1997) and coal slurry impoundments (e.g. Stump, 2001). However, there are also examples where disposal of wastes under a water cover was accompanied by adverse environmental impacts (Poling and Ellis, 1995; Ellis and Robertson, 1999; Martin and Pedersen, 2004; Schultze et al., 2008). Interaction between dumped overburden and lake water, directly or via ground water, is often one of the reasons for acidification and poor water quality in pit lakes (Klapper and Schultze, 1995; Miller et al., 1996). Permanent chemical stratification (meromixis) limits the impact of adverse substances to the bottom layer (monimolimnion) of pit lakes due to the limited exchange between the monimolimnion and the overlying aerated part of the water body (mixolimnion) (Boehrer and Schultze, 2008). Therefore, meromictic appear as a helpful condition in the case waste disposal in pit lakes.

Although disposal of waste is not allowed in surface waters, waste deposits in pit lakes exist and have to be handled. The purpose of the paper is to contribute to the discussion how to handle such existing waste deposits. Firstly, the paper gives a brief overview over processes relevant for the transport of substances between waste and the lake water and between monimolimnion and mixolimnion. Secondly, examples and experiences from Germany and from international literature are presented. Finally, conclusions are drawn.

# 2 Relevant processes

## 2.1 Interaction waste materials – lake water

Disposal of waste in an existing pit lake (① in Figure 1) is accompanied by potential washout of constituents from the waste during sedimentation. As long as disposal is continued, it sustains the direct contact between fresh waste and lake water at the lake bottom. Once the disposal ceased, diagenetic processes change the condition at the interface between lake water and dumped waste.

Natural sedimentation (e.g. of dead plankton, eroded soil from the lake shore or from the catchment area etc., ① in Figure 1) results in the formation of an increasing cover layer when waste disposal ceased. Distances between lake water and waste material buried below the fresh sediment become longer and make diffusion a gradually less efficient transport mechanism. This applies for the diffusion of undesired substances from the waste into the lake water as well as for the diffusion of oxygen towards the surface of the waste. Therefore, natural sedimentation usually supports and stabilises reductive conditions in the waste material and on its surface. However, when the new sediment contains substantial amounts of ferric iron, e.g. when originating from an acidic oxidised water body, the ferric iron acts as oxidising agent (see also Section 3.1).

Molecular diffusion (② in Figure 1) is usually an important mechanism of the transport of dissolved oxidising agents like oxygen or nitrate into the sediment. Likewise, reduced substances like ammonia or ferrous iron diffuse from the sediment into the lake water. The resulting net transport rates depend on the properties of the sediment, on the gradients of concentrations between pore water and lake water, on hydrodynamics. In addition, net transport rates are influenced by interactions of the transported substances along their migration path like adsorption or biogeochemical reactions. On the scale of a lake, diffusion is a very slow process.

Water flow across the interface between lake water and sediment (③ in Figure 1) is basically an exchange between lake and groundwater. It occurs in both directions, i.e. from the groundwater across the sediment into the lake as well as vice versa. It is known that such flows vary widely in space and time, including seasonal changes of directions and flow rate. Fleckenstein et al. (2009) reported flow rates of water from the groundwater into Mining Lake 77 (Germany) ranging from -2.73 $L/d \times m^2$ to 264 $L/d \times m^2$. Negative values mean flow from the lake into the groundwater. Knorr and Blodau (2006) demonstrated that flow rate, flow direction and composition of water flowing across the sediment of pit lakes are decisive for the biogeochemical processes inside the sediment and at the interface between lake water and sediment. The same applies for water flow from deposited wastes into a pit lake or vice versa. The results of Fleckenstein et al. (2009) and Knorr and Blodau (2006) can be considered as relevant for pit lakes in hard rock environments as well although they have been obtained from pit lakes embedded in unconsolidated rock in a former German lignite mine. The waste materials usually behave like porous aquifers. The risk of contamination of the lake water or the groundwater by advective transport of substances out of the waste material can be avoided if the hydraulic gradients are kept as small as possible. Moldovan et al. (2008) demonstrated that the transport of arsenic out of the tailings deposit at Rabbit Lake In-pit Tailings Management Facility (Canada) will be limited to diffusion and will not endanger the surrounding groundwater. The key factor is the construction of the tailings deposit. Of particular importance is its interface to the surrounding rock which prevents high hydraulic gradients and, thus, advective transport out of the tailings. As indicated by the results of Fleckenstein et al. (2009), sites with very different flow conditions occur within a single pit lake including areas of no exchange with groundwater.

Resuspension (④ in Figure 1) requires strong enough currents at the lake bottom. It is also influenced by the density, the grain size, and the degree of consolidation of the material to be resuspended and the slope at the actual location of the respective material. Strong enough currents occur not only in polymictic shallow lakes and in the littoral zone of seasonally stratified lakes, but also at the bottom of the hypolimnion of such lakes (Bloesch, 1995). Resuspension of already settled material is usually accompanied by changes of the biogeochemical conditions for the resuspended particles, their pore water and the material immediately below the resuspended material. This will cause remobilisation of substances, at least temporarily.

Biogeochemical reactions occur in the lake water (⑤ in Figure 1), in sediment (⑥ in Figure 1) and at the interface between lake water and deposited waste. Typical examples are the oxidation and precipitation of iron in oxic lake water and the microbial reduction and dissolution of iron precipitates. The latter includes the re-dissolution of adsorbed substances under reductive conditions in the deposited waste material and the lake sediment. Basically, all biogeochemical reactions known from lake water, lake sediments or waste deposits may occur, if favourable conditions develop.

Bioturbation (⑦ in Figure 1) is the result of the activity of motile organisms migrating through the sediment or feeding on sediment. This process is important only if the deposited waste is not of lethal toxicity for relevant organisms, e.g. benthic invertebrates. The organisms mix the sediment layers and some create tubes, which often are actively flushed with water by their inhabitants. Along the walls of such tubes additional interfaces between lake water and sediment are created. Their consequences for the mobility of substances within the sediment and for the exchange of matter between lake water and sediment have to be considered (Lewandowski and Hupfer, 2005).

Ebullition, i.e. the formation of gas bubbles as a result of microbial activity (⑧ in Figure 1), is relevant especially in deposits of oil sand tailings, where substantial methane production has been observed (e.g. Penner and Foght, 2010; BGC Engineering, 2010). In the other waste materials mentioned above, the content of easily degradable organic substances is usually not high enough for a net microbial gas production exceeding the solubility of the respective gas. An exception was described by Asmussen and Strauch (1998). They reported degassing from the interface between a mixed deposit of municipal and industrial waste and Lake Hufeisensee, a pit lake in the Central German lignite mining district.

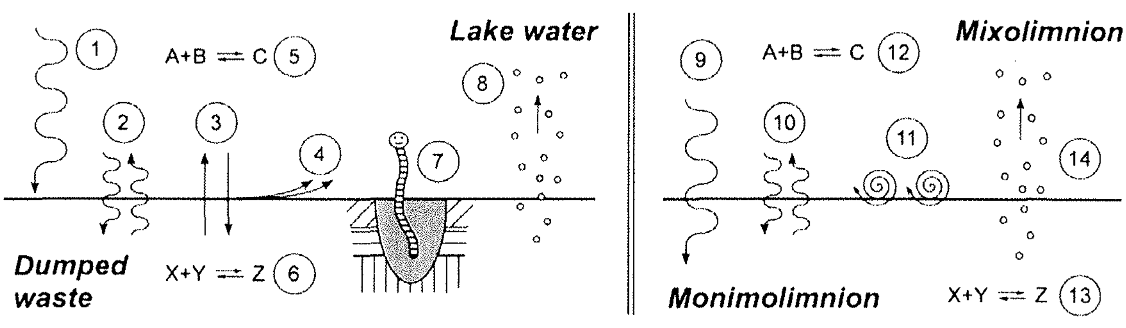


**Figure 1 Main processes relevant for exchange between water body and dumped material (left panel) and between monimolimnion and mixolimnion (right panel): 1+9 – disposal of waste and natural sedimentation, 2+10 – diffusion, 3 – water flow, 4 – resuspension, 5+6+12+13 – biogeochemical reactions, 7 – bioturbation, 8+14 – ebullition, 11 – local turbulent mixing at the chemocline**

## 2.2 Transport through the water column; stratification and meromixis

Released from the sediment, only gas bubbles can escape directly, while being subject to exchange with the water column during the ascent, until the bubbles finally enter the atmosphere (McGinnis et al., 2006). Other solutes are confined to the adjacent thin water layer above the sediment. During periods of density stratification (Boehrer and Schultze, 2008), currents carry these substances away preferably in the horizontal direction. Only turbulent mixing induced by bottom friction forwards these substances vertically into the main body of the lake at a slow pace. Reactive substances will affect local water quality, while being subject to chemical transformation themselves. During the cold season, usually a full overturn takes place which homogenises the entire lake and hence distributes all solutes within the water body.

Not all lakes experience a full overturn at some time during the annual cycle. Some contain a water layer of higher mineralisation in the deepest regions of the lake basin. If sufficiently high, the density of these waters prevents the deep recirculation from including these bottom waters. A chemically different water body can develop called monimolimnion. Such lakes are referred to as meromictic lakes. Mine pit lakes in particular, tend to develop meromixis, more than lakes of natural origin (Boehrer and Schultze, 2006).

Both external forcing as well as internal biogeochemical processes have been claimed to be responsible for meromixis (Boehrer and Schultze, 2008). Especially the presence of iron has proved to be an effective precondition to keep lakes density stratified perennially (Hongve, 1997). Groundwater inflow substances released from the sediment, and decomposing organic materials are the main contributors to the density stratification of a monimolimnion.

For two reasons, meromictic lakes seem to be better suited for waste disposal than other lakes. Firstly, the full overturn in the cold season is not including the monimolimnion. Hence the most effective transport mechanism is missing in these lakes. Secondly, the higher density stratification in the deep water also limits the vertical turbulent transport (von Rohden and Ilmberger, 2001). A meromictic lake has the potential to keep undesirable substances confined to deep waters. Thereby the environmental hazard does not appear at the lake surface, where human beings, land animals and many of the aquatic organisms would be directly affected. In addition, the different chemical milieu in the monimolimnion (e.g. anoxia) can be used for remediation strategies, such as reducing sulphate, or using produced sulphide for removing metal ions from the water column (see following Island Copper Mine). As a consequence, meromixis has been considered as a possible deposition strategy.

Stottmeister et al. (2010) used the meromictic condition of a heavily polluted mine lake. They implemented a remediation strategy mainly in the surface waters, while the deep waters kept a much higher concentration of organic pollutants. The successful outcome is a good indication, that the stratification and circulation pattern of a lake could well be used for remediation. Another pit lake, Island Copper Mine Lake, was even designed to become meromictic. High concentrations of metals could be confined in the deep water for later treatment, while keeping the surface waters in an environmentally acceptable condition (see following).

Stevens and Lawrence (1997) have already pointed out that mixing and vertical transport out of a monimolimnion can be higher than molecular diffusion by orders of magnitude. Von Rohden et al. (2009) showed that the seasonal oscillation of the chemocline depth pumps monimolimnion water into the mixolimnion, including the substances (e.g. tracers) contained within the amplitude of the oscillation. Also at high gradients of density there is a measurable turbulent transport in the vertical direction. Furthermore, transport in the seemingly permanently stratified monimolimnion can be accomplished by double diffusion (Schmid et al., 2004). Convection is driven in layers of a thickness of one or several decimetres by the increasing temperature towards the bottom under the monimolimnion. Between these layers high density gradients are found. Previously only reported in tropical lakes, this phenomenon has been observed in meromictic lakes of the temperate climate zone as well (Sánchez España et al., 2009; von Rohden et al., 2010). At chemoclines, also chemical reactions can control the double diffusive convection, and much thicker convection layers are formed by the changing boundary conditions and the reactivity of the dissolved substances. A complete monimolimnetic overturn was observed in the meromictic pit lake Waldsee (Boehrer et al., 2009). Finally also the risk of catastrophic events must be taken into account when a disposal of waste in or below a monimolimnion is considered. Extremely cold or extremely windy winters can cause stronger erosion of the monimolimnion than in usual years. In these cases, an unexpectedly large amount of dissolved substances could be delivered from the monimolimnion into the upper water body.

In conclusion, it can be reported that higher density stratification inhibits the vertical transport of dissolved substances, and such conditions can be favoured, if material is disposed of in mine lakes. However, there is always a certain flux of solutes from the deep water. Prognostications of these transports have been attempted (Stevens and Lawrence, 1998; Böhrer et al., 1998; Moreira et al., 2011). However only under favourable conditions, a reliable and accurate answer can be given. This fact must be kept in mind when disposing material of below a monimolimnion.

The right panel of Figure 1 summarises the most relevant processes at the chemocline of a meromictic lake. Except bioturbation, the interaction between monimolimnion and the dumped waste is influenced by the same processes as described in section 2.1. More detailed descriptions of the processes briefly summarised in

Figure 1 can be found in Salomons and Förstner (1988), Stumm and Morgan (1996), Boudreau (1996), Boudreau and Jørgennsen (2001), Jacobs and Förstner (2001), Hakanson (2007), and Boehrer and Schultze (2008).

# 3 Examples

## 3.1 Disposal of sludge of biological treatment of AMD

The stimulation of microbial alkalinity production by addition of suitable organic substrates basically can be used to neutralise acidic lakes. This was tested in a number of experiments on different scales (Brugam and Stahl, 2000; McCullough et al., 2008). Our initial approach was to establish a layer for iron and sulphate reduction by addition of organic substances and straw on the sediment surface of an acid pit lake. This approach failed under field conditions in an enclosure (volume (V): 4,242 $m^3$, area (A): 707 $m^2$, maximum depth ($z_{max}$): 6 m; Koschorreck et al., 2007; Geller et al., 2009). The main reasons were the seasonal reoxidation of the sediment surface and the formation of a new oxidised sediment layer above a created reductive layer where sulphate reduction took place. In addition to dissolved oxygen, ferric iron was found to be an important oxidant under the given conditions.

Floating reactors for iron and sulphate reduction were our second approach for neutralising acid pit lakes (Preuß et al., 2007). The resulting sludge of precipitated iron sulphide was planned to be deposited at the lake bottom within an enclosure. The water body inside the enclosure should be made anoxic by repeated addition of whey powder allowing for complete oxygen depletion and iron reduction. This approach also failed (Friese et al., 2010; Koschorreck et al., 2011) because ferric iron within the enclosure was not reduced (about 131 mg/L). The failure of both approaches clearly indicates that it is hardly possible to sustain reduced conditions at the interface between lake water and sediment in a holomictic acidic pit lake.

## 3.2 Creation of a pit lake as remediation strategy for a deposit of wastes

The void of the former lignite mine Großkayna (Central German lignite mining district) was used for the deposition of industrial wastes from 1969 to 1995. The wastes mainly consisted of ashes from lignite combustion in power plants and from production of synthetic gas. However, also waste materials from the production of nitrogen fertiliser and from other chemical productions were deposited among the ashes. The major characteristics of the pore water inside the waste deposit are (median; Schroeter, 1997): pH 10, 1,860 mg/L $SO_4^{2-}$, 495 mg/L $Cl^-$, 625 mg/L Ca, 10 mg/L Mg, 420 mg/L Na, 125 mg/L K, 360 mg/L $NH_4^+$. Differing from initial plans, only the lower part (V: $24.5\times10^6$ $m^3$) of the void (V: $88.5\times10^6$ m) was filled with waste. A pit lake (Lake Runstedt; V: $54\times10^6$ $m^3$, A: 2.33 $km^2$, $z_{max}$: 33 m) was established on top of the waste material by deviating water of river Saale from May 2001 to December 2002. Model simulations showed that this lake has only inflows but no outflow if neighbouring pit lakes will be held on certain levels. The only loss of water is evaporation. In this way, the transport of fluids from the waste into the groundwater is prevented (Schroeter, 1997; Fritz et al., 2001).

The major concern regarding water quality results from the high ammonia concentration in the pore water of the waste. In order to manage the oxygen consumption accompanying the import of ammonia into the lake, three hypolimnetic aerators are operated in the lake (Fritz et al., 2001), mainly during seasonal stratification in summer and during ice cover in winter. Till now, the system works well. Monitoring indicates that denitrification established naturally in stands of emerged macrophytes in the littoral avoiding enrichment of nitrate in the lake water. This case indicates that pit lakes can be used even for the disposal of industrial wastes under certain circumstances.

Creation of water capped lakes or end pit lakes (distinguished by the amount of tailings at the bottom) is also a strategy for permanent storage of mature fine tailings (MFT) from oil sand processing (MacKinnon, 1989; Johnson and Miyanishi, 2008; BGC Engineering, 2010). Both fresh and process-affected waters are used for capping. One function of such lakes is the passive bioremediation of toxic chemicals such as polycyclic aromatic hydrocarbons (PAHs) and naphthenic acids. BGC Engineering (2010) stated that mixing between the MFT and the overlying water cap can be prevented by a sufficient depth of the water layer. Moreover, the lake must not recharge aquifers that are in contact with other sensitive water bodies. A commercial scale demonstration lake is currently being planned. However, regulators have not yet approved this concept, and there are uncertainties how microbial metabolism and gas production will affect long-term water quality (BGC Engineering, 2010). For example, Fedorak et al. (2003) observed enhanced densification rates and dewatering of the MFT in methanogenic mesocosms, due to microbial activity and methane production. Thereby some compounds of the diluent naphtha (applied for oil sands processing) supports $CH_4$ biogenesis (Siddique et al., 2007). However, accelerated densification of fine particles seem to overcompensate the hindering of sedimentation of fine tailings by gas bubbles and can give implications for tailings reclamation and management.

## 3.3 (Re-) Use of deposited wastes to improve water quality of pit lakes

Some mine voids of former lignite mines in the Lusatian mining district of Germany were used as disposal sites for waste lime sludge, ash from power plants and sludge of a liming based mine water treatment plant.

Lake Geierswald (V: $98\times10^6$ m$^3$, A: 6.42 km$^2$) was partially neutralised during five months in 2004 and 2005 by re-excavation and spreading of waste lime formerly deposited at the lake bottom (Benthaus and Uhlmann, 2006). About 500,000 m$^3$ of a 1.9% lime suspension were re-excavated, corresponding to 10,000 t of lime. The treatment raised the pH from 3.0 to 3.3 and reduced the acidity of the lake water from 1.6 to 0.9 mmoleq/L.

Sludge of the mine water treatment plant Schwarze Pumpe has been dumped in Lake Spreetal (V: $97\times10^6$ m$^3$, A: 3.14 km$^2$). The sludge contributed about 30% to the overall improvement of the lake water until the end of 2006 (Uhlmann et al., 2007). This neutralising activity of the sludge was based on remaining reactive lime. The disposal of the sludge is still in operation. Unger-Lindig et al. (2010) found that addition of $CO_2$ could even improve the alkalinity gain for the lake water from added sludge by formation of bicarbonate.

About $26\times10^6$ m$^3$ of fly ash was deposited in the former mine Burghammer (Koch C. et al., 2008). Rising groundwater formed Lake Bernstein (V: $35\times10^6$ m$^3$, A: 4.82 km$^2$) in the remaining space of the former mine void. The use of the ash deposits for lake neutralisation was tested successfully by re-suspension and re-distribution of a small portion of the ash (Koch C. et al., 2008; Koch T. et al., 2008). The full neutralisation of the lake water would require resuspension of about 1% of the deposited ash. Also in this case, the addition of $CO_2$ considerably improved the reactivity of the ash and lead to a higher gain of bicarbonate in the lake water (Koch C. et al., 2008; Koch T. et al., 2008).

## 3.4 South Mine pit lake in the Tennessee copper basin

South Mine pit lake is used as disposal site for sludge from a treatment plant neutralising the North Potato Creek (mean flow 31 m$^3$/min, mean pH 5, 0.5 meq/L acidity, 10 mg/L iron at the treatment site). The creek is a tributary of the Ocoee River, Tennessee, and drains a major part of the Copper Basin in Tennessee (Faulkner et al., 2005; Wyatt et al., 2006). The South Mine pit lake (550 m long, 146 m wide, 61 m deep) is located close to the confluence of the North Potato Creek and the Ocoee River, provides the potential use of the lake for treatment purposes. Since the pit lake is meromictic, it also serves as "provider" of iron as an additional flocculant for the treatment plant: Water from the monimolimnion (average: pH 4.7, 20 meq/L acidity, 600 mg/L iron) is added to the mixing tank of the treatment plant together with lime in order to achieve better removal of metals. The initially acidic mixolimnion of the pit lake (average: pH 3.4, 0.75 meq/L acidity, 3.7 mg/L iron) became neutral within a month after the start of the treatment system (Wyatt et al., 2006). Model simulations proved the stability of the permanent stratification of the lake (Colarusso et al., 2003). An essential step of the overall treatment procedure is the re-dissolution of part of the iron under the reducing conditions in the monimolimnion, i.e. the monimolimnion is not simply used as a safe disposal site but also as a reactor.

## 3.5 Island Copper Mine pit lake

The pit lake in the former Island Copper Mine (Vancouver Island, Canada, V: $241\times10^6$ m$^3$, A: 1.73 km$^2$, $z_{max}$: 350 m; Pelletier et al., 2009) is the only case of intentional artificial meromixis known to the authors: a facility for storage and treatment of acid rock drainage (ARD) as well as a disposal facility for sludge resulting from treatment (Dagenais and Poling, 1997; Fisher and Lawrence, 2006). Additionally, one of the ARD producing waste rock dumps was relocated into the pit before lake flooding (Dagenais and Poling, 1997). Flooding was mainly accomplished with sea water capped by an 8 m thin fresh water layer. This surface layer is recharged by rain, local runoff and circumneutral waste rock drainage and has an outflow into the ground water (Pelletier et al., 2009). Moderately acidic ARD is introduced into the pit lake at 220 m depth. As a result of the flooding and operation of the lake, three layers exist in the lake: brackish water on top (4–6 psu; currently ca. 4 m thick), sea water diluted by ARD as main layer below (ca. 4 m to 220 m depth) and remaining sea water at the bottom (lowermost ca. 130 m) (Pelletier et al., 2009). The deepest part of the lake is acting as disposal site for precipitating metal sulphides formed in the anoxic middle layer and for detritus from the top layer. In order to keep the metal concentrations below given thresholds, the top layer is fertilised with phosphorus and nitrogen. Stimulation of algal growth by fertilisation as measure for metal removal from water in pit lakes was found to work well in experimental approaches (Martin et al., 2003; Poling et al., 2003) and also in practice in the Island Copper Mine pit lakes if done weekly and year-round (Pelletier et al., 2009).

Although no detailed results on the behaviour of the sediments have been published yet, the Island Copper Mine lake can be considered a successful example of a combined treatment and disposal facility. The major factor for success is the stability of the meromixis. The permanent flushing of the brackish layer at the lake surface is essential for the stability of the chemical stratification. It keeps the density gradient steep enough to withstand wind forcing. Otherwise it would be weakened over time by intrusion of salt from the middle layer via diffusion and small scale turbulent mixing at the chemocline and by gradual dilution of the middle layer by the introduction of ARD.

# 4 Discussion and conclusions

The above mentioned examples of subaqueous disposal of waste from mining show that this approach is a good choice under certain circumstances. The risks are also clearly demonstrated by the examples and the cited literature. The major prerequisite is the stability of biogeochemical conditions inside the deposited material and at its interface with surrounding water bodies.

Capping, i.e. the construction of a cover layer consisting of solid material on top of the waste, results in longer diffusion paths, eventually decreasing the diffusive transport rate to practically zero. Additionally, resuspension of the waste is avoided (Jacobs and Förstner, 2001). Experiences in prediction of behaviour, planning and construction of subaqueous capping of disposal sites for dredged material are available (Jacobs and Förstner, 2001; Alshawabkeh et al., 2005; Eek et al., 2007). Therefore, capping is a potential measure for the management of existing waste deposits in pit lakes.

Meromictic conditions have similar effects. However, establishing meromixis artificially is possible only under very special conditions (Schultze and Boehrer, 2009). All necessary measures for establishing and sustaining meromixis must not have adverse impacts on the mixolimnion or the ground water. Using saline water for filling pit lakes analogously to the use of sea water in the case of the Island Copper Mine pit lake does not allow for hydraulic connections between the pit lake and surrounding groundwater. If there are any pathways, the saline water will flow into the groundwater, driven by its high density, and cause unacceptable high salt concentrations there. However, mine water often has high density due to its content of dissolved solids and may cause meromixis in lakes. In almost all meromictic pit lakes this effect is one reason for the occurrence of meromixis. Even natural lakes are sometimes turned meromictic by ARD (Moncur et al., 2006). However, the minerals causing the high density of ARD are subject to long term washout. This results in a gradual decrease of the density of the monimolimnetic water, i.e. a potential destabilisation of meromixis, if the recharge of the monimolimnion by ARD is essential.

In addition to the risks resulting from potential transfer of hazardous substances from waste into lake water or groundwater, one may argue that the installation of devices for monitoring and the implementation of measures for control is complicated once the waste is deposited at the bottom of a pit lake. Basically, lake water and lake sediments are much easier accessible than groundwater. However, installation of instruments inside the waste deposit is usually much simpler without a water cover above the waste, e.g. from a dry and stable surface of a dewatered tailings pond. Such disadvantages have to be evaluated in comparison with alternative disposal options and their risks under the locally given conditions. Potential risks of alternative disposal options, e.g. tailings ponds or dams, are associated with instability of the dams and of the sealing layer (if present) in the long term. Contamination of groundwater is likely if the sealing at the base is not strong enough to withstand the hydraulic head. In the case of ongoing oxidation of sulphide minerals in the deposited material followed by ARD formation, this is especially problematic.

In conclusion, every particular case of potential disposal of waste materials in pit lakes requires careful site-specific evaluation. Many experiences and methods are available for risk assessment. A general preclusion of this approach is not reasonable from a scientific point of view.

## References

Alshawabkeh, A.N., Rahbar, N. and Sheahan, T. (2005) A model for contaminant mass flux in capped sediment under consolidation, Journal of Contaminant Hydrology 78, pp. 147–165.

Amyot, G. and Vezina, S. (1997) Flooding as a reclamation solution to an acidic tailings pond – the Sobec case, Proceedings 4th International Conference on Acid Rock Drainage, 31 May–6 June, Vancouver, Canada, pp. 681–696.

Asmussen, G. and Strauch, G. (1998) Sulfate reduction in a lake and the groundwater of a former lignite mining area studied by stable sulfur and carbon isotopes, Water, Air and Soil Pollution 108, pp. 271–284.

Benthaus, F.C. and Uhlmann, W. (2006) Die chemische Behandlung saurer Tagebauseen in der Lausitz – Erfahrungen zur Kalkschlammresuspension im Tagebausee Koschen, Wissenschaftliche Mitteilungen des Institutes für Geologie Nr. 31/2006, B. Merkel, H. Schaeben, C. Wolkersdorfer, A. Hasche-Berger (eds), Technische Universität Bergakademie Freiberg, Freiberg, pp. 85–96.

BGC Engineering Inc. (2010) Oil Sands Tailings Technology Review, OSRIN Report No. TR-1, Oil Sands Research and Information Network, University of Alberta, School of Energy and the Environment, Edmonton, 136 p.

Bloesch, J. (1995) Mechanisms, measurement and importance of sediment resuspension in lakes, Marine and Freshwater Research 46, pp. 295–304.

Boehrer, B. and Schultze, M. (2006) On the relevance of meromixis in mine pit lakes, in Proceedings 7th ICARD, R.I. Barnhisel (ed), 7th International Conference on Acid Rock Drainage (ICARD), 26–30 March, St. Louis, USA, American Society of Mining and Reclamation (ASMR), Lexington, pp. 200–213.

Boehrer, B. and Schultze, M. (2008) Stratification of lakes, Reviews in Geophysics 46, RG2005, doi:10.1029/2006RG000210, 27 p.

Boehrer, B., Dietz, S., von Rohden, C., Kiwel, U., Jöhnk, K.D., Naujoks, S., Ilmberger, J. and Lessmann, D. (2009) Double-diffusive deep water circulation in an iron-meromictic lake, Geochemistry, Geophysics and Geosystems 10, Q06006, doi:10.1029/2009GC002389.

Böhrer, B., Heidenreich, H., Schimmele, M. and Schultze, M. (1998) Numerical prognosis for salinity profiles of future lakes in the opencast mine Merseburg-Ost, International Journal of Salt Lake Research 7, pp. 235–260.

Boudreau, B. (1996) Diagenic models and their implementation, Springer, Berlin, 414 p.

Boudreau, B. and Jørgennsen, B.B. (2001) The benthic boundary layer – Transport processes and biogeochemistry. Oxford, Oxford University Press, 404 p.

Brugam, R.B. and Stahl, J.B. (2000) The potential of organic matter addition for neutralizing surface mine lakes, Transactions of the Illinois State Academy of Science 93, pp. 127–144.

Colarusso, L.A., Chermak, J.A., Priscu, J.C. and Miller, F.K. (2003) Modeling pit lake water column stability using Ce-Qual-W2, Tailings and Mine Waste '03, Swets & Zeitlinger, Lisse, pp. 213–222.

Dagenais, P.J. and Poling, G.W. (1997) An investigation into the geochemical history of waste rock dump and its effect on water quality of the flooded open pit at Island Copper Mine, Port Hardy, British Columbia, in Proceedings Fourth International Conference on Acid Rock Drainage, 31 May–6 June, Vancouver, Canada, pp. 1709–1725.

Eek, E., Godoy, O., Aagaard, P. and Breedveld, G.D. (2007) Experimental determination of efficiency of capping materials during consolidation of metal-contaminated dredged material, Chemosphere 69, pp. 719–728.

Ellis, D.V. and Robertson, J.D. (1999) Underwater placement of mine tailings: Case examples and principles, Environmental impacts of mining activities, Azcue, J.M. (ed), Springer, Berlin, Germany, pp. 123–141.

Faulkner, B.B., Wyatt, G., Chermak, J.A. and Miller, F.K. (2005) The Largest Acid Mine Drainage Treatment Plant In The World? Proceedings 26th Annual West Virginia Surface Mine Drainage Task Force Symposium, 19 April, Morgantown, USA, viewed 18 February 2009, http://wvmdtaskforce.com/proceedings/05/Faulkner.pdf.

Fedorak, P.M., Coy, D.L-, Dudas, M.J., Simpson, M.J., Renneberg, A.J. and MacKinnon, M.D. (2003) Microbially-mediated fugitive gas production from oil sands tailings and increased tailings densification rates, Journal of Environmental Engineering and Science 2, pp. 199–211.

Fisher, T.S.R. and Lawrence, G.A. (2006) Treatment of Acid Rock Drainage in a meromictic mine pit lake, Journal of Environmental Engineering 132, pp. 515–526.

Fleckenstein, J., Neumann, C., Volze, N. and Beer, J. (2009) Raumzeitmuster des See-Grundwasser-Austausches in einem sauren Tagebaurestsee, Grundwasser 14, pp. 207–217.

Friese, K., Koschorreck, M., Wendt-Potthoff, K., Schultze, M. and Geller, W. (2010) Application of whey to prevent re-oxidation in a passive anoxic in-lake reactor – success and failure, Proceedings Mine Water and Innovative

Thinking, C. Wolkersdorfer, A. Freund (eds), IMWA Symposium 2010, 5–9 September, Sydney, Canada, Cape Breton University Press, Sydney, pp. 225–228.

Fritz, W., Tropp, P. and Meltzer, A. (2001) A Remediation and reclamation strategy for disused brown coal mines in the Geiseltal area, Surface Mining – Braunkohle & Other Minerals 53, pp. 155–166.

Garnett, R.T.H. and Ellis, D.V. (1995) Tailings disposal at a marine placer mining operation by WestGold, Alaska, Marine Georesources and Geotechnology 13, pp. 41–57.

Geller, W., Koschorreck, M., Wendt-Potthoff, K., Bozau, E., Herzsprung, P., Büttner, O. and Schultze, M. (2009) A pilot-scale field experiment for the microbial neutralization of a holomictic acidic pit lake, Journal of Geochemical Exploration 100, pp. 153–159.

Hakanson, L. (2007) Lake environments, in Environmental sedimentology, Perry, C. and Taylor, K. (eds), Blackwell Publishing, Malden, pp. 109–143.

Hongve, D. (1997) Cycling of iron, manganese, and phosphate in a meromictic lake, Limnology and Oceanography 42, pp. 635–647.

Jacobs, P. and Förstner, U. (2001) Managing contaminated sediments. IV. Subaqueous storage and capping of dredged material. Journal of Soils and Sediment 1, pp. 205–212.

Johnson, E.A. and Miyanishi, K. (2008) Creating new landscapes and ecosystems – The Alberta oil sands, Annals of the New York Acadamy of Science 1134, pp. 120–145.

Jones, S.G. and Ellis, D.V. (1995) Deep water STD at the Misima gold and silver mine, Papua, New Guinea, Marine Georesources and Geotechnology 13, pp. 183–200.

Klapper, H. and Schultze, M. (1995) Geogenically acidified mining lakes – living conditions and possibilities of restoration, International Review of Hydrobiology 80, pp. 639–653.

Knorr, K.-H. and Blodau, C. (2006) Experimentally altered groundwater inflow remobilizes acidity from sediments of an iron rich and acidic lake, Environmental Science and Technology 40, pp. 47–59.

Koch, C., Graupner, B. and Werner, F. (2008) Pilotversuch zur Wasserbehandlung im Tagebausee Burghammer unter Verwendung von $CO_2$ und Braunkohlenfilterasche, Wissenschaftliche Mitteilungen des Institutes für Geologie Nr. 37/2008, B. Merkel, H. Schaeben, A. Hasche-Berger (eds), Technische Universität Bergakademie Freiberg, Freiberg, pp. 31–38.

Koch, T., Preuß, V., Schöpke, R. and Koch, R. (2008) Pilotversuch zur kombinierten Förderung und Aktivierung alkalischer Sedimente mittels Druckluftheber (Airlift) und $CO_2$-Einsatz, Wissenschaftliche Mitteilungen des Institutes für Geologie Nr. 37/2008, B. Merkel, H. Schaeben, A. Hasche-Berger (eds), Technische Universität Bergakademie Freiberg, Freiberg, pp. 39–46.

Koschorreck, M., Boehrer, B., Friese, K., Geller, W., Schultze, M. and Wendt-Potthoff, K. (2011) Oxygen depletion induced by adding whey to an enclosure in an acidic mine pit lake, submitted to Ecological Engineering.

Koschorreck, M., Bozau, E., Frömmichen, R., Geller, W., Herzsprung, P. and Wendt-Potthoff, K. (2007) Processes at the sediment water interface after addition of organic matter and lime to an acid mine pit lake mesocosm, Environmental Science and Technology 41, pp. 1608–1614.

Lewandowski, J. and Hupfer, M. (2005) Effect of macrozoobenthos on two-dimensional small-scale heterogeneity of pore water phosphorus concentrations in lake sediments: A laboratory study, Limnology and Oceanography 50, pp. 1106–1118.

MacKinnon, M.D. (1989) Development of the tailings pond at Syncrude's oil sand plant: 1978–1987, AOSTRA Journal of Research 5, pp. 109–133.

Martin, A.J. and Pedersen, T.F. (2004) Alteration to lake trophic status as a means to control arsenic mobility in a mine-impacted lake, Water Research 38, pp. 4415–4423.

Martin, A.J., Crusius, J., McNee, J.J., Whittle, P., Pieters, R. and Pedersen, T.F. (2003) Field-scale assessment of bioremediation strategies for two pit lakes using limnocorrals, in Proceedings 6th International Conference on Acid Rock Drainage, 14–17 July, Cairns, Australia, T. Farell, G. Taylor (eds), The Australian Institute of Mining and Metallurgy, Carlton Victoria, pp. 529–539.

McCullough, C., Lund, M.A. and May, J.M. (2008) Field-scale demonstration of the potential for sewage to remediate acidic mine waters, Mine Water and the Environment 27, pp. 31–39.

McGinnis, D.F., Greinert, J., Artemov, Y., Beaubien, S.E. and Wüest, A. (2006) The fate of rising methane bubbles in stratified waters: How much methane reaches the atmosphere?, Journal of Geophysical Research, 111 (C09007), doi:10.1029/2005JC003183.

Miller, G.C., Lyons, W.B. and Davis, A. (1996) Understanding the water quality of pit lakes, Environmental Science and Technology 30, pp. 118A–123A.

Moldovan, B.J., Hendry, M.J. and Harrington, G.A. (2008) The arsenic source term for an in-pit uranium mine tailings facility and its long-term impact on the regional groundwater, Applied Geochemistry 23, pp. 1437–1450.

Moncur, M.C., Ptacek, C.J., Blowes, D.W. and Jambor, J.L. (2006) Spatial variations in water composition at a northern Canadian lake impacted by mine drainage, Applied Geochemistry 21, pp 1799–1817.

Moreira, S., Boehrer, B., Schultze, M., Dietz, S. and Samper, X. (2011) Modelling geochemically caused permanent stratification in Lake Waldsee, Aquatic Geochemistry 17, pp 265–280.

Pedersen, T.F., McNee, J.J., Flather, D.H., Mueller, B. and Pelletier, C.A. (1998) Geochemical behaviour of submerged pyrite-rich tailings in Canadian lakes, Acidic mining lakes, Geller, W., Klapper, H., Salomons, W. (eds), Springer, Berlin, pp. 87–125.

Pelletier, C.A., Wen, M.E. and Poling, G.W. (2009) Flooding pit lakes with surface water, Mine pit lakes – Characteristics, predictive modeling and sustainability, D.N. Castendyk, L.E. Eary (eds), Society for Mining, Metallurgy and Exploration, Littleton, pp. 239–248.

Penner, T.J. and Foght, J.M. (2010) Mature fine tailings from oil sand processing harbour diverse methanogenic communities, Canadian Journal of Microbiology 56, pp. 459–470.

Poling, G.W. and Ellis, D.V. (1995) Importance of geochemistry: The Black Angel lead-zinc mine, Greenland, Marine Georesources and Geotechnology 13, pp. 101–118.

Poling, G.W., Ellis, D.V., Murray, J.W., Parsons, T.R. and Pelletier, C.A. (2002) Underwater tailings placement at Island Copper Mine – A success story, Society for Mining, Metallurgy, and Exploration, Littleton, USA, 204 p.

Poling, G.W., Pelletier, C.A., Muggli, D., Gerits, J., Hanks, C. and Black, K. (2003) Field studies of semi-passive biogeochemical treatment of acid rock drainage at the Island Copper Mine pit lake, Proceedings 6th International Conference on Acid Rock Drainage, 14–17 July, Cairns, Australia, T. Farell, G. Taylor (eds), The Australian Institute of Mining and Metallurgy, Carlton Victoria, pp. 549–558.

Preuß, V., Horn, M., Koschorreck, M., Luther, G., Wendt-Potthoff, K. and Geller, W. (2007) In-lake bioreactors for the treatment of acid mine water in pit lakes, Advanced Materials Research 20–21, pp. 271–274.

Salomons, W. and Förstner, U. (eds) (1988) Chemistry and biology of solid waste – Dredged material and mine tailings, Springer-Verlag, Berlin, 305 p.

Sánchez España, J., López Pamo, E., Diez, M. and Santofimia, E. (2009) Physico-chemical gradients and meromictic stratification in Cueva de la Mora and other acidic pit lakes of the Iberian Pyrite Belt, Mine Water and the Environment 28, pp. 15–29.

Schmid, M., Lorke, A., Dinkel, C., Tanyileke, G. and Wüest, A. (2004) Double-diffusive convection in Lake Nyos, Cameroon, Deep-Sea Research I 51, pp. 1097–1111.

Schroeter, A. (1997) Modellhafte Sanierung von Altlasten am Beispiel des Tagebaurestloches Großkayna, Umweltbundesamt, Berlin, 723 p.

Schultze, M. and Boehrer, B. (2009) Induced meromixis, Mine pit lakes – Characteristics, predictive modeling and sustainability, D.N. Castendyk, L.E. Eary (eds), Society for Mining, Metallurgy and Exploration, Littleton, pp. 239–248.

Schultze, M., Friese, K., Sánchez, J., Santofimia, E. and López, E. (2008) The Aznalcóllar pit lake – water quality and options of remediation, Investigación y gestión de los recurso del subsuelo. Libro homenaje al Profesor Fernando Pendás Fernández, Publicaciones del Instituto Geológico y Minero de España, Series Hidrogeologia y aguas subterraneas no 27, J.A. López-Geta, J. Loredo Pérez, L. Fernández Ruiz, J.M. Pernía Llera (eds), Instituto Geológico y Minero de España, Madrid, pp. 853–863.

Siddique, T., Fedorak, P.M., McKinnon, M.D. and Foght, J.M. (2007) Metabolism of BTEX and naphtha compounds to methane in oil sands tailings, Environmental Science and Technology 41, pp. 2350–2356.

Stevens, C.L. and G.A. Lawrence (1997) The effect of subaqueous disposal of tailings in standing waters, IAHR Journal of Hydraulic Research 35, pp. 147–159.

Stottmeister, U., Kuschk, P. and Wiessner, A. (2010) Full-scale bioremediation and long-term monitoring of a phenolic wastewater disposal lake, Pure and Applied Chemistry 82, pp. 161–173.

Stumm, W. and Morgan, J.J. (1996) Aquatic chemistry, John Wiley & Sons, New York, 1022 p.

Stump Jr., D.E. (2001) Grants Branch Lake: A successful experiment, Tailings and Mine Waste '01, Balkema, Rotterdam, pp. 435–440.

Uhlmann, W., Gröschke, A., Arnold, I. and Rolland, W. (2007) Wirkung von Eisenhydroxidschlämmen aus der Grubenwasseraufbereitung auf die Gewässerchemie von Tagebauseen in der Lausitz, Management bergbaubedingten Säurepotentials, C. Drebenstedt, M. Struzina (eds). Technische Universität Bergbauakademie Freiberg, Freiberg, pp. 109–121.

Unger-Lindig, Y., Merkel, B. and Schipek, M. (2010) Carbon dioxide treatment of low density sludge: a new remediation strategy for acidic mining lakes? Environmental and Earth Sciences 60, pp. 1711–1722.

von Rohden, C. and Ilmberger, J. (2001) Tracer experiment with sulfurhexafluoride to quantify the vertical transport in a meromictic pit lake, Aquatic Sciences 63, pp. 417–431.

von Rohden, C., Boehrer, B. and Ilmberger, J. (2010) Evidence for double diffusion in temperate meromictic lakes, Hydrology and Earth Systems Sciences 14, pp. 667–674.

von Rohden, C., Ilmberger, J. and Boehrer, B. (2009) Assessing groundwater coupling and vertical exchange in a meromictic mining lake with an SF6-tracer experiment, Journal of Hydrology 372, pp. 102–108.

Wyatt, G., Miller, F. and Chermak, J. (2006) Innovative water treatment plant utilizing the South Mine Pit at the Copper Basin mining site in Tennessee, USA, in Proceedings 7th ICARD, R.I. Barnhisel (ed), 7th International Conference on Acid Rock Drainage (ICARD), 26–30 March, St. Louis, USA, American Society of Mining and Reclamation (ASMR), Lexington, pp. 2529–2539.