# The filling and remediation of pit lakes in former open cast lignite mines

Von der

Fakultät Architektur, Bauingenieurwesen und Umweltwissenschaften der Technischen Universität Carolo-Wilhelmina zu Braunschweig

> zur Erlangung des Grades eines Doktors der Naturwissenschaften (Dr. rer. nat.) genehmigte

#### Dissertation

von Martin Schultze geboren am 07. Juni 1958 aus Treuenbrietzen

Eingereicht am Disputation am 19. Januar 2012 18. April 2012

Berichterstatter/in

Prof. Dr. Antje Schwalb PD Dr. Burkhard Scharf

2012

### Summary

Pit lakes are important features of post-mining landscapes if surface mining was done. This doctoral thesis is focused on pit lakes in former lignite open cast mines. However, selected aspects of pit lakes in former surface hard coal and metal mines are included.

The main concern of water quality in pit lakes is acidification caused by pyrite oxidation. Eutrophication, contamination with trace contaminants originating from waste deposits and former industrial sites, salinization and contamination with pathogens may also affect aquatic live in pit lakes and the use of pit lakes by humans.

The filling of pit lakes with river water is an approach for the abatement of acidification that has been applied particularly in Germany. This thesis represents the first detailed quantification of the contribution of diverse processes to the neutralization and, in this way, to the ultimate water quality in pit lakes. In addition, the use of river water for the filling and management of pit lakes is evaluated and compared with other approaches. Such a comprehensive comparison and evaluation has not been available yet.

The regular filling of Lake Goitsche (May 1999-July 2002) with water of river Mulde serves as an example for the detailed quantification of the contribution of diverse processes to the neutralization. Lake Goitsche is located near Bitterfeld in the Central German lignite mining district. The alkalinity of the river water was the main factor for the neutralization of the partially acid lake water (minimum pH 2.8). An important part of the acidity to be neutralized entered the lake water even during the filling and neutralization. The most important process for eventual concentrations of chloride, sulphate, calcium and other fully or, at least mainly, conservative substances was the dilution by river water. The ultimate concentrations of iron and aluminium were governed by the precipitation of their hydroxides during neutralization. The accompanying co-precipitation was the critical process for the final very low concentrations of trace metals and phosphorus. Therefore, Lake Goitsche was oligotrophic to mesotrophic at the end of its regular filling. Nitrate was imported by the river water into Lake Goitsche.

The comprehensive evaluation of the use of river water in comparison with alternative approaches was done by using other pit lakes of the Central German lignite mining district or from other German lignite mining districts as examples and by including pit lakes from other countries as far as documented in the international literature. The filling as well as the permanent flushing with river water was found to be a very effective, economically efficient approach that is accompanied by only few risks. The main limitation of its application is the

availability of water. In addition to the general climatically and hydrologically given availability of river water in a certain mining region, the ecological needs of the river, the existing rights of water users downstream, the water quality of the available river water and the necessary effort for the diversion of the river water are important. Therefore, the use of river water for filling and management of pit lakes has a high potential in particular in humid climates. However, the use of seasonal floods may allow for the use of river water also under dry conditions.

### Zusammenfassung

Tagebauseen sind eine wichtiger Bestandteil der Bergbaufolgelandschaft im Falle von Bergbau im Tagebaubetrieb. Die Arbeit konzentriert sich auf Tagebauseen in ehemaligen Braunkohletagebauen. Ausgewählte Aspekte von Tagebauseen in ehemaligen Tagebauen des Steinkohle- und Erzabbaus werden aber mit einbezogen.

Das Hauptproblem der Wasserqualität von Tagebauseen ist die Versauerung durch Pyritoxidation. Eutrophierung, Kontamination mit Spurenschadstoffen aus Deponien und Altlasten, Versalzung und die Kontamination mit pathogenen Keimen können ebenfalls zu Beeinträchtigungen der aquatischen Lebensgemeinschaften in Tagebauseen und der menschlichen Nutzung der Tagebauseen führen.

Die Füllung von Tagebauseen mit Flusswasser ist eine insbesondere in Deutschland praktizierte Strategie der Verhinderung und Bekämpfung der Versauerung. In der vorgelegten Dissertationsschrift wird erstmalig eine detaillierte Quantifizierung der Beiträge der verschiedenen Prozesse zur Neutralisation und damit zur letztlich erreichten Wasserqualität eines Tagebausees vorgenommen. Außerdem wird die Nutzung von Flusswasser für die Füllung und das Management von Tagebauseen in bisher nicht verfügbarer Breite mit alternativen Verfahren verglichen und bewertet.

Für die detaillierte Quantifizierung der Beiträge verschiedener Prozesse zur Neutralisation und zur entstehenden Wasserqualität diente die planmäßige Füllung des Goitschesees bei Bitterfeld im Mitteldeutschen Braunkohlerevier mit Wasser aus der Mulde von Mai 1999 bis Juli 2002 als Beispiel. Für die Neutralisation des bei Beginn der Füllung zum Teil sauren Seewassers (Minimum: pH 2,8) war die neutralisierende Wirkung der Alkalinität des Flusswassers entscheidend. Dabei wurde ein großer Teil der zu neutralisierenden Azidität erst während der Füllung in den Goitschesee eingetragen. Für die am Ende der planmäßigen Füllung des Goitschesees erreichten Konzentrationen von Chlorid, Sulfat, Kalzium und anderen ganz oder weitgehend konservativ reagierenden Stoffen war die Verdünnung durch das Flusswasser entscheidend. Die am Ende der planmäßigen Füllung erreichten Konzentrationen von Eisen und Aluminium wurden durch deren nahezu quantitative Ausfällung als Hydroxide im Zuge der Neutralisation bestimmend. Die dabei auftretende Mitfällung war der entscheidende Prozess für die entsprechenden Konzentrationen von Spurenmetallen und Phosphor. Dadurch war der Goitschesee am Ende seiner planmäßigen Füllung oligotroph bis mesotroph. Nitrat wurde mit dem Flusswasser in den Goitschesee importiert.

3

Die umfassende Bewertung der Flusswassernutzung im Vergleich zu alternativen Verfahren erfolgte am Beispiel anderer Tagebauseen im Mitteldeutschen Revier, in anderen Braunkohlerevieren Deutschlands und durch Einbeziehung von in der internationalen Fachliteratur dokumentierten Beispielen aus anderen Ländern. Die Füllung und auch die dauerhafte Spülung mit Flusswasser zeigten sich im Vergleich als sehr effektive, ökonomisch günstige und mit nur wenigen Risiken behaftete Verfahren. Die wichtigste Begrenzung seiner Anwendbarkeit ist die Verfügbarkeit von Wasser. Neben der grundsätzlichen Verfügbarkeit von Flusswasser in Tagebauregionen aufgrund der klimatischen und hydrologischen Gegebenheiten sind die ökologischen Erfordernisse des Flusses, die bestehenden Wassernutzungsrechte von Unterliegern, die Wasserqualität des verfügbaren Flusswassers und der nötige technische Aufwand für die Flusswassernutzung von Bedeutung. Entsprechend hat die Nutzung von Flusswasser für das Wasserqualitätsmanagement von Tagebauseen vor allem in gemäßigten Klimaten ein sehr großes Potential. Bei der Nutzung von saisonalen Hochwassersituationen ist aber auch eine Anwendung unter trockenen Klimabedingungen möglich.

## Acknowledgements

My supervisors, Prof. Dr. Antje Schwalb and Dr. Burkhard Scharf are the first I want to thank for their encouragement and the helpful discussion.

I want to thank the Faculty of Architecture, Construction Engineering and Environmental Science of the Technical University of Braunschweig for giving me the opportunity to submit my doctoral thesis.

I also thank the Helmholtz-Centre for Environmental Research – UFZ for giving me the opportunity to prepare my doctoral thesis using results of my research work done as employee of the UFZ. Special thanks are directed to Prof. Dr. Helmut Klapper and Dr. Kurt Friese as heads of my working group and Prof. Dr. Walter Geller and Dr. Karsten Rinke as heads of the Department Lake Research for their encouragement and their trust. Further more, I want to thank all colleagues from the UFZ in Magdeburg, in particular Dr. Katrin Wendt-Potthoff, Dr. Bertram Boehrer and Dr. Gabriele Packroff for the excellent collaboration and many fruitful discussions, Burkhard Kuehn, Karsten Rahn, Uwe Kiwel and Siegfried Frimel for their work in the field and the staff of the laboratory for chemical water analysis for their diligent laboratory work.

The Federal Ministry for Education and Research of the Federal Republic of Germany funded part of the presented research (BMBF-project numbers: 02 WB 9984/3 and PTJ 0330492).

Further funding was provided by the Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft mbH (LMBV). Additionally, the LMBV provided many data and logistic help. Special thanks I want to give to Walter Laugwitz, Eberhardt Zeh, Dr. Peter Tropp, Karl-Heinz Pokrandt, Wolfram Hille and Dr. Friedrich-Carl Benthaus for their excellent collaboration, their trust and the diverse help and information. Without their encouragement and their sharing information about mining in general as well as about special details of mining technology, mining history and local conditions, my work could not have been conducted on the level I present it here.

In two out of four articles included in this doctoral thesis, English was improved by a professional (Herzsprung et al. 2010, Schultze et al. 2010). Additional improvement of Schultze et al. (2010) resulted from the very helpful comments of Prof. Dr. Devin Castendyk regarding the English as well as the content. Dr. Clint McCullough substantially improved the English of Schultze et al. (2011) and provided also helpful comments regarding the content.

Finally, I want to thank my family for the encouragement and for the endurance.

## Contents

	List of figures					
	List of tables					
1	Intr	Introduction				
2	Properties of pit lakes					
	2.1	Basic factors influencing pit lakes	15			
	2.2	Water quality of pit lakes	16			
3	The	Central German mining district	23			
4	Lak	e Goitsche as an example for the filling with river water	28			
	4.1	Location, geological conditions, mining history, filling and morphometry	28			
	4.2	Neutralization of Lake Goitsche	31			
	4.3	Development of the concentrations of further substances	33			
	4.4	Behaviour of phosphorus during filling of Lake Goitsche and the risk of eutrophication	34			
5	Cor	nparison of different approaches for filling and remediation of pit lakes	36			
	5.1	Approaches for filling and remediation of pit lakes in Germany	36			
	5.2	International aspects	41			
	5.3	Advantages and limitations of the approaches	44			
6	Cor	Conclusions				
7	Ref	erences	49			
	App	pendix 1: Schultze et al. 2010+Schultze et al. 2011a				
	App	pendix 2: Schultze et al. 2012a				
Appendix 3: Herzsprung et al. 2010						

Appendix 4: Schultze et al. 2011b

# List of figures

Figure 1	Relation between age (as indicated by the year of filling start) and surface area for German pit lakes in former lignite mines	15
Figure 2	Scheme of the dewatering of the underground in the vicinity of German lignite surface mines and the classification of the dewatered underground in zones of different extent and duration of dewatering	17
Figure 3	Titration curves of two pit lakes in the Lusatian lignite mining district (Germany) demonstrating the buffering (grey areas) by ferric iron $(Fe^{3+})$ and by aluminium $(Al^{3+})$ under acid conditions	19
Figure 4	Typical ranges for concentrations of the usually most relevant substances in pit lakes resulting from lignite mining and from metal mining	20
Figure 5	Typical frequency distributions of pH-values in pit lakes resulting from lignite mining and from metal mining	20
Figure 6	Relation between pH-value and concentration of sulphate $(SO_4^{2-})$ , calcium (Ca), iron (Fe) and copper (Cu) in the water of pit lakes	21
Figure 7	Map showing the location of the pit lakes in the Central German lignite mining district	24
Figure 8	Frequency distribution of the morphometric data of the pit lakes in the Central German lignite mining district	25
Figure 9	Water quality parameters describing the state of acidification in the pit lakes of the Central German district in spring 2007	26
Figure 10	Concentrations of total phosphorus (TP) and chlorophyll a in the pit lakes of the Central German district in spring 2007	27
Figure 11	Map of Lake Goitsche and its surroundings	29
Figure 12	Rise of water level in the basins of Lake Goitsche from April 1999 to July 2002 and volume of water diverted into Lake Goitsche from river Mulde in that period	30
Figure 13	Development of pH in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion	31
Figure 14	Balance of alkalinity and acidity (expressed as negative values of alkalinity) in Lake Goitsche and its basins and cumulative alkalinity imported into Lake Goitsche by river water during the period of regular filling	32
Figure 15	Changes in the alkalinity inventories of Lake Goitsche between the dates shown in Figure 14 (lake total) in comparison with the amount of alkalinity imported with river water during the respective periods (input Mulde) and the difference "lake total – input Mulde"	32
Figure 16	Inventories of total phosphorus in Lake Goitsche and its basins at selected dates	35

Figure 17	Schematic overview of the approaches for the remediation of pit lakes in Germany	37
Figure 18	Fluxes of river water and mine water diverted into pit lakes in the Lusatian lignite mining district (LD) and in the Central German lignite mining district (CGD) since 1990	37
Figure 19	Distribution system for mine water in the south of Leipzig (Central German lignite mining district)	38
Figure 20	Development of the pH-value in Lake Cospuden (panel D; CO1-CO7 are sampling sites within Lake Cospuden) under the influence of filling and flushing with mine water from different sources	39
Figure 21	Remaining concentration of metals with increasing pH values of acid mine drainage assuming equilibrium with respective metal hydroxides and maximum concentration of each metal 100 mmol $L^{-1}$ ; based on	
	calculations with Wateq4f	43

## **List of Tables**

- Table 1Morphometric data of Lake Goitsche at the three major stages of<br/>filling: before start of regular filling with river water (April 1999), at<br/>the end of regular filling (July 2002) and at its final level (January<br/>2003)
- Table 2Comparative evaluation of different approaches for the remediation of<br/>pit lakes based on the above sections 5.1 and 5.2, Geller and Schultze<br/>(2012 a-e) and additional literature

11

30

45

#### 1 Introduction

Pit lakes are an important element of post-mining landscapes of surface mining. Their ecological functions in the landscape and the human use of the pit lakes require water quality similar to that of natural waters. Since the formation of pit lakes is the result of mining operations, the formation of pit lakes is subject of the respective environmental and mining related legislation. Therefore, pit lakes have to meet the normal environmental standards as e.g. given by the EU Water Framework Directive for artificial and heavily modified water bodies (Nixdorf 2005).

Many mining operations are faced with acid mine drainage (AMD). AMD is the result of the oxidation of geogenic pyrite and other sulphide minerals. The oxidation starts when the dewatering of the mined underground causes the aeration of the underground. AMD is one of the most serious impacts of mining on the environment (Younger et al. 2002; Rice and Herman 2012). When pit lakes form, they may also receive AMD resulting in acidification of the lake water (e.g. Parsons 1964, Campbell and Lind 1969, Pietsch 1979, Klapper and Schultze 1995, Miller et al. 1996, Geller et al. 1998, Castendyk and Eary 2009). The acidification has to be overcome in order to meet the above mentioned requirements for the water quality of the pit lakes.

There are several approaches to abate acidification in pit lakes. One of the approaches is the use of river water for filling pit lakes. Although this approach has been applied for decades no detailed quantification of the contribution of the processes influencing the final water quality in the pit lake is available. Therefore, the filling of this gap is the main goal of this dissertation thesis using Lake Goitsche (Central German lignite mining district) as an example. In addition, this approach will be evaluated in the context of other approaches.

The thesis comprises four publications of the author of the thesis. The full papers are provided in the appendices. The contribution of the author of this doctoral thesis to the particular publications is given at the front sites of the appendices. The publications are:

- Schultze M, Pokrandt K-H, Hille W (2010) Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects. Limnologica 40 (2), 148-155.
  + Schultze M, Pokrandt K-H, Hille W (2011a) Erratum to: "Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects" [Limnologica 40 (2) (2010) 148–155]. Limnologica 41:78. (Appendix 1)
- Schultze M, Boehrer B, Büttner O, Geller W, Herzsprung P (2012a) Processes during filling of pit lakes with river water as indicated by water quality changes in Lake Goitsche, Germany (submitted to Science of the Total Environment) (Appendix 2)

- Herzsprung P, Boehrer B, Duffek A, Friese K, Hupfer M, Schultze M, v. Tümpling jr. W, van der Veen A (2010) Flood effects on phosphorus mobility in a river water filled pit lake – Lake Goitsche (Germany). Limnologica 40:182-190 (Appendix 3)
- Schultze M, Geller W, Benthaus F-C, Jolas P (2011) Filling and management of pit lakes with diverted river and mine water - German experiences. In: McCullough CD (ed.) Mine Pit Lakes: Closure and Management. ACG Australian Centre for Geomechanics, Perth, 107-120 (Appendix 4)

After a basic overview over the properties of pit lakes in section 2, the Central German lignite mining district is presented in section 3 summarizing Schultze et al. (2010; 2011a). The Central German lignite mining district was the regional frame of the presented research and provided diverse examples for the issues discussed in this doctoral thesis Lake Goitsche, located in the Central German lignite mining district, is then used as example for the detailed discussion of the processes occurring during filling of pit lakes with river water in section 4. The neutralization and the risk of eutrophication are the focus of section 4 based on Schultze et al. (2012a) and Herzsprung et al. (2010). In section 5, the approach of filling pit lakes with river water is discussed and evaluated in comparison with alternative approaches of filling and management of pit lakes within Germany and on the international scale based on Schultze et al. (2011b). Finally, conclusions are drawn in section 6.

#### 2 Properties of pit lakes

Before discussing properties of pit lakes, the term "pit lake" has to be defined. Basically, pit lakes are lakes in former excavations. They may result from excavations of sand, of gravel, of clay, of lignite, of coal, of ore and they may also occur in former quarries. Synonymous terms used in the literature are "mining lake", "mine lake", and "mine pit lake". In this thesis, only pit lakes in former lignite, coal or metal surface mines are considered when using the term "pit lake". Within this group of pit lakes, this thesis is focussed on pit lakes in former lignite open cast mines in Germany, in particular in the Central German lignite mining district.

#### 2.1 Basic factors influencing pit lakes

There are many factors influencing the properties and the development of pit lakes. The first and most basic factor is the geographic location. It determines the following factors: climate, regional water balance, regional geology, orology of the catchment area, characteristics of soil and vegetation of the catchment area and linkage to the regional groundwater flow. The connection to the network of surface waters may often be influenced by human activities. Further anthropogenic factors are the land use in the catchment area and the use and management of the lakes themselves. Finally, the form of the lake basin and the lake's exposition to sunlight (the main heat source) and to wind action (a major driving force for mixing) are important factors. All these factors are important for all lakes, natural ones and pit lakes.



Figure 1 Relation between age (as indicated by the year of filling start) and surface area for German pit lakes in former lignite mines (from Schultze et al. 2012b. Database: Nixdorf et al. 2001, the web sites of the Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft (LMBV; www.lmbv.de), and Mitteldeutscher Seenkatalog (www.mitteldeutscheseen.de)

Special aspects apply additionally for pit lakes. The form of the lake basin may strongly depend on the age of the pit lake. Older pit lakes, e.g. in Germany originating from the end of

the ninetieth century and the beginning of the twentieth century, are usually small due to the limited technological options for mining at that time. This directly leads to the second aspect: the applied mining technology which has a decisive influence on the final geometry of the basin of a pit lake. Larger machineries usually leave larger and deeper mining voids at the end of the mining operations and, consequently, result in larger and deeper pit lakes. Figure 1 shows the permanent increase of the average size of German pit lakes over time.

A decisive factor for choosing the mining technology at the beginning of the mining operations and also for the options to shape the lake basin is the local geology of the particular mine site. Solid rock allows steep side walls of the mines resulting in steep side walls of the final lake basin. Unconsolidated rock such as sand, gravel or clay requires much smoother side walls with a much smaller inclination. The relative depth  $z_{rel}$  is a good measure for the shape of a lake basin. It is the ratio of the maximal depth of a lake  $z_{max}$  to the mean diameter of the lake surface, expressed in percent. It is calculated as follows (Wetzel 2001):

$$z_{rel} = \frac{z_{\max} \cdot \sqrt{\pi} \cdot 100\%}{2 \cdot \sqrt{A}} \tag{1}$$

with A – area of the lake surface.

The relative depth of pit lakes in hard rock environments is usually higher compared to the relative depth of lakes located in unconsolidated rock. E.g., the relative depth of the pit lakes in the Iberian Pyrite Belt in southern Spain – a hard rock environment – range from 6.3% to 45% with a median of 23.4% (database: Sanchez Espana et al. 2008). In the Central German lignite mining district, the relative depth of the pit lakes range from 0.57% to 5.4% with a median of 1.8% due to their imbedding in sand, gravel, loam and clay (database: the same as used in Schultze et al. 2010). Mining sites with strongly fractured hard rock or considerable covers of unconsolidated rock and soil on top of solid rock provide conditions between the two example regions.

#### 2.2 Water quality of pit lakes

The water quality of a lake is the result of import of diverse substances into the lake, of export of substances of the lake and the transformations inside the lake. An important factor determining the import of substances beside water balance is the mineralogical composition of the catchment area. In the case of pit lakes, sulphide minerals play often a dominant role in this context. They are oxidized when the dewatering operations of the mines and the excavation allow for contact between these minerals and the atmosphere. Figure 2 shows a schematic depiction of the dewatering and consequent aeration of the underground in the vicinity of lignite surface mines in Germany according to Grützmacher et al. (2001).



Figure 2 Scheme of the dewatering of the underground in the vicinity of German lignite surface mines and the classification of the dewatered underground in zones of different extent and duration of dewatering (from Grützmacher et al. 2001, adapted. For details on the zones see text.)

The zones of groundwater lowering are characterized as follows (Grützmacher et al. 2001):

- In the side walls and the bottom of the mine void below the pre-mining groundwater level with intensive contact with oxygen for a long period
- Ib fully drained underground areas between zone Ia and the dewatering wells
- II area where the Tertiary material below the lignite remained saturated except for a small cone of depression close to the wells and the Tertiary materials above the lignite was only drained partially, while the Quaternary material was fully drained
- III area where the Tertiary material remained saturated with water and the Quaternary material was partially drained
- IV area without groundwater lowering

The separation of Tertiary and Quaternary strata in German lignite mines is important since Quaternary material usually does not contain pyrite whereas the presence of pyrite is common in Tertiary strata. Extension of the zones generally depends on the local hydrogeological conditions of a particular mine site. For the conditions of the Central German lignite mining district, Grützmacher et al. (2001) reported 1 to 6 m thickness for zone Ia, up to 500 m extension for zone II, and 500 to 2000 m extension for zone III.

The oxidation of sulphide minerals, in particular pyrite, results in the formation of acidity and AMD as already mentioned in the Introduction. Sulphide minerals are often the target of metal mining. However, pyrite (chemically  $FeS_2$ ) also occurs in coal, lignite and in the strata around the respective seams. In the case of coal and lignite deposits, the pyrite formed as the result of microbial reduction of iron and sulphate under the anoxic conditions occurring below, inside and above the coal and lignite seams. The mechanisms of this sedimentary

pyrite formation were described in detail by Berner (1970; 1984). The oxidation of pyrite is described by the following Equations 2-5:

Oxidation of pyritic sulphur by oxygen  $FeS_2 + 3.5O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+$  (2) Oxidation of ferrous iron by oxygen  $14Fe^{2+} + 3.5O_2 + 14H^+ \rightarrow 14Fe^{3+} + 7H_2O$  (3) Oxidation of pyritic sulphur by ferric iron  $FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+$  (4) Ferric iron that was not consumed by further oxidation of pyritic sulphur hydrolyses and precipitates if the pH is above 3.5, according to the following reaction:  $Fe^{3+} + 3H_2O \leftrightarrow Fe(OH)_3 + 3H^+$  (5)

The pyrite oxidation occurs not only at the surfaces inside the mines but also inside the underground as far as oxygen can enter the dewatered pore volume. Usually, oxidation fronts form (Prein and Mull 1998). However, the surfaces in the mines are sites of highest oxidation rates (Wisotzky 1994). The burial of temporary surfaces by dumping further overburden often results in a very heterogenous pattern of degree of oxidation in overburden dumps (Wisotzky 1994; Wiegand et al. 2003). The formed acidity reacts with present buffering substances in the water and with minerals at the sites of its formation as well as along the flow paths of water, both surface and underground. The most common reactions are summarized in Equations 6 to 13:

Buffering by dissolved bicarbonate in the water	
$HCO_3^- + H^+ \leftrightarrow CO_2 + H_2O$	(6)
Dissolution of carbonate minerals (here e.g. calcite)	
$CaCO_3 + 2H^+ \leftrightarrow CO_2 + H_2O + Ca^{2+}$	(7)
Ion exchange at diverse surfaces of minerals and organic matter	
$XM + nH^+ \leftrightarrow XH_n + M^{n+}$	(8)
Dissolution of colloidal or mineral aluminium hydroxide	
$Al(OH)_3 + 3H^+ \leftrightarrow Al^{3+} + 3H_2O$	(9)
Dissolution of colloidal or mineral ferric iron hydroxide	
$Fe(OH)_3 + 3H^+ \leftrightarrow Fe^{3+} + 3H_2O$	(10)
Dissolution of goethite	
$FeOOH + 3H^+ \leftrightarrow Fe^{3+} + 2H_2O$	(11)
Dissolution of feldspars and formation of clay minerals	
$2KAlSi_{3}O_{8} + 2H^{+} + 9H_{2}O \leftrightarrow Al_{2}Si_{2}O_{5}(OH)_{4} + 4H_{4}SiO_{4} + 2K^{+}$	(12)
Total dissolution of feldspars	
$KAlSi_3O_8 + 4H^+ + 7H_2O \leftrightarrow Al^{3+} + 3H_4SiO_4 + K^+ + 3H_2O$	(13)

As indicated by the bidirectional arrows in the above equations, many of these reactions are reversible and form a buffering system. In the water of the pit lakes, the hydrolysis of aluminium and ferric iron are important buffering systems under acid conditions. The titrations curves in Figure 3 demonstrate the buffering function of the hydrolysis of ferric iron and aluminium in two pit lakes in the Lusatian lignite mining district (Germany). In Lake 111, iron provides the dominant contribution to the buffering under acid conditions whereas aluminium dominates in Lake 110 due to the lower concentration of ferric iron in this lake compared to Lake 111.



Figure 3 Titration curves of two pit lakes in the Lusatian lignite mining district (Germany) demonstrating the buffering (grey areas) by ferric iron (Fe<sup>3+</sup>) and by aluminium (Al<sup>3+</sup>) under acid conditions (from Schultze 2010. Data by courtesy of O. Totsche)

The reaction of the acidity with minerals results in the consumption of acidity on one hand and in the mobilization of diverse substances on the other hand as shown in the above equations. That is the reason for the typical high concentrations of dissolved substances in AMD. Figure 4 shows typical concentration ranges in pit lakes from lignite mining and from metal mining. The concentrations are usually higher in pit lakes resulting from metal mining since sulphide minerals are often the target of mining there. Hence, more acidity can be formed since much more sulphide minerals are present. Typical pH-values in such pit lakes are shown in Figure 5.

For many substances, there is a strong relation between pH-value of the water and their concentrations. In particular, heavy metals and often sulphate show increasing concentrations with decreasing pH (Klapper and Schultze 1995; Eary 1999). However, there are also substances which do not follow such simple rules like calcium, sodium, potassium and chloride (Klapper and Schultze 1995). Figure 6 shows as examples the concentrations of sulphate, calcium, iron and copper in pit lakes in former lignite and former metal mines.



Figure 4 Typical ranges for concentrations of the usually most relevant substances in pit lakes resulting from lignite mining and from metal mining (from Schultze 2010. Data for lignite mining from Nixdorf et al. 2001, data for metal mining originate from Sanchez Espana et al. 2008). Lines within the boxes are median values, box limits are 25 and 75 percentiles, and whiskers show the 10- and 90-percentile values. Single dots indicate data below 10-percentile or above 90-percentile, respectively (ACY-acidity, ALK-alkalinity, SRP-soluble reactive phosphorus, TP-total phosphorus, EC-electrical conductivity, nd-not detected).



Figure 5 Typical frequency distributions of pH-values in pit lakes resulting from lignite mining and from metal mining (from Schultze 2010. Data for lignite mining from Nixdorf et al. 2001, data for metal mining from Sanchez Espana et al. 2008 and from Shevenell et al. 1999)

The reasons for the different behaviour are well known from the basics of geochemistry. Firstly, many metals form hydroxides when pH increases. Secondly, diverse substances may also form other minerals completely independent from pH like the precipitation of gypsum which influences the concentrations of calcium and sulphate. For that reason, pit lakes with sulphate concentrations above 1500 mg/L are rare if enough calcium is available. Thirdly there are substances which are neither influenced by acidification and pH nor by the

precipitation of minerals under the typical conditions in pit lakes. Examples for this group are chloride and to a wide extent sodium.



Figure 6 Relation between pH-value and concentration of sulphate (SO<sub>4</sub><sup>2-</sup>), calcium (Ca), iron (Fe) and copper (Cu) in the water of pit lakes (from Schultze 2010. Data for Data for lignite mining from Nixdorf et al. 2001, data for metal mining from Sanchez Espana et al. 2008 and from Shevenell et al. 1999)

In addition to hydroxides as described so far, ferric iron may also form hydroxo-sulphates at low pH and high concentrations of sulphate. The respective minerals are jarusite and schwertmannite. Both minerals were found in pit lakes (e.g. Göttlicher and Gasharova 2000; Peine et al. 2000) and are stable only at low pH. If pH increases, they transform into ferrihydrite or goethite (Nordstrom and Alpers 1999; Nordstrom 2009). Therefore, they act as a sink for acidity and sulphate when they form but as a source for acidity and sulphate when they transform into more stable minerals.

The acidification of the water results also in restrictions for aquatic life. Many organisms which are common in neutral natural water bodies cannot survive in acidic pit lakes. Fish is generally not present in acidic pit lakes (Duis et al. 2001; Rümmler 2004). Benthic invertebrates occur only in much reduced diversity (Rodrigues and Scharf 2001). The pelagic foodweb is often very simple consisting only of bacteria, very few species of algae (often dominate mixotrophs) and rotifers and/or protozoa (Lessmann et al. 1999; Woelfl 2000;

Wollmann et al. 2000; Kamjunke et al. 2004). The different composition of macrophyte communities can be used as indicator system for estimation of the extent of acidification (Pietsch 1998).

According to the described chemical and biological conditions in pit lakes, their use for diverse purposes like recreation, fishery, water supply or water storage is often very limited or even impossible. This requires strategies for the remediation and management of pit lakes and their water quality.

In addition to acidification also other threats for the water quality in pit lakes may occur. Klapper and Schultze (1995) summarized the potential threats in the order of decreasing importance as follows:

- Acidification caused by pyrite oxidation and the accompanying mobilization of acidity, iron and sulphate
- Eutrophication caused by excessive import of phosphorus and nitrogen via river water for filling or flushing the lake or inappropriate lake use (e.g. excessive feeding in aquaculture)
- Contamination with industrial pollutants caused by groundwater inflow from industrial sites or waste deposits in the vicinity of pit lakes
- Salinization by highly saline ground water
- Infection by import of pathogens via river water which in turn was impacted by waste water

### 3 The Central German mining district

The Central German lignite mining district is one of the four German lignite mining districts where mines are still in operation (others: Rhineland district, district of Helmstedt and Lusatian district; for location see inset of Figure 7). In the last years, the lignite production of this district was about  $20 \cdot 10^6$  t/yr, i.e. about 11.8% of the German lignite production (DEBRIV 2011). However, it hosts about 140 pit lakes representing 28% of all pit lakes in former lignite surface mines in Germany. Figure 7 shows the location of the pit lakes and the state of their filling in September 2011. The numbered pit lakes are mentioned in the text of this thesis. Consequently, Figure 7 is an updated and adapted version of the respective map in Schultze et al. (2010) in Appendix 1.

The lignite in the Central German district is of Tertiary age. The oldest lignite was mined in the Geiseltal deposit. It formed in the Middle Eocene. The seams mined in the southern part of the Central German lignite mining district originated from Upper Eocene to Lower Oligocene. In the vicinity of Halle, the mined lignite formed from Lower to Middle Oligicene. Finally, the mined seams around Bitterfeld originated from Lower Miocene (Eissmann 2002a).

The lignite seams are embedded and overlain by unconsolidated Tertiary and Quaternary rock like clay, silt, sand and gravel. Dissolution of Permian and Triassic salts (mainly chlorides of sodium, potassium and manganese and gypsum) resulted in the deformation of all overlaying strata, including the lignite seams. If the dissolution and the resulting subsidence occurred syngenetically, the lignite seams developed exceptional thickness. The pressure of Quaternary glaciations and the erosive activity of melt water caused further deformations and changes of the initial forms of the lignite seams (Eissmann 2002a,b; Litt and Wansa 2008).

Natural groundwater rebound after decommissioning of the dewatering operations was the only way of filling the pit lakes in the Central German lignite mining district until the 1970s. Beginning with Lake Muldestausee in 1975/76, diversion of river water became the main way of filling. In the 1990s, the use of water from dewatering operations of active mines (referred to as mine water in the following) was established as a third option for the filling of pit lakes in the Central German district.



Figure 7 Map showing the location of the pit lakes in the Central German lignite mining district (from Schultze et al. (2010), adapted). The indicated status of filling reflects the state of September 2011 (according to the information given by the LMBV on its web site http://www.lmbv.de in November 2011). Only pit lakes mentioned in the main part of this thesis are marked. Their names are: 1-Bergwitz, 2-Golpa IV, 3-Goitsche, 4-Muldereservoir, 5-Seelhausen, 6-Hufeisensee, 7-Wallendorf, 8-Raßnitz, 9-Geiseltal, 10-Runstedt, 11-Cospuden, 12-Zwenkau, 13-Kahnsdorf, 14-Hain, 15-Haubitz, 16-Bockwitz, 17-Harthsee, 18-Borna, 19-Haselbach, 20-Vollert Süd. Insert: RD-Rhineland lignite mining district, HeD-lignite mining district of Helmstedt, CGD-Central German lignite mining district, OD-lignite mining district of Oberpfalz, LD-Lusatian lignite mining district.

The size of the pit lakes varies over a wide range as shown in Figure 8. The small lakes are usually old ones resulting from the end of the ninetieth or from the first decades of the twentieth century. However, caused by technological reasons, small pit lakes may also form today.



Figure 8 Frequency distribution of the morphometric data of the pit lakes in the Central German lignite mining district (from Schultze et al. 2011a; for data sources and more details, please see Appendix 1)

Pit lakes deeper than 10 m show a thermal stratification in the Central German district. Since full ice covers form only exceptionally, the majority of them can be classified as monomictic. However, there are also meromictic pit lakes in the central German district. Meromixis results from the intrusion of saline groundwater rising up from Permian and Triassic salt deposits under dissolution. Examples are Lake Hufeisensee, Lake Raßnitz and Lake Wallendorf (Maiss et al. 1998; Böhrer et al. 1998; Stottmeister et al. 1999; Trettin et al. 1999, 2006; Schultze and Boehrer 2008). The intrusion of highly mineralised water which originates from reactions of AMD with the minerals along the flow paths caused meromixis in Lake Goitsche (see section 4 and Appendix 2). Lake Vollert Süd became meromictic because of the filling with waste water of coke production and the applied remediation technology (Stottmeister et al. 2010).

The water quality of the pit lakes of the Central German district is the result of the regional and local conditions, of remediation measures and of water management. Therefore, the data presented in Figure 9 may suggest that acidification of pit lakes is not an important issue in this region. This is misleading since the remediation measures and the water management strongly contributed to the abatement of acidification in the last 20 years. Without those measures, there would be many more acidic pit lakes in the Central German district, in particular among the large, young pit lakes filled after 1990. In addition, many of the pit lakes are at risk to become acidic again (referred to as re-acidification in the following) due to the continuing inflow of acidity bearing water. If the rate of inflow of acidity is bigger than the summarized rates of inflow of alkalinity and of internal alkalinity generation in the lake (e.g. by sulphate reduction), the alkalinity present in the lake water is consumed over time and the lake becomes re-acidified. Lake Bockwitz is an example where the permanent inflow of acidity requires continued measures to keep the lake neutral (Neumann et al. 2008; Rönicke et al. 2010; Heinrich et al. 2011). This lake was neutralized by addition of soda ash in 2007. The

oxidation of pyrite in portions of the underground, which remained permanently aerated after groundwater rebound, and the interflow through those portions of the underground were found to be the main sources for acidity of Lake Bockwitz (Heinrich et al. 2011). Further examples are Lake Hain, Lake Haubitz, Lake Kahnsdorf and Lake Zwenkau. Lake Raßnitz and Lake Wallendorf came close to re-acidification but stabilised at a low level of alkalinity (about 0.5 mmol/L; Schultze and Boehrer 2008).



Figure 9 Water quality parameters describing the state of acidification in the pit lakes of the Central German district in spring 2007 (from Schultze et al. 2010. For data sources and the concentrations of further water constituents, please see Appendix 1.)

According to the list of potential threats for the water quality of pit lakes in Klapper and Schultze (1995), eutrophication has to be discussed briefly. The concentrations of total phosphorus (TP) and chlorophyll a (Figure 10) indicate that the pit lakes of the Central German lignite mining district are mainly oligotrophic to mesotrophic. Only the Muldereservoir and Lake Golpa IV are eutrophic. The entire Mulde River flows permanently through the Muldereservoir and causes the high load of nutrients and the respective eutrophication. The load of nutrients and organic matter was even higher in the past (e.g. Junge and Jendryschik 2003; Klemm et al. 2005). Lake Golpa IV served as storage for cooling water of a power plant. Therefore, the water in this lake was replaced several times per year with water from Mulde River. Until the beginning of the 1990s, the concentration of soluble reactive phosphorus (SRP) was in the range of 500 mg/L causing the eutrophication of Lake Golpa IV and the high amounts of phosphorus in its sediment (Hupfer et al. 1998).



Figure 10 Concentrations of total phosphorus (TP) and chlorophyll a in the pit lakes of the Central German district in spring 2007. The inserts provide a finer resolution of the smallest class of the main histogram. (from Schultze et al. 2010. For data sources and the concentrations of further nutrients, please see Appendix 1.)

The other potential threats listed in Klapper and Schultze (1995) are of less importance. The intrusion of saline groundwater caused meromixis in some pit lakes but did not result in salinisation of many lakes as already described above. The contamination of pit lakes by contaminated groundwater from waste deposits and contaminated industrial sites required remedial measures only in Lake Runstedt and in Lake Hufeisen. In Lake Runstedt where a deposit of ash and industrial waste forms the bottom of the lake, the creation of the pit lake was part of the remediation strategy for the waste deposit (Schroeter 1997; Schultze et al. 2011c). The contamination of Lake Hufeisensee by a deposit of municipal waste was always limited due to the accumulation of the hazardous substances in the monimolimnion and its sediments and further minimized by the implementation of a sealing cover on top of the waste deposit (Stottmeister et al. 1999). The risk resulting from the import of pathogens with river water during filling of pit lake was found to be very small since the pathogens were removed from the water body of the lakes rapidly (Pusch et al. 2005; Wolf 2005).

The pit lakes of the Central German district are mainly used for recreation (swimming, sailing, diving, etc.). Flood protection (Lake Borna, Lake Zwenkau) and the providence of habitat for aquatic and other water related wildlife, including nature protection areas, are further important functions of pit lakes in the Central German lignite mining district. Commercial fishery is of small importance while hobby fishing is of relevance for the local population.

# 4 Lake Goitsche as an example for the filling with river water

Lake Goitsche is discussed here in detail because it was the first pit lake filled with river water where the filling was accompanied not only by a standard monitoring but by a research project. The goal of this project was getting a deeper insight into the processes occurring during filling pit lakes with river water.

# 4.1 Location, geological conditions, mining history, filling and morphometry

Lake Goitsche is located in the former mine Goitsche which was operated from 1949 to 1991 south-east of Bitterfeld in the Central German lignite mining district. It is embedded in a large post-mining landscape resulting from about 100 years of surface mining for lignite (Figure 11, panels B and C). The mined lignite originated from the Lower Miocene (Eissmann 2002a). Dissolution of Permian salt deposits in the deeper underground caused the formation of local depressions in the lignite seams which resulted in local depressions in the bottom of Lake Goitsche (Figure 11, panel A). The area of one of the islands in Lake Goitsche (Figure 11, panel B) was not mined because of the absence of lignite there. The reason was the occurrence of paleodunes at the time the formation of the lignite striking roughly from southwest to north-east (Eissmann 2002a; Wimmer et al. 2006). Another important aspect of the Bitterfeld region was the occurrence of amber bearing strata below the lignite and the mining for amber from 1975 to 1991 (Quasdorf 1998; Fuhrmann 2005; Wimmer et al. 2006). The amber bearing strata extended only in the area of the current basins Muehlbeck and Niemegk of Lake Goitsche (Wimmer et al. 2006; Figure 11, panel A). The facility separating the amber from its silt matrix was located in the current basin Niemegk. Therefore, the waste silt was dumped also in basin Niemegk. This is important since this silt contained up to 5% pyrite (Grützmacher et al. 2001) and was the main source of acidification for Lake Goitsche (Kringel et al. 2000). Today, Lake Goitsche has three basins. This is the result of the geological conditions, the mining history and technological conditions. E.g., the peninsula separating basin Muehlbeck and basin Doebern (Figure 11, panel B) is the result of dumping overburden on already mined area as close to actual excavation sites as possible while leaving amber bearing areas uncovered for later exploitation (Quasdorf, 1998).



Figure 11 Map of Lake Goitsche and its surroundings. Panel A: bathymetric map of Lake Goitsche including sampling sites for lake water (D5, M3, N3, N5, P4); panel B: mining district of Bitterfeld (1 - former mine Goitsche with Lake Goitsche, 2 - former mines Holzweißig with pit lakes, 3 – former mine Rösa with Lake Seelhausen, 4 – former mine Muldenstein with Muldereservoir; areal extension of pit lakes indicates planned final water level whereas state of filling indicates state for period 1999-2002); panel C: location of study area in Germany (from Trettin et al, 2007, adapted).

The filling of Lake Goitsche with river water started at May 7, 1999. Water was diverted from Mulde River downstream the Muldereservoir. A remaining section of the initial bed of Mulde River was used for filling connecting Mulde River with basin Muehlbeck (Figure 11, panel B). Since 1976, Mulde River was completely diverted into the Muldereservoir which resulted from the former mine Muldenstein (Müller 1998). The natural course of Mulde River including the respective flood plains were mined as eastern part of mine Goitsche from 1976 to 1991 (Quasdorf 1998).

Figure 12 shows the stepwise rise of the water level in the basins of Lake Goitsche. The basins were connected with each other by channels. When filling with river water began, there were already small water bodies at the bottom of the basins referred to as precursor lakes in the following. They resulted from groundwater inflow, local runoff and precipitation due to limited dewatering. The initial water levels were different according to the technological needs of preparation of the mine void for filling and the needs of geomechanical stability of the side walls. For more details, please see Appendix 2 and 3.



Figure 12 Rise of water level in the basins of Lake Goitsche from April 1999 to July 2002 and volume of water diverted into Lake Goitsche from river Mulde in that period (from Schultze and Geller 2004, adapted)

Table 1 summarizes the morphometric data of Lake Goitsche at all important stages of filling: April 1999 immediately before filling with river water began, July 2002 when regular filling finished, and January 2003 when the final water level was reached after a flood event in August 2002 (Boehrer et al. 2005; Klemm et al., 2005). The flood event is visible in Figure 12 as sudden increase of the water level in August 2002. The depiction of the amount of inflowing river water was not continued after July 2002 because the respective data are not available. During the flood event, measurements were not possible. The only possible quantification was that the inflow of river water exceeded 90·10<sup>6</sup> m<sup>3</sup> since there was an overflow of Lake Goitsche when its volume had reached  $256 \cdot 10^6$  m<sup>3</sup> at August 17, 2002 (Laugwitz 2004). Between August 2002 and January 2003, the water level of Lake Goitsche was lowered to the level of 75 m a.s.l.

Table 1Morphometric data of Lake Goitsche at the three major stages of filling: before start of regular<br/>filling with river water (April 1999), at the end of regular filling (July 2002) and at its final level<br/>(January 2003) (from Schultze et al. 2012a, see also Appendix 2)

	Water level	Volume	Surface area	Maximal depth
	m a.s.l. <sup>a</sup>	$10^{6} \mathrm{m}^{3}$	$10^{6} \text{ m}^{2}$	m
April 1999				
Basin Muehlbeck	53.5	1.7	1.08	3.5
Basin Niemegk	40.0	4.0	0.68	16.0
Basin Doebern	35.0	1.2	0.35	13.0
July 2002				
Lake Goitsche	71,5	166	10.5	44.7
January 2003				
Lake Goitsche	75,0	213	13.3	47.9

<sup>a</sup> m a.s.l. – meter above sea level

The focus of this section is the regular filling of Lake Goitsche. However, the year 2003 was included here because the investigations of the behaviour of phosphorus in the sediments of Lake Goitsche after the flood gave important insights into the long-term behaviour of phosphorus in pit lakes after filling with river water in general. For more details on the flood, please see Herzsprung et al. (2010) in Appendix 3. Further detail can be found in Boehrer et al. (2005), Klemm et al. (2005) and Laugwitz (2004).

#### 4.2 Neutralization of Lake Goitsche

Figure 13 shows the development of the pH in Lake Goitsche during regular filling separated for the basins and for the epilimnion and the hypolimnion. Since the inflowing river water had a lower density than the already existing lake water, the river water preferably flew into the epilimnia during seasonal stratification. This caused considerable chemical differences between epilimnion and hypolimnion. The epilimnia were always neutralized earlier than the hypolimnia. Basin Doebern was temporarily acidified from the end of 1999 to February 2001 by acidic water flowing from basin Niemegk into basin Doebern (Figure 13). The development of acidity and alkalinity provided basically the same information. However, there was some evidence that additional acidity entered the lake during its filling, in particular basin Niemegk (see Appendix 2).



Figure 13 Development of pH in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion (from Schultze et al. 2012a, see Appendix 2)

In order to get a deeper insight into this issue, an acidity-alkalinity balance was established for selected dates of the filling period (Figure 14; for details of the calculation see Appendix 2). The differences between the inventories of alkalinity at the selected dates confirmed, that substantial amounts of acidity entered Lake Goitsche and in particular basin Niemegk until neutralization of the lake water in February 2001 (Figure 15). The sources of this acidity were

mainly the amber bearing strata and the dumped wastes of amber separation at the bottom of basin Niemegk. The later changes in the inventories of alkalinity (after February 2001) were probably not caused by pyrite oxidation but rather by accumulation of carbon dioxide in the hypolimnion during periods of seasonal stratification.



Figure 14 Balance of alkalinity and acidity (expressed as negative values of alkalinity) in Lake Goitsche and its basins and cumulative alkalinity imported into Lake Goitsche by river water during the period of regular filling (Lake Goitsche represents the sum of the basins) (from Schultze et al. 2012a, see Appendix 2)



Figure 15 Changes in the alkalinity inventories of Lake Goitsche between the dates shown in Figure 14 (lake total) in comparison with the amount of alkalinity imported with river water during the respective periods (input Mulde) and the difference "lake total – input Mulde" (from Schultze et al. 2012a, see Appendix 2)

The detailed analysis of the differences shown in Figure 15 and further considerations (for details see Appendix 2) indicated that about 60% of the overall acidity to be neutralized until February 2001 entered the lake during filling. The import of alkalinity with the river water was by far the most important neutralizing process. The import of alkalinity with river water

was 112% of the estimated overall acidity to be neutralized until February 2001 indicating that part of the entering acidity was obviously not included into the estimate of the overall acidity. The reason was that some potentially relevant processes could not be quantified. Direct measurements were not done (erosion and microbial alkalinity generation) or not possible in the necessary spatial and temporal resolution (import of acidity and alkalinity with inflowing groundwater). Land slides did not play an important role in Lake Goitsche since only very small ones occurred during regular filling. However, when they occur in a relevant dimension, they strongly influence the development of the water quality in pit lakes as e.g. reported by Gröschke et al. (2002) for Lake Gräbendorf (Lusatian lignite mining district). It is not likely that much more detailed quantifications of the contribution of particular processes are possible than presented here. The heterogeneity of the underground surrounding a pit lake is usually very high. Therefore, a complete detection of all fluxes of substances

a pit lake is usually very high. Therefore, a complete detection of all fluxes of substances entering and leaving a filling pit lake is probably not possible because of the enormous effort necessary for that.

#### 4.3 Development of the concentrations of further substances

The final statement regarding the possibility of quantification of the contribution of different processes to neutralization in subsection 4.2 applied also for the contribution of the different processes to the final concentration of divers substances not mentioned above. However, for several substances estimates or at least semi-quantitative estimates were possible.

Precipitation and co-precipitation as one kind of reactions in the lake water were responsible for the removal of more than 95% of iron, aluminium, arsenic, trace metals and phosphorus. Since phosphorus was the main reason for eutrophication in lakes in the last decades (e.g. Schindler et al. 2008) and expected to be a major risk of filling pit lakes with river water (Klapper and Schultze 1995), its behaviour will be discussed in more detail in subsection 4.4.

Dilution was relevant for all substances which had initially higher concentrations in the lake water than in the river water, in particular regarding (almost) non-reactive substances. The replacement of lake water by river water because of outflow of water into the surrounding aquifers was of low relevance. The amount of water leaving the lake was small compared with that entering the lake. Accordingly, the respective fluxes of substances could be assumed being also small compared to fluxes caused by other processes. The transport into the lake with inflowing groundwater was relevant for substances mobilized by acidification in the acidified portions of the underground and for ammonia. The interaction with the bottom and the side walls of the former mine void was relevant for the same substances. However, the distinction between import via inflowing groundwater and via erosion and elution of the

bottom and the side walls of the former mine void was not possible. For nitrate, the import with river water was the main source. The interaction with the lake sediment was found to be restricted to sedimentation. Re-dissolution of substances during regular filling was very unlikely. Primary production consumed nutrients and, therefore, was probably relevant for phosphorus, nitrogen (ammonia and nitrate) and silicon.

# 4.4 Behaviour of phosphorus during filling of Lake Goitsche and the risk of eutrophication

Salts of iron and aluminium are widely used in water technology for flocculation (e.g. ATV 1997; Weigelt 2002) and are also well established in the abatement of eutrophication in lakes (Cooke et al. 2005). Therefore, the findings on the binding of phosphorus by iron in acidc pit lakes were not unexpected (e.g. Kleeberg and Grüneberg, 2005; Grüneberg 2006, Kleeberg et al. 2008). Due to the precipitation of iron and aluminium during neutralisation, the accompanying co-precipitation of phosphorus is an important process during filling and neutralization of pit lakes with river water. Figure 16 shows the inventories of total phosphorus in the water of Lake Goitsche compared with the import of phosphorus with river water from April 1999 to December 2003, i.e. during regular filling and after the flood event in 2002. The import of phosphorus with flood water could not be calculated for the period after July 2002 since respective data are not available. The development of the concentrations is presented in detail in Appendix 2 and 3. The phosphorus was rapidly removed from the water during regular filling as well as after the flood. Duffek and Langner (2002) demonstrated that the phosphorus was bound to iron at pH below 4, to aluminium in a pHrange from 4 to 7 and again to iron at pH above 7 during neutralization of Lake Goitsche. However, the potential re-mobilization of phosphorus from the lake sediment is a phenomenon well known as "internal load" from many natural eutrophic lakes. Since there is plenty of sulphate present in pit lakes usually, the competitive binding of iron as sulphide after reduction of sulphate (Caraco et al. 1989; Golterman 2001; Steinberg 2003) was seen as a potential long-term mechanism of re-mobilisation of phosphorus in the sediment. Further more, eutrophication of pit lakes was reported by Axler et al. (1996) and by Hupfer et al. (1998) indicating that the binding capacity of sediments of pit lakes for phosphorus is not unlimited.

Herzsprung et al. (2010, see Appendix 3) showed that the combined redox-cycles of iron and manganese prevented the release of phosphorus from the sediment. The dissolved phosphorus present in the pore water of the sediment is precipitated by ferric iron in the upper layers of
the sediment. The iron is oxidized there by manganese which is permanently re-cycled within the upper layers of the sediment by oxidation and precipitation at the sediment surface and reductive dissolution in the sediment. The risk of excessive sulphate reduction in the sediment was found to be small as long no extraordinary load with organic material happens. In addition, the thicker the sediment becomes on top of the layers formed during filling with river water the longer the transport path for dissolved phosphorus becomes. This makes the phosphorus originating from filling with river water less relevant over time. The importance of this factor was demonstrated by Lewandowski et al. (2003) for an opposite case: In a eutrophic natural lake, a layer of aluminium hydroxide still providing free adsorption sites for phosphorus fixation was no longer binding phosphorus since it was buried by too thick layers of younger sediment.



Figure 16 Inventories of total phosphorus in Lake Goitsche and its basins at selected dates (from Schultze and Geller 2004, adapted)

Consequently, the risk of eutrophication by filling pit lakes with river water is small. Prikryl et al. (2011) reported that the filling of a pit lake with river water did not result in eutrophication in the Czech Republic although no acidification occurred. The more important long term risk for eutrophication has to be seen in long-lasting high phosphorus loads resulting from aquaculture as reported by Axler et al. (1996), permanent through-flow of river water with high phosphorus concentrations as reported by Hupfer et al. (1998) or other situations of inadequate high loads of phosphorus.

# 5 Comparison of different approaches for filling and remediation of pit lakes

### 5.1 Approaches for filling and remediation of pit lakes in Germany

Figure 17 summarizes the approaches which were applied in Germany to neutralize pit lakes. The first approach was "Doing nothing". That means, no measures were taken to neutralize the pit lakes. This was the only approach applied until the end of the 1960s. Because of natural attenuation processes, several initially acidic pit lakes became neutral over time. The relevant processes were the natural washout of acidity from the catchment and the lake, slow buffering processes along the flow paths of acidity bearing water (e.g. dissolution of silicates) and the microbial reduction of iron and sulphate along the flow paths of acidity bearing water (as well as in the pit lakes. The occurrence of sulphate reduction was reported for overburden dumps (Hoth et al. 2005; Storch et al. 2007) and for pit lakes (e.g. Peine and Peiffer 1996; Peine et al. 2000). In the pit lakes, seasonal overturn and the permanent addition of ferric iron precipitates to the sediment re-oxidized the majority of the formed sulphide (Peine et al. 2000; Koschorreck et al. 2007). Koschorreck and Tittel (2007) found a re-oxidation rate of about 97% as an average of values reported in the literature.

However, the result of "Doing nothing" depended on the local conditions. The neutralisation of Lake Harthsee required about 5 years after reaching its final water level whereas Lake Bergwitz required about 25 years (Jordan and Weder 1995). Some of the pit lakes of the Muskauer Faltenbogen (Lusatian lignite mining district) remained acidic for more that 50 years (Gockel, personal communication 1993). Today, the exclusive waiting for natural attenuation is only accepted for small old pit lakes or where natural attenuation can be expected to be successful within 5 years or less, given there is no adverse impact on other water bodies downstream.

A detailed description of the approach of filling pit lakes with river water was presented in section 4 with the example Lake Goitsche. Figure 18 shows the annual use of river water for the filling of pit lakes in Germany in the last 20 years. From 1990 to 2010, diversion of river water contributed about 64.4% (ca.  $2.317 \cdot 10^9$  m<sup>3</sup>) to the filling of the pit lake volume in that period.



Figure 17 Schematic overview of the approaches for the remediation of pit lakes in Germany (after Schultze et al. 2009, adapted). The style of the frames indicate the current importance: thick dotted line – most important in the past, thick solid lines – currently most important, thick dashed line – currently of increasing importance, thin solid line – most important in future, thin dashed line – potentially important in future, no line – no importance til today or expected for future.



Figure 18 Fluxes of river water and mine water diverted into pit lakes in the Lusatian lignite mining district (LD) and in the Central German lignite mining district (CGD) since 1990. (\*The data for river water from the Lusatian lignite mining district for 2009 and 2010 also include the amount of mine water used in the respective river catchment areas. From Schultze et al. 2011b.)

A very similar approach is the filling with mine water. This approach contributed also considerably to the filling of pit lakes in the last about 20 years (Figure 18; ca.  $0.763 \cdot 10^9 \text{ m}^3$  corresponding to 21.2% of the pit lake volume filled from 1990 to 2010). In the southern part of the Central German lignite mining district, almost all new pit lakes were filled with mine water since 1990. A pipeline system was constructed (ca. 60 km long) for the transport of the mine water from the still operating mines to the pit lakes to be filled (Jolas 1998; Schultze et al. 2010, 2011 b,d; Figure 19; for more details see Appendix 4). Figure 20 shows the

development of the pH in the first pit lake filled with mine water, Lake Cospuden. Water from the mines Zwenkau (today Lake Zwenkau is filled in this mine) and Profen was used for filling and flushing Lake Cospuden. The quality of the water varied depending on the sources and over time. In the mine Zwenkau, three dewatering systems (Zwenkau 1-3) were operated and contributed to the filling of Lake Cospuden. In particular, water from water accumulations in local depressions of the overburden dump in the northern part of mine Zwenkau was a source of highly acidic water. The neutral water from mine Profen was the key for the successful neutralization. This applies for the initial neutralization as well as for the stabilisation of pH after 2002. The decrease of the acid neutralization capacity in the water of mine Profen resulted from an increasing contribution of overburden dumps to the catchment area of the dewatering system of mine Profen.



Figure 19 Distribution system for mine water in the south of Leipzig (Central German lignite mining district). VSH- Vereinigtes Schleenhain; Names of the pit lakes: 1- Cospuden, 2-Markkleeberg, 3-Zwenkau, 4-Stöhna, 5-Störmthal, 6-Werben, 7-Kahnsdorf, 8-Hain, 9-Haubitz, 10-Großstolpen, 11-Witznitz, 12-Borna, 13-Bockwitz, 14-Hauptwasserhaltung, 15-Südkippe, 16-Haselbach, 17-Harthsee (from Schultze et al., 2011b; for more details see Appendix 4)

The flushing of pit lakes with mine water is a rather exceptionally applied approach for the abatement for acidification of pit lakes in Germany. The more important reason for the flushing with mine water is the maintenance of the water level in pit lakes which are influenced by dewatering operations of neighbouring mines (examples: Lake Cospuden and Lake Haselbach). On the contrary, the flushing with river water is well established for the maintenance of neutral conditions in Lake Senftenberg (Werner et al. 2001) and planned to be used for many other German pit lakes, in particular in the Lusatian district (Schultze et al.

2011b. For more detail see Appendix 4). A temporal interruption of the inflow of river water into Lake Senftenberg in 1995 caused a drop of the pH in the lake water (see Figure 5 of Appendix 4; Werner 1999; Werner et al. 2001) underlining the necessity of the permanent import of alkalinity with the river water. However, such events should be avoided in order to prevent adverse effects of outflowing acid water in receiving water bodies downstream. In pit lakes which are used as reservoirs or for flood protection, the flushing with river water and the respective avoidance of re-acidification is a welcome side effect.



Figure 20 Development of the pH-value in Lake Cospuden (panel D; CO1-CO7 are sampling sites within Lake Cospuden) under the influence of filling and flushing with mine water from different sources (panels A-C; ANC-Acid neutralisation capacity, analysed by titration with 0.1N NaOH to pH 4.3). For more details see text (after Schultze et al. 2011d, adapted).

In the Lusatian district, a model based system for the combined management of the rivers Spree, Neiße and Schwarze Elster was implemented to reach the most efficient use of the available water for the filling and flushing of pit lakes (Fritz and Benthaus 2000; Grünewald 2001; Koch et al. 2005; Schlaeger et al. 2003; Luckner 2006a). This included exchange of water between the different, naturally separated river catchments (Luckner 2006b). The inclusion of the diversion of water from Elbe River into this system is under discussion (Koch et al. 2009). Further more, mine water was treated to allow its use for filling pit lakes.

Since river water and mine water was not available in the required quantity and quality to overcome acidification in all German pit lakes, also the application of alkaline materials like limestone, lime or soda ash was used for neutralization. Examples from the Central German district are Lake Bockwitz (soda ash; Neumann et al. 2008; Rönicke et al. 2010; Heinrich et al. 2011), Lake Haselbach (limestone), Lake Hain, Lake Haubitz and Lake Zwenkau (lime in addition to filling with mine water; see appendices 2 and 4 for more details). Further examples are available from the Rhineland linite mining district (Hemm et al. 2002), the lignite mining district of Oberpfalz (Hemm et al. 2002) and the Lusatian lignite mining district (Rabe and Uhlmann 2006; Pust et al. 2010; Schipek et al. 2011). Waste sludge from mine water treatment plants was used to promote the neutralization of Lake Spreetal (Lusatian district; Unger 2007; Uhlmann et al. 2007). Re-excavated waste lime previously deposited at the bottom of mine Koschen was used to promote the neutralisation of Lake Geierswald (Lusatian district; Benthaus and Uhlmann 2006). Similarly, waste fly ash that had been deposited at the bottom of the former mine Bärwalde previously was successfully tested for its potential use via re-excavation and spreading on the lake surface at field scale (Koch T et al. 2008; Koch C et al. 2008). Those field tests also included the addition of carbon dioxide in order to improve the gain of alkalinity in the water. The respective findings of improvement in lab studies (Werner et al. 2006; Unger 2007; Unger-Lindig et al. 2010; Schipek et al. 2011) could be reproduced in the field (Koch T et al. 2008; Koch C et al. 2008).

The development of in-situ technologies based on biological processes, in particular microbial sulphate reduction, was not successful in Germany. The addition of organic materials directly on the lake bottom which was tested successfully in the lab at different scales (e.g. Frömmichen et al. 2004; Fyson et al. 1998a) failed in the field. The re-oxidation of the formed sulphides could not be prevented under field conditions (Koschorreck et al. 2007; Geller et al. 2009). Artificial eutrophication as source for organic substances which in turn should serve as substrate for sulphate reducing bacteria (Fyson et al. 1998b; Nixdorf et al. 2010) was shown by to be not efficient (Fyson et al. 2006; Totsche et al. 2006). The necessary rates of primary

production were not realistic in pit lakes (Totsche et al. 2006). Floating reactors reached not the necessary rates of sulphate reduction (Preuß et al. 2007). In addition, the required reductive conditions for the planned deposition of produced sulphidic sludge at the bottom of the lakes could not be established (Koschorreck et al. 2011). Ex-situ reactors for sulphate reduction were more successful (Glombitza et al. 2001) but were not applied in full scale yet.

The treatment of inflowing or outflowing surface water was not applied in Germany except the above mentioned treatment of mine water for filling pit lakes in the Lusatian district. The treatment of groundwater flowing into pit lakes or out of them was successfully tested in the Lusatian district (Schöpke et al. 2006, 2011; Gast et al. 2010). This approach uses the aquifer as reactor for sulphate reduction by adding organic substances and nutrients to the groundwater. The implementation in remediation practice is not yet done. However, this approach not only appears to have the potential for being a valuable addition to the already applied approaches for pit lakes remediation. It also may be an approach to abate the deterioration of streams and rivers by rising, iron rich ground water in post-mining landscapes.

For future, the prevention of pyrite oxidation probably will play an important role. Selective handling and dumping of overburden rich in pyrite is a well established practice in German lignite mines today (Eyll-Vetter 2007; Rolland 2007; Jolas 2007). A second strategy is the addition of alkaline substances to the overburden during dumping. This approach has the potential to prevent acidification of future pit lakes to a wide extent. However, it took about 20 years from first successful tests (Fischer et al. 1987, 1999) and further research (Wisotzky 1998, 2003; Wisotzky and Obermann 2001) to the full scale application in the mine Garzweiler since 1998 (Wisotzky and Lenk, 2006; Eyll-Vetter 2007).

### 5.2 International aspects

Long lasting acidification as well as neutralisation of acid pit lakes by natural attenuation was known not only from Germany but also from the USA for many decades (Parsons 1964; Campbell and Lind 1969). Comparable to Germany, natural attenuation was obviously the only approach to handle acid pit lakes internationally until the last quarter of the twenties century.

Filling of pit lakes with river water is known from the Czech Republic (Sixta 1998; Svoboda et al. 2008), from Spain (Arnhold et al. 2002; Delgado 2005) and from Australia (Lund and McCullough 2008; Salmon et al., 2008). In Spain and in the Czech Republic, the main goal of the diversion of river water for the filling of pit lakes is the stabilisation of the side walls and the early use of the pit lakes. Due to the geological conditions, acidification is not likely in the

majority of Czech pit lakes in former lignite mines in North Bohemia (Svoboda et al. 2008). However, the available amount of water is limited requiring an integrated river basin management for the two main rivers of that region, rivers Ohre and Bilina (Svoboda et al., 2008). The results of the water quality development of the pit lakes in the Czech Republic appear satisfying and have met the expectations in almost all cases so far (Prikryl, 2010; Prikryl et al. 2011; Vrzal et al. 2011). In the Collie region in Western Australia, partial neutralization was achieved. A further special goal in that region was the storage of saline river water at the beginning of the wet season (McCullough et al. 2010). The high salt content in the first flush of rivers after the dry season results from agricultural land use and occurs in many regions of Australia (e.g. Jolly et al. 2001). Re-acidification was observed, when the inflow of river water finished (McCullough et al. 2010). Filling of pit lakes in former coal mines in north-west Spain is underway yet.

The use of mine water is known only from one case in the Collie region in Western Australia. The treated lake re-acidified when the inflow of mine water ceased (McCullough et al. 2010). Kalin et al. (2001) reported the filling of the B-Zone pit lake with water from the neighbouring natural Lake Wollastone in winter 1991/1992. The excellent water quality of the pristine Lake Wollaston resulted in good water quality in the B-Zone pit lake too. The pit lake water quality even improved over time (1992–1998) as indicated by changes in the plankton community (Kalin et al. 2001).

Although addition of alkaline substances was successfully applied for the neutralization of lakes which were acidified by acid atmospheric deposition (e.g. Olem 1991; Henrikson and Brodin 1995) or by industrial waste water (Lake Orta, Italy; Calderoni and Tartari 2001) only few neutralizations of acid pit lakes using this approach were reported. The respective cases are from Sweden (Lu 2004; Öhlander, 2007), Canada (Aube 2007, 2009) and USA (Lewis et al. 2003; Dowling et al. 2004). May be, this approach appears too simple to be subject of many internationally available publications.

The stimulation of sulphate reduction was tested in several cases (Brugam et al. 1995; Brugam and Stahl 2000; Lu 2004; Lewis et al. 2003; Park et al. 2006; McCullough et al. 2008a,b; Kumar et al. 2011). There is no example with well documented long term success. However, results from Australia (McCullough et al. 2008a,b) show some evidence for potential success in warm climate.

Beside neutralisation, the metal load of pit lakes in former metal mines is an important goal for the remediation of pit lakes outside of Germany. Whereas heavy metals and arsenic are usually removed from the lakes water by co-precipitation with iron and aluminium in Germany pit lakes during neutralization, metals like zinc or nickel may occur in lake water of pit lakes also after neutralization in considerable concentrations. The reasons are on one hand the very high concentrations of heavy metals and arsenic in such lakes (see Figure 4). Hydroxides of ferric iron and aluminium cannot remove all metals under such conditions by co-precipitation. On the other hand, many heavy metals require alkaline pH for precipitation as hydroxide (Figure 21). Such elevated pH-values may damage freshwater biota.



Figure 21 Remaining concentration of metals with increasing pH values of acid mine drainage assuming equilibrium with respective metal hydroxides and maximum concentration of each metal 100 mmol L<sup>-1</sup>; based on calculations with Wateq4f (after Cortina et al. 2003)

Therefore, artificial eutrophication was tested and applied not only for neutralization but also for the removal of metals from the water body. The growing algae take up the metals from the water body and settle. Successful pilot scale tests were reported by Martin et al. (2003), Dessouki (2005) and Wen et al. (2006). In the Island Copper Mine pit lake, the thresholds for heavy metals were met because year-round fertilisation of the lake for several years (Pelletier et al. 2009).

In the case of the Berkeley pit lake, the copper concentration was high enough to operate an ex situ cementation of copper using iron scrap economically. In this way, the copper concentration could be reduced in the last years. However, this approach resulted not in a gain of alkalinity because dissolved copper was replaced by dissolved iron in the lake water (Gammons and Tucci 2012). Since the water was extracted for treatment from the monimolimnion and the treated water was released into the mixolimnion, meromixis disappeared in the Berkeley pit lake in 2009 (Gammons and Tucci 2012).

Other pit lakes turned meromictic as a result of treatment. Examples are the Anchor Hill pit lake (Park et al. 2006) and Lake Rävlidmyran (Lu 2004; Ölander et al. 2007). Already existing meromixis was used for the improvement of the water quality in the mixolimnion of

the South Pit Lake which is part of a complex treatment system for the protection of the Ocoee River Tennessee from heavy metal load resulting from historical mining in the Tennessee Copper Basin (Wyatt et al. 2006; for a more detailed description see also Appendix 4). The Island Copper Mine pit lake was even designed as a meromictic pit lake serving as treatment facility for AMD (Fisher and Lawrence 2006; Pelletier et al. 2009; for a more detailed description see also Appendix 4).

### 5.3 Advantages and limitations of the approaches

Höppner et al (2006) showed that the diversion of river water and the use of mine water were by far the cheapest options to remediate German pit lakes at that time. The advantages and limitations of the different approaches for the remediation of pit lakes comprised in Table 2 also indicate that diversion of river water and the use of mine water are excellent approaches. The even better option, prevention, is not yet relevant since the pit lakes which may benefit from this approach will form in future. For pit lakes which are in filling or already existing today, diversion of river water appears as the best option currently if river water is available. Compared to the use of mine water, the long term potential of flushing with river water is a clear advantage. Although the use of mine water is similar to the diversion of river water in many aspects, the use mine water is much more limited by the existence and life time of mines in the region of the pit lakes to be remediated and managed. The most important limitation of diversion of river water is the availability of river water regarding quantity and quality. In arid regions in particular, this may exclude the diversion of river water from the list of options for pit lake remediation. On the other hand, pit lakes may provide options for the storage of water during wet season in combination with the management of water quality of the pit lakes under respective climate conditions. Duthe et al. (2011) provide an example from Botswana. They point out the importance of good water quality in pit lakes because of the enormous pressure for the use of any surface water by humans and by wild life.

Since the diversion of river water and the use of mine water are limited in applicability, the addition of alkaline substances is an important alternative approach. It is robust and well established. The treatment of inflows may be an efficient approach if the sources of AMD are concentrated and easy to collect and treat. The treatment of discharge from pit lakes is just an ultimate option to protect water sources downstream. The respective pit lake remains without improvement. Biological approaches for the neutralization of pit lakes are probably limited to warm climates. However, even there they still need further development and approvals before they can become a well established approach.

Approach	Advantages	Limitations
Prevention	<ul> <li>very limited formation of acidity</li> <li>almost no maintenance necessary after mine closure</li> <li>high sustainability</li> </ul>	<ul> <li>large grain size may limit interaction between alkaline material and sulphide bearing material</li> <li>alkaline material has to be mined somewhere and has to be provided in big amounts</li> <li>more complicated handling of mine waste and more complicated logistics in the mine</li> <li>very detailed exploration necessary</li> </ul>
Filling and flushing with river water	<ul> <li>only limited maintenance necessary after neutralization of pit lake</li> <li>high sustainability</li> </ul>	<ul> <li>availability of river water (regarding quantity as well as quality) may be limited, in particular in arid regions</li> <li>requirements of river ecology and existing water use have to be considered</li> </ul>
Filling and flushing with mine water	<ul> <li>only limited maintenance necessary after neutralization of pit lake</li> <li>pit lake may serve as treatment facility for mine water under special circumstances</li> </ul>	<ul> <li>operating mines are needed in an acceptable distance to pit lakes</li> <li>quality of the mine water may require treatment</li> <li>requirements of regional water balance and existing water use have to be considered</li> <li>use of mine water is temporarily limited by the life time of the water providing mines</li> </ul>
Addition of alkaline substances	<ul> <li>safe technology with many experiences available from full scale application over decades</li> <li>different industrial alkaline products (e.g. limestone, lime, soda ash) are available with different solubility and solution kinetics (depending on chemical composition, grain size and potential addition of CO<sub>2</sub>) allowing for optimal adaptation to local requirements</li> <li>cheap alkaline waste products may be used</li> </ul>	<ul> <li>cost may become high if alkaline substances are not regionally available, e.g. due to regional geology or if pit lakes are located in remote regions</li> <li>different solubility and solution kinetics (depending on chemical composition, grain size and potential addition of CO<sub>2</sub>) have to be considered in designing particular applications</li> <li>already established aquatic communities may be damaged when applied against re-acidification by high pH (overdosage of highly alkaline materials like lime) or by overdosage of CO<sub>2</sub></li> <li>alkaline waste products may contain unacceptable high concentrations of potentially toxic trace elements</li> </ul>

 Table 2
 Comparative evaluation of different approaches for the remediation of pit lakes based on the above sections 5.1 and 5.2, Geller and Schultze (2012 a-e) and additional literature (signed by references)

Approach	Advantages	Limitations
Biological treatment	- naturally occurring processes are used and	- very limited success in full scale applications yet due to re-
using sulphate reduction	stimulated	oxidation of formed sulphides
	- organic waste materials and sewage may be	- probably limited to warm climates except application in
	used which are usually available even in	meromictic pit lakes
	remote regions	- high dosage of organic materials necessary
		- overdosage may result in the formation of toxic $H_2S$
Biological treatment	- successful application for metal removal in full	- safe burial of removed metals in the sediment of pit lakes only
using artificial	scale for one decade	demonstrated for meromictic pit lake in full scale application
eutrophication		- part of the settled metals and metalloids may be remobilized
		due to reductive dissolution of ferric iron minerals in the
		sediment
		- no case of successful neutralization has been reported yet
Treatment of inflowing		
and outflowing water		
a) Surface water	- receiving waters downstream are protected	- big reserves may be necessary for safe handling of high flow in
	- many experiences available from treatment of	terms of treatment capacity (inflow and outflow treatment) or
		storage capacity (outflow treatment)
	- all technologies for AMD treatment are	- pit lake remains actors in the case of treatment of outflow
h) Crown drugton	basically applicable	reactive homions were successfully tested and emplied only for
b) Groundwater	- groundwater down-gradient is protected	- reactive barriers were successfully tested and applied only for
		Gibert et al. 2002: Ludwig et al. 2002: Williams et al. 2002:
		Jarvis et al. 2002, Eudwig et al. 2002, Williams et al. 2002, Jarvis et al. 2006)
		- biological treatment by injection of organic substances and
		nutrient is not vet implemented in remediation practice and
		respective long-term experiences are missing
		- very limited options for detailed manipulation of flow and
		distribution of added substances
		- pit lake remains acidic in the case of treatment of outflow

Local conditions may often modify the above given general ranking. In many cases, combinations of different approaches may be the best option, applied in parallel or sequentially. E.g., the limited availability of river water may require the additional application of alkaline substances while the risk of re-acidification may be successfully handled by flushing with river water. The planned remediation and management of a chain of pit lakes in the Lusatian lignite mining district (Germany) is basically an example for this approach. This chain of pit lakes comprises from east to west the lakes Spreetal (V: 97·10<sup>6</sup> m<sup>3</sup>), Sabrodt (V: 27·10<sup>6</sup> m<sup>3</sup>), Bergen (V: 3·10<sup>6</sup> m<sup>3</sup>), Bluno (V: 64·10<sup>6</sup> m<sup>3</sup>), Neuwiese (V: 56·10<sup>6</sup> m<sup>3</sup>), Partwitz (V: 133·10<sup>6</sup> m<sup>3</sup>), and Sedlitz (V: 212·10<sup>6</sup> m<sup>3</sup>). River water and treated mine water is used for filling in addition to rising groundwater. Liming will be applied for initial neutralization and for the first phase of maintaining neutral conditions. The water for long term flushing will be diverted from the Spree River in the east. The Schwarze Elster River will receive the outflow in the west (Schultze et al. 2011c, see also Appendix 4).

# 6 Conclusions

The detailed investigation of the development of the water quality of Lake Goitsche during its filling indicated that the alkalinity of the river water provided the by far the most important contribution to the neutralization of the lake water. Other processes were not essential regarding alkalinity.

Regarding acidity, the inflow of groundwater and the erosion and elution of the bottom and the side walls of the former mine void were found to be important sources of acidity during filling. About 60% of the acidity to be neutralized until the lake water became fully neutral entered Lake Goitsche in that period.

For other pit lakes, the contribution of the different processes to neutralization may be different depending on local conditions, in particular when river water contributes much less to the filling of the lake than in the case of Lake Goitsche.

Inflowing groundwater, erosion and elution of the bottom and the side walls, the precipitation of iron and aluminium and the accompanying co-precipitation of phosphorus, trace metals and arsenic were the most important processes for the mentioned substances. Dilution by river water was relevant for (almost) non-reactive substances.

Eutrophication by phosphorus originating from the river water was not observed. It could be shown that future eutrophication caused by this phosphorus is very unlikely. The binding of the phosphorus in the sediment is sustained by the very high iron content of the sediment and a redox-driven cycle of iron and manganese. However, a long-lasting excessive load of phosphorus may cause eutrophication. This risk requires proper lake management in future.

Lake Goitsche, further examples from the Central German lignite mining district and examples from other mining regions indicate that filling with river water is a very efficient strategy to create pit lakes and to abate their acidification. Compared to alternative approaches, the filling with river water is probably the best option if enough river water is available. Otherwise, combination with other approaches should be considered.

## 7 References

- ATV (1997) ATV-Handbuch Biologische und weitergehende Abwasserreinigung. 4<sup>th</sup> ed. Ernst & Sohn, Berlin
- Aube B (2009) Treatment and monitoring of pit lakes. In: Proceedings Securing the Future 2009 & 8<sup>th</sup> ICARD, June 22-26, 2009, Skelleftea, Sweden, http://www.proceedingsstfandicard-2009.com/pdfer/Bernard\_Aube\_B4\_T3\_Treatment-and-monitoring-of-pitlakes.pdf
- Axler R, Larsen C, Tikkanen C, McDonald M, Yokom S, Aas P (1996) Water quality issues associated with aquaculture: A case study in mine pit lakes. Water Environment Research 68:995-1011
- Benthaus F-C, Uhlmann W (2006) Die chemische Behandlung saurer Tagebauseen in der Lausitz. Erfahrungen zur Kalkschlammresuspension im Tagebausee Koschen. In: Merkel B, Schaeben H, Wolkersdorfer C, Hasche-Berger A (eds) Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wissenschaftliche Mitteilungen Institut für Geologie Nr. 31/2006, TU Bergakademie Freiberg, Freiberg, Germany, 85-96.
- Berner RA (1970) Sedimentary pyrite formation. American Journal of Science 268:1-23
- Berner RA (1984) Sedimentary pyrite formation: an update. Geochimica Cosmochimica Acta 48:605-615
- Boehrer B, Schultze M, Ockenfeld K, Geller W (2005) Path of the 2002 Mulde flood through Lake Goitsche, Germany. Verhandlungen Internationale Vereinigung Limnologie 29:369-372
- Böhrer B, Heidenreich H, Schimmele M, Schultze M (1998) Numerical prognosis for salinity profiles of future lakes in the opencast mine Merseburg-Ost. International Journal of Salt Lake Research 7:235-260
- Campbell RS, Lind OT (1969) Water quality and aging of strip-mine lakes. Journal Water Pollution Control Federation 41:1943-1955
- Caraco NF, Cole JJ, Likens GE (1989) Evidence for sulfate-controlled phosphorus release from sediments of aquatis systems. Nature 341:316-18
- Castendyk DN, Eary LE (2009) The nature and the global distribution of pit lakes. In: Castendyk DN, Eary LE (eds) Mine pit lakes: Characteristics, predictive modeling, and sustainability. Society for Mining, Metallurgy, and Exploration SME, Littleton, 1-11
- Cooke GD, Welch EB, Peterson SA, Nichols SA (2005) Restoration and management of lakes and reservoirs. 3<sup>rd</sup> ed. Taylor & Francis, Boca Raton
- Cortina JL, Lagreca I, de Pablo J, Cama J, Ayora C (2003) Passive in situ remediation of metal-polluted water with caustic magnesia: Evidence from column experiments. Environmental Science Technology 37:1971–1977
- DEBRIV (2011) Braunkohle in Deutschland 2010. DEBRIV-Bundesverband Braunkohle, Köln.
- Dessouki TCE, Hudson JJ, Neal BR, Bogard MJ (2005) The effects of phosphorus additions on the sedimentation of contaminants in a uranium mine pit-lake. Water Research 39:3055-3061
- Duffek A, Langner C (2002) P-retention in an acidic mining lake under the influence of flooding with river water. Verhandlungen Internationale Vereinigung Limnologie 28:1717-1720
- Duis K, Prawitt O, Mattukat F (2001) Grenzen der Säuretoleranz einheimischer Fischarten unter den hydrochemischen Bedingungen der Lausitzer Tagebaurestseen. In: Dethlefsen V, Hilge V (eds) Aktuelle Probleme der Gewässerverschmutzung - Arbeiten des Deutschen Fischereiverbandes e.V., Heft 77. Deutscher Fischereiverband e.V., Hamburg, 67-85

- Duthe D, Mahoney JJ, Shchipansky AA, Terrell CL (2011) Assessment of the process of pit lake formation and associated geochemistry in open pits – Mapune gold mine, Botswana. In: Rüde TR, Freund A, Wolkersdorfer C (eds) Mine water - Managing the Challenge, Proceedings of the International Mine Water Association Congress 2011, 4-11 September, Aachen, Germany. RWTH Aachen University, Institute of Hydrogeology on behalf of the International Mine Water Association, Aachen, 511-516
- Eary LE (1999) Geochemical and equilibrium trends in mine pit lakes. Applied Geochememistry 14:963-987
- Eissmann L (2002a) Tertiary geology of the Saale-Elbe Region. Quaternary Science Reviews 21:1245-1274
- Eissmann L (2002b) Quaternary geology of eastern Germany (Saxony, Saxon-Anhalt, South Brandenburg, Thuringia), type area of the Elsterian and Saalian Stages in Europe. Quaternary Science Reviews 21:1275-1346
- Eyll-Vetter, M (2007) Betriebliche Umsetzung der Maßnahmen zur Reduzierung bergbaubedingten Säurepotentials in Kippen im Tagebau Garzweiler. Drebenstedt C, Struzina M (eds) Management bergbaubedingten Säurepotenzials. Institut für Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, Freiberg, 79-90
- Fischer R, Reißig H, Peukert D, Hummel J (1987) Untersuchungen zur Beeinflussung der Markasitverwitterung. Neue Bergbautechnik 17:60-64
- Fischer R, Luckner L, Reißig H, Peukert D, Roßbach B (1999) Hemmung der Pyrit- und Markasitverwitterung im Lausitzer und Mitteldeutschen Braunkohlebergbau. Hydrologie und Wasserbewirtschaftung 43:82-83
- Fisher TSR, Lawrence GA (2006) Treatment of Acid Rock Drainage in a meromictic mine pit lake. Journal Environmental Engineering 132:515-526
- Frömmichen R., Wendt-Potthoff K, Friese K, FischerR (2004) Microcosm Studies for Neutralization of Hypolimnic Acid Mine Pit Lake Water (pH 2.6). Environ Sci Technol 38:1877-1887
- Fuhrmann R (2005) Die Bernsteinlagerstätte Bitterfeld, nur ein Höhepunkt des Vorkommens von Bernstein (Succinit) im Tertiär Mitteldeutschlands. Zeitschrift deutsche Gesellschaft Geowissenschaften 156:517-530
- Fyson A, Nixdorf B, Steinberg CEW (1998a) Manipulation of the sediment-water interface of extremely acidic mining lakes with potatoes: Laboratory studies with intact sediment cores. Wat Air Soil Pollut 108:353-363
- Fyson A, Nixdorf B, Kalin M, Steinberg CEW (1998b) Mesocosm studies to assess acidity removal from acidic mine lakes through controlled eutrophication. Ecol Engin 10:229-245
- Fyson A, Nixdorf B, Kalin M (2006) The acidic lignite pit lakes of Germany Microcosm experiments on acidity removal through controlled eutrophication. Ecol Engin 28:288-295.
- Gammons CH, Tucci NJ (2012) The Berkeley Pit Lake, Butte, Montana. In: Geller W, Schultze M (eds) Acidic pit lakes – The legacy of coal and metal surface mining, Springer, Berlin (in press)
- Gast M, Schöpke R, Waklko M, Benthaus CF (2010) In situ aquifer treatment by microbial sulphate reduction. In: Mine water and innovative thinking, Proceedings of the International Mine Water Association Symposium 2010, 5-9 September, Sydney, Nova Scotia, Canada. Cape Breton University Press, Sydney, 119-122
- Geller W, Schultze M (2012a) Goals and conditions of remediation and management. In: Geller W, Schultze M (eds) Acidic pit lakes – The legacy of coal and metal surface mining. Springer, Berlin (in press)

- Geller W, Schultze M (2012b) Hydrological management and chemical in-lake treatments. In: Geller W, Schultze M (eds) Acidic pit lakes – The legacy of coal and metal surface mining. Springer, Berlin (in press)
- Geller W, Schultze M (2012c) Biological in-lake treatment. In: Geller W, Schultze M (eds) Acidic pit lakes – The legacy of coal and metal surface mining. Springer, Berlin (in press)
- Geller W, Schultze M (2012d) Treatments of in- and outflows. In: Geller W, Schultze M (eds) Acidic pit lakes – The legacy of coal and metal surface mining. Springer, Berlin (in press)
- Geller W, Schultze M (2012e) Conclusions and lessons learned. In: Geller W, Schultze M (eds) Acidic pit lakes The legacy of coal and metal surface mining. Springer, Berlin (in press)
- Geller W, Klapper H, Schultze M (1998) Natural and anthropogenic sulfuric acidification of lakes. In: Geller W, Klapper H, Salomons W (eds) Acidic mining lakes Acid mine drainage, limnology and reclamation. Springer, Berlin, 3-14
- Gibert O, de Pablo J, Cortina JL, Ayora C (2002) Treatment of acid mine drainage by sulphate-reducing bacteria using permeable reactive barriers: A review from laboratory to full-scale experiments. Re/Views Environmental Science and Bio/Technology 1:335-343
- Glombitza F (2001) Treatment of acid lignite mine flooding water by means of microbial sulfate reduction. Waste Management 21:197-203
- Golterman HL (2001) Phosphate release from anoxic sediments or 'What did Mortimer really write?'. Hydrobiologia 450:99-106
- Göttlicher J, Gasharova B (2000) Interactions of iron and sulfur bearing solid phases with water in surface coal mining pits and acidic mining lakes. In: Rammlmair D, Mederer J, Oberthür TH, Heimann RB, Pentinghaus H (eds) Applied Mineralogy in Research, Economy, Technology, Ecology and Culture 2. Balkema, Rotterdam 557-560
- Gröschke A, Uhlmann W, Rolland W, Grünewald U (2002) Hydrochemische Entwicklung Lausitzer Tagebauseen während der Flutung - Beispiel Gräbendorfer See. Hydrologie und Wasserbewirtschaftung 46:256-267
- Grüneberg B (2006) Akkumulation, Transformation und Remobilisierung von Phosphor aus Sedimenten saurer Tagebauseen. Doctoral thesis, Brandenburg Technical University, Cottbus
- Grützmacher G, Hindel R, Kantor W, Wimmer R (2001) Chemical investigations of aquifers affected by pyrite oxidation in the Bitterfeld lignite district. Waste Management 21:127-137
- Hemm M, Schlundt A, Kapfer M, Nixdorf B (2002) Beispiele für Neutralisierungsversuche am Steigenberger See (Bayern) und Zieselmaar (Nordrhein-Westfalen) - aus der UBA-Studie "Tagebauseen in Deutschland". In: Deneke R, Nixdorf B (eds) Gewässerreport Nr. 7 – Aktuelle Reihe 3/2002. Eigenverlag der BTU Cottbus, Cottbus, Germany, 37-42
- Herzsprung P, Boehrer B, Duffek A, Friese K, Hupfer M, Schultze M, v. Tümpling jr. W, van der Veen A (2010) Flood effects on phosphorus mobility in a river water filled pit lake Lake Goitsche (Germany). Limnologica 40:182-190.
- Höppner S, Rieger K, Scholz G (2006) Wirtschaftliche Bewertung innovativer, aktiver und passiver Wasseraufbereitungssysteme für saure Bergbaugewässer. In: Merkel B, Schaeben H, Wolkersdorfer C, Hasche-Berger A (eds) Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wissenschaftliche Mitteilungen Institut für Geologie Nr. 31/2006, TU Bergakademie Freiberg, Freiberg, 105-112
- Hoth N, Feldmann H, Rinker A, Glombitza F, Häfner F. (2005) Reductive processes within lignite dumps--chance of a long-term natural attenuation process. Geoderma 129:19-31

- Hupfer M, Fischer P, Friese K (1998). Phosphorus retention mechanisms in the sediment of an eutrophic mining lake. Water, Air, and Soil Pollution 108:341-352
- Jarvis AP, Moustafa M, Orme PHA, Younger PL (2006) Effective remediation of grossly polluted acidic, and metal-rich, spoil heap drainage using a novel, low-cost, permeable reactive barrier in Northumberland, UK. Environmental Pollution 143:261-268
- Jolas P (2007) Analyse der geplanten Bergbaufolgelandschaft der MIBRAG-Tagebaue in Bezug auf mögliche Umweltauswirkungen durch die Folgen von Stoffumwandlungsprozessen in Kippenmischböden. In: Drebenstedt C, Struzina M (eds) Management bergbaubedingten Säurepotenzials. Institut für Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, Freiberg, 101-108
- Junge FW, Jendryschik K (2003) Investigations into the Distribution of Element Concentrations in a Recent Dam and their Seasonal and Hydrographical Correlation (Bitterfelder Muldestausee, Saxony-Anhalt). Acta hydrochim. hydrobiol. 31:378-390
- Kamjunke N, Gaedke U, Tittel J, Weithoff G, Bell EM (2004) Strong vertical differences in plankton composition of extremely acidic lakes. Archiv Hydrobiologie 161:289-306
- Klapper H, Schultze M (1995) Geogenically acidified mining lakes living conditions and possibilities of restoration. International Revue ges. Hydrobiologie 80:639-653
- Kleeberg A, Grüneberg B (2005) Phosphorus mobility in sediments of acid mining lakes, Lusatia, Germany. Ecological Engineering 24:89-100
- Kleeberg A, Schapp A, Biemelt D (2008) Phosphorus and iron erosion from non-vegetated sites in a post-mining landscape, Lusatia, Germany: Impact on aborning mining lakes. CATENA 72:315-324
- Klemm W, Greif A, Broeckaert JAC, Siemens V, Junge FW, van der Veen A, Schultze M, Duffek A (2005) A study on arsenic and the heavy metals in the Mulde River system. Acta hydrochimica et hydrobiologica 33 (5), 475-491
- Koch H, Kaltofen M, Grünewald U, Messner F, Karkuschke M, Zwirner O, Schramm M (2005) Scenarios of water resources management in the Lower Lusatian mining district, Germany. Ecol Engin 24:49-57
- Koch H, Grünewald U, Kaltofen M, Kaden S (2009) Anpassungsstrategien für die Wasserbewirtschaftung auf den globalen Wandel im Einzugsgebiet der Spree. Korrespondenz Wasserwirtschaft 2:600-605
- Koch C, Graupner B, Werner F (2008) Pit Lake Treatment Using Fly Ash Deposits and Carbon Dioxide. In: Rapantova N, Hrkal Z (eds.) Mine Water and the Environment -Proceedings of the 10th International Mine Water Association 2008, June 2-5Congress, Karlovy Vary, Czech Republic. VSB-Technical University of Ostrava, Ostrava, 579-582
- Koch, T., V. Preuß, R. Schöpke and R. Koch. 2008. Pilotversuch zur kombinierten Förderung und Aktivierung alkalischer Sedimente mittels Druckluftheber (Airlift) und CO2-Einsatz. p. 39-46. In: B. Merkel et al. (eds) Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wiss. Mitt. Inst. f. Geologie Nr. 37/2008. TU Bergakademie Freiberg, Freiberg.
- Koschorreck M; Tittel J (2007) Natural alkalinity generation in neutral lakes affekted by acid mine drainage. J Environ Qual 36:1163-1172
- Koschorreck M, Bozau E, Frömmichen R, Geller W, Herzsprung P, Wendt-Potthoff K (2007) Processes at the Sediment Water Interface after Addition of Organic Matter and Lime to an Acid Mine Pit Lake Mesocosm. Env Sci Technol 41:1608-1614
- Koschorreck M, Boehrer B, Friese K, Geller W, Schultze M, Wendt-Potthoff K (2011) Creating and managing an anoxic water body in an acidic mine pit lake by addition of whey. Ecol Engin 1983-1989

- Kringel R, Nitsche C, Haferkorn B (2000) Limnology of sulfur-acidic lignite mining lakes III: approach and limitation in predicting chemical properties. Verhandlungen Internationale Vereinigung Limnologie 27:416-419
- Kumar RN, McCullough CD, Lund MA, Newport M (2011) Sourcing organic materials for pit lake bioremediation in remote mining regions. Mine Water Environ 30:296-301
- Laugwitz W (2004) Sanierungsgebiet Goitzsche Bericht zum Hochwasserereignis im August 2002. Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft mbH, Bitterfeld
- Lessmann D, Deneke R, Ender R, Hemm M, Kapfer M, Krumbeck H, Wollmann K, Nixdorf B (1999) Lake Plessa 107 (Lusatia, Germany) an extremly acidic shallow mining lake. Hydrobiologia 408/409:293-299.
- Lewandowski J, Schauser I, Hupfer M (2003) Long term effects of phosphorus precipitation with alum in hypereutrophic Lake Süsser See (Germany). Water Research 37:3194-3204.
- Lewis NM, Wangerud KW, Park BT, Fundingsland SD, Jonas JP (2003) Status of in situ treatment of Anchor Hill pit lake, Gilt Edge Mine Superfund site, South Dakota, USA.In: Farell T, Taylor G (eds), Proc, 6th ICARD, Australian Inst of Mining and Metallurgy, Carlton Victoria, Australia, p 779-788
- Litt T, Wansa S (2008) Quartär. In: Bachmann GH, Ehling B-C, Eichner R, Schwab M (eds) Geologie von Sachsen-Anhalt. E. Schweitzerbart#sche Verlagsbuchhandlung, Stuttgart, 293-325
- Lu M (2004) Pit lakes from sulphide ore mining, geochemical characterization before treatment, after liming and sewage sludge treatment cases studies at Rävlidmyran and Udden, Sweden. PhD Thesis, Lulea University of Technology, Lulea, Sweden.
- Luckner L (2006a) Lausitz: Sanierung des Wasserhaushalts. Teil 2: Das Problem mit der Wasserqualität. Die Neutralisation der sauren Restseen. Wasserwirtschaft-Wassertechnik 4/2006:10-16
- Luckner L (2006b) Lausitz: Sanierung des Wasserhaushalts. Teil 1: Die Lösung des Wassermengenproblems. Wasserwirtschaft-Wassertechnik 3/2006:33-37
- Ludwig RD, McGregor RG, Blowes DW, Benner SG, Mountjoy K (2002) A permeable reactive barrier for treatment of heavy metals. Ground Water 40:59-66
- Maiss M, Walz V, Zimmermann M, Ilmberger J, Kinzelbach W, Gläßer W (1998) Experimentelle Tracerstudien und Modellierungen von Austauschprozessen in einem meromiktischen Restsee (Hufeisensee), UFZ-Bericht 1/1998. Umweltforschungszentrum Leipzig-Halle, Leipzig
- Martin A, Crusius J, McNee JJ, Whittle P, Pieters R, Pedersen TF (2003) Field-scale assessment of bioremediation strategies for two pit lakes using limnocorrals. In: Farell T, Taylor G (eds), Proc, 6th ICARD, Australian Inst of Mining and Metallurgy, Carlton Victoria, Australia, p 529-539
- McCullough C, Lund MA, May JM (2008a) Field Scale Trials Treating Acidity in Coal Pit Lakes Using Sewage and Green Waste. In: Rapantova N, Hrkal Z (eds.) Mine Water and the Environment - Proceedings of the 10th International Mine Water Association 2008, June 2-5Congress, Karlovy Vary, Czech Republic. VSB-Technical University of Ostrava, Ostrava, 599-603
- McCullough C, Lund MA, May JM (2008b) Field-scale demonstration of the potential for sewage to remediate acidic mine waters. Mine Wat Environ 27:31-39
- Miller GC, Lyons WB, Davis A (1996) Understanding the water quality of pit lakes. Environmental Science Technology 30:118A-123A
- Müller H-S (1998) Tagebau Muldenstein. In: Chronik des Braunkohlebergbaus im Revier Bitterfeld. Bitterfelder Berglaute e.V., Bitterfeld, 199-223

- Nixdorf B, Hemm M, Schlundt A, Kapfer M, Krumbeck H (2001) Braunkohlentagebauseen in Deutschland: Gegenwärtiger Kenntnisstand über wasserwirtschaftliche Belange von Braunkohlentagebaurestlöchern, UBA Texte 35-01. Umweltbundesamt, Berlin
- Nixdorf B, Lessmann D, Deneke R (2005) Mining lakes in a disturbed landscape: Application of the EC Water Framework Directive and future management strategies. Ecological Engineering 24:67-73
- Nixdorf B, Uhlmann W, Lessmann D (2010) Potential for remediation of acidic mining lakes evaluated by hydrogeochemical modelling: Case study Grünewalder Lauch (Plessa 117, Lusatia/Germany). Limnologica 40:167-174
- Nordstrom DK (2009) Pitfalls and limitations of mineral equilibrium assumptions for geochemical modeling of water-rock interactions at mine sites. In: Proceedings Securing the Future 2009 & 8<sup>th</sup> ICARD, June 22-26, 2009, Skelleftea, Sweden, http://www.proceedings-stfandicard-2009.com/pdfer/Kirk\_Nordstrom\_B7\_T2\_Pitfalls-and-Limitations-of-Mineral-Equilibrium-Assumptions-for-Geochemical-Modeling-of-Water-Rock-Interactions-at-Mine-Sites.pdf
- Nordstrom DK, Alpers CN (1999) Geochemistry of acid mine waters. Reviews Economic Geology 6A:133-160
- Ölander B, Lu M, Alakangas L (2007) Effects of Liming a Permanently Stratified Pit Lake, Rävlidmyran, Northern Sweden. In: Beckett PJ, Spiers GA (eds) Proceedings (CD) of the 5<sup>th</sup> International Mining and the Environment Conference Sudbury 2007, October 22-24, 2007, Sudbury, Ontario, Canada
- Park B, Wangerud KW, Fundingsland SD, Adzic ME, Lewis MN (2006) In-situ chemical and biological treatment leading to successful water discharge from Anchor Hill pit lake, Gilt Edge Mine Superfund site, South Dakota, U.S.A. In: Barnhisel RI (ed), Proceedings 7<sup>th</sup> ICARD, American Soc of Mining and Reclamation, Lexington, KY, USA, 1065-1069
- Parsons JD (1964) Comperative limnology of strip-mine lakes. Verhandlungen Internat. Vereinigung Limnologie 15:293-298
- Peine A, Tritschler A, Küsel K, Peiffer S (2000) Elecron flow in an iron-rich acidic sediment evidence for an acidity-driven iron cycle. Limnology Oceanography 45:1077-1087
- Pelletie CA, Wen ME, Poling GW (2009) Flooding pit lakes with surface water. In: Castendyk DN, Eary LE (eds) Mine pit lakes: Characteristics, predictive modeling, and sustainability. Society for Mining, Metallurgy, and Exploration SME, Littleton, 187-202
- Pietsch W (1979) Zur hydrochemischen Situation der Tagebauseen des Lausitzer Braunkohlen-Reviers. Archiv Naturschutz Landschaftsforschung 19:97-115
- Pietsch W (1998) Colonization and development of vegetation in mining lakes of the Lusatian lignite area depending on water genesis. In: Geller W, Klapper H, Salomons W (eds) Acidic mining lakes - Acid mine drainage, limnology and reclamation. Springer, Berlin, 169-193
- Prein A, Mull R (1998) Oxygen as a limiting factor for pyrite weathering in the overburden of open pit lignite areas. In: Geller W, Klapper H, Salomons W (eds) Acidic mining lakes Acid mine drainage, limnology and reclamation. Springer, Berlin, 237-250
- Preuß V, Horn M, Koschorreck M, Luther G, Wendt-Potthoff K, Geller W (2007) In-lake bioreactors for the treatment of acid mine water in pit lakes. Advanced Material Research 20-21: 271-274
- Prikryl I (2010) Kvalita vody jezera Chabarovice na konci napousteni. In: Povodi Ohre (ed) Proceedings 14<sup>th</sup> Magdeburger Gewässerschutzseminar, October 4–6, 2010, Teplice, Czech Republic. Povodi Ohre, Chomutov, 97–100
- Prikryl I, Špacek J, Koza V (2011) First large mine lake successfully filled up in Czech republic. In: Rüde TR, Freund A, Wolkersdorfer C (eds) Mine water Managing the Challenge, Proceedings of the International Mine Water Association Congress 2011, 4-

11 September, Aachen, Germany. RWTH Aachen University, Institute of Hydrogeology on behalf of the International Mine Water Association, Aachen, 539-543

- Pusch D, Oh D-Y, Wolf S, Dumke R, Schröter-Bosin U, Höhne M, Röske I, Schreier E (2005) Detection of enteric viruses and bacterial indicators in German environmental waters. Archives of Virology 150:929-947
- Pust C, Schüppel B, Merkel B, Schipek M, Lilja G, Rabe W, Scholz G (2010) Advanced Mobile Inlake Technology (AMIT) – An efficien process for neutralisation of acid open pit lakes. In: Mine water and innovative thinking, Proceedings of the International Mine Water Association Symposium 2010, 5-9 September, Sydney, Nova Scotia, Canada. Cape Breton University Press, Sydney, 175-178
- Quasdorf P (1998) Der Tagebau "Goitsche" und seine Bernsteinlagerstätte. In: Chronik des Braunkohlebergbaus im Revier Bitterfeld. Bitterfelder Berglaute e.V., Bitterfeld, 224-39
- Rabe W, Uhlmann W (2006) Neue Erkenntnisse zur Anwendung von In-Lake-Verfahren für die Neutralisation saurer Bergbaufolgeseen. In: Merkel B, Schaeben H, Wolkersdorfer C, Hasche-Berger A (eds) Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wissenschaftliche Mitteilungen Institut für Geologie Nr. 31/2006, TU Bergakademie Freiberg, Freiberg, 97-104
- Rice KC, Herman JS (2012) Acidification of Earth: An assessment across mechanisms and scales. Applied Geochemistry 27:1-14
- Rodrigues GG, Scharf BW (2001) Review of benthic invertebrate fauna in extremely acidic environments (pH≤3). Mine Water Environment 20:114-121
- Rolland, W. 2007. Maßnahmen der VATTENFALL Europe Mining AG zur aktuellen und langfristigen Minderung der bergbaulichen Belastung von Grund- und Oberflächenwässern durch saure Kippenwässer. In: Drebenstedt C, Struzina M (eds) Management bergbaubedingten Säurepotenzials. Inst für Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, Freiberg, 91-100
- Rümmler F, Ritterbusch D, Schiewe S, Weichler F (2004) Fischereiliche Bewirtschaftung von Braunkohletagebauseen - Möglichkeiten und Besonderheiten. In: von Lukowicz M, Hilge V (eds) Fischereiliche Bewirtschaftung künstlich angelegter stehender Gewässer -Arbeiten des Deutschen Fischereiverbandes e.V., Heft 80. Deutscher Fischereiverband, Hamburg 29-57
- Sanchez Espana J, Lopez Pamo E, Santofimia Pastor E, Diez Ercilla M (2008) The acidic mine pit lakes of the Iberian Pyrite Belt: An approach to their physical limnology and hydrogeochemistry. Applied Geochemistry 23:1260-1287
- Schindler DW, Hecky RE, Findlay DL, Stainton MP, Parker BR, Paterson MJ, Beaty KG, Lyng M, Kasian SEM (2008) Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. PNAS 105:11254-11258
- Schipek M, Merkel B, Scholz G, Rabe W, Clauß D, Lilja G (2011) Recent results of the research project OILL (Optimizing In-Lake Liming). In: Rüde TR, Freund A, Wolkersdorfer C (eds) Mine water - Managing the Challenge, Proceedings of the International Mine Water Association Congress 2011, 4-11 September, Aachen, Germany. RWTH Aachen University, Institute of Hydrogeology on behalf of the International Mine Water Association, Aachen, 457-461
- Schöpke R, Koch R, Mangold S, Preuß V, Regel R, Striemann A (2006) Entwicklung und Erprobung eines Verfahrens zur Untergrundentsäuerung von Kippengrundwässern. Grundwasser 11:270-275
- Schöpke R, Gast, M, Walko M, Regel R, Koch R, Thürmer K (2011) Wissenschaftliche Auswertung von Sanierungsversuchen zur Untergrundsulfatreduktion im ehemaligen Lausitzer Bergbaurevier. Schriftenreihe Siedlungswasserwirtschaft und Umwelt Bd.21

- Schroeter A (1997) Modellhafte Sanierung von Altlasten am Beispiel des Tagebaurestloches Großkayna. Umweltbundesamt, Berlin
- Schultze, M. (2010). Pit lake. In: Bhattacharia J (ed) Project Environmental Clearance -Enginieering and management aspects for fuel and non-fuel minerals producing, user and construction industries. Wide Publishing, Kolkata, 1-56
- Schultze M, Boehrer B (2008) Development of Two Meromictic Pit Lakes a Case Study from the Former Lignite Mine Merseburg-Ost, Germany. In: Rapantova N, Hrkal Z (eds.) Mine Water and the Environment - Proceedings of the 10th International Mine Water Association 2008, June 2-5Congress, Karlovy Vary, Czech Republic. VSB-Technical University of Ostrava, Ostrava, 611-614
- Schultze M, Geller W (2004) Seewassergüte nach einer Flusswasserflutung Ergebnisse vom Goitschesee bei Bitterfeld -. Zeitschrift Angwandte Umweltforschung Sonderheft 14/2004:71-82
- Schultze M, Geller W, Wendt-Potthoff K, Benthaus F-C (2009) Management of water quality in German pit lakes. In: Proceedings Securing the Future 2009 & 8<sup>th</sup> ICARD, June 22-26, 2009, Skelleftea, Sweden, http://www.proceedings-stfandicard-2009.com/pdfer/Martin\_Schultze\_B4\_T3\_Management-of-water-quality-in-Germanpit-lakes.pdf
- Schultze M, Pokrandt KH, Hille W (2010) Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects. Limnologica 40:148-155
- Schultze M, Pokrandt K-H, Hille W (2011a) Erratum to: "Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects" [Limnologica 40 (2) (2010) 148–155]. Limnologica 41:78
- Schultze M, Geller W, Benthaus F-C, Jolas P (2011b) Filling and management of pit lakes with diverted river and mine water - German experiences. In: McCullough CD (ed) Mine Pit Lakes: Closure and Management. ACG Australian Centre for Geomechanics. Perth, 107-120
- Schultze M, Boehrer B, Friese K, Koschorreck M, Stasik S, Wendt-Potthoff K (2011c)
  Disposal of waste materials at the bottom of pit lakes. In: Fourie A, Tibbett M, Beersing
  A (eds) Proceedings of the 6<sup>th</sup> International Conference on Mine Closure, September 18-21, 2011, Lake Louise, Alberta, Canada, Volume 1: Mine Site Reclamation. ACG
  Australian Centre for Geomechanics, Perth, 555-564
- Schultze, M., Pokrandt, K.-H., Scholz, E., Jolas, P. (2011d): Use of mine water for filling and remediation of pit lakes. In: Rüde TR, Freund A, Wolkersdorfer C (eds) Mine water -Managing the Challenge, Proceedings of the International Mine Water Association Congress 2011, 4-11 September, Aachen, Germany. RWTH Aachen University, Institute of Hydrogeology on behalf of the International Mine Water Association, Aachen, 545-550.
- Schultze M, Boehrer B, Büttner O, Geller W, Herzsprung P (2012a) Processes during filling of pit lakes with river water as indicated by water quality changes in Lake Goitsche, Germany. (submitted to Science of the Total Environment)
- Schultze M, Boehrer B, Geller W (2012b) Morphology, age, and development of pit lakes. In: Geller W, Schultze M (2012) Acidic pit lakes – The legacy of coal and metal surface mines. Springer, Berlin (in press)
- Shevenell L, Connors KA, Henry CD (1999) Controls on pit lake water quality at sixteen open-pit mines in Nevada. Applied Geochemistry 14:669-687
- Steinberg, CEW (2003) Die bottom-up-Steuerung der Seeneutrophierung: Waldemar Ohle legt frühzeitig die Mechanismen der Seeneutrophierung in Vom Wasser niere. Von Wasser 101:91-102

- Storch A, Tröger K, Hoth N, Knöller K (2007) First evidence of active sulphate reduction in a lignite mine dump site at low pH values – the Plessa site. Advanced Materials Research 20-21:201-204
- Stottmeister U, Glässer W, Klapper H, Weissbrot E, Eccarius B, Kennedy C, Schultze M, Wendt-Potthoff K, Frömmichen R, Schreck P, Strauch G (1999) Strategies for remediation of former opencast mining areas in eastern Germany. In: Azcue JM (ed) Environmental impacts of mining activities: emphasis on mitigation and remediation. Springer, Berlin, 263-296
- Stottmeister U, Kuschk P, Wiessner A (2010) Full-scale bioremediation and long-term monitoring of a phenolic wastewater disposal lake. Pure Applied Chemistry 82:161-173
- Totsche O, Fyson A, Steinberg CEW (2006) Microbial alkalinity production to prevent reacidification of neutralized mining lakes. Mine Water Environment 25:204-213
- Trettin R, Freyer K, Strauch G, Treutler H-C, Gläßer W (1999) Isotopenuntersuchungen zur Wechselwirkung von Grund- und Oberflächenwasser in der Entstehungsphase von Bergbauseen. Grundwasser 4:3-10
- Trettin R, Gläßer W, Lerche I, Seelig U, Treutler H-C (2006) Flooding of lignite mines: isotope variations and processes in a system influenced by saline groundwater. Isotopes Environmental Health Studies 42:159-179
- Trettin R, Glaser HR, Schultze M, Strauch G (2007) Sulfur isotope studies to quantify sulfate components in water of flooded lignite open pits Lake Goitsche, Germany. Applied Geochemistry 22:69-89
- Uhlmann W, Gröschke A, Arnold I, Rolland W. (2007) Wirkungen von Eisenhydroxidschlämmen aus der Grubenwasseraufbereitung auf die Gewässerchemie von Tagebauseen in der Lausitz. In: Drebenstedt C, Struzina M (eds) Management bergbaubedingten Säurepotenzials. Inst für Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, Freiberg, 109-121
- Unger Y 2007. Using carbon dioxide to remediate acidic mining lakes. In: Cidu R, Frau F (eds) Water in mining environments of the International Mine Water Association 2007, 27-31 May 2007, Cagliari Italy. Universita degli Studi di Cagliari, Dipartimento di Scienze della Terra, Cagliari, 267-271
- Unger-Lindig Y, Merkel B, Schipek M (2010) Carbon dioxide treatment of low density sludge: a new remediation strategy for acidic mining lakes? Environmental Earth Sciences 60:1711-1722
- Vrzal D, Prikryl I, Truszyk A, Fafílková V (2011) Complicated water quality and stratification at the post-mining pit lake Medard near Sokolov, Czech Republic. In: Rüde TR, Freund A, Wolkersdorfer C (eds) Mine water - Managing the Challenge, Proceedings of the International Mine Water Association Congress 2011, 4-11 September, Aachen, Germany. RWTH Aachen University, Institute of Hydrogeology on behalf of the International Mine Water Association, Aachen, 551-555
- Weigelt R (2002) Wasseraufbereitung. In: Mutschmann J, Stimmelmayr F (eds) Taschenbuch der Wasserversorgung. 13<sup>th</sup> ed. Vieweg Verlag, Wiesbaden, 135-296
- Wen ME, Poling GW, Pelletier CA, Chapman J, Bingham ELJ (2006) Biological treatment of acid mine drainage at the Island Copper Mine pit lake: Results from an enclosure study. In: Barnhisel RI (ed), Proceedings 7th ICARD, American Soc of Mining and Reclamation, Lexington, KY, USA, 2332
- Werner F, Graupner B, Merkel B, Wolkersdorfer C (2006) Assessment of a treatment scheme for acidic mining lakes using CO<sub>2</sub> and calcium oxides to precipitate carbonates. In: Barnhisel RI (ed) Proceedings of 7th International Conference on Acid Rock Drainage, March 26-30, 2006, St. Louise. American Society of Mining and Reclamation (ASMR), Lexington, 2344-2353
- Wetzel R (2001) Limnology. Academic Press, San Diego

- Wiegand U, Schreck P, Schreiter P, Lerche I, Glaesser W (2003) Restoration of open pit lignite mining in the former GDR: Lessons learnt from Zwenkau. Energy Environment 14:437-450
- Williams RL, Mayer KU, Amos RT, Blowes DW, Ptacek CJ, Bain JG (2007) Using dissolved gas analysis to investigate the performance of an organic carbon permeable reactive barrier for the treatment of mine drainage. Applied Geochemistry 22:90-108
- Wimmer R, Pester L, Eissmann L (2006) Das bernsteinführende Tertiär zwischen Leipzig und Bitterfeld. Mauritania 19:373-421
- Wisotzky F (1994) Untersuchungen zur Pyritoxidation in Sedimenten des Rheinischen Braunkohlereviers und deren Auswirkungen auf die Chemie des Grundwassers.
   Besondere Mitteilungen zum Deutschen Gewässerkundlichen Jahrbuch Nr. 58. Landesumweltamt Nordrhein-Westfalen, Essen
- Wisotzky F (1998) Assessment of the extent of sulfate reduction in lignite mining dumps using thermodynamic equilibrium models. Water Air Soil Pollution 108:285-296
- Wisotzky F (2003) Saure Bergbauwässer (Acid Mine Drainage) und deren Qualitätsverbesserung durch Zugabe von alkalisch wirkenden Zuschlagstoffen zum Abraum – Untersuchungen im Rheinische Braunkohlerevier. Besondere Mitteilungen zum Deutschen Gewässerkundlichen Jahrbuch Nr. 61. Landesumweltamt Nordrhein-Westfalen, Essen
- Wisotzky F, Obermann P (2001) Acid mine groundwater in lignite overburden dumps and its prevention the Rhineland lignite mining area (Germany) Ecological Engineering 17:115-123
- Wisotzky F, Lenk S (2006) Darstellung und Evaluierung der Minderungsmaßnahmen zur Kippenwasserversauerung im Tagebau Garzweiler der RWE Power AG (Rheinisches Braunkohlerevier, Germany). In: Merkel B, Schaeben H, Wolkersdorfer C, Hasche-Berger A (eds) Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wissenschaftliche Mitteilungen Institut für Geologie Nr. 31/2006, TU Bergakademie Freiberg, Freiberg, 133-138
- Woelfl S, Tittel J, Zippel B, Kringel R (2000) Occurrence of an algal mass development in an acidic (pH 2.5), iron and aluminium-rich coal mining pond. Acta hydrochim. hydrobiol. 28:305-309
- Wolf S (2005) Evaluierung der hygienischen Wasserqualität unter besonderer Berücksichtigung von Bakteriophagen am Beispiel eines Tagebausees. PhD Thesis, Technische Universität Dresden, Dresden, Germany
- Wollmann K, Deneke R, Nixdorf B, Packroff G (2000) Dynamics of planktonic food webs in three mining lakes across a pH gradient (pH 2-4). Hydrobiologia 433:3-14
- Wyatt G, Miller F, Chermak J (2006) Innovative water treatment plant utilizing the South Mine Pit at the Copper Basin mining site in Tennessee, USA. In: Barnhisel RI (ed), Proceedings 7th ICARD, American Soc of Mining and Reclamation, Lexington, KY, USA, 2529-2539
- Younger PL, Banwart SA, Hedin RS (2002) Mine Water: Hydrology, Pollution, Remediation. Kluwer, Dordrecht

# **Appendix 1**

Schultze, M., Pokrandt, K.-H. and Hille, W. (2010): Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects. Limnologica 40 (2), 148-155. Schultze, M., Pokrandt, K.-H. and Hille, W. (2011): Erratum to: ''Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects'' [Limnologica 40 (2) (2010) 148–155]. Limnologica 41 (1), 78.

Contributions of M. Schultze: Concept 90% Writing 90% Figures 100%

Contents lists available at ScienceDirect

### Limnologica



journal homepage: www.elsevier.de/limno

### Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects

### Martin Schultze<sup>a,\*</sup>, Karl-Heinz Pokrandt<sup>b</sup>, Wolfram Hille<sup>b</sup>

<sup>a</sup> UFZ Helmholtz Centre for Environmental Research, Department of Lake Research, Brückstraße 3a, D-39114 Magdeburg, Germany <sup>b</sup> Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft, Leipzig, Germany

#### ARTICLE INFO

Article history: Received 9 September 2009 Accepted 13 November 2009

Dedicated to Prof. Dr. Walter Geller on the occasion of his 65th birthday anniversary

Keywords: Pit lake Lignite mining Acidification Remediation Eutrophication Trace contaminants Saline groundwater Water management

#### Introduction

Lakes and rivers are dominant inland surface waters and important parts of the landscape. Because of their properties, lakes have particular ecological and socio-economic functions as habitat for aquatic organisms, sites for fishery and recreation, etc. Artificial lakes and reservoirs should fulfil those functions as far as possible.

The region called "Central Germany", i.e., the area described by circles around Leipzig and Halle/Saale of about 150 km in diameter, is poor in natural lakes. Surface mining has changed this situation, mainly within the last 120 years. Many artificial lakes formed in former excavations of clay, sand and gravel, in former quarries and in former open-cast lignite mines when the respective operations ceased. This paper deals only with lakes in former open-cast lignite mines. For simplification, these lakes are generally called pit lakes in this paper.

There are about 140 pit lakes in the Central German lignite mining district today, i.e., about 28% of the pit lakes in all German lignite mining districts. Fig. 1 shows a map of the mining district including the lakes which are planned to form in the currently operating lignite mines.

#### ABSTRACT

About 140 pit lakes exist in Central Germany. These have resulted from lignite mining and are important parts of the post-mining landscape in the Central German lignite mining district. Their water quality is mainly influenced by the consequences of pyrite oxidation, i.e., acidification or results of natural or artificial neutralization. The major way of filling as well as a cheap and successful measure against acidification was the diversion of river water into the lakes or their filling with neutral water from mines still operating. Eutrophication, contamination by industrial pollutants and infection with pathogens imported with river water were found to be unimportant threats for the pit lakes in the Central German lignite mining district. Intrusion of naturally saline groundwater from deeper underground resulted in some cases in elevated concentrations of sodium chloride and in meromixis. The diverse uses of the lakes (e.g. recreation, nature conservation, water management) indicate that the pit lakes fulfil widely the typical functions of lakes in a landscape. The creation, the current state of water quality and lessons learned in water quality management are reported upon for the pit lakes of the Central German lignite mining district.

© 2009 Elsevier GmbH. All rights reserved.

When managing pit lakes, one may be faced with one or more of the following concerns with respect to water quality (Klapper and Schultze, 1995):

- Acidification caused by pyrite oxidation and the accompanying mobilization of acidity, iron and sulphate.
- Eutrophication caused by excessive import of phosphorus and nitrogen via river water for filling or flushing the lake or inappropriate lake use (e.g. excessive feeding in aquaculture).
- Contamination with industrial pollutants caused by groundwater inflow from industrial sites or waste deposits in the vicinity of pit lakes.
- Salinization by highly saline ground water.
- Infection by import of pathogens via river water which in turn was impacted by waste water.

In this paper, we report on the creation and the current characteristics of the lakes in the former lignite open-cast mines of the Central German lignite mining district. In addition to the description of the current conditions, lessons learned are presented.

#### Data sources

The data presented below originate from lake monitoring in spring 2007, as far as possible. Morphometric data were taken



<sup>\*</sup> Corresponding author. Tel.: +49 391 8109400; fax: +49 391 8109150. *E-mail address:* martin.schultze@ufz.de (M. Schultze).

<sup>0075-9511/\$ -</sup> see front matter  $\circledcirc$  2009 Elsevier GmbH. All rights reserved. doi:10.1016/j.limno.2009.11.006



**Fig. 1.** Map of the Central Germany mining district showing the pit lakes resulting from lignite mining, the major rivers and some towns for orientation. The lakes mentioned in the text are: 1 – Lake Concordia, 2 – Lake Bergwitz, 3 – Lake Gremmin, 4 – Lake Golpa IV, 5 – Lake Gröbern, 6 – Muldereservoir, 7 – Lake Goitsche, 8 – Lake Paupitzsch, 9 – Lake Seelhausen, 10 – Lake Hufeisen, 11 – Lake Grabschütz, 12 – Lake Zwochau, 13 – Lake Werbelin, 14 – Lake Wallendorf, 15 – Lake Rassnitz, 16 – Lake Geiseltal, 17 – Lake Runstedt, 18 – Lake Zwenkau, 19 – Lake Störmtal, 20 – Lake Hain-Haubitz, 21 – Lake Bockwitz, 22 – Lake Vollert Süd, 23 – Lake Borna, 24 – Lake Harthsee. The dotted black line west of Halle indicates the natural Lake Salziger See, which disappeared in 1890 due to dewatering operations of underground mining for copper. Dewatering of the lake basin is still performed although copper mining ceased and the underground galleries are flooded. The lake will be re-established in the future. Lignite mining districts in Germany: RD – Rhineland district, HeD – district of Hessen, HsD – district of Helmstedt, OD – district of Oberpfalz, CDG – Central German district, LD – Lusatian district.

from the web sites of Mitteldeutscher Seenkatalog (community of governmental and non-governmental institutions concerned with and interested in the sustainable development of lakes in Central Germany; www.mitteldeutscheseen.de) and of Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft (LMBV; state company responsible for the remediation of the former lignite mines in eastern Germany; www.lmbv.de). Water quality data were provided by the LMBV as far as the LMBV was responsible for water quality monitoring in 2007. The Landestalsperrenverwaltung des Freistaates Sachsen (authority of the federal state of Saxony responsible for the management of reservoirs and lakes in Saxony) provided data for pit lakes according to its responsibility. Data of other lakes were taken from web-published results of lake monitoring in the federal state of Saxonia-Anhalt in 2007 (www.sachsen-anhalt.de/LPSA/index.php?id=27953) or for some cases from a report on the status of German pit lakes by Nixdorf et al. (2001). The data from Nixdorf et al. (2001) can still be considered as representative since the respective pit lakes are older and show mainly seasonal variations in water quality but nearly no inter-annual trends. Less data were available from small and/or old lakes. Therefore, the small and/or old pit lakes are generally underrepresented in this study.

#### Creation

In the 19th century or in some cases even earlier, lignite mining began where lignite seams reached the surface. The exploitation was performed by hand with very simple technical devices. Accordingly, the first surface mines were very small. The same applies for the first pit lakes forming in such abandoned mines at the end of the 19th or at the beginning of the 20th century, e.g. the Gniester lakes south of Lake Bergwitz. The surface areas of these lakes are in the range of some 100 m<sup>2</sup> only and the depths are in the range of 10 m or less. The pressure of Pleistocene glaciation resulted in folding of Tertiary layers including lignite seams in the region of the Gniester lakes. The tops of the lignite folds reached the surface (Litt and Wansa, 2008). Larger excavators and improved dewatering technology resulted in larger mines and, consequently, larger pit lakes. However, relatively small lakes may form due to technical reasons and efficient overburden handling even today, e.g. Lake Zwochau and Lake Grabschütz.

Until the 1970s, the filling of the lakes was exclusively based on local runoff and rebound of the groundwater level in the Central German lignite mining district after ceasing dewatering operations. The so-called Muldereservoir with a maximal depth of 26 m and a total volume of  $118 \times 10^6$  m<sup>3</sup> was the first lake in the Central German lignite mining district, which was filled with river water in 1975 and 1976. For that process, a new artificial river bed was constructed on a 10 km long trench including two residual voids of the former mine Muldenberg (the latter forming the Muldereservoir today). The River Mulde still flows completely through this lake (Junge and Jendryschik, 2003). Due to this redirection of the River Mulde, mining became possible in the former floodplain. It resulted in the subbasins Muehlbeck and Doebern of Lake Goitsche and the so-called Pouch peninsula between them, which consists of an overburden dump inside the former mine (Trettin et al., 2007).

As a consequence of German re-unification, many open-cast lignite mines were closed at the beginning of the 1990s. Therefore, many pit lakes were created in the last nearly 20 years or are currently in filling (Krüger et al., 2002). For fast filling of the pit lakes, river water or water from mines still operating (Jolas, 1998) was used as far as possible. The initial reason for fast pit lake filling was the stabilization of the side walls of the mining voids which became the slopes of the lake basins. If the filling happens faster, they can remain steeper than in the case of slow filling, particularly if the water level in the rising lakes can be kept higher than the groundwater level in the surrounding aquifers until the final water level of the lakes is reached. The limitation or even prevention of lake acidification due to pyrite oxidation in the mined material became a second reason for fast filling (Klapper and Schultze, 1995; Jolas, 1998; Zschiedrich et al., 2007). This aspect will be discussed in more detail in the "Water chemistry" section.

The diversion of river water has been used for pit lake filling as follows: River Saale for Lake Runstaedt and Lake Geiseltal. River Weiße Elster for Lake Wallendorf. Lake Rassnitz and Lake Werbelin, River Mulde for the Muldereservoir, Lake Goitsche, Lake Seelhausen, Lake Gremmin and Lake Gröbern and River Selke for Lake Concordia. The diverted amount of water has been limited due to requirements of other already established water use (e.g. extraction of water for industrial purposes) and of ensuring the ecological functioning of the rivers. The overall amount of diverted river water was  $792 \times 10^6$  m<sup>3</sup>, accounting for 56.6% of the filling of the lakes in the period 1993–2007. In the same period,  $433 \times 10^6$  m<sup>3</sup> of water from dewatering of mines still operating was used for pit lake filling, accounting for 30.9% of the lake filling. Groundwater contributed 12.5%, i.e.,  $175 \times 10^6 \text{ m}^3$ (LMBV, 2008). The filling of the Muldereservoir is not included in the data above because it happened earlier. The flood water which passed into Lake Goitsche in August 2002 (Klemm et al., 2005) also had to be excluded due to the lack of possibilities for measuring its amount.

#### Morphometry and stratification

Fig. 2 gives an overview of the volumes, surface areas and maximal depths of the pit lakes in the Central German lignite mining district. The largest lake is Lake Geiseltal which is still in filling (75% of lake volume was filled as of December 31, 2008). Its final volume will be  $423 \times 10^6$  m<sup>3</sup>, its final surface area will be 18.5 km<sup>2</sup> and its maximal depth will be 80 m. In Fig. 2, Lake Geiseltal is included with its final morphometric data like all other lakes, which are still in filling or which are planned to be filled in the next few decades when the currently operating mines will be mined out.

The morphometry of the pit lakes is the combined result of geological conditions of the lignite deposits, the excavation technology, the filling rate and some local aspects such as planned use of the lake insofar as known and considered during mining. Eissmann (2002a, b) gives an overview of the geological conditions in the Central German lignite mining district. Fig. 3 shows a simplified schematic depiction of the typical arrangement of pit lakes, overburden dumps and untouched underground. The lignite is of Tertiary age and is imbedded and overlain by layers of clay, sand and gravel from the Tertiary and the Quaternary. Consequently, the steepness of the slopes of the mining voids and the lakes is limited because all these materials are unconsolidated rock. This is an important difference to pit lakes in hard rock environments, which are common in many other coal mining regions of the world and which usually occur at metal mining sites (e.g. Doyle and Runnells, 1997; Shevenell et al., 1999; Sanchez Espana et al., 2008).

An important aspect of the geology of the Central German lignite mining district is the occurrence of Permian and Triassic salt deposits (mainly sodium chloride and calcium sulphate) in the deeper underground, often more than 100 m below the lignite seams. In some cases, the widespread subrosion of these salt deposits formed wide depressions which were filled with lignite



**Fig. 3.** Schematic cross-section of a former open-cast lignite mine showing the spatial relation of pit lake, overburden dump and untouched underground, the typical relation of groundwater and lake level after groundwater rebound, and the location of pyrite.



Fig. 2. Frequency distribution of the morphometric data of the pit lakes in the Central German lignite mining district (as far as data are available from the sources mentioned above). All data represent the final water level of the pit lakes. The current data are smaller for lakes which are still in filling.

over a long period syn-genetically. In other cases, local depressions formed syn- or post-genetically resulting eventually in local depressions of the pit lake bottom if the lignite in these local depressions was excavated and the depressions were not filled with overburden. For more details see Eissmann (2002a, b).

Usually the pit lakes are dimictic or monomictic, depending on the occurrence of an ice cover in winter. However, meromictic lakes also developed. In some cases, the intrusion of saline water from the above-mentioned salt deposits is the reason for meromixis: Lake Hufeisensee. Lake Rassnitz and Lake Wallendorf (Stottmeister et al., 1999: Heidenreich et al., 1999: Trettin et al., 2006). In other cases, inflowing groundwater with high concentrations of dissolved solids due to pyrite oxidation and its consequences caused the formation and persistence of monimolimnia. Examples are Lake Goitsche (Boehrer et al., 2003) and Lake Störmtal. In Lake Vollert Süd, meromixis resulted from the filling of the lake with waste waters of coke production containing very high concentrations of organic contaminants (260-690 mg/L DOC, mainly consisting of poly-aromatic, humic-like substances and phenolic compounds; Stottmeister et al., 1998) and from sheltering the lake from wind impact by surrounding side walls and forest. The applied remediation technology (a sequence of flocculation with iron, neutralization and fertilization) stabilised the meromixis via accumulation of the organic contaminants in the monimolimnion and the sediment (Stottmeister, 2008).

#### Water chemistry

As already mentioned above, acidification due to pyrite  $(FeS_2)$  oxidation is the major problem of water quality in pit lakes. Pyrite occurs in the Tertiary strata and, consequently, also in the overburden dumps (Fig. 3) and is the result of microbial reduction of iron and sulphur during and after the formation of the lignite. Pyrite is stable as long it remains under anoxic conditions. The dewatering of the underground and the excavation of the overburden and the lignite cause aeration of the pyrite and thus its microbial oxidation during and after mining. The grain size of the pyrite particles is usually in the range of millimetres. The following equations describe the pyrite oxidation and the production of acidity (only simplified net-reactions; for more details see e.g. Evangelou, 1995)

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4SO_4^{2-} + 4H^+$$
(1)

$$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+$$
(2)

$$4Fe^{2+} + O_2 + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O$$
(3)

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^+$$
 (4)

After start of the oxidation process via reaction (1), usually reaction (2) becomes the dominating pathway of pyrite oxidation (Evangelou, 1995). At least a part of the acidity produced by pyrite oxidation is already neutralized underground along its transport by trickling water and groundwater before the water enters the pit lake. The following reactions and processes are the most important ones consuming acidity along the path from pyrite oxidation sites to pit lakes (only simplified net-reactions; for more details see e.g. Stumm and Morgan, 1996)

$$HCO_3^- + H^+ \rightarrow H_2O + CO_2 \tag{5}$$

$$CaCO_3 + 2H^+ \rightarrow Ca^{2+} + H_2O + CO_2$$
 (6)

$$XM + nH^+ \rightarrow XH + M^{n+} \tag{7}$$

$$KAlSi_{3}O_{8} + 7H_{2}O + H^{+} \rightarrow Al(OH)_{3} + 3HSiO_{4} + K^{+}$$
(8)

Reaction (5) happens in the groundwater and in the aqueous phase of the soil (buffering of acidity by bicarbonate) whereas Eqs. (6)–(8) describe buffering reactions between the underground solids and the acid trickling water or groundwater: buffering by carbonates (6), by ion exchange ((7), whereby X is e.g. a clay mineral and M is a metal) or by silicates (8). However, since iron is mainly transported as ferrous iron by the groundwater entering the pit lake due to the availability of oxygen in the pit lakes.

Initially, about 50% of the pit lakes in the Central German lignite mining district were acid or had the potential to turn acidic without special measures. The other about 50% of the pit lakes never had acid water due to local geological conditions (relatively low pyrite content combined with relatively high content of carbonates in the overburden). Fig. 4 shows that today a much higher percentage of the lakes are neutral. This is mainly the result of filling with river water and well-buffered water from dewatering of operating mines. Lake Bockwitz was neutralized by addition of soda ash (Na<sub>2</sub>CO<sub>3</sub>; Neumann et al., 2008; Rönicke et al., this issue) whereas Lake Hain-Haubitz was neutralized by a combination of filling with water from operating mines and addition of lime. A number of initially acid pit lakes became neutral without special remediation measures. Experiences from these pit lakes showed that the neutralizations without special measures may require periods varying from a few years to some decades. Examples are Lake Nenkersdorf, which became neutral within 5 years, and Lake Bergwitz, where neutralization required about 25 years after its filling by groundwater and local runoff (Jordan and Weder, 1995). The mechanisms of neutralization in such cases are (i) natural washout and flushing of the lakes and of the underground where the acidifying substances originate from, (ii) the neutralization by inflow of bicarbonate with naturally neutral groundwater and (iii) microbial sulphate reduction. The natural occurrence of sulphate reduction was reported for German pit lakes e.g. by Peine and Peiffer (1998) and for ground water in overburden dumps by Hoth et al. (2005). However, the observed net rates of sulphate reduction are small and, therefore, the contribution to neutralization is limited.

Although the majority of pit lakes are neutral today (Fig. 4) and the remaining acid lakes in the Central German lignite mining district are expected to be neutralized within the next few years, acidification is still an important threat for the lakes. Ongoing inflow of groundwater with high loads of products of pyrite oxidation (mainly sulphate and iron) into the lakes may cause socalled re-acidification after initially successful neutralization of the lake water (via reactions (3) and (4)). Concentrations of up to 3 g/L iron were found in groundwater in overburden dumps (Mansel et al., 2007). Re-acidification occurs if the inflow of acidity exceeds the stock of alkalinity of the lakes and its natural recharge (inflow of neutral and buffered water, internal production of alkalinity by iron and sulphate reduction). This may require continued or frequently repeated lake neutralization in the future, e.g. the continued addition of soda ash for Lake Bockwitz (Rönicke et al., this issue). Another option might be the permanent flushing of the respective lakes by neutral river water as has been done for Lake Senftenberg in the Lusatian lignite mining district since 1977 (Werner et al., 2001), given that rivers for permanent diversion of water are nearby and have appropriate water quality. About 20% of the pit lakes of the Central German lignite mining district are threatened by re-acidification, based on current knowledge. The time frame of potential re-acidification can be expected to be in a range similar to the natural neutralization of pit lakes without special measures as mentioned above.



Fig. 4. Frequency distribution of chemical water properties related to acidification of the pit lakes in the Central German lignite mining district. Iron and manganese are given as total dissolved iron or manganese, respectively.

Acidity (detected by titration with 0.1 M NaOH to pH 8.2) and alkalinity (detected by titration with 0.1 M HCl to pH 4.3) indicate the degree of acidification present in spring 2007, or the buffering capacity of the lake waters insofar as they were neutral at that time, respectively, (Fig. 4). The alkalinity is usually low due to consumption of carbonates during neutralization and due to continuing inflow of acidity-bearing groundwater.

The concentrations of iron and aluminium also reflect the remaining acidification (Fig. 4). Higher concentrations are restricted to still-acid lakes or monimolimnia of meromictic lakes in the case of iron (not included in Fig. 4). However, iron still plays an important role for the water quality and its future development in the pit lakes because of its interaction with the phosphorus cycle in the lakes.

The manganese concentrations in the pit lakes are elevated under acid conditions but very low under neutral conditions (Fig. 4). Compared to the concentrations of iron and aluminium, the manganese concentrations are too low to make manganese an important contributor to acidity, even under acid conditions.

Calcium and magnesium concentrations reflect the consequences of buffering the acidity produced by pyrite oxidation (dissolution of carbonates, ion exchange, dissolution of silicates, etc.), the initial composition of minerals in the mined area and the source area of the current lake water (Fig. 4). Silicon concentrations are elevated in acid pit lakes (Fig. 4). Once the lake water is neutralized, silicon concentrations decrease, at least partially due to the improved living conditions for diatoms. Sulphate is clearly the dominating anion in the lake water (Fig. 4). It results mainly from pyrite oxidation, even in lakes which were never acid. Its concentrations are high compared to waters not impacted by lignite mining in the region.

The higher concentrations of chloride are the result of the previously mentioned intrusion of saline groundwater (Fig. 4). As with iron, the monimolimnia of the meromictic lakes are not included in Fig. 4. The chloride concentrations in the monimolimnia of Lake Hufeisensee, Lake Wallendorf and Lake Rassnitz reach 8.8, 48.1 and 14.2 g/L, respectively.

The electrical conductivity indicates the generally relatively high concentrations of dissolved solids in the pit lake water, mainly sulphate, calcium and magnesium and, in the case of acid lakes with pH < 2.8, iron (Fig. 4). The highest values of electrical conductivity originate from the lakes, which are influenced by intrusion of saline groundwater. In these lakes, chloride and sodium dominate the electrical conductivity. The values of the monimolimnia of the meromictic examples of these lakes are again not included in Fig. 4.

Phosphorus concentrations of the lake water are low (Fig. 5), even in the case of filling with river water containing more than 100 µg/L of total phosphorus (TP). The relatively high concentration of iron in the lake sediment and in its pore water and the permanent import of iron with inflowing groundwater are the major reasons: the phosphorus is adsorbed onto the surfaces of precipitates of iron and buried in the sediment when the iron precipitates settle to the sediment (Duffek and Langner, 2002; Kleeberg and Grüneberg, 2005). According to the usually low phosphorus concentrations, the concentrations of chlorophyll a are also small (Fig. 5). As a consequence, in most German pit lakes eutrophication does not pose a risk to water quality requiring special measures and strategies (Lessmann et al., 2003; Schultze et al., 2005), in contrast to initial assumptions by Klapper and Schultze (1995). The only eutrophic pit lake is Lake Golpa IV in the Central German lignite mining district. In this case, eutrophication results from using the lake for intermediate storage of river water from the River Mulde for cooling purposes in a power plant over a period of decades (Hupfer et al., 1998).

The concentration of ammonium (Fig. 5) is often relatively high in acid pit lakes due to inhibition of nitrification (Prosser, 1987). The low concentrations of nitrate (Fig. 5) are the result of missing sources such as explosives or intense agriculture. Mining is done without blasting and intense agriculture is not common in the post-mining landscape of Central Germany due to limited fertility of the soils at the dump sites. Even in the case of filling with nitrate-bearing river water, rapid removal of nitrate from the lake water resulted in low nitrate concentrations after finishing input of river water. Primary production and denitrification in the lake sediment are suspected to be the most important processes for this removal. Although the water of the pit lakes is usually well oxygenated throughout the year (except the permanently anoxic monimolimnia of meromictic pit lakes), anoxic conditions were found to be common in pit lake sediments (Peine and Peiffer, 1998; Meier et al., 2004; Blodau, 2006).

The concentrations of dissolved organic carbon (DOC) are in the range of eutrophic natural lakes (3–34 mg/L; Thurman, 1985). Since the ranges of concentrations of phosphorus and chlorophyll a are low, the DOC found is probably not the result of autochthonous primary production. The DOC may originate from remains of lignite spread in the overburden dumps or in the aquifers which are feeding the pit lakes.

Trace contaminants are not presented in detail here. Heavy metals and trace elements mobilized due to acidification may reach considerable concentrations in acid pit lakes (Zn and Ni:  $< 500 \ \mu g/L$ ; As, Cu, Cd, Cr and Pb:  $< 10 \ \mu g/L$ ; Hg  $< 1 \ \mu g/L$ ). Under neutral conditions, however, they are usually precipitating or removed by co-precipitation with iron or aluminium (Klapper and Schultze, 1995). Organic contamination of groundwater in the neighbourhood of Lake Goitsche did not affect the lake water quality due to changes in flow direction of the groundwater as a result of the flood event in 2002 (Wycisk et al., 2005). Contaminants leaving a waste deposit at the southern margin of



Fig. 5. Frequency distribution of concentrations of substances related to the biological productivity of the pit lakes in the Central German lignite mining district (SRP – soluble reactive phosphorus, TP – total phosphorus).

Lake Hufeisensee via groundwater are accumulated in the monimolimnion of the lake and the respective sediment (Stottmeister et al., 1999). Therefore, they do not affect the mixolimnion. Ammonium entering Lake Runstedt from the waste deposit at the lake bottom (>300 mg/L ammonium were found in the pore water of the waste) is oxidized by naturally occurring microbial communities in the lake. Hypolimnetic aeration was implemented in order to ensure the availability of sufficient oxygen. Lake Vollert Süd, its contaminants and its remediation were already described in the section on stratification. The enormous amounts of organic carbon originally restricted the Secchi depth to 3 cm (Stottmeister et al., 1999). Today, the mixolimnion of the lake has a good water quality. The contaminants are accumulated in the lake sediment and in the monimolimnion (Stottmeister, 2008).

#### Functioning of pit lakes in the landscape

Due to their total surface area of around 140 km<sup>2</sup> today, the sustainable use of the pit lakes is an important ecological and socio-economical factor for the Central German lignite mining district. Typical functions of lakes in a landscape are as habitat for aquatic and amphibian organisms, for larval stages of insects and for water fowl, as water source for wildlife, as geochemical sink and as providers of sites for human use, e.g. water-borne recreation, fishery, aquaculture, flood management and water storage.

The pit lakes of the Central German lignite mining district fulfil these functions widely, as indicated by their use. The major use of the pit lakes is for recreational purposes (Linke and Schiffer, 2002). The above-mentioned concern of infection, caused by the import of pathogens with river water, was found to be a much smaller risk than initially expected. Pathogens are removed from lake water relatively quickly by sedimentation, decay and dilution (Pusch et al., 2005; Wolf, 2005). Many lakes also became part of protected areas for nature conservation (e.g. Lake Rassnitz, Lake Paupitzsch, Lake Gremmin). A few lakes are used for fish breeding (e.g. the Muldereservoir). In the majority of the pit lakes, however, fishery is done only on the basis of the natural development of the fish community after an initial stocking. A few lakes are used for purposes of water management, mainly flood protection (e.g. Lake Borna, Lake Zwenkau after reaching planned water level). The function as geochemical sink is most impressive in the Muldereservoir due to the permanent through-flow of the complete River Mulde. It acts as a sink for heavy metals originating from abandoned ore mines in the Ore Mountains at the Czech-German border. In this way, the River Elbe is protected from strong contamination (Klemm et al., 2005).

#### Summary and lessons learned

Surface mining for lignite resulted in a new artificial lake district in the Central German lignite mining district which is poor in natural lakes. The pit lakes are an important part of the postmining landscape and fulfil ecological as well as socio-economic functions. The major concern with respect to water quality, acidification resulting from mining-induced pyrite oxidation, has been managed successfully for the majority of the pit lakes. The filling with water from external sources, i.e., rivers or dewatering operations of mines still operating, was the most important strategy and by far the cheapest one (Höppner et al., 2006).

However, some lakes required additional or alternative measures in order to become neutral. The addition of soda ash and of lime at the lake surface was applied. The lessons learned from these treatments are that (i) a complete investigation and quantification of all sources of acidity is necessary for successful planning of neutralization, (ii) the parts of the underground which were aerated during mining or remain aerated in the long run in the immediate vicinity of the lakes may be a major source of acidity for a long time, and (iii) the wide spreading of the applied alkaline substances may improve the efficiency of neutralization (Neumann et al., 2008).

"Doing nothing", i.e., simply waiting for natural neutralization as in earlier times, is no longer accepted by the responsible authorities for new acid pit lakes or acid pit lakes having an outflow. Today, such a strategy is restricted to lakes which are not acidified.

Eutrophication, contamination with industrial hazardous substances and infection of lake water by pathogens via inflow of river water were found to be of low importance in the Central German lignite mining district. Salinization by intrusion of naturally saline groundwater is restricted to a few cases according to the geological conditions. The monimolimnia formed by the intruding saline groundwater act as sinks for phosphorus and other contaminants (Stottmeister et al., 1999; Schultze and Boehrer, 2008).

In some cases, the long-term inflow of acidifying groundwater from dump sites may result in re-acidification. Common treatment options (addition of alkaline substances, diversion of river water) may be applied in order to maintain the neutral state of the lakes, or new, more efficient options may be developed based on current research and scientific evaluation of the experiences made in Germany and worldwide (Schultze et al., 2009; Castendyk and Eary, 2009). However, further learning from past and current management practice for future management in Germany or other countries requires detailed documentation of applied methods, adequate monitoring of currently treated lakes and publication of results and lessons learned.

#### Acknowledgements

The study is kindly dedicated to Prof. Dr. W. Geller who promoted research on water quality of pit lakes during his work as head of the UFZ Department of Lake Research in Magdeburg, Germany from 1992 to 2008.

The Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft mbH (LMBV) and the Landestalsperrenverwaltung des Freistaates Sachsen (LTV) are acknowledged for providing data on water quality of many pit lakes. Kathleen Rogge is kindly acknowledged for her help in handling and analyzing data. Dr. Wolf von Tümpling made very helpful comments on an earlier version of this paper. The authors also wish to thank the reviewers for their comments, especially Prof. Dr. Devin Castendyk, whose very helpful comments considerably improved the paper.

#### References

- Blodau, C., 2006. A review of acidity generation and consumption in acidic coal mine lakes and their watersheds. Sci. Total Env. 369, 307–332.
- Boehrer, B., Schultze, M., Liefold, S., Behlau, G., Rahn, K., Frimel, S., Kiwel, U., Kuehn, B., Brookland, I., Büttner, O., 2003. Stratification of mining Lake Goitsche during flooding with river water. In: Tailings and Mine Waste '03. Swets & Zeitlinger, Lisse, pp. 223–231.
- Castendyk, D.N., Eary, L.E. (Eds.), 2009. Management Technologies for Metal Mining Influenced Water, vol. 3 – Mine Pit Lakes: Characteristics, Predictive Modeling, and Sustainability. Society for Mining, Metallurgy, and Exploration, Littleton.
- Doyle, G.A., Runnells, D.D., 1997. Physical limnology of existing mine pit lakes. Min. Eng. 49, 76–80.

- Duffek, A., Langner, C., 2002. P retention in an acidic mining lake under the influence of flooding with river water. Verh. Internat. Verein. Limnol. 28, 1717–1720.
- Eissmann, L., 2002a. Tertiary geology of the Saale-Elbe Region. Quat. Sci. Rev. 21, 1245–1274.
- Eissmann, L., 2002b. Quaternary geology of eastern Germany (Saxony, Saxon-Anhalt, South Brandenburg, Thuringia), type area of the Elsterian and Saalian Stages in Europe. Quat. Sci. Rev. 21, 1275–1346.
- Evangelou, V.P., 1995. Pyrite Oxidation and its Control. CRC Press, Boca Raton.
- Heidenreich, H., Boehrer, B., Kater, R., Henning, G., 1999. Gekoppelte Modellierung geohydraulischer und limnophysikalischer Vorgänge in Tagebaurestseen und ihrer Umgebung. Grundwasser 4, 49–54.
- Höppner, S., Rieger, K., Scholz, G., 2006. Wirtschaftliche Bewertung innovativer, aktiver und passiver Wasseraufbereitungssysteme für saure Bergbaugewässer.
  In: Merkel, B., Schaeben, H., Wolkersdorfer, C., Hasche-Berger, A. (Eds.), Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen und Entwicklungen. Wiss. Mitt. Inst. f. Geologie Nr 31/2006. TU Bergakademie Freiberg, Freiberg, pp. 105–112.
- Hoth, N., Feldmann, H., Rinker, A., Glombitza, F., Häfner, F., 2005. Reductive processes within lignite dumps – chance of a long-term natural attenuation process. Geoderma 129, 19–31.
- Hupfer, M., Fischer, P., Friese, K., 1998. Phosphorus retention mechanisms in the sediment of a eutrophic mining lake. Water Air Soil Pollut. 108, 341–352.
- Jolas, P., 1998. Utilization of drainage water to fill residual lakes. BRAUNKOHLE Surf. Min. 50, 337–345.
- Jordan, H., Weder, H.-J., 1995. Hydrogeologie Grundlagen und Methoden. F. Enke Verlag, Stuttgart.
- Junge, F.W., Jendryschik, K., 2003. Investigations into the distribution of element concentrations in a recent dam and their seasonal and hydrographical correlation (Bitterfelder Muldestausee, Saxony-Anhalt). Acta Hydrochim. Hydrobiol. 31, 378–390.
- Klapper, H., Schultze, M., 1995. Geogenically acidified mining lakes living conditions and possibilities of restoration. Int. Rev. ges. Hydrobiol. 80, 639–653.
- Kleeberg, A., Grüneberg, B., 2005. Phosphorus mobility in sediments of acid mining lakes, Lusatia, Germany. Ecol. Eng. 24, 89–100.
   Klemm, W., Greif, A., Broeckaert, J.A.C., Siemens, V., Junge, F.W., van der Veen, A.,
- Klemm, W., Greif, A., Broeckaert, J.A.C., Siemens, V., Junge, F.W., van der Veen, A., Schultze, M., Duffek, A., 2005. A study on arsenic and the heavy metals in the Mulde River system. Acta Hydrochim. Hydrobiol. 33, 475–491.
- Krüger, B., Kadler, A., Fischer, M., 2002. The creation of post-mining landscapes of lignite mining in the new federal states. Surf. Min. – Braunkohle other miner. 54, 161–169.
- Lessmann, D., Uhlmann, W., Grünewald, U., Nixdorf, B., 2003. Sustainability of the flooding of lignite mining lakes as a remediation technique against acidification in the Lusatian mining district, Germany. In: Farell, T., Taylor, G. (Eds.), Proceedings of the 6th International Conference on Acid Rock Drainage (Cairns, Queensland, July 14–17, 2003). The Australian Institute of Mining and Metallurgy Publication Series No 3/2003. The Australian Institute of Mining and Metallurgy, Carlton Victoria, pp. 521–527.
- Linke, S., Schiffer, L., 2002. Development prospects for the post-mining landscape in Central Germany. In: Mudroch, A., Stottmeister, U., Kennedy, C., Klapper, H. (Eds.), Remediation of Abandoned Surface Coal Mining Sites. Springer, Berlin, pp. 111–145.
- Litt, T., Wansa, S., 2008. Quartär. In: Bachmann, G.H., Ehling, B.-C., Eichner, R., Schwab, M. (Eds.), Geologie von Sachsen-Anhalt. E. Schweizerbart'sche Verlagsbuchhandlung, Suttgart, pp. 293–325.
- LMBV, 2008. Sanierungsbericht 2007. Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft, Senftenberg.
- Mansel, H., Büttcher, H., Tienz, B.-S., Jolas, P., 2007. Exfiltration eisenbelasteter Grundwässer in die Pleiße südlich von Leipzig - Ergebnisse einer vorbereitenden Studie. In: Drebenstedt, C., Struzina, M. (Eds.), Management bergbaubedingten Säurepotenzials. Institut f. Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, pp. 155–165.
- Meier, J., Babenzien, H.-D., Wendt-Potthoff, K., 2004. Microbial cycling of iron and sulfur in sediments of acidic and pH-neutral mining lakes in Lusatia (Brandenburg, Germany). Biogeochemistry 67, 135–156.
- Neumann, V., Nitsche, C., Pokrandt, K.-H., Tienz, B.-S., 2008. Quantifizierung des Aziditätsstroms in einen neutralisierten Tagebausee zu Beginn der Nachsorge. In: Merkel, B., Schaeben, H., Hasche-Berger, A. (Eds.), Behandlungstechnologien für bergbaubeeinflusste Wässer + GIS-Geowissenschaftliche Anwendungen

und Entwicklungen. Wiss. Mitt. Inst. f. Geologie Nr. 37/2008. TU Bergakademie Freiberg, Freiberg, pp. 73–80.

- Nixdorf, B., Hemm, M., Schlundt, A., Kapfer, M., Krumbeck, H., 2001. Braunkohlentagebauseen in Deutschland: Gegenwärtiger Kenntnisstand über wasserwirtschaftliche Belange von Braunkohlentagebaurestlöchern. UBA-Texte 35-01. Umweltbundesamt, Berlin.
- Peine, A., Peiffer, S., 1998. In-lake neutralization of acid mine lakes. In: Geller, W., Klapper, H., Salomons, W. (Eds.), Acidic Mining Lakes. Springer-Verlag, Berlin, pp. 47–64.
- Prosser, J.I., 1987. In: Nitrification. IRL Press, Oxford.
- Pusch, D., Oh, D.-Y., Wolf, S., Dumke, R., Schröter-Bosin, U., Höhne, M., Röske, I., Schreier, E., 2005. Detection of enteric viruses and bacterial indicators in German environmental waters. Arch. Virol. 150, 9292–9947.
- Rönicke, H., Schultze, M., Nitsche, C., Neumann, V., Tittel, J., 2010. Changes of the plankton community composition during chemical neutralisation of pit lake Bockwitz. Limnologica, 40, this issue, doi:10.1016/j.limno.2009.11.05.
- Sanchez Espana, J., Lopez Pamo, E., Santofimia Pastor, E., Diez Ercilla, M., 2008. The acidic mine pit lakes of the Iberian Pyrite Belt: an approach to their physical limnology and hydrogeochemistry. Appl. Geochem. 23, 1260–1287.
- Schultze, M., Boehrer, B., 2008. Development of two Meromictic Pit Lakes a case study from the Former Lignite Mine Merseburg-Ost, Germany. In: Rapantova, N., Hrkal, Z. (Eds.), Mine Water and the Environment – Proceedings of the 10th IMWA Congress, 2–5 June, 2008 Karlovy Vary, Czech Republic. VSB-Technical University of Ostrava, Faculty of Mining and Geology, Ostrava, pp. 611–614.
- Schultze, M., Boehrer, B., Duffek, A., Herzsprung, P., Geller, W., 2005. Introduction of river water as a tool to manage water quality in pit lakes. In: Loredo, J., Pendas, F. (Eds.), Proceedings of the 9th International Mine Water Association Congress, September 5–7, 2005, Oviedo, Spain. Departemento de Explotacion y Prospeccion de Minas, University of Oviedo, Oviedo, pp. 273–279.
- Schultze, M., Geller, W., Wendt-Potthoff, K., Benthaus, F.-C., 2009. Management of German pit lakes. In: Öhlander, B. et al. (Eds.), Proceedings of Securing the Future and 8th ICARD, June 23–26, 2009, Skellefteå, Sweden, <a href="http://www.proceedings-stfandicard-2009.com">http://www.proceedings-stfandicard-2009.com</a>.
- Shevenell, L., Connors, K.A., Henry, C.D., 1999. Controls on pit lake water quality at sixteen open-pit mines in Nevada. Appl. Geochem. 14, 669–687.
- Stottmeister, U., 2008. Altlastensanierung mit Huminstoffsystemen. Chem. Unserer Zeit 42, 24–41.
- Stottmeister, U., Weißbrodt, E., Becker, P.M., Pörschmann, J., Kopinke, F.D., Martius, G.G.M., Wießner, A., Kenedy, C., 1998. Analysis, behaviour and fate of a lignite pyrolysis wastewater deposit, Contaminated Soil '98. Thomas Telford Publishing, London.
- Stottmeister, U., Glässer, W., Klapper, H., Weissbrot, E., Eccarius, B., Kennedy, C., Schultze, M., Wendt-Potthoff, K., Frömmichen, R., Schreck, P., Strauch, G., 1999. Strategies for remediation of former opencast mining areas in eastern Germany. In: Azcue, J.M. (Ed.), Environmental Impacts of Mining Activities: Emphasis on Mitigation and Remediation. Springer-Verlag, Berlin, pp. 263–296.
- Stumm, W., Morgan, J.J., 1996. Aquatic Chemistry Chemical Equilibria and Rates in Natural Waters. 3rd edition, John Wiley and Sons, New York.
- Thurman, E.M., 1985. Organic geochemistry of natural waters, Martinus Nijhoff/Dr. W. Junk Publishers, Dordrecht.
- Trettin, R., Gläßer, W., Lerche, I., Seelig, U., Treutler, H.-C., 2006. Flooding of lignite mines: isotope variations and processes in a system influenced by saline groundwater. Isot. Environ. Health Stud. 42, 159–179.
- Trettin, R., Gläser, H.R., Schultze, M., Strauch, G., 2007. Sulfur isotope studies to quantify sulfate components in water of flooded lignite open pits – Lake Goitsche, Germany. Appl. Geochem. 22, 69–89.
- Werner, F., Bilek, F., Luckner, L., 2001. Impact of regional groundwater flow on the water quality of an old post-mining lake. Ecol. Eng. 17, 133–142.
- Wolf, S., 2005. Evaluierung der hygienischen Wasserqualität unter besonderer Berücksichtigung von Bakteriophagen am Beispiel eines Tagebausees. Ph.D. thesis, Technical University Dresden.
- Wycisk, P., Neumann, C., Gossel, W., 2005. Flooding induced effects from the Mining Lake Goitzsche on groundwater and land-use in the Bitterfeld area. Acta Hydrochim. Hdrobiol. 33, 507–518.
- Zschiedrich, K., Benthaus, F.-C., Dahlke, M., 2007. Flutung und Bewirtschaftung von Bergbaufolgeseen. In: Drebenstedt, C., Struzina, M. (Eds.), Management bergbaubedingten Säurepotentials. Inst. f. Bergbau und Spezialtiefbau der TU Bergakademie Freiberg, Freiberg, pp. 166–174.



Contents lists available at ScienceDirect

### Limnologica



journal homepage: www.elsevier.de/limno

#### Erratum

### Erratum to: "Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects" [Limnologica 40 (2) (2010) 148–155]

Martin Schultze<sup>a,\*</sup>, Karl-Heinz Pokrandt<sup>b</sup>, Wolfram Hille<sup>b</sup>

<sup>a</sup> UFZ Helmholtz Centre for Environmental Research, Department of Lake Research, Brückstraße 3a, D-39114 Magdeburg, Germany <sup>b</sup> Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft, Leipzig, Germany

In Fig. 2, a wrong unit  $(10^6 \text{ m}^2)$  was given for the surface area of the pit lakes. The right unit is ha. The correct version of Fig. 2 is shown below.



Fig. 2. Frequency distribution of the morphometric data of the pit lakes in the Central German lignite mining district (as far as data are available from the sources mentioned above). All data represent the final water level of the pit lakes. The current data are smaller for lakes which are still in filling.

DOI of original article: 10.1016/j.limno.2009.11.006

<sup>\*</sup> Corresponding author. Tel.: +49 391 8109400; fax: +49 391 8109150. *E-mail address:* martin.schultze@ufz.de (M. Schultze).

<sup>0075-9511/\$ -</sup> see front matter  $\circledcirc$  2010 Elsevier GmbH. All rights reserved. doi:10.1016/j.limno.2010.07.001
# Appendix 2

Schultze, M., Boehrer, B., Büttner, O., Geller, W., Herzsprung, P. (2012): Processes during filling of pit lakes with river water as indicated by water quality changes in Lake Goitsche, Germany. (submitted to Science of the Total Environment)

Contributions of M. Schultze: Concept 80% Writing 75% Figures 100%

Processes during filling of pit lakes with river water as indicated by water quality changes in Lake Goitsche, Germany

Schultze, M.<sup>a\*</sup>, Boehrer, B.<sup>a</sup>, Büttner, O.<sup>a</sup>, Geller, W.<sup>a</sup>, Herzsprung, P.<sup>a</sup>

<sup>a</sup> UFZ – Helmholtz-Centre for Environmental Research, Department Lake Research, Brückstraße 3a, 39114 Magdeburg, Germany

martin.schultze@ufz.de bertram.boehrer@ufz.de olaf.buettner@ufz.de walter.geller@ufz.de peter.herzsprung@ufz.de

\* corresponding author

# Abstract

Diversion of river water is a well established strategy for filling and for the abatement of acidification of pit lakes. Although often used, no detailed quantification of the processes relevant for the formation of the final water quality is available. The aim of this study was to establish such quantification. The filling of Lake Goitsche near Bitterfeld (Germany) with water from Mulde River was used as example. It lasted from May 1999 until July 2002.

The initially acid water was neutralised February 2001. The major contribution to neutralization (>90%) was the import of bicarbonate with the river water. Other processes like microbial sulphate reduction or dilution were of small relevance or even negligible regarding neutralization. An important finding was that more than 50% of the acidity to be neutralized entered the lake during its filling. Erosion and elution of the bottom and the side walls of the mine void and inflowing groundwater were the main sources for this acidity. Where the side walls contained carbonates, their erosion contributed to neutralization. The seasonal thermal stratification of Lake Goitsche was accompanied by temporal chemical stratification. This influenced the temporal progress of the neutralization but not its result.

Dilution was relevant for chloride, sulphate, calcium and other (almost) non-reactive substances. Precipitation and sedimentation removed almost all iron and aluminium which were initially present in high concentrations due to the acidification caused by pyrite oxidation. Co-precipitation removed almost all trace metals and more than 98% of the phosphorus imported with the river water. Therefore, eutrophication was not an issue in Lake Goitsche. Findings from other pit lakes confirmed the results from Lake Goitsche. Therefore, filling pit lakes with river water can be applied with high potential for success wherever enough river water is available.

# 1 Introduction

Pit lakes exist in many countries and their number is increasing (Castendyk and Eary, 2009). They represent characteristic features of the post-mining landscape (Linke and Schiffer, 2002; Doupe and Lymbery, 2005; Lienhoop and Messner, 2009; McCullogh and van Etten, 2011). Where mining extended over a wide area, artificial lake districts may form like in the Central German lignite mining district (Schultze et al., 2010, 2011). Pit lakes form where surface mining extends below groundwater level and the resulting mine void is not refilled with overburden or other material over the level of groundwater rebound after mine closure.

Acidification caused by pyrite oxidation and its consequences are the main concerns regarding water quality in pit lakes (Campbell et al., 1964; Klapper and Schultze, 1995; Miller et al., 1996; Castro and Moore, 2000; Blodau, 2006; Castendyk and Eary, 2009). In order to avoid acidification, to stabilize the side walls of the mine voids, and to allow for early public use of pit lakes, fast filling is desired in many cases. Once dewatering operations have ceased, surface water or water from mines still in operation may be used to expedite the filling of pit lakes. Such filling strategies have been applied in Germany (Fritz and Benthaus, 2000; Zschiedrich, 2007; Schultze et al., 2011), in the Czech Republic (Halir and Zizka, 2008; Svoboda et al., 2008; Prikryl et al., 2011), in Spain (Arnhold et al., 2002; Delgado, 2006), Canada (Kalin et al., 2001) and in Australia (Lund and McCullough, 2008). An exceptional case is the Island Copper Mine pit lake, which was filled mainly with sea water and got only a thin cap of fresh water on its surface to establish meromixis artificially (Fisher and Lawrence, 2006; Pelletier et al., 2009).

Although filling with river water was applied in many cases, no detailed study has been published yet on the processes occurring during filling with river water and on the contribution of the diverse processes to the final water quality. The purpose of this paper is filling this gap. The most important processes are from a theoretical point of view:

- dilution in its purely physical sense, i.e. without chemical reactions
- replacement of lake water as a result of outflow into the groundwater
- import of substances with water flowing into the lake (including alkalinity and acidity)
- chemical reactions in the lake water (including sedimentation of resulting precipitates)
- interaction with the bottom and the sidewalls of the mine void (including land slides and erosion)
- interaction with lake sediment
- primary production in the lake and microbial alkalinity generation

Based on detailed monitoring of the water quality of Lake Goitsche (Germany) during filling with water from river Mulde from 1999 to summer 2002, the above mentioned processes are evaluated regarding their contribution to the water quality of Lake Goitsche at the end of the regular filling period in July 2002. As far as possible, the contribution of the processes was quantified. The uncontrolled final step of filling, caused by a dyke breach during an exceptional flood of river Mulde in August 2002 (Boehrer et al., 2005; Klemm et al., 2005), is not subject of this paper since it is exceptional and cannot serve as a basic example for controlled filling of pit lakes with river water.

## 2 Site Description

#### 2.1 Location and mining history

Lake Goitsche is part of the post mining landscape around Bitterfeld (Germany; Figure 1). It resulted from the lignite mine Goitsche which was operated from 1949 to 1991 (Quasdorf, 1998). Since the mine removed part of the natural Mulde riverbed, the river was diverted into

the residual void of the neighbouring mine Muldenstein forming the Muldereservoir (Figure 1, panel B) in 1975 and 1976 (Müller, 1998). Amber was produced from a layer immediately below the mined lignite seam in the area of the current lake basins Muehlbeck and Niemegk (Figure 1, panel A) from 1975 to 1991 (Quasdorf, 1998). The processing facility for separating amber from silt was located in basin Niemegk. As a consequence, the waste material was dumped there.

The mine void was prepared for filling from 1991 to 1998. In particular, the inclination of the side walls had to be adjusted for stability requirements. Quaternary overburden was placed in front of the food of the sidewalls originating from the mine Rösa which is now covered by Lake Seelhausen (Figure 1, panel B).



Figure 1 Map of Lake Goitsche and its surroundings. Panel A: bathymetric map of Lake Goitsche including sampling sites for lake water (D5, M3, N3, N5, P4); panel B: mining district of Bitterfeld (1 - former mine Goitsche with Lake Goitsche, 2 - former mines Holzweißig with pit lakes, 3 - former mine Rösa with Lake Seelhausen, 4 - former mine Muldenstein with Muldereservoir; areal extension of pit lakes indicates planned final water level whereas state of filling is shown for period 1999-2002); panel C: location of study area in Germany (from Trettin et al, 2007, adapted).

## 2.2 Geological setting

The lignite mined in the former mine Goitsche originated from the Tertiary. It was embedded and overlain by unconsolidated rock like clay, sand and gravel of Tertiary and Quaternary age (Eissmann, 2002). Lignite seams formed between paleo-dunes, which often remained as areas without lignite. Consequently, tey were not mined completely and formed one of the islands in the southern part of Lake Goitsche (Figure 1, panel B). Local depressions of the lignite seams resulted from dissolution of Permian salt in the deeper underground (Eissmann, 2002). The amber bearing layer formed only in the current lake basins Muehlbeck and Niemegk

(Wimmer et al., 2006). The deposits consisted of a sequence of thin layers of silt and fine sand and included high concentrations of pyrite (up to 5%; Grützmacher et al., 2001).

As indicated by the large extension of mined area in the vicinity of Bitterfeld (Figure 1), the original geological conditions were heavily disrupted by surface mining. Today, the mined

area is widely covered by overburden dumps including substantial portions of the lake beds. The dumped overburden consists of mixtures of the initial aquifers and aquitards. Mostly, the hydraulic transmissivity of the dumps is lower than the initial hydraulic transmissivity of the mined natural package of aquifers and aquitards.

For more details on the geological setting see Grützmacher et al. (2001), Eissmann (2002) and Wimmer et al. (2006).

#### 2.3 Filling and modelled exchange with groundwater

The first step of filling was the formation of small water bodies at the deepest locations of the basins as the result of limited dewatering during preparation of the mine void for filling with river water (referred to as precursor lakes in the following). The sources of water for the precursor lakes were groundwater, precipitation and local runoff. According to the requirements of preparation work and to the goal of minimal dewatering costs, the water level of the precursor lakes varied between 1991 and 1999. However, it did not exceed the levels given in Table 1 for April 1999.

The main step of regular filling was the diversion of water from river Mulde. The water was diverted using the remaining northern part of the natural bed of river Mulde which had got out of function in 1976 when river Mulde had been fully diverted into the Muldereservoir. This section of the old river bed is visible in Figure 1 (panel B) as connection between river Mulde downstream the Muldereservoir and basin Muehlbeck. In order to test the inflow channel in basin Muehlbeck, a first trial was conducted from July 6 until August 25, 1998.  $1.67 \times 10^6$  m<sup>3</sup> flew into basin Muehlbeck in that period elevating the water level from 49.0 to 53.5 m above sea level (m a.s.l.).



Figure 2 Rise of water level in the basins of Lake Goitsche from April 1999 to July 2002 and volume of water diverted into Lake Goitsche from river Mulde in that period (after Schultze and Geller, 2004, adapted)

During regular filling from May 1999 to July 2002, the maximal inflow rate of river water was 3 m3 s<sup>-1</sup> according to the permission of the environmental authorities. However, the inflow rate had to be reduced or filling had to be fully stopped in periods of low flow in river Mulde or during flood events. The restrictions during low flow periods aimed at protecting the river ecosystem. Deviation of river water was stopped during flood events was based on the experience that the load of hazardous substances is usually elevated in rivers under such conditions, in particular in the phase of increasing flow (e.g. Baborowski et al., 2004).

The basins of Lake Goitsche were connected with channels on a certain level. Therefore, the filling developed stepwise. Firstly, basin Muehlbeck was filled up to the level of 57.5 m a.s.l. Then, the overflow into basin Niemegk started and went on up to the level of 59.0 m a.s.l., when the overflow into basin Doebern set on. In May 2000, all basins had the same water level resulting in a uniform rise further on.  $140 \times 10^6$  m<sup>3</sup> of river water were diverted into Lake Goitsche until the end of July 2002. I.e., the average inflow rate of river water was 1.37 m<sup>3</sup>/s during that period. Figure 2 summarises the development of the water level in all basins during the regular filling with river water.

After dewatering had stopped in the precursor lake in basin Doebern, rising groundwater, precipitation, local runoff and dewatering operations along the eastern margin of basin Doebern raised the water level in 1999. Dewatering at the eastern margin of basin Doebern were continued in order to prevent uncontrolled formation of springs at the respective side walls and accompanying risk of collapse of the side walls.

## 2.4 Morphology of the lake basin

Lake Goitsche consists of three basins as shown in Figure 1 (panel A). This is caused by the geological conditions (occurrence of ridges at the base of the lignite seams and of local depressions of the lignite seams) as well as by the technology and progress of mining. E.g., the peninsula separating basin Muehlbeck and basin Doebern is the result of dumping overburden on already mined area as close to actual excavation sites as possible while leaving amber bearing areas uncovered for later exploitation (Quasdorf, 1998). The local depressions in basin Niemegk and basin Doebern are the result of lignite exploitation in the geologically caused local depressions of the seams.

Table 1 shows the morphometric data of Lake Goitsche at three important stages: before start of regular filling with river water (April 1999), at the end of regular filling (July 2002) and at its final level (January 2003). The data are presented separately for the basins of Lake Goitsche at the beginning of regular filling because the basins were separated at that time and had different water levels. The changes in maximal depth do not agree with the changes in water level in Table 1. The reasons for that are local land slides and the deposition of about 500.000 m<sup>3</sup> of eroded soil from the flood plain into basin Doebern during the flood event in August 2002 (Boehrer et al., 2005).

(January 2003)					
	Water level	Volume	Surface area $106^{-2}$	Maximal depth	
	m a.s.1."	10° m°	10° m <sup>-</sup>	m	
April 1999					
Basin Muehlbeck	53.5	1.7	1.08	3.5	
Basin Niemegk	40.0	4.0	0.68	16.0	
Basin Doebern	35.0	1.2	0.35	13.0	
July 2002					
Lake Goitsche	71,5	166	10.5	44.7	
January 2003					

213

13.3

47.9

Table 1Morphometric data of Lake Goitsche at the three major stages of filling: before start of regular<br/>filling with river water (April 1999), at the end of regular filling (July 2002) and at its final level<br/>(January 2003)

<sup>a</sup> m a.s.l. – meter above sea level

75.0

Lake Goitsche

## 2.5 Groundwater quality and modelled exchange with groundwater

Groundwater was not subject of this investigation. Information on groundwater was taken from other studies: Trettin et al. (2007) published data on groundwater quality. Observation wells around Lake Goitsche and two sampling devices in the bottom of basin Niemegk were employed. The water quality varied widely between aquifers as well as within particular aquifers. The calculation of representative averages for particular observation wells or aquifers was not possible. The special variability resulted from the heterogeneity of the aquifers. The groundwater at the bottom of basin Niemegk lies significantly out of the range of the groundwater quality around Lake Goitsche and varied widely between the two sampling devices and depending on the depth of sampling. Therefore, Table 2 provides only ranges for the water quality. For more details, we refer to Trettin et al. (2007).

	Observation wells around	Grundwater at the bottom of
	Lake Goitsche	basin Niemegk
рН	4.7 – 7.6	1.18 - 2.66
acidity	0.4 - 24.0	na
alkalinity	0.1 – 26.3	na
SO4 <sup>2-</sup>	57 - 2290	6610 - 31750
Cl	15 - 241	na
Na	10-219	9.8-48.4
Mg	9 - 130	89 – 129
K	2.2 – 27	77 – 314
Ca	46 - 789	422 - 506
NO <sub>3</sub> <sup>-</sup>	<2-5	na
Fe	0.4 - 474	1800 - 11000
A1	na	482 - 2513

Table 2Ranges of groundwater quality in the vicinity of Lake Goitsche and in the bottom of basin<br/>Niemegk according to Trettin et al. (2007). Acidity and alkalinity in mmol L<sup>-1</sup>, others except pH<br/>in mg L<sup>-1</sup>; na – not analysed

The exchange of water between the rising lake and the groundwater changed considerably during the investigation period. Before filling with river water had started, the precursor lakes were the main sinks for groundwater since their water level was kept on a low level by pumping excess water out of the mine void. Therefore, the groundwater level in the vicinity of the mine void was equal or higher than the water level of the precursor lakes. The inflow of river water changed this situation. Water began to flow from the lake into the groundwater partly filling the dewatered aquifers and overburden dumps. The lake water mainly flew into the upper, fully dewatered aquifers. The lower aquifers which were already re-saturated and which had contributed to the formation of the precursor lakes continued to feed into the lake. Figure 3 shows a simplified scheme of the interaction between groundwater and a pit lake during its filling under the typical hydro-geological conditions of the Central German lignite mining district.



Figure 3 Scheme of the interaction of groundwater and a pit lake during its filling with river water under the typical geological conditions of the Central German lignite mining district. Different patterns indicate different aquifers. Aquitards between aquifers are not shown.

This complex interaction between rising lake and groundwater was simulated already for the prediction of the filling process and the potential water quality in Lake Goitsche by the consulting company Ingenieurbüro für Grundwasser Leipzig (IBGW) (Kringel et al., 1999). After regular filling of Lake Goitsche, a new simulation was carried out by the IBGW (IBGW, 2003) using the monitored groundwater level data and the real inflow data for the river water as input data for the simulation. Figure 4 shows the calculated inflows of groundwater into the lake and outflows of lake water into the groundwater for the period 1999 to summer 2002. The flows were separately calculated for the different aquifers and the overburden dumps and the different basins. Figure 4 shows the results summarized for the basins. The amount of water lost to the groundwater was rather small in basin Muehlbeck. It was always smaller than the inflow of groundwater into this basin. On the contrary, the flow rate of water leaving basin Doebern into the groundwater exceeded considerably the inflow rate of groundwater into this basin once filling with overflowing water from basin Niemegk had begun. The development in basin Niemegk was an intermediate one compared to the other basins. It had phases with dominating inflow of groundwater and, as well, phases with dominating outflow into the groundwater.



Figure 4 Calculated exchange of water between rising Lake Goitsche and groundwater according to IBGW (2003); inflow means flow of groundwater into the rising lake, outflow means flow of water from the lake into the groundwater; B-Mb – basin Muehlbeck, B-Ni – basin Niemegk, B-Do – basin Doebern)

# 3 Methods

## 3.1 Field measurements and sampling

In the first month of filling with river water, sampling was done weekly in basin Muehlbeck and in basin Niemegk. In summer 1999, the frequency of sampling was reduced to fortnightly and later on to monthly. Basin Doebern was generally sampled only monthly, based on the experiences collected in the other basins.

Temperature, electrical conductivity, oxygen concentration and pH-value were measured with a multi-parameter probe (IDRONAUT Ocean Seven, Milano, Italy) from the lake surface to the lake bottom in 20 cm steps. Based on the obtained results, depths of sampling for chemical analysis were selected in the field in order to take representative samples of all relevant layers (epilimnion, hypolimnion and monimolimnion, if present).

The measurements revealed high gradients in pH in space and time, which had to be included in calculating conductivity at reference temperature. Acidic waters show a temperature behaviour different to neutral waters (see Boehrer 2012). As a consequence, pH profiles were used to determine the portion of the measured electrical conductivity delivered by acidity. This portion was temperature compensated by implementing the proper specific coefficients for the hydronium ion from physical chemistry (e.g. Sorensen and Glass 1987) to evaluate conductance  $\kappa_{25}$  at 25°C.

High gradients between oxic and anoxic water layers could not be resolved by profiling with relatively robust oxygen sensors. Hence to achieve a realistic representation of concentration profiles of dissolved oxygen, oxygen measurements were compensated numerically for the response time of the sensor of about 6.0 s as described in Koschorreck et al. (2011).

## 3.2 Chemical analysis

Acidity and alkalinity were measured by an automatic titrator (Methrom). Calcium (Ca<sup>2+</sup>), magnesium (Mg<sup>2+</sup>), sodium (Na<sup>+</sup>), potassium (K<sup>+</sup>), sulphate (SO<sub>4</sub><sup>2-</sup>), and chloride (Cl<sup>-</sup>) were analyzed by single column ion chromatography (SCIC) (GAT Analysentechnik, Germany) using a two channel autosampler and conductivity detector (Herzsprung et al., 2006). TOC, DOC, and TIC were analyzed IR-spectrometrically by injection of the water sample in a C-analyzer (Dimatec, Germany) (Herzsprung et al., 1998). Nitrate (NO<sub>3</sub><sup>-</sup>-N) (Herzsprung et al., 2005; EN ISO 13395), ammonium (NH<sub>4</sub><sup>+</sup>-N) (Krom, 1980; ISO 11732), soluble reactive phosphorus (SRP) (Mecozzi, 1995; EN 1189), and silica (Si) (Schmith and Milne, 1981) were measured by continuous flow analysis (CFA, Skalar, The Netherlands) (Herzsprung et al., 2006). Total phosphorus (TP) was measured after addition of sulphuric acid and potassium peroxodisulfate by digestion (EN 1189; Rönicke et al., 2008). Chlorophyll-a concentrations (Chl-a) were measured by HPLC (DIONEX, Germany) using ethanol as extracting agent after filtration onto Whatman GF/F glass fiber filters (nominal pore size 0.7mm; vacuum filtration method, with 200 mbar) (Baborowski et al., 2011).

Aluminum (Al), iron (Fe), manganese (Mn), and zinc (Zn) were determined by optical emission spectroscopy with inductively coupled plasma (ICP-OES, Perkin-Elmer, OPTIMA 3000, Germany) (Baborowski et al., 2011). Cadmium (Cd), copper (Cu), lead (Pb) were measured using mass spectroscopy (MS) with inductively coupled plasma (ICP-MS, Agilent Technologies, 7500c series, Germany) (Baborowski et al., 2011).

## 3.3 Calculation of inventories and import of substances

Inventories of substances were the amounts of particular substances present in Lake Goitsche, one of its basins or in one of the compartments of the respective water body during

stratification (epilimnion, hypolimnion) at a certain date. They were calculated according to equation 1.

$$Inv_{i,j,d} = c_{i,j,d} \cdot V_{j,d} \tag{1}$$

with  $Inv_{i,j,d}$  - inventory of substance i in compartment j at date d

 $c_{i,i,d}$  - concentration of substance i in compartment j at date d

 $V_{i,d}$  - volume of compartment j at date d

The inventories of the basins were calculated as sum of the inventories of epilimnion and hypolimnion during periods of stratification. The inventories of the lake were calculated as sum of the inventories of the basins.

Changes in the inventories of compartments were calculated as difference between the inventories in the respective compartment at the end of a period and the beginning of a period according to equation 2.

$$Cha_{i,j,p} = Inv_{i,j,d2} - Inv_{i,j,d1}$$

(2)

with  $Cha_{i,j,p}$  - change of the inventory of substance i in compartment j in period p

 $Inv_{i,i,d1}$  - inventory of substance i in compartment j at the beginning of a period d1

 $Inv_{i,j,d2}$  - inventory of substance i in compartment j at the end of a period d2

p = d2 - d1 - duration of the period lasting from date d1 until date d2

The cumulative import of a substance with river water was calculated according to equation 3.  $cIMP_{i,d} = c_{i,med} \cdot Q_{j,d}$ (3)

with  $cIMP_{i,d}$  - cumulative import of substance i with river water from May 7, 1999 until date d

 $c_{i,med}$  - median of concentration of substance i in the river water

 $Q_d$  - volume of river water flown into Lake Goitsche in the period from May 7, 1999 until date d

The import of a substance with river water for a certain period was calculated according to equation 4.

$$IMP_{i,p} = IMP_{i,d2} - IMP_{i,d1}$$
<sup>(4)</sup>

with  $IMP_{i,p}$  - import of substance i with river water in the period p from date d2 until date d1

 $cIMP_{i,d1}$  - import of substance i with river water from May 7, 1999 until date d1

 $cIMP_{i,d2}$  - import of substance i with river water from May 7, 1999 until date d2

The alkalinity gain from microbial activity within a certain period was calculated according to equation 5.

$$Alk_{microb,p} = r_{microb} \cdot A_{av,p} \cdot p \tag{5}$$

with  $Alk_{microb,p}$  - alkalinity gain from microbial activity in the period p from date d2 until date d1

 $r_{microb}$  - area specific rate of microbial alkalinity generation

 $A_{av,p}$  - average of the surface area for the period p

The volume weighted average concentrations in the water of the precursor lakes at the beginning of filling with river water (i.e. in April 1999) were calculated according to equation 6.

$$c_{i,av,V} = \frac{\sum c_{i,j} V_j}{\sum V_j} \tag{6}$$

with  $c_{i,av,V}$  - volume weighted average concentrations in the water of the precursor lakes in April 1999

 $c_{i,j}$  - concentration of substance i in the precursor lake in basin j in April 1999

 $V_i$  - volume of the precursor lake in basin j in April 1999

### 4 Results

#### 4.1 Stratification

The precursor lakes were not stratified in spring 1999 before filling with river water began. Supported by rising atmospheric temperatures, river water could form a clearly separated layer on top of the previously present water of the precursor lake in basin Muehlbeck as well as in basin Niemegk in summer 1999. This top layer had a higher temperature and was chemically different. At the end of August 1999, full overturn happened in basin Muehlbeck after a period of relatively low air temperatures. Afterwards, no considerable chemical difference was observed between epilimnion and hypolimnion in basin Muehlbeck during periods of seasonal stratification in the years 2000 to 2002.

In basin Niemegk, deep recirculation started in September 1999. The full overturn was accomplished in January 2000 at sampling site N5. At sampling site N3, the mixing did not reach the lake bottom. A chemically different layer of about 2 m thickness persisted in the depression around sampling site N3 until July 2002. In 2000, seasonal circulation and perennial stratification showed the same features as 1999. The thermal stratification was accompanied by a chemical stratification.



Figure 5 Selected profiles for temperature (T), electrical conductivity ( $\kappa_{25}$ ) and concentration of dissolved oxygen (O<sub>2</sub>)

The same applied for basin Doebern in 2000. In 1999, this basin had shown only a thermal stratification. A permanently chemically different water body appeared at the lake bottom of sampling site P4 in 2000. It was 4 m thick and persisted until the end of July 2002. Except for both mentioned monomolimnia at sampling sites N3 and P4, all basins showed only thermal stratification in summer and full overturn from autumn to spring in the years 2001 and 2002.

Thermal stratification in winter could not develop because of the absence of a full ice cover. More details on the development of the stratification of Lake Goitsche during its regular filling were described in Boehrer et al. (2003). Figure 5 shows only a few selected profiles of temperature, electrical conductance ( $\kappa_{25}$ ) and concentration of dissolved oxygen for illustration of the main phenomena mentioned above: Data from sampling sites M3 in July 1999 and D5 in July 2000 illustrate thermal stratification accompanied by only temporal chemical stratification. In contrast, data from sampling sites N3 and P4 in February 2002 demonstrate meromixis.

#### 4.2 pH and concentrations of selected substances

The chemical water quality aspects of diverted river water varied over time with flow rate of the river and also seasonally (Figure 6). In particular, low concentrations of nutrients like phosphorus and nitrogen were observed usually from spring to autumn according to seasonally higher biological activity. However, changes in flow rate disturbed simple seasonal patterns. The temporary interruption of diversion of water into Lake Goitsche made the patterns even more complex. Therefore, box plots were used in Figure 6.



Figure 6 Chemical water quality of the river water flowing into Lake Goitsche at sampling site M1. Lines within the boxes are median values, box limits are 25 and 75 percentiles, and whiskers show the 10- and 90-percentile values. Single dots indicate data below 10-percentile or above 90-percentile, respectively.

Table 3 summarizes the changes in the chemical water quality of Lake Goitsche during its regular filling with river water. March 2002 was chosen for the characterisation of the final water quality because lake waters had been homogenized by seasonal overturn. Only two small monimolimnia showed different water quality. Because of there very small volume, the monimolinmia were not included in the further considerations.

	April 1999			March 2002
	Basin			
	Mühlbeck	Basin Niemegk	Basin Doebern	Lake Goitsche
pН	3.3	3.1	7.5	7.6
acidity	4.29	9.61	0.23	0.04
alkalinity	0.00	0.00	1.29	0.64
$SO_4^{2-}$	851	1580	760	262
Cl	40.2	100.0	37.2	43.7
Na	29.8	71.3	29.7	35.6
Mg	30.0	74.7	52.0	16.2
Κ	7.6	13.8	5.4	7.0
Ca	129	386	409	94
Fe	48.6	157	0.31	0.013
Al	12.4	21.7	< 0.1	< 0.1
Mn	1.92	4.82	1.8	0.085
Cd	0.0008	0.0005	< 0.0002	0.0002
Cu	0.011	0.005	< 0.002	< 0.002
Pb	0.0026	0.003	< 0.002	< 0.002
Zn	0.180	0.149	0.015	0.020
$NH_4^+-N$	1.00	1.14	1.27	0.05
$NO_3$ -N	0.49	0.05	0.49	3.55
SRP	0.005	0.027	< 0.002	< 0.003
TP	0.006	0.787	< 0.006	0.007
Si	14.5	18.9	8.7	2.1
DOC	1.0	4.2	3.4	2.7
TOC	1.1	4.3	3.5	2.7
TIC	< 0.5	1.6	19.6	7.4

Table 3Chemical water quality in Lake Goitsche in April 1999 and in March 2002. Acidity and<br/>alkalinity in mml  $L^{-1}$ , others (except pH) in mg  $L^{-1}$ .

#### 4.2.1 Concentrations related to acidification

Figure 7 shows the development of the pH in the water of Lake Goitsche separated for the three basins and the different layers during periods of stratification. The water of the epilimnion of basin Muehlbeck was neutralized in June 1999. The hypolimnion of basin Muehlbeck was neutralized by late August 1999. In basin Niemegk, the neutralization of the epilimnion was accomplished at the beginning of June 2000 while the hypolimnion took until February 2001. The water in basin Doebern turned acidic in January 2000. The epilimnion returned to neutral conditions in June 2000 while the hypolimnion returned to neutral conditions in June 2000 while the hypolimnion returned to neutral conditions in June 2001 and 2002, the pH dropped slightly in the hypolimnion of all basins during summer.



Figure 7 Development of pH in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion

The development of the acidity in the water of Lake Goitsche (Figure 8) was basically complementary to that of the pH value. There was an increase in the hypolimnion every summer. In basin Niemegk, there was an increase also in the epilimnion in summer 1999 after an initial drop in May and June 1999.



Figure 8 Development of acidity in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion

Basically, the development of the alkalinity of the water of Lake Goitsche (Figure 9) again reflected the neutralization similar to the development of the pH value. A remarkable temporary increase of alkalinity occurred in basin Doebern in summer 1999. In summer 2001 and 2002, the alkalinity decreased slightly in the hypolimnion of all basins.



Figure 9 Development of alkalinity in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion

In order to get a deeper insight into the overall progress of neutralization, inventories of alkalinity were calculated for the different basins at different dates according to equation 1. Figure 10 compares the inventories of alkalinity Lake Goitsche and its basins with the cumulative import of alkalinity into Lake Goitsche with the river water for selected dates (calculated according to equation 2).



Figure 10 Inventories of alkalinity and acidity (expressed as negative values of alkalinity) in Lake Goitsche and its basins and cumulative alkalinity imported into Lake Goitsche by river water during regular filling (Lake Goitsche represents the sum of the basins; for more details see text)

Figure 10 shows that the inventories of acidity (negative values of alkalinity) increased in Lake Goitsche until the end of 1999, in particular in basin Niemegk. This indicates substantial inputs of acidity into Lake Goitsche in that period. In order to quantify these inputs of acidity, the changes of the inventories of alkalinity were calculated according to equation 2 for Lake Goitsche for the periods between the dates shown in Figure 10. The results were compared with the input of alkalinity with river water in the respective periods (calculated according to equation 4; Figure 11). Furthermore, the differences between the changes of alkalinity inventories in the lake water and the import of alkalinity with river water were also calculated for all periods (Figure 11). The resulting decreases of alkalinity represented the input of acidity into Lake Goitsche in the respective periods and all neutralizing processes which were not related directly to the alkalinity of river water. I.e. the calculated differences represented a

lower limit for the input of acidity since processes like neutralization by interaction with carbonates in the banks of Lake Goitsche or by inflowing net-neutral groundwater were not considered due to the limited availability of respective data (see subsection 5.3). The sum of the calculated decreases of alkalinity was  $68.5 \times 10^6$  mol for the period from May 7, 1999 until the final neutralization of the lake water in February 2001. Together with the initial stock of alkalinity in the precursor lakes, the minimum of acidity to be neutralized amounted to  $111.4 \times 10^6$  mol until February 2001.



Figure 11 Changes in the alkalinity inventories of Lake Goitsche between the dates shown in Figure 10 (lake total) in comparison with the imports of alkalinity with river water during the respective periods (input Mulde) and the difference "lake total – input Mulde". Calculated changes are shown at the centre of the considered periods (for more details see text)

#### 4.2.2 Concentrations related to trophic state

In basin Muehlbeck, the concentration of total phosphorus (TP) increased after the beginning of the inflow of river water in May 1999. Concentrations remained at an elevated level until low could establish agin in basin Muehlbeck when a uniform epilimnion for all basins formed in summer 2001. The TP-concentrations dropped rapidly in basin Niemegk in the epilimnion in May and June 1999. In the hypolimnion of basin Niemegk, relatively high concentrations occurred in summer 1999 and 2000. The TP-concentrations in basin Doebern remained low in the epilimnion and in the hypolimnion over the entire investigation period.



Figure 12 Development of total phosphorus (TP) in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion

In order to get a deeper insight into the behaviour of phosphorus, inventories for selected dates and the cumulative import with river water were calculated according to equation 1 and equation 3. The results are shown in Figure 13. There was almost no difference between the inventories of TP at the beginning and the end of regular filling of Lake Goitsche.



Figure 13 Inventories of total phosphorus (TP) in Lake Goitsche during regular filling compared with the cumulative input of TP with river water (from Schultze and Geller, 2004, adapted)

The initially elevated concentration of ammonia decreased in Lake Goitsche over the investigation period whereas the concentration of nitrate increased (Figure 14). The increase of the nitrate was higher than the decrease of ammonia on a molar basis. The ammonia concentrations of the hypolimnetic water were higher than in the epilimnion. For nitrate, the concentrations of the epilimnetic water were the higher ones.



Figure 14 Development of nitrogen occurring as ammonia (NH<sub>4</sub><sup>+</sup>-N) and of nitrate (NO<sub>3</sub><sup>-</sup>-N) in basin Muehlbeck (B-Mb), basin Niemegk (B-Ni) and basin Doebern (B-Do) separated for epilimnion and hypolimnion

## **5** Discussion

#### 5.1 Stratification

The parallel occurrence of thermal and chemical stratification in basin Muehlbeck in 1999, in basin Niemegk in 1999 and in 2000 and in basin Doebern in 2000 was the consequence not only of the typical seasonal development. The lower concentration of dissolved substances in the river water also contributed to the differences between density of river water and lake water resulting in the preferred inflow of river water into the epilimnion in summer. The chemical differences between epilimnion and hypolimnion caused an unusual long stable summer stratification. Typically, overturn is completed in natural lakes of comparable morphometry in northern Germany in late November or early December. In basin Niemegk, completion of overturn took until following January in 1999 and 2000. I.e., thermal and chemical stratification stabilized each other.

The chemical stratification, discussed above can not be called meromixis since it did occur only seasonally. On the contrary, the formation of the monimolimnia at the sampling sites N3 and P4 clearly allows for calling Lake Goitsche meromictic. The monimolimnia withstood the overturn for three (two) winters at sampling site N3 (at P4) within the investigation period, respectively. In both cases, the inflow of groundwater is the most probable reason for the formation of the monimolimnia. For sampling site P4, observed concentrations of radon gave

additional evidence of strong groundwater inflow at that site (A. Schmidt (UFZ), unpublished data). The high concentrations of substances in the rising groundwater in the bottom of basin Niemegk (Table 2) basically supported the hypothesis of the groundwater related formation of the monimolimnion at sampling site N3.

The sampling sites N3 and P4 are located in former local depressions of the lignite seams. Since the depressions obviously formed after deposition of the lignite bearing strata (Eissmann, 2002), all strata were stretched. This made the aquitards less resistant against differences in hydraulic head. Several break-through events occurred at the flanks of the depressions during mining and remediation (W. Laugwitz (LMBV), personal communication). These hydrogeological conditions support the hypothesis that the monimolimnia at the sampling sites N3 and P4 resulted from preferential inflow of groundwater. This groundwater inflow was probably not only responsible for the formation of the monimolimnia but also the main process sustaining the monimolimnia.

# 5.2 Basic interpretation of observed development of concentrations

#### 5.2.1 Concentrations related to acidification

The development of the pH (Figure 7) reflects the progress of neutralization including the influence of the temporal thermal and chemical stratification. Even after the complete neutralization in February 2001, the pH dropped slightly in the hypolimnia during stratification in summer and early fall. This is also reflected by acidity (Figure 8) and alkalinity (Figure 9). Elevated pH-values above eight occurred only in basin Muehlbeck and were limited to spring and summer 2000 and 2001. At that time, the phosphorus concentrations were higher in the epilimnion of basin Muehlbeck compared to the other basins (Figure 12). Probably, photosynthetic activity of algae was responsible for raining pH beyond 8.0. The temporal acidification of basin Doebern (Figure 7) resulted from the inflow of acidic water from basin Niemegk. The epilimnetic water of basin Niemegk was acidic until early June 2000. Being the last in the chain of lake basins, basin Doebern epilimnion took longest for final neutralization. The neutralization of the hypolimnia was generally controlled by the stratification. During stratification, river water did not enter the hypolimnion. Therefore, the neutralization of the hypolimnia was restricted to the periods of overturn.

The development of the acidity is complementary to that of pH as expected. In the epilimnion of basin Niemegk, there was an increase in acidity after an initial drop in 1999. This increase probably has to be attributed to the influence of the pyrite rich amber bearing material and the waste of the amber extraction. The increase of acidity in the hypolimnion of basin Doebern in 1999 has to be interpreted in a different way since there was a parallel increase in alkalinity. This indicated that the acidity did not result from pyrite oxidation but from inflow of groundwater rich in carbon dioxide (causing the acidity) as well as in bicarbonate (causing the alkalinity).

As more neutralization progressed as more carbon dioxide contributed to the acidity. After neutralization, carbon dioxide was the dominating chemical species representing acidity. Therefore, the increases in acidity in the hypolimnia in 2001 and 2002 and partly in 2000 have to be attributed to the accumulation of carbon dioxide in the hypolimnion during stratification.

The development of alkalinity one more time reflected the neutralization. The main aspects were already discussed above. An interesting additional detail was the switch from drop to increase of the alkalinity in the hypolimnion of basin Muehlbeck in summer 2000. The reason for this temporal development was probably the exchange of water with basin Niemegk via the connecting channel. The exchange was stratified and, therefore, bidirectional. I.e. water

was flowing from basin Muehlbeck into basin Niemegk while the Niemegk water was flowing in the opposite direction at the same time. Karakas and Boehrer (2003) reported respective measurements and discussed this issue from a theoretical point of view. The stratified flow was caused by the differences in chemistry and, consequently, in density between the water in basin Muehlbeck and basin Niemegk. The very small inclination of the channel between the basins supported the observed phenomenon. Until the neutralisation of the epilimnion of basin Niemegk in early June 2000, acidic water entered basin Muehlbeck as counter flow through the connecting channel. Later, both flow directions transported neutral water causing the neutralization of the accumulated acidity in the hypolimnion of basin Muehlbeck.

Periods of increasing alkalinity before February 2001 indicated that for some periods neutralizing processes which were not considered in the above calculations dominated over the input of acidity (e.g. from December 2000 to February 2001; Figure 11). The changes of alkalinity after February 2001 were more complicated to interpret. In that period, accumulation of  $CO_2$  in the hypolimnion became relevant whereas in the period before acidity was almost exclusively caused by dissolved iron and aluminium. Different processes were potential reasons for the accumulation of  $CO_2$  in the hypolimnion: (a) reaction of bicarbonate of the lake water with acidity entering the lake, (b) decay of organic material (e.g. settling dead plankton and allochthonous detritus), (c) import of  $CO_2$  with groundwater flowing into the hypolimnion. The relative evaluation of these processes was not possible based on the available data.

#### 5.2.2 Concentrations related to trophic state

The origin of the high concentration of TP (Figure 12) in basin Niemegk at the beginning of the regular filling is unknown. In the years before regular filling, the high phosphorus concentrations caused temporarily blooms of algae (Woelfl, 2000; Zippel, 2005). However, the TP concentrations dropped rapidly when neutralization set in caused by co-precipitation with iron and aluminium (Duffek and Langner, 2002). In contrast, the TP concentration in basin Muehlbeck increased considerably. This was caused by the inflowing river water, which entered Lake Goitsche in basin Muehlbeck. Therefore, basin Muehlbeck served like a cleaning pond. However, TP concentration decreased also in basin Muehlbeck in 2001 when Lake Goitsche formed a uniform epilimnion. This decrease in TP was a result of homogenization of the water body as well as of the removal of TP by settling detritus, plankton and by chemical co-precipitation. Co-precipitation of phosphorus with iron and aluminium is well known from pit lakes (Hupfer et al., 1998; Duffek and Langner, 2002; Kleeberg and Grüneberg, 2005; Grüneberg and Kleeberg, 2005; Herzsprung et al., 2010).

The concentrations of ammonia (Figure 14) basically reflected the progress in neutralization including the influences of the stratification. Relatively high ammonia concentrations in acidic pit lakes are common (Klapper and Schultze, 1995). They probably result from the inhibition of nitrification under acidic conditions and the release of ammonia from lignite. Accordingly, ammonia concentrations decreased with progressing neutralisation. Dilution and assimilative consumption certainly also contributed to the decrease of ammonia concentrations of ammonia in the hypolimnion of basin Doebern in 1999 and in 2000 and in the hypolimnion of basin Niemegk in 2000 probably resulted from inflowing groundwater, interaction with the sediment and the former bottom and side walls of the mine void and the decay of settling detritus and the rests of terrestrial vegetation on the former bottom and side walls of the mine void.

The concentrations of nitrate (Figure 14) mainly reflected the inflow of river water since the water from Mulde River was the major source of nitrate for Lake Goitsche during its regular filling.

## 5.3 Quantification of potentially relevant processes

Many processes contribute to water quality of pit lakes during their filling with river water. However, only a limited number is of them are responsible for the main aspects of water quality: neutralization, risk of eutrophication and risk of import of hazardous substances into the lake. A basic assumption of the following paragraphs is that filling of a pit lake with river water does not start with a dry mine void but with precursor lakes formed by inflowing groundwater, precipitation and local runoff in the deepest locations of the mine void. The water quality of such precursor lakes is usually very similar to that of the local mine water after oxygenation.

#### 5.3.1 Dilution

The term dilution is used here only for its physical meaning without any chemical reaction. Usually, mine water has high concentrations of dissolved substances (1,500-5,800 mg  $L^{-1}$  total dissolved substances (TDS) on average, depending on mined minerals and local conditions; Banks et al., 1997; Nordstrom and Alpers, 1999; Geller and Schultze, 2010). Compared to such water, rivers usually have much lower concentrations of almost all substances (50–1,300 mg  $L^{-1}$  TDS on average, depending on climate, regional geology and human impact; Meybeck, 2005). Therefore, dilution is a potentially important process when filling a pit lake with river water.

The contribution of dilution to the neutralization of pit lakes cannot be quantified following directly the simple rules known for the dilution of pure acids because the water of acidic pit lakes is buffered. The quantification has to be based on acidity remaining to be neutralized. Assuming this acidity as pure acid present in the volume of the lake water at the beginning of filling with river water allows for the calculation of the necessary dilution to reach pH 7.

In April 1999 immediately before filling with river water began in Lake Goitsche, the precursor lakes had in sum a volume of  $6.9 \times 10^6$  m<sup>3</sup>. Following the above considerations, the acidity to be neutralized until February 2001 ( $111 \times 10^6$  mol) was equivalent to an acid with pH 1.79. Diluting by a factor of more than 100.000 was required to reach pH 7. The relation between the initial volume of the precursor lakes and the river water flowing into Lake Goitsche until February 2001 ( $102 \times 10^6$  m<sup>3</sup>) allowed only for a dilution factor of 15.8. Accordingly, the dilution by river water contributed less than 0.016% to the neutralization. Since the inflow of groundwater contributed a much smaller volume than the inflow of river water, the potential dilution effect was accordingly smaller. In addition, many groundwater inflows were not able to contribute to neutralization by dilution because of their acidity.

Dilution by river water was relevant for substances which were non-reactive at the given conditions, i.e. sulphate, chloride, sodium, magnesium, potassium and calcium. Based on the median concentrations in the river water as shown in Figure 6 and the calculation of the volume weighted average concentration in the precursor lakes at the beginning of filling with river water (according to equation 6 and based on the data given in Table 1 and Table 3), the following dilution factors were calculated: 12.6 for sulphate, 1.9 for chloride, 1.8 for sodium, 5.0 for magnesium, 1.7 for 6.5 for calcium. Because its high concentrations of above substances, groundwater did not contribute to the dilution. For most reactive substances, concentration changes were dominated by chemical transformations and only to a negligible portion by dilution.

#### 5.3.2 Outflow into the groundwater

As discussed in subsection 2.3, the outflow from Lake Goitsche into the groundwater occurred in the upper aquifers. Therefore, water from the epilimnion was flowing out. Based on the observed acidity in the epilimnia of the basins of Lake Goitsche and the calculated outflow of lake water into the groundwater according to IBGW (2003), the amount of acidity exported into the groundwater was estimated. For the period from May 1999 (begin of regular filling) to October 2000 (total neutralization of all epilimnia), about  $1.88 \times 10^6$  mol of acidity were exported into the groundwater. This was about 1.7% of the overall acidity to be neutralized in Lake Goitsche according to the estimates presented in subsection 5.2.

#### 5.3.3 Import of substances with water flowing into the lake

The river water carried  $124 \times 10^6$  mol of alkalinity into Lake Goitsche until neutralization was complete in February 2001. This value represented 112% of the estimated acidity to be neutralized indicating that the estimated acidity to be neutralized did not include all imported acidity. However, the import of alkalinity with river water was obviously the most important contribution to the neutralization of Lake Goitsche.

The import of alkalinity and acidity could not be quantified because the knowledge on the inflow rates and the respective groundwater quality was not detailed enough in space and time. However, the known data on the groundwater quality, in particular in the bottom of basin Niemegk clearly indicated, that the inflow of groundwater caused a substantial import of acidity into Lake Goitsche. The increase of the acidity inventory in basin Niemegk from May to December 1999 (Figure 10 and Figure 11) was caused probably mainly by groundwater.

Although the import of substances with inflowing groundwater could not be quantified

The average quantity of precipitation was  $0.087 \text{ m}^3 \text{ s}^{-1}$ , i.e. very small compared to inflows. Surface runoff was also small compared to the inflow of groundwater and river water (Figure 15). Therefore, these components of water inflow could be considered irrelevant without detailed investigations of their water quality.

Concentrations of non-reactive substances in the river water were already included into the calculation of dilution. The import by inflowing groundwater was doubtless important for sulphate although it could not be quantified. For the other non-reactive substances, concentration ranges in the groundwater were similar to the volume weighted average concentrations in the precursor lakes. Therefore, the inflow of groundwater could not change those concentrations considerably indicating that the inflow of groundwater was of low relevance regarding non-reactive substances except sulphate. Reactive substances were more influenced by reactions in the lake water, except nitrate. The river water was the main source for nitrate in Lake Goitsche.



Figure 15 Components of water inflow into Lake Goitsche according to IBGW (2003)

#### 5.3.4 Chemical reactions in the lake water

The most important chemical reactions were the consumption of bicarbonate by acidity and the precipitation of iron and aluminium as hydroxides. The consequences of the consumption of bicarbonate originally existing in the lake water (only basin Doebern) or entering the lake with groundwater or river water was already discussed above. Additionally, quantifying the contribution of the reaction of bicarbonate with acidity would mainly mean quantifying the contribution of alkalinity imported with inflowing water again. This does not make sense despite the extraordinary importance of the reaction of bicarbonate with acidity.

The precipitation of iron and aluminium was on one hand identical with the removal of acidity. On the other hand, the precipitation of iron and aluminium was accompanied by coprecipitation of diverse substances. Examples are trace metals and metalloids like zinc, copper, nickel, cadmium and arsenic. They occurred in the initially acidic lake water as the result of their leaching from the soil along the flow path of the acidic water feeding the precursor lakes. The initially existing pyrite may have been also a source of trace metals since sedimentary formed pyrite contains often impurities of trace metals. A further source of trace metals and metalloids was the river water. The Muldereservoir is known to be an important sink for trace metals and arsenic originating from the catchment of Mulde River including the remaining consequences of ore mining, smelting and metal based manufacturing in the last about 800 years (Zerling et al., 2001; Klemm et al., 2005). However, the removal of metals and arsenic ranged between 32% for nickel and 84% for lead depending on the chemical nature of the different elements (Zerling et al., 2001). The remaining load reached Mulde River downstream the Muldereservoir including Lake Goitsche. Without co-precipitation, the removal of e.g. zinc as hydroxide required a pH of above 10 whereas the adsorption onto precipitating hydroxide of ferric iron or aluminium worked under slightly acidic to neutral conditions as given in Lake Goitsche during its neutralization and later on. Therefore, coprecipitation was considered the main mechanism of removal for arsenic and trace metals during regular filling of Lake Goitsche.

Phosphorus was also subject of co-precipitation with iron and aluminium in Lake Goitsche during neutralization and in the following (Duffek and Langner, 2002). At the beginning, phosphorus was mainly bound by ferric iron, then to aluminium and, once circum-neutral conditions were reached, again by ferric iron (Duffek and Langner, 2002). The co-precipitation of phosphorus resulted in a removal of 98.6% of the total phosphorus that was imported into Lake Goitsche with the river water. Therefore, the inventory of total

phosphorus did not change much while the cumulative input of total phosphorus with river water was increasing continuously during regular filling (Figure 13).

The almost complete removal of phosphorus as a result of co-precipitation was also reported for other pit lakes which were filled with river water (Lessmann et al., 2003; Grüneberg and Kleeberg, 2005). Even without initial acidification, the phosphorus concentration in a Czech pit lake remained very low during its filling with river water and allowed for oligotrophic to mesotrophic conditions at the end of the filling (Prikryl et al., 2011).

#### 5.3.5 Interaction with the bottom and the sidewalls of the mine void

During filling of Lake Goitsche, erosion at the shore by wave action was observed. The heights of the temporarily forming cliffs ranged between about 5 cm and >30 cm. At the beginning of filling with river water, this erosion mainly affected the pyrite rich, amber bearing material or the waste of the amber extraction. At that time, the erosion was a source of acidity. Certainly, it contributed to the increase of acidity in basin Niemegk in summer 1999. Once the water level reached the Quaternary material dumped at the foot of the sidewalls and constituting the upper portion of the sidewalls where no overburden dumps form the shore, erosion became a source of alkalinity since the Quaternary material naturally contained carbonates. However, since detailed knowledge on erosion was not available, an estimation of the contribution of erosion was not possible separated from other processes.

Although land slides of dimensions relevant for neutralization did not occur in Lake Goitsche during its regular filling, land slides have to be considered as a generally important threat for the progress of neutralization in filling pit lakes. Gröschke et al. (2002) reported for Lake Graebendorf (Lusatian lignite mining district, Germany) that the acidity mobilised by land slides consumed previous progress in neutralization of several months repeatedly.

#### 5.3.6 Interaction with lake sediment

Typical lake sediment existed only at the bottom of the precursor lakes at the beginning of the regular filling of Lake Goitsche which covered only 20% of the lake surface of July 2002 (Table 1). This lake sediment was dominated by iron that contributed 20 to 40% to the dry weight of the sediment (Langner, 2001). The sediments which formed during regular filling and neutralization were similar (Duffek and Langner, 2002; Herzsprung et al., 2010). During regular filling of Lake Goitsche, the iron precipitated and settled mainly as poorly crystalline hydroxides of ferric iron (Duffek and Langner, 2002). This is comparable with sediment s of many other pit lakes in former lignite mines in Germany (Friese et al., 1998; Peine and Peiffer, 1998). This dominance of ferric iron precipitates strongly influenced the interaction between lake water and sediment.

As shown by Herzsprung et al. (2010), the oxic conditions at the interface between sediment and lake water prevented the release of iron and all substances typically adsorbed by precipitates of ferric iron into the lake water. Typically adsorbed substances were phosphorus and trace metals as discussed in subsection 5.3.4. The only exceptions were the small areas covered by anoxic monimolimnia at sampling sites N3 and P4 (Herzsprung et al., 2010). However, oxidation and precipitation of iron in the chemocline prevented release of substances into the mixolimnion as well (Herzsprung et al., 2010). Consequently, the interactions between sediment and water were basically restricted to sedimentation and burial in the sediment in Lake Goitsche during its regular filling.

#### 5.3.7 Primary production and microbial alkalinity generation

Primary production has direct effects on neutralization only via nitrogen assimilation. If nitrate is assimilated, a gain of alkalinity occurs. If ammonia is assimilated, acidity is formed. During the regular filling of Lake Goitsche, the decrease of concentration of nitrate-nitrogen was approximately equal to that of ammonia-nitrogen: roughly 1 mg L<sup>-1</sup> (Figure 14). That means that in summary there was neither a gain of alkalinity nor a formation of acidity in considerable dimensions caused by nitrogen assimilation.

Increase of pH caused by algal blooms due to consumption of carbon dioxide and bicarbonate during photosynthesis is well known from many eutrophic lakes. If the pH is elevated by algal blooms considerably, i.e. above 9, calcite precipitates. This causes a loss of alkalinity for the water column since bicarbonate as the main constituent of alkalinity in neutral lake water is removed by the precipitation of calcite. Because of its low phosphorus concentrations and the resulting oligotrophic character of Lake Goitsche, algal blooms influencing the pH considerably did not occur (Dreher, 2007). Therefore, the loss of alkalinity caused by algal blooms was not relevant in Lake Goitsche.

Primary production may influence also indirectly the alkalinity-acidity-balance in a pit lake as source of organic material. The organic material may be used by bacteria as substrate for the reduction of nitrate, iron and sulphate. These processes occur in the sediment if enough labile organic material is present. Sulphate reduction is usually not possible as long considerable amounts of ferric iron are present. Because of the pyrite oxidation, usually plenty of ferric iron precipitates and plenty of sulphate are present in the sediments of pit lakes. The reduction of ferric iron alone is not able to result in sustainable alkalinity generation. The formed ferrous iron diffuses back into the water body of the pit lake and is oxidized and precipitated there. This removes the initial gain in alkalinity as e.g. described by Peine et al. (2000). Therefore, the rate of sulphate. Only the formation and burial of ferrous sulphides results in sustainable alkalinity gain.

The alkalinity generation processes were not measured in Lake Goitsche during regular filling. However, the potential contribution of these processes was estimated based on rates of microbial alkalinity generation from literature. Koschorreck and Tittel (2007) reviewed the literature for net-rates of alkalinity production in neutral lakes impacted by AMD. They considered denitrification, burial of nitrogen, net sulphate reduction and potential sulphate reduction due to inhibition of methanogenesis in sulphate rich water. For low productive lakes, i.e. oligotrophic lakes like Lake Goitsche, they found a median rate of 387 mmol m<sup>-2</sup> yr<sup>-1</sup> for alkalinity production. The net-rates for sulphate reduction under acid conditions are comparable to that under neutral conditions (Koschorreck, 2008) if the same relation between gross-rates and net rates is assumed as given in Koschorreck and Tittel (2007). To apply the above mentioned rate from Koschorreck and Tittel (2007), the contribution of sulphate reduction has to be doubled in order to include the reduction of ferric iron and assuming, that ferrous mono-sulphide is the result of both iron and sulphate reduction.

The upper limit of microbial alkalinity generation was estimated by using the average lake surface for the period from May 1999 to February 2001. The application of equation 5 gave:

 $Alk_{microb} = 0.387 mol \times yr^{-1} \cdot 4.56 \times 10^{6} m^{2} \cdot 1.75 yr = 5.1 \times 10^{6} mol$ 

This was about 4.6% of the total acidity which had to be neutralized according to the estimate in subsection 4.2.1. However, more than 50% of the lake bottom was not covered with well developed lake sediment in February 2001. It was terrestrial material, covered by sparse vegetation and just at the beginning of soil development until it became the bottom of Lake Goitsche due to rising water level. Considering only the area of the precursor lakes for the estimation of sulphate reduction gave the lower limit of the contribution of microbial activity to the neutralization of Lake Goitsche:  $Alk_{microb} = 0.387 mol \times yr^{-1} \cdot 2.4 \times 10^{6} m^{2} \cdot 1.75 yr = 5.1 \times 10^{6} mol$ 

This was about 2.1% of the total acidity which had to be neutralized according to the estimate in subsection 4.2.1. Therefore, microbial alkalinity production was probably relevant but not decisive for the neutralization of Lake Goitsche.

# 6 Conclusions

The contribution of diverse processes to the neutralization and water quality of Lake Goitsche in July 2002 evaluated. For many processes, the contribution could be quantified. Table 4 summarizes the respective results.

Table 4Summarised quantification and evaluation of the relevance of processes for the neutralization<br/>and water quality of Lake Goitsche in July 2002. The percentages quantify the contribution<br/>to neutralization in addition to verbal evaluation of relevance.

Processes	Neutralization	Other aspects of water quality
Dilution in its purely physical sense	0.016%	relevant for all substances
	very low	which had lower concentrations
	relevance	in the river water that in the
		initial lake water, in particular
		for (almost) non-reactive
		substances
Replacement of lake water as a result	1.7%	low relevance due to small
of outflow into the groundwater	low relevance	volume of outflowing water
Import of substances with water	112% alkalinity	most important for nitrate and
flowing into the lake	(river water)	relevant for products of pyrite
	most important	oxidation and ammonia
Chemical reactions in the lake water	very high	most important for iron,
	relevance (but	aluminium, trace metals, arsenic
	contribution	and phosphorus (>95%
	already included	precipitated)
	in the estimates of	
	other processes)	
Interaction with the bottom and the	relevant	relevant for products of pyrite
sidewalls of the mine void		oxidation
Interaction with lake sediment	low relevance	low relevance (sedimentation is
		included in chemical reactions
		in the lake water)
Primary production in the lake and	2.1 - 4.6%	relevant for phosphorus and
microbial alkalinity generation	low relevance	nitrogen

The import of bicarbonate clearly turned out as the most important process regarding neutralization. The import of acidity during filling by inflowing groundwater and interaction with the former bottom and sidewalls of the mine void are also very important for the neutralization.

Regarding other aspects of water quality dilution was found to be very important for (almost) non-reactive substances. Reactive substances were most strongly influenced by reactions in the lake water, in particular precipitation and sedimentation.

Future studies should focus on the contribution of groundwater inflow and of the interaction with the bottom and the side walls of the mine void since they were the most important processes which could not be quantified in our study. The typically very high heterogeneity of

mine sites has to be considered in such studies. This will require a very high effort for measurements in the necessary spatial and temporal resolution. The respective costs may limit the chances that such studies can be conducted.

The study also demonstrated that the stratification had an important influence of the temporal progress of the neutralization and of the evolution of the water quality in July 2002. However, the stratification was not essential for the final result since the formed monimolimnia were very small compared to the total volume of Lake Goitsche.

Based on the development in Lake Goitsche and experiences reported from other pit lakes in the Central German lignite mining district (Schultze et al., 2005, 2010, 2011), in the Lusatian lignite mining district (Lessmann et al., 2003) and in the Collie region in Western Australia (Lund and McCullough, 2008; McCullough et al., 2010), the filling of pit lakes can be considered a useful strategy for the abatement of acidification. Where river water is available this strategy has a high potential for successful application.

# 7 Acknowledgements

The presented research was funded by the Federal Ministry for Education and Research of the Federal Republic of Germany (BMBF-project number: 02 WB 9984/3) and the Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft mbH (LMBV). Particular support we received from W. Hille, W. Laugwitz, P. Tropp and E. Zeh (all LMBV) by providing data and information and support in administrative issues. We would like to thank Siegfried Frimel, Burkhard Kuehn, Karsten Rahn and Uwe Kiwel for their field work and the staff of the water-chemical laboratory for carrying out the chemical analysis.

# 8 References

- Abel A, Michael A, Zartl A, Werner F. Impact of erosion-transported overburden dump materials on water quality in Lake Cospuden evolved from a former open cast lignite mine south of Leipzig, Germany. Environmental Geology 2000; 39: 683-8.
- Arnold I, Rivas Rodenas F, Rolland W, Strempel P. Flooding-project of the open cast mine As Pontes, Spain. Surface Mining Braunkohle & Other Minerals 2002; 54: 321-31.
- Baborowski M, von Tümpling W, Friese K. Behaviour of suspended particulate matter (SPM) and selected trace metals during the 2002 summer flood in the River Elbe (Germany) at Magdeburg monitoring station. Hydrology and Earth System Sciences, 2004; 8: 135–50.
- Baborowski, M, Büttner O, Einax JW. Assessment of Water Quality in the Elbe River at Low Water Conditions Based on Factor Analysis. Clean Soil, Air, Water 2011; 39: 437–43.
- Banks D, Younger PL, Arnesen R-T, Iversen ER, Banks SB. Mine-water chemistry: the good, the bad and the ugly. Environment Geological 1997; 32: 157-74.
- Blodau C. A review of acidity generation and consumption in acidic coal mine lakes and their watersheds. Science Total Environment 2006; 369: 307-32.
- Boehrer B. Physical properties of acidic pit lakes. In: Geller W, Schultze M, editors. Acidic pit lakes The legacy of coal and metal surface mines. Berlin: Springer; 2012 (in press).
- Boehrer B, Schultze M, Liefold S, Behlau G, Rahn K, Frimel S, Kiwel U, Kuehn B, Brookland I, Büttner O. Stratification of mining Lake Goitsche during flooding with river water. Tailings and Mine Waste '03. Lisse: Swets & Zeitlinger; 2003. p. 223-31.
- Boehrer B, Schultze M, Ockenfeld K, Geller W. Path of the 2002 Mulde flood through Lake Goitsche, Germany. Verh Intern Verein Limnol 2005; 29: 369-72.
- Campbell RS, Lind OT, Geiling WT, Harp GL. Recovery from acid pollution in shallow stripmine lakes in Missouri. Purdue University Engineering Bulletin 1964; 49: 17-26.

- Castendyk DN, Eary LE. The nature and the global distribution of pit lakes. In: Castendyk DN, Eary LE, editors. Mine pit lakes: Characteristics, predictive modeling, and sustainability. Littleton: Society for Mining, Metallurgy, and Exploration SME; 2009. p. 1-11.
- Castro JM, Moore JN. Pit lakes: their characteristics and the potential for their remediation. Environmental Geology 2000; 39: 1254-60.
- Delgado JL. Rehabilitation of Meirama pit lake. In: Loredo J, Pendas F, editors. Proceedings of the 9th International Mine Water Association Congress - 5th-7th September 2005 Oviedo, Spain. Oviedo: Departemento de Explotacion y Prospeccion de Minas, University of Oviedo, Spain on behalf of the International Mine Water Association; 2005. p. 267-72.
- Doupe RG, Lymbery AJ. Environmental risks associated with beneficial end use of mine pit lakes in Southwestern Australia. Mine Water Environment 2005; 24: 134-8.
- Dreher NS. Entwicklung des pelagischen Nahrungsnetzes in einem neu entstandenen Tagebausee. Doctoral thesis, University of Potsdam; 2007.
- Duffek A, Langner C. P-retention in an acidic mining lake under the influence of flooding with river water. Verh Intern Verein Limnol 2002; 28: 1717-1720.
- Eissmann L. Tertiary geology of the Saale-Elbe Region. Quaternary Science Reviews 2002; 21: 1245-74.
- EN 1189. Water quality Determination of phosphorus Ammonium molybdate spectrometric method. Standard EN 1189:1996
- EN ISO 13395. Water quality Determination of nitrite and nitrate nitrogen and the sum of both by flow analysis (CFA and FIA) and spectrometric detection (Standard ISO 13395:1996); German version of Standard EN ISO 13395:1996
- Fisher TSR, Lawrence GA. Treatment of Acid Rock Drainage in a meromictic mine pit lake. Journal Environmental Engineering 2006; 132: 515-526.
- Friese K, Hupfer M, Schultze M. Chemical Characteristics of water and sediment in acid mining lakes of the Lusatian lignite district. In: Geller W, Klapper H, Salomons W, editors. Acidic mining lakes - acid mine drainage, limnology and reclamation. Berlin: Springer-Verlag; 1998. p. 25-45.
- Fritz W, Benthaus F-C. Application of new techniques to create post mining landscape suitable for future generations. Braunkohle/Surface Mining 2000; 52: 261-5.
- Geller W, Schultze M. Acidification. In: Likens GE, editor. Encyclopedia of Inland Waters Volume 3. Oxfort: Academic Press/Elsevier; 2009. p. 1-12.
- Gröschke A, Uhlmann W, Rolland W, Grünewald U. Hydrochemische Entwicklung Lausitzer Tagebauseen während der Flutung - Beispiel Gräbendorfer See. Hydrologie und Wasserbewirtschaftung 2002; 46: 256-67.
- Grüneberg B, Kleeberg A. Benthic phosphorus forms and transformations during neutralization of acid mining lakes. In: Serrano L, Golterman HL, editors. Proceedings of the 4th International Symposium on Phosphate in Sediments. Kerkwerve: Backhuys Publishers; 2005. p. 127-37.
- Grützmacher G, Hindel R, Kantor W, Wimmer R. Chemical investigations of aquifers affected by pyrite oxidation in the Bitterfeld lignite district. Waste Management 2001; 21: 127-37.
- Halir J, Zizka L. Residual Mining Pits in Central Part of North Bohemian Brown Coal Basin.
  In: Rapantova N, Hrkal Z, editors. Mine Water and the Environment Proceedings of the 10th IMWA Congress, 2-5 June, 2008 Karlovy Vary, Czech Republic. Ostrava; VSB-Technical University of Ostrava, Faculty of Mining and Geology; 2008. p. 575-8.
- Herzsprung P, Friese K, Packroff G, Schimmele M, Wendt-Potthoff K, Winkler M. Vertical and annual distribution of ferric and ferrous iron in acidic mining lakes. Acta hydrochim hydrobiol 1998; 26: 253-62

- Herzsprung P, Duffek A, Friese K, de Rechter M, Schultze M, von Tümpling W. Modification of a continuous flow method for analysis of trace amounts of nitrate in iron-rich sediment pore-waters of mine pit lakes. Water Research 2005; 39: 1887-95
- Herzsprung P, Bozau E, Büttner O, Duffek A, Friese K, Koschorreck M, Schultze M, von Tümpling W, Wendt-Potthoff K. Routine analysis of sediment pore water of high ionic strength, Acta hydrochim hydrobiol 2006; 34: 593-607.
- Herzsprung P, Schultze M, Hupfer M, Boehrer B, von Tümpling W, Duffek A, van der Veen A, Friese K. Flood effects on phosphorus immobilisation in a river water filled pit lake--Case study Lake Goitsche (Germany). Limnologica 2010; 40: 182-90.
- Hupfer M, Fischer P, Friese K. Phosphorus retention mechanisms in the sediment of an eutrophic mining lake. Water Air Soil Pollution 1998; 108: 341-52.
- IBGW. Endbericht zu IBGW-Projekt-Nr. 034/2002 (unpublished report to Umweltforschungszentrum Leipzig-Halle UFZ). Leipzig: Ingenieurbüro für Grundwasser; 2003.
- ISO 11732. Water quality determination of ammonium nitrogen by flow analysis (CFA and FIA) and spectrometric detection (Standard ISO 11732: 1997); German version of Standard EN ISO 11732:1997
- Kalin M, Cao Y, Smith M, Olaveson, MM. Development of the phytoplankton community in a pit-lake in relation to water quality changes. Water Research 2001; 35: 3215–25.
- Karakas G, Boehrer B. Estimation of Effectiv Vertical Diffusivity in Turbulent Exchange Flows by Numerical Experiments. Progress of Theoretical Physics Supplement 2003; 150: 353-6.
- Klapper H, Schultze M. Geogenically acidified mining lakes living conditions and possibilities of restoration. Int Revue ges Hydrobiol 1995; 80: 639-653.
- Kleeberg A, Grüneberg B. Phosphorus mobility in sediments of acid mining lakes, Lusatia, Germany. Ecological Engineering 2005; 24: 89-100.
- Klemm W, Greif A, Broeckaert JAC, Siemens V, Junge FW, van der Veen A, Schultze M, Duffek A. A study on arsenic and the heavy metals in the Mulde River system. Acta hydrochim hydrobiol 2005; 33: 475-91.
- Koschorreck M, Tittel J. Natural alkalinity generation in neutral lakes affekted by acid mine drainage. J Environ Qual 2007; 36: 1163-71.
- Koschorreck M. Microbial sulphate reduction at a low pH. FEMS Microbiology Ecology 2008; 64: 329-42.
- Koschorreck M, Boehrer B, Friese K, Geller W, Schultze M, Wendt-Potthoff K. Oxygen depletion induced by adding whey to an enclosure in an acidic mine pit lake. Ecological Engineering 2011; 37: 1983-9.
- Kringel R, Nitsche C, Haferkorn B. Limnology of sulfur-acidic lignite mining lakes III: Approach and limitation in predicting chemical properties. Verh Intern Verein Limnol 2000; 27: 416-9.
- Krom M. Spectrophotometric determination of ammonia; a study of modified Berthelot reaction using salicylate and dichloroisocyanurate. Analyst 1980; 109: 305–16.
- Langner C. Frühdiagenetische Prozesse in jungen Sedimenten von entstehenden Seen des Braunkohletagebaus - am Beispiel der Flutung des Goitschekomplexes in Mitteldeutschland. Doctoral thesis, University of Bremen; 2002.
- Lessmann D, Uhlmann W, Grünewald U, Nixdorf B. Sustainability of the flooding of lignite mining lakes as a remediation technique against acidification in the Lusatian mining district, Germany. In: Farell T, Taylor G, editors. Sixth International Conference on Acid Rock Drainage, 14-17 July 2003, Cairs, Queensland. The Australian Institute of Mining and Metallurgy Publication Series No 3/2003. Carlton Victoria: The Australian Institute of Mining and Metallurgy; 2003. p. 521-7.

- Lienhoop N, Messner F. The economic value of allocating water to post-mining lakes in East Germany. Water Resources Management 2009; 23: 965-80.
- Linke S, Schiffer L. Development Prospects for the Post-Mining Landscape in Central Germany. In: Mudroch A, Stottmeister U, Kennedy C, Klapper H, editors. Remediation of abandoned surface coal mining sites. Berlin: Springer-Verlag; 2002. p. 111-45.
- Lund MA, McCullough CD. Limnology and ecology of low sulphate, poorly-buffered, acidic coal pit lakes in Collie, Western Australia. In: Rapantova N, Hrkal Z, editors. Mine Water and the Environment - Proceedings of the 10th IMWA Congress, 2-5 June, 2008 Karlovy Vary, Czech Republic. Ostrava: VSB-Technical University of Ostrava, Faculty of Mining and Geology; 2008. p. 591-594.
- McCullough CD, van Etten EJB. Ecological restoration of novel lake districts: New approaches for new landscapes. Mine Water Environment 2011; 30: 312-9.
- McCullough CD, Zhao LYL, Lund MA. Mine voids management strategy (I): pit lake resources of the Collie Basin, MiWER/Centre for Ecosystem Management Report 2009–10 (unpublished report to Department of Water, Western Australia). Perth: Edith Cowan University; 2010.
- Mecozzi M. Robust regression methods to calibrate a continuous flow analyzer in the colorimetric analysis of inorganic phosphorus in seawater. Talanta 1995; 42: 1239-44.
- Meybeck M. Global occurrence of major elements in rivers. In: Drever JI, editor. Surface and ground water, weathering, and soil (Volume 5 of Treatise on geochemistry). Amsterdam: Elsevier; 2005. p. 207-23.
- Miller GC, Lyons WB, Davis A. Understanding the water quality of pit lakes. Environmental Science Technology 1996; 30: 118A-23A.
- Müller H-S. Tagebau Muldenstein. In: Chronik des Braunkohlebergbaus im Revier Bitterfeld. Bitterfeld: Bitterfelder Berglaute e.V.; 1998. p. 199-223.
- Nordstrom DK, Alpers CN. Negative pH, efflorescent mineralogy, and consequences for environmental restoration at the Iron Mountain Superfund site, California. PNAS 1999; 96: 3455-62.
- Peine A, Peiffer S. In-lake neutralization of acid mine lakes. In: Geller W, Klapper H, Salomons W, editors. Acidic mining lakes acid mine drainage, limnology and reclamation. Berlin: Springer-Verlag; 1998. p. 47-64.
- Peine A, Tritschler A, Küsel K, Peiffer S. Electron flow in an iron-rich acidic sediment evidence for an acidity-driven iron cycle. Limnology Oceanography 2000; 45: 1077-87.
- Pelletier CA, Wen ME, Poling GW. Flooding pit lakes with surface water. In: Castendyk DN, Eary LE, editors. Mine pit lakes: Characteristics, predictive modeling, and sustainability. Littleton: Society for Mining, Metallurgy, and Exploration SME; 2009. p. 187-202.
- Prikryl I, Špacek J, Koza V. First large mine lake successfully filled up in Czech republic. In: Rüde T, Freund A, Wolkersdorfer C, editors. Mine Water – Managing the Challenges. Proceedings of the 11th IMWA Congress, 4-11 September, 2011, Aachen, Germany. Aachen: RWTH Aachen University, Institute of Hydrogeology; 2011. p. 539-43.
- Quasdorf P. Der Tagebau "Goitsche" und seine Bernsteinlagerstätte. In: Chronik des Braunkohlebergbaus im Revier Bitterfeld. Bitterfeld: Bitterfelder Berglaute e.V.; 1998. p. 224-39.
- Rönicke H, Doerffer R, Siewers H, Büttner O, Lindenschmidt KE, Herzsprung P, Beyer M, Rupp H. Phosphorus input by nordic geese to the eutrophic Lake Arendsee, Germany. Fundamental Applied Limnology 2008; 172: 111–9.
- Schultze M, Pokrandt K-H, Hille W. Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects. Limnologica 2010; 40: 148-55.

- Schultze M, Pokrandt K-H, Hille W. Erratum to: "Pit lakes of the Central German lignite mining district: Creation, morphometry and water quality aspects" [Limnologica 40 (2) (2010) 148–155]. Limnologica 2011a; 41: 78.
- Schultze M, Geller W, Benthaus F-C, Jolas P. Filling and management of pit lakes with diverted river and mine water - German experiences. In: McCullough CD (ed.) Mine Pit Lakes: Closure and Management. Perth: ACG Australian Centre for Geomechanics; 2011b. p. 107-20.
- Schulz M, Bischoff M. Variation in riverine phosphorus between 1994 and 2003 as affected by land-use and loading reductions in six medium-sized to large German rivers. Limnologica 2008; 38: 126-38.
- Smith JD, Milne PJ. Spectrophotometric determination of silicate in natural waters by formation of small alpha, Greek-molybdosilicic acid and reduction with a tin(IV)-ascorbic acid-oxalic acid mixture. Analytica Chimica Acta 1981; 123: 263-270.
- Sorenson JA, Glass GE. Ion and temperature dependence of electrical conductance for natural waters. Analytical Chemistry 1987; 59: 1594-7.
- Svoboda I, Pecherova E, Prikryl I, Kasparova I. The development of future lakes in opencast mine residual pits in the Krušné Mountain region of the Czech Republic. In: Rapantova N, Hrkal Z, editors. Mine Water and the Environment - Proceedings of the 10th IMWA Congress, 2-5 June, 2008 Karlovy Vary, Czech Republic. Ostrava: VSB-Technical University of Ostrava, Faculty of Mining and Geology; 2008. p. 619-22.
- Trettin R, Glaser HR, Schultze M, Strauch G. Sulfur isotope studies to quantify sulfate components in water of flooded lignite open pits Lake Goitsche, Germany. Applied Geochemistry 2007; 22: 69-89.
- Wagner, H. Dynamik der Windwellenerosion an Uferböschungen verbunden mit dem Wasserspiegelanstieg bei der Restlochfüllung. In: Proceedings des Dresdner Grundwasserforschungszentrums e.V., Heft 13. Dresden: Dresdner Grundwasserforschungszentrum e.V.; 1998. p. 209-10.
- Wimmer R, Pester L, Eissmann L. Das bernsteinführende Tertiär zwischen Leipzig und Bitterfeld. Mauritania 2006; 19: 373-421.
- Woelfl S. Limnology of sulphur-acidic lignite mining lakes. Biological propperties: plankton structure of an extreme habitat. Verh Intern Verein Limnol 2000; 27: 2904-7.
- Zerling L, Müller A, Jendryschik K, Hanisch C, Arnold A. Der Bitterfelder Muldestausee als Schadstoffsenke. Abhandlungen der Sächsischen Akademie der Wissenschaften zu Leipzig, Mathematisch-naturwissenschaftliche Klasse, Band 59, Heft 4. Leipzig: Verlag der Sächsischen Akademie der Wissenschaften zu Leipzig; 2001.
- Zippel B. Einfluss von Intraguild Predation auf die Dynamik der Planktonsukzession in einem sauren Bergbausee. Doctoral thesis, University of Potsdam; 2005.
- Zschiedrich K, F-C, Dahlke M. Benthaus Flutung und Bewirtschaftung von Bergbaufolgeseen. In: Drebenstedt С, Struzina Management M, editors. bergbaubedingten Säurepotentials. Freiberg: Institut für Bergbau und Spezialtiefbau der TU Bergakademie Freiberg; 2007. p. 166-74.

# Appendix 3

Herzsprung, P., Boehrer, B., Duffek, A., Friese, K., Hupfer, M., Schultze, M., v. Tümpling jr., W., van der Veen, A. (2010):

Flood effects on phosphorus mobility in a river water filled pit lake – Lake Goitsche (Germany). Limnologica 40 (2), 182-190.

Contributions of M. Schultze:Concept35%Writing35%Figures66%
Contents lists available at ScienceDirect

## Limnologica



journal homepage: www.elsevier.de/limno

## Flood effects on phosphorus immobilisation in a river water filled pit lake—Case study Lake Goitsche (Germany)

Peter Herzsprung<sup>a,\*</sup>, Martin Schultze<sup>a</sup>, Michael Hupfer<sup>b</sup>, Bertram Boehrer<sup>a</sup>, Wolf v. Tümpling jr.<sup>c</sup>, Anja Duffek<sup>a,1</sup>, Andrea Van der Veen<sup>c,2</sup>, Kurt Friese<sup>a</sup>

<sup>a</sup> Helmholtz Centre for Environmental Research—UFZ, Department of Lake Research, Brueckstrasse 3a, 39114 Magdeburg, Germany

<sup>b</sup> Leibniz-Institute of Freshwater Ecology and Inland Fisheries, RG Biogeochemistry Mueggelseedamm 301, 12587 Berlin, Germany

<sup>c</sup> Helmholtz Centre for Environmental Research—UFZ, Department of River Ecology, Brueckstrasse 3a, 39114 Magdeburg, Germany

#### ARTICLE INFO

Article history: Received 11 September 2009 Accepted 17 November 2009

Dedicated to Prof. Dr. Walter Geller on the occasion of his 65th birthday anniversary

Keywords: Phosphorus retention Eutrophication Pore water Manganese Redox cycling Iron

#### ABSTRACT

Between 1999 and 2002, a former open-cast mine was filled with river water forming the recent Lake Goitsche. During filling initially acid water was neutralised. Phosphorus (P) imported from Mulde River was nearly completely removed from the water column by co-precipitation with iron (Fe) and aluminium (Al) and deposited in the sediment.

During extremely high waters of the Mulde River in 2002, a dike breach facilitated a second high import of P into Lake Goitsche with suspended and dissolved matter. The analysis of total phosphorus (TP), however, showed that P again had been eliminated from the water body a few months after the flood event. Sediment investigations before filling with river water, during filling, and after the flood event were used to analyse the process of P immobilisation in a lake with acid mine drainage history.

The ratios of Fe to soluble reactive P (SRP) of sediment pore water were up to three orders of magnitudes higher than in natural lakes and can serve as an indicator for potential internal P loading from sediments. The SRP concentrations at the oxic/anoxic boundary were near or below the limit of quantification (  $< 0.2 \,\mu$ mol/L). Fe and manganese (Mn) redox cycling were responsible for hindering P dissolution from sediment to lake water.

Finally it can be stated, that the risk of eutrophication for such a lake seems to be low.

© 2009 Elsevier GmbH. All rights reserved.

#### 1. Introduction

Pit lakes are of growing importance as resources for diverse use and due to their increasing number (Miller et al., 1996; Castro and Moore, 2000; Klapper and Geller, 2001; Stottmeister et al., 2002). In the eastern part of Germany, the diversion of river water is the main way to create about 120 new pit lakes, which have been formed in former lignite mines in recent years. A rapid filling with river water has been found to be by far the cheapest way to limit or even to prevent acidification and to stabilize side walls of mining voids. However, river water often carries phosphorus (P) loads that suffice to put lakes at risk of eutrophication (Schindler, 2006; Schulz and Bischoff, 2008).

Previous investigations indicated that the eutrophication had occurred only in few pit lakes, which were filled with river water (Schultze and Klapper, 2004). In most cases, the P from the river water was co-precipitated with Fe and Al (Duffek and Langner, 2002). Possible P remobilization due to reductive dissolution of Fe minerals in the sediment poses the question of the long-term perspective when gradual wash out of the Fe sources, e.g. in overburden dumps, limit the availability of Fe (Kleeberg and Grüneberg, 2005).

In general the P release from lake sediments into the lake water is controlled by two main pathways: (i) remobilisation from the sediment particle surface at the contact phase with the lake water and (ii) sediment pore water flux into the lake water.

The pore water composition itself is a sensitive indicator of involved processes and the P release (Boers and De Bles, 1991; Eckerrot and Petterson, 1993) and the dissolved Fe:P ratio (further Fe:SRP ratio; SRP-soluble reactive P) in pore waters is reported to act as an indicator of P release to overlying oxic water (Lehtoranta and Heiskanen, 2003; Zak et al., 2004)). Furthermore the capacity of sediment surface to retain P is strongly linked to Fe species and their mobility (Roden and Edmonds, 1997).

Flood events are often accompanied by enormous transport of suspended and dissolved matter. When entering a lake, most of the suspended material settles on the lake bottom. In August

<sup>\*</sup> Corresponding author. Tel.: +49 391 8109 330; fax: +49 391 8109 150. E-mail address: peter.herzsprung@ufz.de (P. Herzsprung).

Present address: Federal Environment Agency-UBA, Laboratory for Water Analysis, Bismarckplatz 1, 14193 Berlin, Germany.

Present address: ECT Oekotoxikologie GmbH, Boettgerstrasse 2-14, 65439 Floersheim/Main, Germany.

<sup>0075-9511/\$ -</sup> see front matter © 2009 Elsevier GmbH. All rights reserved. doi:10.1016/j.limno.2009.11.007

2002, Lake Goitsche was impacted by a huge flood (Klemm et al., 2005). Beside arsenic and heavy metals (Klemm et al., 2005), there was a high import of P into this pit lake.

Based on the conception that the P cycling depends on the high Fe budget in sediments originating from the former lake acidification by pyrite weathering the purposes of the study were (i) to investigate the impact of the flood on the chemical composition of the sediment pore water, (ii) to evaluate the Fe:SRP ratios in sediment pore water of Lake Goitsche as an indicator for the stability of P-binding in the sediment and (iii) to identify mechanisms for efficient P-fixation at the sedimentwater interface of a flooded mine pit lake.

#### 1.1. Study area

The open-cast lignite mine Goitsche near Bitterfeld (Germany; Fig. 1) was operated from 1949 to 1991. From 1991 to 1999, the residual void was prepared for filling with water from River Mulde. Small water bodies formed in the deepest depressions of the void due to rising groundwater, precipitation and surface runoff.

Due to local geological conditions, the mining void consisted of three sub-basins referred to as Muehlbeck, Niemegk and Doebern. Differences of the geochemical composition of the soils resulted in differing chemical conditions in the water bodies at the bottom of the sub-basins: the water in the sub-basins Muehlbeck and Niemegk was acidic, while the water in sub-basin Doebern was neutral although high concentrations of sulphate indicated the occurrence of pyrite oxidation in this sub-basin, too (Kringel et al., 2000; Grützmacher et al., 2001).

The geochemistry of initial water bodies at the bottom of the sub-basins Niemegk and Doebern was investigated by Langner (2001). The regular filling with river water started in May 1999 and was stopped at a water level of 71.5 m above sea level in spring 2002. During filling, chemical stratification accompanied the thermal stratification at times. Finally, monimolimnia formed at two sites (Boehrer et al., 2003). After neutral conditions were reached in Lake Goitsche in 2001 and further filling until spring 2002, more than 90% of the imported P was deposited in the surface layers of the lake sediment (Schultze and Geller, 2004). The elimination of P by co-precipitation with Fe and Al during

filling and the P species in solid material were described by Duffek and Langner (2002).

In August 2002, a flood from River Mulde entered Lake Goitsche as the result of a dike breach. The water level was raised from 71.5 to 78.5 m above sea level within two days but lowered again to its final state of 75 m above sea level in late fall 2002 (final lake surface 13.3 km<sup>2</sup>, final lake volume  $213 \cdot 10^6$  m<sup>3</sup>). The flood caused an import of more than 4.3 t P into Lake Goitsche, which was more than twice of the P stock in June 2002 (1.7 t) or in August 2003 (2.1 t).

The investigated sampling sites represent different parts of the profundal of Lake Goitsche: at site M3. a small new sediment laver with 1-2 cm thickness had been formed during the flood. This site is representative for about 99.5% of the sub-basins Muehlbeck and Niemegk (about 60% of lake area). Site N3 represents a small region (about 0.3% of lake area) with permanently anoxic bottom water before and after the flood event and, like M3, with a thin flood caused sediment layer (Boehrer et al., 2003). The sites D5 and D6 are representative for subbasin Doebern which was covered with a thick new sediment layer after the flood event (up to 5 m thickness; about 40% of lake area) (Boehrer et al., 2005). The main characteristics of the three sampling sites are summarized in Table 1. Further details of the flood and its effects regarding arsenic and heavy metals are described by Klemm et al. (2005). The sampling site was changed in subbasin Doebern after the flood due to the formation of the new thick and relatively uniform sediment layer by the flood.

#### 2. Methods

#### 2.1. Pore water and lake water sampling

Sediment cores were obtained with a gravity corer (UWITEC, Mondsee, Austria) and sampled at the deepest points in the subbasins Muehlbeck, Niemegk and Doebern (sampling sites M3, N3, D5 and D6, see Fig. 1). Immediately after collection, the cores were transferred into a glove box with an oxygen-free argon atmosphere, sliced in 1–2 cm thick layers and transferred into 50 mL polyethylene centrifuge tubes. In the laboratory the samples were homogenised, and pH values were measured with electrodes (ORION). The samples



Fig. 1. Study area of Lake Goitsche with pore water sampling sites. The bathymetric map of Lake Goitsche shows the situation after the flood event of August 2002.

#### Table 1

Main characteristics of the three sub-basins under consideration.

Sub-basin	Muehlbeck	Niemegk	Doebern
Sampling site	M3	N3	D5, D6
Depth of sampling site	21.9 m (July 2002)	41.3 m (July 2002)	44.1 m (D5, July 2002)
	25.4 m (2003)	44.8 m (2003)	44.3 m (D5, 2003)
			40.0 m (D6, July 2002)
Maximum depth of subbassin	21.9 m (July 2002)	44.7 m (July 2002)	44.1 m (July 2002)
	25.4 m (2003)	48.2 m (2003)	44.3 m (2003)
Mixis	Monomictic	Meromictic	Monomictic
sediment after flood event	< 10 cm	< 10 cm	> 5 m
Sediment-water contact zone	Anoxic/oxic (2002)	Anoxic/anoxic (2002)	Anoxic/oxic (2002)
	Anoxic/oxic (2003)	Anoxic/anoxic (2003)	Anoxic/oxic (2003)

were centrifuged for 20 min at 4000 rpm. The supernatant was passed through 0.45  $\mu$ m cellulose acetate filters.

Water samples from the epilimnion were taken with a 3.5 L sampler from Limnos (Finland). Lake water samples close to the sediment surface were taken with syringes from the water column inside the corer about 5 cm above the surface of the sediment. The lake water samples were also passed through 0.45  $\mu$ m cellulose acetate filters.

#### 2.2. Analytical methods

The chemical analysis of pore waters was described in detail by Herzsprung et al. (2006). Briefly, Fe<sup>2+</sup>, NH<sub>4</sub><sup>+</sup>-N, soluble reactive P (SRP), Si, and NO<sub>3</sub><sup>-</sup>-N were analysed photometrically by continuous flow analysis (SKALAR, The Netherlands). Total dissolved Fe, Al, Mn, Ca, Mg, Na, and K were analysed by ICP-OES (Perkin Elmer, USA). SO<sub>4</sub><sup>2-</sup> and Cl<sup>-</sup> were analysed by single column ion chromatography (GAT Analysentechnik, Germany). The same analytical methods were applied for analysis of SRP and Cl<sup>-</sup> in lake water samples, except the dilution before analysis. In sediment pore water samples, the limit of quantification (LOQ) of SRP was 0.1  $\mu$ mol/L times dilution factor.

For analysis of sedimentary Al, K, Mg, Ca, Fe, and Mn dried core samples were digested with aqua regia following the National German Standard Methods (DIN 38414-7). The measurements of these elements were performed using the ICP-OES technique. Sedimentary P was determined as described by Rönicke et al. (2008).

#### 2.3. Calculation of Fe:SRP molar ratio in pore waters

The molar ratio of Fe:SRP was calculated for each sediment depth, where both Fe and SRP had been measured above LOQ. All pairs of values from all sediment sampling data (all depths) and all sampling sites were evaluated. There was no data pair with Fe below LOQ and SRP above LOQ. The minimum, maximum, and median values for the ratio of Fe:SRP were calculated from each pore water profile. To condense data, in some cases the Fe and SRP values of several sediment cores were pooled. That means, the overall Fe:SRP maximum of Fe:SRP maximum values originating from different sediment cores was calculated and accordingly the overall minimum and median. Pooling is indicated in Table 4 by specification of the number of cores.

#### 3. Results

#### 3.1. Lake water

The investigation of TP and SRP in pelagic water of Lake Goitsche describes the import of P by the flood event and the



**Fig. 2.** Variation of TP concentration in the epilimnion of Lake Goitsche during filling and after the flood event.

Table 2

Chloride concentrations of the lake water about 5 cm above the sediment surface (mmol/L).

	June 2002	August 2003
M3	1.23	0.87
N3	1.41	4.65
D5/6	1.24	0.85

successive elimination of P. Fig. 2 shows the development of TP with minimum values less or near LOQ at the end of the regular filling and several months after the flood event. In contrast, the TP values were significantly elevated at the first sampling time after the flood event. The concentration of SRP (epilimnion, data not shown) increased from levels with less LOQ (0.1  $\mu$ mol/L) to levels of more than 0.16  $\mu$ mol/L in all sub-basins initiated by the flood event and declined after about four months again to levels with less LOQ.

Table 2 shows the results of analysis of chloride in the lake water close to the sediment in June 2002 and in August 2003.

#### 3.2. Pore waters and sediments

At sampling site M3, sediment pore water concentrations of  $NH_4^+$ -N, Ca, total dissolved Fe,  $SO_4^{2-}$ , and SRP increased with sediment depth before and after the flood event (Table 3, Fig. 3). While the existence of concentration gradients kept unchanged after the flood event, the level of total dissolved Fe and  $SO_4^{2-}$  concentrations decreased slightly. The level of SRP concentration remained unchanged (Fig. 3). The vertical course of the Mn pore water profiles peaking near the sediment–water interface was equal before and after the flood event. The Fe:SRP ratios in pore water were quite similar before and after the flood event (Table 4). From the

Table 3	
Pore water chemistry of Lake Goitsch	e sediments.

Flood event	Sampl. site	Depth (cm)	Acidity (mmol/L)	рН	SRP (mmol/L)	HCO <sub>3</sub>	Al	Fe	$SO_4^{2-}$	$\rm NH_4^+$	Ca <sup>2+</sup>	Mg <sup>2+</sup>	Na <sup>+</sup>	K*	Mn <sup>2+</sup>	Cl-
Before	M3	0-1	-	-	< 0.0002	-	< 0.02	0.005	3.2	0.093	2.9	0.83	1.7	n.d.	0.14	n.d.
	17/6/02	11-13	+11.8	-	0.0004	9.0	< 0.02	10.3	9.9	0.41	2.8	0.59	1.5	0.56	0.045	1.26
After	M3	0-1	-0.5	6.4	< 0.0002	1.0	< 0.02	0.003	2.3	0.086	1.8	0.48	1.0	0.29	0.19	0.81
	19/8/03	11-13	+7.5	5.9	0.0005	6.8	< 0.02	7.1	8.3	0.29	3.5	0.55	1.2	0.43	0.051	0.94
Before	N3	0-1	+58	-	0.0018	8.4	< 0.02	33	42	1.4	8.4	2.9	3	0.73	0.11	1.6
	18/6/02	11-13	+139	-	0.0063	7.1	< 0.02	73	87	2	10.6	4.5	3.4	1.3	0.089	2.0
After	N3	0-1	+157	6.5	0.0044	8.4	< 0.02	82	94	4.1	10.2	3.8	3.0	0.85	0.12	4.6
	18/8/03	11-13	+181	6.2	0.0051	5	< 0.02	93	108	4.6	11.8	3.6	3.5	0.77	0.06	4.8
Before	D6	0-1	-0.2	-	< 0.0002	0.4	0.03	< 0.001	4.3	0.086	3.2	0.84	1.7	0.27	0.08	1.3
	18/6/02	11-13	+5.4	-	< 0.0002	4.6	< 0.02	5.0	10.8	0.29	6.1	1.3	2.1	0.28	0.025	1.3
After	D5	0-1	-2.2	6.7	< 0.0002	2.7	< 0.02	< 0.001	2.2	0.093	2.4	0.63	1.1	0.22	0.24	0.87
	18/8/03	11-13	-10.8	6.6	0.0055	15	< 0.02	1.9	3.4	0.27	6.7	1.6	1.3	0.24	0.19	0.83

 $HCO_{2}^{-}$  was calculated as follows:

 $HCO_3 \approx 2 [Fe^{2^*}] + 2 [Ca^{2^*}] + 2 [Mg^{2^*}] + [Na^*] + [K^*] + 2 [Mn^{2^*}] + [NH_4^*] - 2 [SO_4^{2^-}] - [CI^-]$ Protons, nitrate, fluoride, and [AI<sup>3+</sup>] were neglected. It is assumed that total iron measured by ICP-OES approximates the ferrous iron concentration. n.d.-no data.

Acidity =  $[Cl^{-}]+[F^{-}]+2$   $[SO_{4}^{2-}]+[NO_{3}^{-}]-[Na^{+}]-[K^{+}]-2$   $[Ca^{2+}]-2$   $[Mg^{2+}]-[NH_{4}^{+}].$ 

The concentrations of fluoride and nitrate were negligible calculating acidity and ionic balance in pore water.

sedimentary data the peak concentration of Mn in the uppermost sediment layer must be emphasized (Table 5).

The sediment of site N3 is quite different from that of the other two sampling sites, since site N3 represents one of the two monimolimnia of Lake Goitsche. At site N3, the ionic strength and SRP and total dissolved Fe concentration in pore water were much higher than at the other two sampling sites as shown by Table 3 and Fig. 3. Only a very small Mn peak was found in pore water (Fig. 3). Furthermore, the peak Mn concentration of solids was not significantly elevated in the uppermost sediment layer compared to the other two sub-basins.

The new sediment layer at sampling site D5 was characterised by a significant change of vertical course of SRP pore water profile after the flood event. Pore water Mn peaking was observed like at site M3. The concentration of hydrogen carbonate as calculated from Table 3 in sub-basin Doebern pore water arose after the flood event and exceeded that of site M3 at comparable sediment depths in 2003.

The minimum, median, and maximum values of Fe:SRP in the new arisen sediment of sub-basin Doebern were small compared to those in Doebern sediments before the flood event and compared to those in all sediments of the other two sub-basins before and after the flood event.

The chemical composition of the solid phase of the newly formed sediment layer in sub-basin Doebern was more homogenous compared to the other two sampling sites. This was confirmed by sedimentary concentrations of Ca, Mg, Na, and K (Na and K data not shown), which are assumed to participate only marginally in biogeochemical reactions.

The peak concentration of solid phase Mn in the uppermost sediment laver (Table 5) was observed in sub-basin Doebern and at site M3 in all investigated sediment cores before and after the flood event.

#### 4. Discussion

#### 4.1. Changes in pore water composition

The chloride  $(Cl^{-})$  concentrations in the lake water (Table 2) and in the pore water (Table 3) indicate an exchange of pore water between June 2002 and August 2003 because Cl<sup>-</sup> can be considered to be a non-reactive tracer without sources in the sediment (e.g.

soluble Cl<sup>-</sup> bearing minerals or considerable amounts of Cl<sup>-</sup> present at the surface of sediment particles due to adsorption or ion exchange). At sampling site M3, the decrease in Cl<sup>-</sup> concentration can be explained by a replacement of pore water by lake water. In June 2002 and again in August 2003, the concentrations of Cl<sup>-</sup> in the pore water were in the range of the lake water concentrations (  $\approx$  1.25 mmol/L or  $\approx$  0.85 mmol/L, respectively; Tables 2 and 3). Concentration profiles found in November 2002 and in April 2003 (data not shown) suggest that the exchange required some months. Accordingly, the changes in the concentrations of other substances in the pore water of M3 and in the shape of concentration profiles have to be considered not only as the result of biogeochemical reactions and of diffusion in the sediment but also as a consequence of the replacement of initial pore water by lake water. However, the concentration gradients of pore water profiles indicate mobilization of SRP, Fe,  $SO_4^{2-}$  and Mn (Fig. 3).

The increase in Cl<sup>-</sup> concentrations at site N3 (Table 3) can be interpreted as an inflow of groundwater into the lake as already concluded from stratification observation by Boehrer et al. (2003). The increase in Cl<sup>-</sup> concentrations in the pore water and in the monimolimnion above the sediment (Tables 2 and 3) suggest a stronger inflow of groundwater in August 2003. This stronger inflow of groundwater in August 2003 is probably a part of the general changes in the groundwater flow regime in the vicinity of Lake Goitsche after the flood, which was described in detail by Wycisk et al. (2005) for the area north of Lake Goitsche. The Clconcentrations in the pore water and in the monimolimnion at site N3 in August 2003 agreed well with each other and were in line with the Cl<sup>-</sup> concentrations of the groundwater at the western edge of subbasin Niemegk (3.75-6.79 mmol/L; Trettin et al., 2007), indicating inflow of water from the aquifer below the excavated lignite from western direction. According to Knorr and Blodau (2006), the assumption of groundwater inflow is also supported by the almost vertical shape of the concentration profiles (Fig. 3). A mobilization can clearly be deduced only for Mn in the upper part of the sediment in August 2003 (Fig. 3).

At sampling sites D5/6, the Cl<sup>-</sup> concentrations in the pore water were in the range of the lake water concentrations in June 2002 (1.24–1.3 mmol/L) and again in August 2003 (0.85 mmol/L) (Tables 2 and 3), similar to site M3. In contrast to site M3, Cl<sup>-</sup> concentrations of pore water are available for June 2002 and August 2003 only, but not for the period in between. Therefore,



**Fig. 3.** Pore water concentrations of SRP, total dissolved Fe, SO<sub>4</sub><sup>2-</sup>, and Mn at different sampling sites before (6/02, black symbols) and after (8/03, open symbols) the flood event. In sub-basin Doebern, sampling site D6 was used in June 2002 (6/02) and sampling site D5 was used in August 2003 (8/03).

two alternatives exist explaining the decrease in pore water Cl<sup>-</sup> concentrations: (i) there is a flow of lake water into the sediment as described for site M3. (ii) lake water was enclosed in the forming sediment during the flood in August 2002 and altered only by biogeochemical reactions and diffusion in the sediment later on. Checking of the second interpretation would require Cl<sup>-</sup> concentration data for the lake water during the flood. Such data are not available. Taking the hypolimnetic Cl<sup>-</sup> concentration of subbasin Doebern at site D5 from August 29, 2002 (0.84 mmol/L, measured as part of the basic lake monitoring program in samples taken with the Limnos sampler) as an approximation for the water quality during the flood, the second interpretation is still possible. Therfore, no final decision between the two alternatives can be given. However, the SRP concentration profile of the pore water clearly

indicates a mobilization of SRP between 3 and 8 cm sediment depth (Fig. 3). The concentration profiles of Mn suggest a mobilization of Mn in the upper part of the sediment before and after the flood and, in addition, in the deeper part of the sediment in August 2003 (Fig. 3). The mobilization of Fe and  $SO_4^{2-}$  was much smaller in August 2003 than in June 2002, probably due to the different origin and composition of the sediment before and after the flood.

The different flow directions between lake water and pore water at the three sampling sites can be explained by the complicated groundwater system around and below Lake Goitsche. It is characterised by the occurrence of special structural elements like local depressions in almost all geological strata caused by subrosion in deeper salt deposits, Quaternary channels cutting deep into the Tertiary strata including the lignite seam,

#### Table 4

Dissolved Fe:SRP molar ratios in sediment pore waters from lakes of different trophic status.

Lake	Sampling date	Trophic status <sup>a</sup>	pH-status <sup>a</sup>	Formation	MIN <sup>b</sup> Fe:SRP	Median Fe:SRP	MAX <sup>b</sup> Fe:SRP	Ref. <sup>c</sup>
M3 before flood event	24/10/00	-	-	mining	3600 (0-1)	12,000	15,000 (5-7)	Х
	17/6/02	ol	N	mining	5500 (3-4)	17,000	25,000 (11-13)	Х
M3 after flood event	18/8/03	ol	N	mining	5700 (4-5)	15,000	15,000 (7-9)	Х
N3 before filling (4 cores)	1997, 1998	eu	Ac	mining	460 (2-3)	17,000	26,000 (5-6)	[1]
N3 before flood event (2 cores)	17/6/02	ol	Ν	mining	2500 (29-31)	18,000	21,000 (15-17)	X
N3 after flood event	18/8/03	ol	Ν	mining	16,000 (2-3)	18,000	20,000 (23-25)	Х
D5 before filling (4 cores)	1997, 1998	ol	Ν	mining	460 (1-2)	10,000	20,000 (18-21)	[1]
D6 before flood event	18/6/02	ol	Ν	mining	5200 (25-27)	8200	13,000 (13-15)	Х
D5 after flood event	18/8/03	ol	Ν	mining	7 (1-2)	270	530 (17–19)	Х
Lake 117		ol	Ac	mining	2400 (18-30)	19,000	21,000 (2-3)	Х
Lake 111	(9 cores)	ol	Ac	mining	110 (25-28)	5500	16,000 (15-18)	Х
Lake 107		me	Ac	mining	1400 (10-11)	3800	18,000 (3-4)	Х
Lake 112		ol	Ac	mining	330 (18-20)	590	2300 (0-0.5)	[2]
Golpa IV		eu	Ν	mining	-	-	4	[3]
Lake Auensee		eu	Ν	gravel pit	-	-	0.4	[4]
Lake Arendsee		eu	Ν	natural	-	-	0.05	[5]
Lake Müggelsee		eu	Ν	natural	0.5	0.8	3 (29-30)	Х
Lake Schliersee		me	Ν	natural	0.08 (0-1)	0.8	7 (27–28)	[6]
Lake Spitzingsee		me	Ν	natural	5 (0)	36	68 (13-14)	[6]
Lake Lustsee		ol	Ν	natural	1.7 (23-24)	3.5	11(5-6)	[6]
Lake Zeller See		ol	Ν	natural	16 (3-4)	45	230 (0-1)	[7]

<sup>a</sup> ol: oligotrophic; me: mesotrophic; eu: eutrophic; N: neutral; Ac: acidic.

<sup>b</sup> In bracket: sediment depth in cm.

<sup>c</sup> X: data from own investigations; [1]: Langner (2001), [2]: Kleeberg and Grüneberg (2005), [3]: Hupfer et al. (1998), [4]: Lewandowski et al. (2002), [5]: Hupfer et al. (2000), [6]: Hupfer (2001), [7]: Hupfer and Schneider (2001).

Table 5											
Sedimentary	concentrations	of TP, Fe, M	/In, in	sediment	cores	from	different	sampling	sites in	August	2003.

Depth (cm)	TP (µm	nol/g)		Fe (mn	nol/g)		Mn (µm	ol/g)		Ca (mmol/g)		Mg (mmol/g)			
	M3	N3	D5	M3	N3	D5	M3	N3	D5	M3	N3	D5	M3	N3	D5
1	51.5	-	52.6	1.06	-	1.06	186	2.9	106	0.134	0.0047	0.146	0.214	0.094	0.263
2	38.4	-	59.1	1.57	3.52	1.00	38.5	2.8	41.1	0.099	0.0041	0.141	0.22	0.114	0.29
3	54.9	-	30.5	1.79	2.18	0.82	9.5	3.6	10.0	0.085	0.012	0.123	0.14	0.228	0.304
4	81.6	44.9	30.1	2.41	3.70	0.83	9.0	2.7	9.8	0.073	0.0035	0.143	0.132	0.093	0.288
5	48.3	30.4	29.6	1.89	3.43	0.83	5.8	2.9	9.8	0.055	0.012	0.137	0.159	0.111	0.268
7	41.3	32.1	28.1	3.63	3.04	0.78	4.5	1.7	9.9	0.041	0.0064	0.141	0.107	0.116	0.253
9	14.6	33.4	29.2	2.91	3.13	0.81	3.8	1.6	10.0	0.052	0.0064	0.144	0.136	0.107	0.265
11	12.0	36.0	29.5	4.14	3.25	0.89	3.0	1.5	10.3	0.053	0.005	0.148	0.091	0.1	0.293
13	7.0	31.6	25.9	3.54	3.34	0.87	2.4	1.5	10.0	0.065	0.0057	0.145	0.091	0.104	0.281
15	13.2	25.5	29.9	2.46	3.30	0.84	1.6	1.5	9.3	0.054	0.0057	0.139	0.115	0.095	0.274
17	10.5	30.3	26.3	1.37	3.91	0.81	1.3	1.5	9.2	0.076	0.006	0.15	0.122	0.106	0.252
19		29.6	26.2		3.73	0.82		1.4	8.9		0.002	0.141		0.096	0.254

etc. (Grützmacher et al., 2001; Eissmann, 2002a, b; Wycisk et al., 2005). The common inhomogeneity of sedimentary geological strata like the aquifers and aquitards in the vicinty of Lake Goitsche also has to be considered. This inhomogeneity is confirmed by the variability of the groundwater composition around Lake Goitsche as documented by Trettin et al. (2007).

Consequently, the conditions at the three sampling sites differ not only with respect to the thickness of the flood-borne sediment layer and the redox potential at the sediment–lake water interface but also with respect to the flow direction of the water exchange between lake water and sediment. However, the biogeochemical mobilisation processes in the sediment appear similar, at least for the sites M3 and D5/6.

#### 4.2. Evaluation of the molar Fe:SRP ratios in pore waters

The molar Fe:SRP ratios of pore water concentrations in Lake Goitsche before the flood were in the same order of magnitude as found in other acidified pit lakes. Table 4 shows that extremely high Fe:SRP ratios are typical in pore waters of lakes that have been influenced by pyrite weathering. Lake 117, Lake 112, Lake 111, and Lake 107 are highly acidic mine pit lakes located in Lower Lusatia (Brandenburg, Germany). Both sub-basins Niemegk and Muehlbeck were acidic before regular filling. Sub-basin Doebern was neutral before filling. However, its sediment clearly showed the influence of pyrite weathering. The content of Fe in sediment was more than 3.5 mmol/g dry weight in 1997 (Langner, 2001). As calculated in Table 3 pore water had an excess of acidity in N3 sediments and deeper layers of M3 and D6 sediments. Only the new deposited sediment in sub-basin Doebern (D5) showed lower acidity (at 11-13 cm sediment depth) in pore water compared to all other samples. The reason for the lower acidity might be the induced import of buffering sediment material (carbonates) from flood, which is exclusively of allochtonous origin and without any impact of lignite mining. This can be confirmed by the elevated particulate Ca and Mg concentrations of the solid phase in sub-basin Doebern sediments compared to the other sampling sites (Table 5).

After the flood, the molar Fe:SRP ratios at sampling site M3 and site N3 of Lake Goitsche remained nearly unchanged. The molar Fe:SRP ratios in the high sedimentation subbasin Doebern were rather small. The detected minimum Fe:SRP levels (Table 4) reached the order of magnitude found in oligotrophic or mesotrophic natural lakes. High Fe:SRP values (>1000) seem to be typical for acidic pit lakes or former acidic pit lakes like Lake Goitsche as shown in Table 4. In natural lakes such high Fe:SRP values were not found. Lake Golpa IV was the only reported pit lake having Fe:SRP ratios in its sediment pore water comparable to ratios of eutrophic natural lakes. This lake served as a shortterm storage of cooling water for a power plant. The water in Lake Golpa IV was replaced several times a year with water from River Mulde over many years. The river water had SRP concentrations in the range of 2.1–12.9 umol/L (Hupfer et al., 1998). This history and P load is completely different from the situation in Lake Goitsche and not representative for pit lakes whereas the conditions of Lake Goitsche can be considered to be typical for pit lakes.

The high Fe:SRP values of the pit Lake Goitsche (before and after the flood event) agree well with the low P remobilisation tendency, which was supported by pore water profiles.

# 4.3. The role of Fe in P-adsorption at the interface of the anaerobic zone and aerobic zone

The interface of the anaerobic zone and aerobic zone can be located at the sediment–water interface of sub-basins Muehlbeck, Doebern, and Niemegk and near the monimolimnion mixolimnion boundary of a small part of sub-basin Niemegk at site N3. At this interface, forming Fe oxides serve as adsorbing surfaces for P and stop it from being carried into the overlying water (Steinberg, 2003). The capacity of this P sink depends on the stoichiometric Fe:SRP ratio in the dissolved phase of the anaerobic zone (Baccini, 1984). Additionally, the dissolved Fe:SRP ratio in pore waters is reported to act as an indicator of P release to overlying oxic water (Lehtoranta and Heiskanen, 2003).

The predominant Fe:P surface complexation ratio of Fe(OOH) is suggested to be 2 mol:1 mol (corresponding to a mass ratio of 3.6). If the Fe:SRP ratio is lower than 2 in pore water or near bottom water, the recently formed Fe(OOH) at the oxic sediment surface does not suffice to bind all the diffusing SRP. Thus SRP is released into overlying lake water (Gächter and Müller, 2003). All calculated Fe:SRP ratios of sediment pore water of Lake Goitsche were above the necessary minimum of 2, which is required to ensure the co-precipitation of P with Fe after pore water comes into contact with oxygenated lake water or any other oxidising agent, e.g. nitrate.

#### 4.4. The role of Mn in P cycling

Allochthonous Mn (III, IV) is reduced in sediments under anoxic conditions either by organic carbon or ferrous iron ( $Fe^{2+}$ ) or sulphide (Davison, 1993). If Fe<sup>2+</sup> acts as reducing agent, ferrihydrite or other ferric iron mineral phases are precipitated and Mn ions are released into pore water. Mn can diffuse to the top of the sediment where it is re-oxidised and precipitated under oxic conditions. Thus, Mn operates as electron shuttle from Fe<sup>2+</sup> about 3 cm below the sediment surface to oxygen in bottom water (Davison, 1993). By this process Mn is enriched within the uppermost sediment layers. The portion of Mn oxides, which is not rapidly reduced at the sediment surface is buried in the uppermost sediment layers. Coming in contact with Fe<sup>2+</sup> under anoxic conditions, Mn is released again into pore water. By this Mn cycling, Fe precipitation and P immobilisation is enhanced due to redox coupling of Fe oxidation and Mn reduction (Hongve, 1997). The precipitated Mn in the upper sediment layer may adsorb P as an additional option for P retention in the sediment (Christensen et al., 1997).

The pore water profiles at sites M3 and D5 suggest an upward Mn flux to the bottom water caused by diffusion. At the sediment–water interface manganous ion concentration is small. Results from analysis of element concentrations in sediment material confirm the supposed diffusive transport of Mn to the sediment surface layers. Indeed, Mn was found to be enriched by a considerable factor (D5: > 10-fold; M3: > 40-fold; Table 5) within the uppermost cm of D5, and M3 sediment. The low pore water concentrations of SRP and Fe within the uppermost sediment layers also confirm that the hypothesis is reliable that P is coprecipitated with Fe, which was oxidised by Mn(III) and/or Mn(IV).

In N3 pore water, the concentration gradients of Mn were smaller than in D5 and M3 pore water. The solid phase concentration of Mn in the uppermost sediment layers was smaller compared to D5 and M3 sediments. The bottom water of site N3 is anoxic. Solid Mn (III, IV) can be reduced and dissolved already during precipitation within the monimolimnion. However, the oxicline at the interface anoxic monimolimnion/oxic mixolimnion is very similar to the upper layers of the sediment at the other sampling sites: reduced Fe and Mn diffuse into the mixolimnion, become oxidised and precipitated, and finally they settle back into the monimolimnion. During precipitation, they adsorb P preventing a P-transport into the mixolimnion.

#### 4.5. P and Fe cycling in Lake Goitsche

The low concentration of TP in pelagic water did only change temporary after the flood event. P remains buried in the sediment. As shown by pore water profiles, SRP is not released into lake water. However, SRP and Fe were released into pore water potentially by either reduction of ferric minerals or by mineralisation of organic P and precipitated potentially by oxidation of Fe<sup>2+</sup> near the sediment–water interface. This process is additionally confirmed by the observation sedimentary P and Fe to be accumulated in the uppermost sediment layers of sub-basin Doebern after the flood event.

The relative amount of Fe accumulation is small compared to Mn accumulation. This is because of the higher level of redox potential of Mn(III) and Mn(IV) reduction compared to Fe(III) reduction. However, the absolute amount of dissolved Fe in pore water and sedimentary Fe in sediments exceeds dissolved and sedimentary Mn by at least one order of magnitude. Thus, Fe is the most important redox active binding partner of P. Both in pore water and in solid material, the molar concentration of Fe exceeds that of P. Under this condition exclusively  $SO_4^{2-}$  reduction and generating of FeS would initialise P release and lake eutrophication. However, reoxidation of sulphide or pyrite would be thermodynamically favoured in Lake Goitsche sediments, which contain amorphous ferric minerals arising continuously by Fe redox cycling.

Compared to natural lakes, extremely high  $SO_4^{2-}$  concentrations occur in Lake Goitsche pore water typical for lakes which have received acid mine drainage in their history. In natural lakes, pore water  $SO_4^{2-}$  is often depleted by  $SO_4^{2-}$  reduction. Unlike natural lakes,  $SO_4^{2-}$  pore water concentrations increase with sediment depth in former acidified pit lakes. There exist two possibilities for the supply of  $SO_4^{2-}$  in pore water.  $SO_4^{2-}$  may either originate from groundwater inflow supplied by leaching of dumps or may originate from schwertmannite or jarosite dissolution (Herzsprung et al., 2002; Blodau, 2005; Trettin et al., 2007). Even in the newly deposited sediment of sub-basin Doebern, increasing  $SO_4^{2-}$  concentrations towards deeper sediment layers were

observed (Fig. 3). Since the conditions for the existence and formation of Schwertmannite (and jarosite) are no longer given after Lake Goitsche neutralisation, these minerals can be assumed of an intermediate state whose relevance is temporally limited. Pit lakes like Lake Goitsche are an excellent example for aquatic systems, where high supply of organic material and sulphate do not necessarily cause P release into lake water via Fe and sulphate reduction and formation of FeS. As discussed by Golterman (2001) P release from Fe(OOH)  $\approx$  P would only occur if more than 75% of the Fe(OOH) was converted into FeS.

A considerable pool of organic carbon exists in the sediment (in the range between 2.1 and 7.9 mmol/g) to be theoretically able to reduce more of the sedimentary Fe and Mn and parts of the pore water  $SO_4^{2-}$ . However, biogeochemical oxidation of sedimentary organic carbon and dissolved organic carbon is kinetically limited and all processes of reoxidation (of Fe<sup>2+</sup>, Mn<sup>2+</sup>, and sulphides) at the oxic/anoxic boundary have to be considered. Usually only a part of organic matter is biogeochemically available, depending on the degree of humification (Senesi and Plaza, 2007). If there is an excess of Fe(OOH) a small change of sedimentary ferric iron by reduction with organic carbon will have no great influence on P release (Golterman, 2001). The complex Fe(OOH)–P is thermodynamically more stable than the (excess of) free Fe(OOH) (Golterman, 2001).

#### 4.6. Risk of eutrophication in Lake Goitsche

During the regular filling, river water and the flood in 2002 have been the major sources of P in Lake Goitsche. Smaller contributions have come from precipitation (rain, dust, snow), local runoff, erosion, decay of terrestrial vegetation (grass and trees growing in the former mining void before filling) and groundwater. The previously minor sources continue to act beyond the end of the filling period of the void. In addition, a small outflow of the new, neighbouring Lake Seelhausener See was connected to Lake Goitsche (Fig. 1). In future, new pit lakes southwest to Lake Goitsche (Fig. 1) will also have an overflow into Lake Goitsche. Additional P will be contributed by public use of the lake, e.g. as a sports and swimming location.

Major sinks of P in the past were the co-precipitation with Fe and Al during neutralization and also to Fe after neutralization (Duffek and Langner, 2002). Sedimentation and burial of dead biomass also contributed to the removal. Still available Fe may result from groundwater inflow into the lake and from release of Fe from sediment. The described formation of an oxidised barrier (Steinberg, 2003) in the upper sediment by cycling of Mn and Fe is obviously an important factor for preventing P from being released from the sediment into the lake water. There has been no recent evidence for a relevant level of SO<sub>4</sub><sup>2–</sup> reduction that would remove Fe from the water by forming Fe sulphides. However, such a development cannot be completely excluded in future.

Kleeberg and Grüneberg (2005) discussed the potential development of pit lakes with respect to P cycling and eutrophication. Although they did not include the coupled cycling of Mn and Fe, our findings and the discussion above confirm their conclusions. A clear tendency with respect to eutrophication cannot be deduced due to the counteracting mechanisms. As long as the use and management of the pit lake prevents an overload with P as reported by Axler et al. (1998) and Hupfer et al. (1998), the risk of eutrophication is apparently small. Consequently, the sedimentary P of Lake Goitsche originating from regular filling and the flood has recently not to be considered as a source for eutrophication.

#### References

- Axler, R., Yokom, S., Tikkanen, C., McDonald, M., Runke, H.M., Wilcox, D., Cady, B., 1998. Restoration of a mine pit lake from aquacultural nutrient enrichment. Restoration Ecology 6, 1–19.
- Baccini, P., 1984. Phosphate interactions at the sediment-water interface. In: Stumm, Werner (Ed.), Chemical Processes in Lakes. Wiley, Ney York 189-204.
- Blodau, C., 2005. Groundwater inflow controls acidity fluxes in an iron rich and acidic lake. Acta Hydrochim. Hydrobiol. 33, 104–117.
- Boehrer, B., Schultze, M., Liefold, S., Behlau, G., Rahn, K., Frimel, S., Kiwel, U., Kuehn, B., Brookland, I., Büttner, O., 2003. Stratification of mining Lake Goitsche during flooding with river water, Tailings and Mine Waste 2003. Swets and Zeitlinger, Lisse 223-231.
- Boehrer, B., Schultze, M., Ockenfeld, K., Geller, W., 2005. Path of the 2002 Mulde flood through Lake Goitsche, Germany. Verh. Internat. Verein. Limnol. 29, 369–372.
- Boers, P., De Bles, F., 1991. Ion concentrations in interstitial water as indicators for phosphorus release processes and reactions. Water Res. 25, 591–598.
- Castro, J.M., Moore, J.N., 2000. Pit lakes: their characteristics and the potential for their remediation. Environ. Geol. 39, 1254–1260.
- Christensen, K.K., Andersen, F.O., Jensen, H.S., 1997. Comparison of iron, manganese, and phosphorus retention in freshwater littoral sediment with growth of Littorella uniflora and benthic microalgae. Biogeochem. 38, 149–171.
- Davison, W., 1993. Iron and manganese in lakes. Earth. Sci. Rev. 34, 119-163.
- DIN 38414-7. Aufschluß mit Königswasser zur nachfolgenden Bestimmung des säurelöslichen Anteile von Metallen. In Wasserchemischen Gesellschaft-Fachgruppe in der Gesellschaft Deutscher Chemiker in Gemeinschaft mit dem Normenausschuß Wasserwesen (NAW) im DIN Deutsches Institut für Normung e.V. Deutsche Einheitsverfahren zur Wasser, Abwasser- und Schlammuntersuchung. Wiley-VCH, Weinheim.
- Duffek, A., Langner, C., 2002. P retention in an acidic mining lake under the influence of flooding with river water. Verh. Internat. Verein. Limnol. 28, 1717–1720.
- Eckerrot, A., Petterson, K., 1993. Pore water phosphorus and iron concentrations in a shallow, eutrophic lake—indications of bacterial regulation. Hydrobiologia 253, 165–177.
- Eissmann, L., 2002a. Tertiary geology of the Saale-Elbe Region. Quat. Sci. Rev. 21, 1245–1274.
- Eissmann, L., 2002b. Quaternary geology of eastern Germany (Saxony, Saxon-Anhalt, South Brandenburg, Thuringia), type area of the Elsterian and Saalian Stages in Europe. Quat. Sci. Rev. 21, 1275–1346.
- Gächter, R., Müller, B., 2003. Why the phosphorus retention of lakes does not necessarily depend on the oxygen supply to their sediment surface. Limnol. Oceanogr. 48, 929–933.
- Golterman, H.L., 2001. Phosphate release from anoxic sediments or "What did Mortimer really write?". Hydrobiologia 450, 99–106.
- Grützmacher, G., Hindel, R., Kantor, W., Wimmer, R., 2001. Chemical investigations of aquifers affected by pyrite oxidation in the Bitterfeld lignite district. Waste Manage. 21, 127–137.
- Herzsprung, P., Friese, K., Frömmichen, R., Göttlicher, J., Koschorreck, M., Schultze, M., Tümpling jr., W.v., Wendt-Potthoff, K., 2002. Chemical changes in sediment pore waters of an acidic mining lake after addition of organic substrate and lime for stimulating lake remediation. Water Air Soil Pollut.—Focus 2 (3), 123–140.
- Herzsprung, P., Bozau, E., Büttner, O., Duffek, A., Friese, K., Koschorreck, M., Schultze, M., Tümpling jr., W.v., Wendt-Potthoff, K., 2006. Chemical analysis of sediment pore water with high ionic strength under routine analytical condition. Acta Hydrochim. Hydrobiol. 34, 593–607.
- Hongve, D., 1997. Cycling of iron, manganese, and phosphate in a meromictic lake. Limnol. Oceanogr. 42, 635–647.
- Hupfer, M., Fischer, P., Friese, K., 1998. Phosphorus retention mechanisms in the sediment of an eutrophic mining lake. Water Air Soil Pollut. 108, 341–352.
- Hupfer, M., Pöthig, R., Brüggemann, R., Geller, W., 2000. Mechanical resuspension of autochthonuous calcite (Seekreide) failed to control internal phosphorus cycle in an eutrophic lake. Water Res. 34, 859–867.
- Hupfer, M., Schneider, J., 2001. Angewandte Limnogeologie am Beispiel des Zeller sees (Land Salzburg, Österreich). Zeitschrift d. Deutschen Geolog. Gesellschaft 152, 701–716.
- Hupfer, M., 2001. Chemische und physikalische Charakterisierung von Sedimenten. Kleinseen in Bayern- Ökologische Bewertung von Freiwasser, Sediment, Ufer und Einzugsgebiet.- Informationsberichte des Bayerischen Landesamtes für Wasserwirtschaft Heft 1/01, 93–124.
- Klapper, H., Geller, W., 2001. Water quality management of mining lakes —a new field of applied hydrobiology. Acta Hydrochim. Hydrobiol. 29, 363–374.
- Kleeberg, A., Grüneberg, B., 2005. Phosphorus mobility in sediments of acid mining lakes, Lusatia, Germany. Ecol. Eng. 24, 89–100.
- Klemm, W., Greif, A., Broeckaert, J.A.C., Siemens, V., Junge, F.W., van der Veen, A., Schultze, M., Duffek, A., 2005. A study on arsenic and the heavy metals in the Mulde river system. Acta Hydrochim. Hydrobiol. 33, 475–491.
- Knorr, K.H., Blodau, C., 2006. Experimentally altered groundwater inflow remobilizes acidity from sediments of an iron rich and acidic lake. Environ. Sci. Technol. 40, 2944–2950.

- Kringel, R., Nitsche, K., Haferkorn, B., 2000. Limnology of sulfur-acidic lignite mining lakes III: approach and limitation in predicting chemical properties. Verh. Internat. Verein. Limnol. 27, 416–419.
- Langner, C., 2001. Early diagenesis in young sediments of arising lakes of lignite mining exemplified by the flooding of Goitsche in Mid-Germany (Frühdiagenetische Prozesse in jungen Sedimenten von entstehenden Seen des Braunkohletagebaus am Beispiel der Flutung des Goitschekomplexes in Mitteldeutschland). Ph.D. Thesis, Universität Bremen, Fachbereich 5, 138 pp.
- Lehtoranta, J., Heiskanen, A.S., 2003. Dissolved iron: phosphate ratio as an indicator of phosphate release to oxic water of the inner and outer coastal Baltic Sea. Hydrobiologia 492, 69–84.
- Lewandowski, J., Schauser, I., Hupfer, M., 2002. Die Bedeutung von Sedimentuntersuchungen bei der Auswahl geeigneter Sanierungs- und Restaurierungsmaßnahmen. Hydrologie und Wasserbewirtschaftung 46, 2–13.
- Miller, G.C., Lyons, W.B., Davis, A., 1996. Understanding the water quality of pit lakes. Environ. Sci. Technol. 30, 118A-123A.
- Roden, E.E., Edmonds, J.W., 1997. Phosphate mobilisation in iron-rich anaerobic sediments: microbial Fe(III)oxide reduction versus iron-sulfide formation. Arch. Hydrobiol. 139, 347–378.
- Rönicke, H., Doerffer, R., Siewers, H., Büttner, O., Lindenschmidt, K.E., Herzsprung, P., Beyer, M., Rupp, H., 2008. Phosphorus input by nordic geese to the eutrophic Lake Arendsee, Germany. Fundam. Appl. Limnol. 172, 111–119.
- Schindler, D.W., 2006. Recent advances in the understanding and management of eutrophication. Limnol. Oceanogr. 51, 356–363.
- Schultze, M., Geller, W., 2004. Seewassergüte nach einer Flusswasserflutung— Ergebnisse vom Goitschesee bei Bitterfeld. Z. angewandte Umweltforschung. Sonderheft 14, 71–82.

- Schultze, M., Klapper, H., 2004. Flusswassereinleitung als Restaurierungs- und Bewirtschaftungsstrategie für Bergbauseen—Möglichkeiten und Grenzen. Studia Quaternaria 21, 87–99.
- Schulz, M., Bischoff, M., 2008. Variation in riverine phosphorus between 1994 and 2003 as affected by land-use and loading reductions in six medium-sized to large German rivers. Limnologica 38, 126–131.
- Senesi, N., Plaza, C., 2007. Role of humification processes in recycling organic wastes of various nature and sources as soil amendments. Clean 35, 26–41.
- Steinberg, C.E.W., 2003. Die Bottom-up-steuerung der Seeneutrophierung: Waldemar Ohle legt frühzeitig die Mechanismen der Seeneutrophierung in VOM WASSER nieder. Vom Wasser 101, 91–102.
- Stottmeister, U., Mudroch, A., Kennedy, C., Matiova, Z., Sanecki, J., Svoboda, I., 2002. Reclamation and regeneration of landscapes after brown coal opencast mining in six different countries. In: Mudroch, A., Stottmeister, U., Kennedy, C., Klapper, H. (Eds.), Remediation of Abandoned Surface Coal Mining Sites. Springer-Verlag, Berlin Heidelberg, pp. 4–36.
- Trettin, R., Gläser, H.R., Schultze, M., Strauch, G., 2007. Sulfur isotope studies to quantify sulfate components in water of flooded lignite open pits—Lake Goitsche, Germany. Appl. Geochem. 22, 69–89.
- Wycisk, P., Neumann, C., Gossel, W., 2005. Flooding induced effects from the Mining Lake Goitsche on groundwater and land-use in the Bitterfeld Area. Acta Hydrochim. Hydrobiol. 33, 507–518.
- Zak, D., Gelbrecht, J., Steinberg, C.E.W., 2004. Phosphorus retention and changes of water chemistry at the redox interface of fens in Northeast Germany. Biogeochemistry 70, 359–370.

## Appendix 4

Schultze, M., Geller, W., Benthaus, F.-C., Jolas, P. (2011):

Filling and management of pit lakes with diverted river and mine water - German experiences. In: McCullough, C.D. (ed.): Mine Pit Lakes: Closure and Management. ACG Australian Centre for Geomechanics. Perth, 107-120.

Contributions of M. Schultze:Concept80%Writing75%Figures90%

# Filling and management of pit lakes with diverted river water and with mine water — German experiences

M. Schultze UFZ Helmholtz Centre for Environmental Research, Germany
 W. Geller UFZ Helmholtz Centre for Environmental Research, Germany
 F-C. Benthaus LMBV Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft, Germany
 P. Jolas MIBRAG Mitteldeutsche Braunkohlen Gesellschaft, Germany

## Abstract

Dependent upon climatic and hydrologic conditions, the duration of filling of pit lakes may range from years to many decades following mine closure. There are three main reasons to prefer fast filling for pit lakes: (i) reduction of slope erosion and landslide frequency, (ii) abatement of acidification, and (iii) shorter waiting time for beneficial end-use of the post-mining landscape including the pit lakes. These reasons have driven the use of river and active mine dewatering water for filling of pit lakes in former lignite strip mines in Germany over the last 35 years. River water has been the most important water source for filling pit lakes created after 1990. From 1990 to 2010, ca. 2.317 10<sup>9</sup> m<sup>3</sup> of river water and ca. 763 10<sup>6</sup> m<sup>3</sup> of water from dewatering operations of active mines was diverted into mine voids to create pit lakes in eastern Germany.

Fast filling allowed for steeper slopes of the mine voids before lake filling, saving money by shaping slopes of shallower inclination. Acidification was prevented or neutralised as well. Diversion of river water was also successfully applied as a strategy to prevent acidification of already filled pit lakes although, in some cases, additional chemical neutralisation was still necessary.

The most relevant processes are introduced and the German experiences in using river water and mine water for filling and management of pit lakes are presented. Examples from other countries are also presented briefly and summarising conclusions are drawn.

## 1 Introduction

For the purposes of this paper, German pit lakes are residual lakes in former surface lignite mines. Lignite has been an important contributor to the production of heat and electricity in Germany (25.2% in 2008; Statistisches Bundesamt, 2010). A similar contribution is expected for the next few decades. Since all lignite mines in Germany extend below groundwater level, pit lakes form after mine closure in the residual mine voids. Full backfilling with overburden is not possible.

Lignite has been mined in three larger and some additional smaller mining districts in Germany. Mines are currently operated in four of the districts: Rhenish district, district of Helmstedt, Central German district and Lusatian district. In the other districts, mining ceased during the last three decades. The mined lignite is of Tertiary age. The lignite seams are embedded and overlain by unconsolidated rock like gravel, sand, loam and clay. However, the geological conditions vary among the districts and also within the single districts. For details on the geology of the three main German lignite mining districts, see Seifert et al. (1993) for the Lusatian district, Eissmann (2002) for the Central German district, and Briechle et al. (1998) for the Rhenish district.

Pit lakes exist in all of the German lignite mining districts, in summary about 575 today (Nixdorf et al., 2001). The majority (about 370) of pit lakes are located in the Central German and in the Lusatian district. About 120 new lakes formed in these two districts during the last 20 years (Krüger et al., 2002). The experiences reported in this chapter mainly originate from the Central German and the Lusatian district.

Fast filling of pit lakes with water is desired for mainly three reasons:

- Fast filling stabilises the side walls of the mine voids.
- Fast filling is an approach to acidification abatement.
- Fast filling allows for early beneficial end use of the post mining landscape including the pit lakes and new socio-economic development after mine closure.

Natural filling of pit lakes by groundwater rebound may require decades or even more than 100 years in Germany. In addition to the geological conditions, the climate strongly influences the duration of filling of a pit lake. Except for the Rhenish district, evaporation from a lake surface (about 750 mm yr<sup>-1</sup>) is higher than precipitation in all other German lignite mining districts. The Central German district and the Lusatian district are located in the regions of lowest precipitation in Germany (500–600 mm yr<sup>-1</sup>). Therefore, diversion of river water and the use of water of mines still in operation (referred to as mine water in the following) became the most important strategies for filling pit lakes in Germany during the last about 35 years.

There are also a number of reasons for a permanent diversion of river water into a pit lake: (a) a mine occupied part of the course of a river and the river is diverted back into its natural bed after mine closure, (b) the pit lake shall be used as a reservoir or for flood protection, (c) river water may be necessary to maintain a minimum water level or minimum water quality in the lake; and, (d) the pit lake can be used as treatment facility to improve the water quality of the river.

The third option, i.e. the flushing of pit lakes with river water to maintain the water quality of the lakes, is the second issue which will be discussed in addition to the fast filling of pit lakes. The aim of this chapter is to describe the processes most relevant for the filling and flushing of pit lakes with river water and mine water, to present examples and the lessons learned in Germany, and to discuss briefly options for the application of fast filling and flushing with river water and mine water relevant to other countries.

## 2 Relevant processes

The most important processes are when filling or flushing pit lakes with river water or mine water (Schultze et al., 2011a):

- dilution of existing pit lake water in a purely physical sense, i.e. without chemical reaction
- displacement of existing pit lake water resulting in outflows into groundwater or into rivers downstream
- import of chemicals into the lake (including alkalinity and acidity)
- chemical reactions in the lake water (including sedimentation of resulting precipitates)
- interaction with the bottom and walls of the mine void (including landslides and erosion)
- interaction with the lake sediment
- primary production in the lake and microbial alkalinity generation.

Concentrations of almost all substances are usually higher in existing mine water (1,500–5,800 mg L<sup>-1</sup> total dissolved substances (TDS) on average, depending on mined minerals and local conditions; Banks et al., 1997; Nordstrom and Alpers, 1999) than in river water (50–1,300 mg L<sup>-1</sup> TDS on average, depending on climate, regional geology and human impact; Meybeck, 2005). Therefore, filling and flushing with river water typically results in dilution initially. Moreover, dilution also contributes to neutralisation. 1 mL of  $10^{-3}$  N hydrochloric acid has a pH of 3. The pH can be elevated to 4 by adding 9 mL of pure water. However, full neutralisation to pH 7 requires 9,999 L of pure water. That is, the contribution of dilution to neutralisation is theoretically considerable but rather limited in practical use.

In the case of fast filling with river or mine water, none or only little groundwater is able to enter the pit lakes while water is flowing from the lake into the dewatered aquifers (Figure 1). This is a major difference between filling of pit lakes exclusively by groundwater rebound and filling by river and/or mine pit dewatering water (Figure 1). The water loss from the lake to groundwater is lost from the lakes but helps faster groundwater rebound. The first consequence of the flow of lake water into the dewatered aquifers is that transport of substances from refilling aquifers into the filling lake is minimised. For example, erosive transport of particles at springs forming on pit void side walls, or washout of small particles from the refilling aquifers leading to instability of the pit wall slopes is minimised. This is a substantial advantage regarding safety and helps avoiding landslides. A second advantage for slope stability is that the pressure of the lake water against the pit walls compensates for the pressure of the groundwater. The latter alone may cause destabilisation of the slopes and landslides.



Figure 1 Interaction between pit lake and groundwater without (left), and with fast filling by diversion of river water or mine water (right), in addition to groundwater rebound. Long dash – water level and flow during mining, short dash - water level and flow during filling of pit lake, solid - water level and flow after establishing a new long-term hydrological balance (after Schultze et al., 2011a, simplified)

A further benefit of rapid filling is that products of pyrite oxidation are transported only slowly into the filling lake. That is, any acidity which has to be neutralised during fast filling of the pit lake is smaller. In addition, more time is available for natural attenuation of acidity in the groundwater which was found to be particularly relevant in older overburden dumps of German lignite mines (Hoth et al., 2005; Storch et al., 2007). Smaller inflow rates of acidity into the pit lakes also allow for proportionally greater contribution of slow in situ processes to the abatement of acidification, e.g. sulphate reduction. Koschorreck and Tittel (2007) reported mean neutralisation rates of 182.5 meq m<sup>-2</sup> yr<sup>-1</sup> for neutral lakes of relatively low productivity and 1,878 m<sup>-2</sup> yr<sup>-1</sup> for lakes of high productivity. Another slow neutralisation process is weathering of silicates (e.g. feldspars, clay minerals) which may become relevant over long periods. However, when flushing pit lakes, the effluent water should have a quality that does not damage the receiving ecosystem of the downstream stretch of the river and that fulfils legal requirements.

The import of chemicals with river water and mine water has two opposite aspects. The import of alkalinity, in particular bicarbonate, is one of the main reasons for diversion of river/mine water to rapid fill pit voids. In the case of diversion of river water, the import of aquatic organisms may accelerate developing pit lake colonisation and the establishment of a more diverse aquatic biotic community. On the other hand, river water and mine water may contain several elevated chemicals which may be undesired in pit lakes such as phosphorus, organic trace pollutants and/or toxic metals. Therefore, prior treatment may be necessary for river/mine water or, in situ treatment of the pit lake water.

The reaction of river/mine water bicarbonate with pit lake acidity is the most important chemical reaction removing lake acidification. It is accompanied by the precipitation of dissolved iron and aluminium as the main contributors to acidity in former lignite mines. In former metal mines, other metals may also have to be removed from the lake water. The success of metal removal may be limited since some metals require pH above 8 (e.g. copper, nickel, zinc) which is usually not reached without specific addition of alkaline chemicals. Co-precipitation with iron and aluminium are important mechanisms of the removal of substances from the lake water during neutralisation. Typically co-precipitated substances are phosphorus and trace metals. A similar mechanism of removal is flocculation of particles and microorganisms by iron and aluminium.

Filling and management of pit lakes with diverted river water and with mine water — German experiences

During pit lake filling, the mine void bottom and pit walls are subject of intense elution and, in the case of unconsolidated rock, of erosion resulting from pit wall runoff and wave action. Because of the rising water table, the entire pit walls are affected by erosion. The thickness of the eroded surface layer depends on the rising rate of the water table, the geomechanical properties of the pit walls and the exposure to runoff and wave action. The erosion causes an intense elution of all soluble substances (Abel et al., 2000). The pit walls and mine void bottom typically show the most intense oxidation rates of pyrite and other sulphides since they are dewatered and exposed to the atmosphere for a long time (e.g. Grützmacher et al., 2001). In hard rock environments, mining operations such as blasting may increase the number and the size of pores, fissures and fractures allowing more intense interaction between rock, oxygen and water. Therefore, mine void bottom and pit walls are often the main sources of acidity for the pit lakes during filling. In the long term, products of pyrite oxidation stored in overburden dumps and ongoing pyrite oxidation there become the most important sources of acidity for pit lakes. Even worse events compared to erosion are landslides. They may cause the total reversal of the neutralisation successes of several months as reported by Gröschke et al. (2002). A number of landslides resulted in a drop of pH from about 5 to about 4 accompanied by an increase of acidity by about 0.1 mmol L<sup>-1</sup> and of the sulphate concentration from about 600 mg L<sup>-1</sup> to temporarily 800 mg L<sup>-1</sup> in Lake Gräbendorf (volume (V):  $93 \times 10^6$  m<sup>3</sup>, Lusatian mining district) in late summer 1997 (Gröschke et al., 2002).

The filling and neutralisation of pit lakes is accompanied by the formation of high amounts of precipitates of iron and aluminium, resulting in temporarily high sedimentation rates. The interaction between lake water and lake sediment may be limited due to this fast accumulation of sediment. The interaction between lake and groundwater varies in space and time as the lignite mines are usually in contact with more than one aquifer. The lowest aquifer of the filling lake often has a higher hydraulic head than the lake during filling since this aquifer has never been fully dewatered. This causes permanent inflow into the lake from this aquifer across the forming lake sediment. Upper aquifers may be still unsaturated with water at the same time and receive water from the filling pit lake. However, stable biogeochemical conditions in the sediment are important to prevent the remobilisation of undesired substances, e.g. by reductive dissolution of ferric iron precipitates.

The potential role of microbial sulphate reduction as a source of alkalinity was already discussed above. Primary production is a considerable source of organic material which contributes organic carbon as a substrate for sulphate reduction in the lakes' sediment. However, Totsche et al. (2006) demonstrated that artificial eutrophication, through stimulation of primary production, is limited by phosphorus fixation to iron minerals in the lake sediment and cannot be considered an effective approach for pit lake neutralisation. Accordingly, the contribution of primary production to the neutralisation in the case of fast filling with river water or mine water has to be expected as rather small. In the case of ongoing lake flushing with river water, the contribution of primary production becomes more relevant over longer terms.

## 3 Filling and flushing of pit lakes with river water and mine water in Germany

The first German pit lake filled with river water was Lake Senftenberg (V:  $98 \times 10^6 \text{ m}^3$ , Lusatian district). Its filling with water from river Schwarze Elster lasted from 1967 to 1972. The lake is still flushed by diversion of part of the river water. The Muldereservoir (V:  $110 \times 10^6 \text{ m}^3$ , Central German district) was the second pit lake filled with river water (1975–1976). The entire Mulde River still flows through the lake. From 1990 to 2010, diversion of river water contributed about 64.4% (ca.  $2.317 \times 10^9 \text{ m}^3$ ) of the pit lake volume filled in that period. About 763 ×  $10^6 \text{ m}^3$  of mine water were used for filling of pit lakes, corresponding to 21.2% of the pit lake volume filled in that period. Figure 2 shows the amounts of river water and mine water used in the Lusatian district and in the Central German district over individual years. Fast filling and flushing of pit lakes with river water and/or mine water has also become the main strategy for the abatement of acidification of pit lakes in Germany in the last 20 years. The large future pit lakes in the Rhenish mining district (Lake Garzweiler, V:  $1.18 \times 10^9 \text{ m}^3$ ; Lake Hambach, V:  $5.3 \times 10^9 \text{ m}^3$ ) will also be filled with river water including water from the Rhine River.





The filling of the pit lakes was planned based on predictions of water balance and water quality for all individual pit lakes. First estimates were based on the local hydrogeology and limnological knowledge from existing pit lakes. These estimates were followed up by elaborated predictions with models. The predictions have been updated based on collected field data as filling and lake development proceeded in order to adapt filling and management to real development.

#### 3.1 Primary filling with river water



Figure 3 Fluxes of river water diverted into pit lakes in the Lusatian and Central German Districts since 1990. (The data from the Lusatian District for 2009 and 2010 also contain the amount of mine water used in the respective river catchment area)

Since 1990, 36 pit lakes received river water for filling. That is 30% of the pit lakes to be filled after 1990. These lakes comprise  $3,490 \times 10^6$  m<sup>3</sup> after complete filling, representing 77.5% of the final volume of pit lakes to be filled since 1990. Figure 3 shows the contribution of the different rivers in the Lusatian District and in the Central German District to the filling of pit lakes.

Although the first diversion of river water was done in the Lusatian District, the most detailed study on the consequences of filling pit lakes with river water was conducted at Lake Goitsche (V:  $213 \times 10^{6}$  m<sup>3</sup>) in the Central German District (Duffek and Langner, 2002; Schultze et al., 2002; Boehrer et al., 2003; Trettin et al., 2007; Schultze et al., 2011a). Table 1 summarises changes in Lake Goitsche water quality during its regular filling from May 1999 to July 2002. The flood event of August 2002 (Klemm et al., 2005) is not shown in Table 1 since it is not typical for the use of river water for filling pit lakes in Germany. Lake Goitsche consists of three basins. At the bottom of the basins, small water bodies (referred to as precursor lakes in the following) formed before the diversion of river water began. The precursor lakes were fed by precipitation, local surface runoff and groundwater. Their water levels were kept stable by dewatering operations.

Table 1Filling of Lake Goitsche from May 1999 to July 2002. ANC – acid neutralisation capacity, analysed by titration<br/>with 0.1N HCl to pH 4.3, BNC – base neutralisation capacity, analysed by titration with 0.1N NaOH to pH 8.2;<br/>TP – total phosphorus. \* Amount of river water diverted from Mulde River which has a mean flow rate of<br/>64 m³ s<sup>-1</sup> (period 1961–2007)

	Water level (m a.s.l.)	Volume (10 <sup>6</sup> m <sup>3</sup> )	рН	ANC (mmol L <sup>-1</sup> )	BNC (mmol L <sup>-1</sup> )	ΤΡ (μg L <sup>-1</sup> )
Lake Goitsche						
April 1999						
Basin Muehlbeck	53.5	1.7	3.2	0.0	4.3	<6
Basin Niemegk	40.0	4.0	2.9	0.0	9.6	290
Basin Doebern	35.0	1.2	7.6	1.1	0.1	<6
July 2002						
Entire lake	71.5	166	7.7	0.7	0.04	<6
River Mulde May 1999 – July 2002	-	140*	6.6–7.6	1.2–1.5	0.02–0.16	19–135

Because of the poor water quality in the rivers in eastern Germany in the first half of the 1990s, in particular their high phosphorus concentrations (>100  $\mu$ g L<sup>-1</sup>), eutrophication was suspected to be an important issue in the filling pit lakes. However, as indicated by the low phosphorus concentration in July 2002 (Table 1), expected eutrophication did not occur in Lake Goitsche. Instead, phosphorus was removed rapidly from pit lake water without causing eutrophication (Duffek and Langner, 2002; Herzsprung et al., 2010). The same loss of phosphorus was found for the other lakes filled with river water (Lessmann et al., 2003; Schultze et al., 2005) because of high lake sediment and pore water iron concentrations (Kleeberg and Grüneberg, 2003; Kleeberg et al., 2008; Herzsprung et al., 2010). Additionally, the inflow of groundwater is a permanent source of iron for pit lakes over a long term. The risk of contamination with pathogens by river water was also not found to be relevant (Pusch et al., 2005; Wolf, 2005).

The rapid filling of the lakes Wallendorf (V:  $38 \times 10^6 \text{ m}^3$ ) and Raßnitz (V:  $66 \times 10^6 \text{ m}^3$ , Central German district) demonstrated the stabilising effect of filling with river water on meromixis (Schultze and Boehrer, 2008). The meromixis in Lake Goitsche was probably also a result of the filling with river water (Boehrer et al., 2003).

In the early 1990s, the potential use of diverting floods to fill pits was discussed. The outcome of this discussion was that the use of floods for filling pit lakes would be not sufficient in Germany. Firstly, the occurrence of flood events is unpredictable. Secondly, the lignite mined in Germany is embedded in unconsolidated rock. Enormous constructions were necessary to protect the side walls against flood waves flowing into empty voids. Therefore, limited but constant, diversion rates in the range of a few cubic metres per second were found to be a better option. However, the diversion of river water for filling pit lakes always requires the consideration of already existing water use downstream and the ecological needs of

the river; including seasonal variability of flow rate and the frequency and magnitude of floods. Therefore, the diversion of river water had to be reduced or even interrupted during periods of low river flow. Environmental authorities required further interruptions during flood events in rivers considerably contaminated with hazardous substances. The flushing of the catchment area caused remobilisation of contaminated sediments often leading to exceptionally high concentrations of these chemicals in the phase of rising flow rates (e.g. Baborowski et al., 2004).

### 3.2 Primary filling with mine water

Figure 4 shows the southern part of the Central German lignite mining district. The majority of the pit lakes which formed in that region during the last 20 years were filled with mine water. No river water was used since it was needed in other parts of the Central German District and enough mine water was available from the two mines which are still in operation.



Figure 4 Distribution system for mine water in the south of Leipzig (Central German Mining District).
VSH – Vereinigtes Schleenhain; Names of the lakes: 1-Cospuden (V: 109 × 10<sup>6</sup> m<sup>3</sup>), 2-Markkleeberg (V: 60 × 10<sup>6</sup> m<sup>3</sup>), 3-Zwenkau (V: 176 × 10<sup>6</sup> m<sup>3</sup>), 4-Stöhna (V: 11 × 10<sup>6</sup> m<sup>3</sup>), 5-Störmthal (V: 157 × 10<sup>6</sup> m<sup>3</sup>), 6-Werben (V: 9 × 10<sup>6</sup> m<sup>3</sup>), 7-Kahnsdorf (V: 22 × 10<sup>6</sup> m<sup>3</sup>), 8-Hain (V: 73 × 10<sup>6</sup> m<sup>3</sup>), 9-Haubitz (V: 25 × 10<sup>6</sup> m<sup>3</sup>), 10-Großstolpen (V: 0.3 × 10<sup>6</sup> m<sup>3</sup>), 11-Witznitz (V: 53 × 10<sup>6</sup> m<sup>3</sup>), 12-Borna (V: 97 × 10<sup>6</sup> m<sup>3</sup>), 13-Bockwitz (V: 19 × 10<sup>6</sup> m<sup>3</sup>), 14-Hauptwasserhaltung (V: 1.1 × 10<sup>6</sup> m<sup>3</sup>), 15-Südkippe (V: 1.6 × 10<sup>6</sup> m<sup>3</sup>), 16-Haselbach (V: 24 × 10<sup>6</sup> m<sup>3</sup>), 17-Harthsee (V: 5.4 × 10<sup>6</sup> m<sup>3</sup>) (from Schultze et al., 2011b)

Four main aspects stimulated consideration of mine water as a source of water for filling pit lakes in the Central German lignite mining district in the first half of the 1990s:

- The water quality of the river water was very poor, with high concentrations of biological and chemical oxygen demand, phosphorus and (at least in winter) ammonia as wastewater treatment was in an early stage of implementation. The experience from a well monitored pit lake filling with river water e.g. like that of Lake Goitsche, was not yet available. Therefore, treatment of the river water was discussed to ensure an adequate water quality of the new pit lakes suiting the requirements of the planned use, mainly recreation.
- 2. Treatment facilities for the mine water of the still operating mines were required by the regional authorities in order to improve the water quality of the rivers. In particular, the iron load of the mine water, (<10 mg L<sup>-1</sup>), impacted the rivers with oxygen consumption and turbidity.
- 3. The iron concentration of the mine water of the operating mines was low (<10 mg L<sup>-1</sup>) compared to the acid waters occurring in the abandoned mine voids (>200 mg L<sup>-1</sup> of iron). The water from the operating mines was net-alkaline as it was mainly natural groundwater originating from

aquifers which were not influenced by mining before. Estimates indicated that the bicarbonate of the mine water would be enough to neutralise the majority of the pit lakes to be filled.

4. The use of the mine water provided an economic benefit for the ones responsible for the filling of the pit lakes as well as for the operators of the active mines. Construction and operation of treatment facilities could be saved by both parties. Therefore, the pipeline system shown in Figure 4 was constructed (ca. 60 km).

Mine water quality from the two operating mines used for filling pit lakes in the south of Leipzig differed at the beginning. The bicarbonate concentration of the water from mine Profen was higher (6.2 mmol L<sup>-1</sup>  $HCO_3^-$  on average in 1998) than of the water from mine Vereinigtes Schleenhain (VSH; 0.46 mmol L<sup>-1</sup>  $HCO_3^-$  on average in 2000). Over time, the contribution of water from mining affected areas and overburden dumps increased for both mines causing a gradual decrease of the bicarbonate concentration in the mine water. The resulting bicarbonate concentrations were 4.8 mmol L<sup>-1</sup>  $HCO_3^-$  for mine Profen and 0.02 mmol L<sup>-1</sup>  $HCO_3^-$  and pH 4.0 for mine VHS on average in 2009. Due to the decrease in water quality, the use of the water from mine VSH for filling pit lakes was stopped in 2010, and a treatment plant for this mine water was implemented. The water of mine Profen is still a valuable source for filling and neutralising pit lakes.

The first pit lake filled with mine water was Lake Cospuden. In 1993, i.e. the year before filling with mine water began, several small precursor lakes had formed at the bottom of the mine void. Half of the precursor lakes were acidic, half of them neutral. The filling of Lake Cospuden started in 1994 with water from the neighbouring mine Zwenkau which was in operation until 1999. The majority of water for filling came from mine Zwenkau (Table 2) since the construction of the pipeline from mine Profen took some time. However, water from mine Profen also contributed substantially to the filling of Lake Cospuden and in particular to its neutralisation (Table 2). The quality of water from mine Zwenkau varied widely from neutral to acidic depending on the sites within mine Zwenkau where the water came from. The mine Profen provided well-buffered neutral water from 1998 to 2000.

Table 2	Filling of Lake Cospuden. Q – amount of mine water for filling from 1994 to June 30, 2000; ANC – acid
	neutralisation capacity, analysed by titration with 0.1N HCl to pH 4.3; BNC - base neutralisation capacity,
	analysed by titration with 0.1N NaOH to pH 8.2; water quality data are median values (minimum-maximum)

Water	Q (10 <sup>6</sup> m <sup>3</sup> )	рН	ANC (mmol L <sup>-1</sup> )	BNC (mmol L <sup>-1</sup> )
Mine Zwenkau	42,15	6.6 (2.4–8.5)	2.0 (0.0–3.5)	0.57 (0.1–6.1)
Mine Profen	31,86	7.3 (7.0–7.8)	5.9 (3.2–6.5)	0.8 (0.5–3.5)
Lake Cospuden				
1993	-	4.9 (3.1–8.1)	0.38 (0–3.75)	1.64 (0.1–59.3)
June 2000	-	7.6	1.6	0.25

When filling of Lake Cospuden was completed in summer 2000, an excellent water quality was achieved fully suiting the requirements of the planned end use for recreation (e.g. swimming, scuba diving, sailing). Because of its location at the margin of the city of Leipzig (ca. 0.5 million inhabitants), Lake Cospuden became a popular recreation site. Basically, the filling with mine water resulted in substantial removal of acidity in all lakes filled with mine water in addition to Lake Cospuden in the south of Leipzig since 1990: Markkleeberg, Zwenkau, Störmthal, Werben, Großstolpen, Kahnsdorf, Hain, Haubitz and Haselbach.

In some pit lakes the neutralisation capacity of the used mine water was not high enough to overcome the acidity already present in the mine voids and entering the rising pit lakes by groundwater, interflow, local runoff and erosion. The lakes Hain and Haubitz had to be treated with lime (8,614 t for Lake Hain, 1,390 t for Lake Haubitz). Lake Haselbach is the only exception since cessation of filling by water from the VSH mine and still receives water from this mine to maintain the water level. Nevertheless, Lake Haselbach was still treated with limestone (314 t) to maintain an acceptable pH buffering capacity. More details on the filling of pit lakes in the south of Leipzig can be found in Schultze et al. (2011b).

In some cases, dewatering wells had to be operated around filling pit voids even during pit lake filling to avoid local destabilisation of the side walls by high groundwater inflow rates. For example, at part of the eastern side walls of Lake Goitsche, dewatering wells were necessary due to the high local groundwater level. The water from such dewatering operations was generally diverted into the rising pit lakes but was not treated since it was usually not acidic and the amount of water was generally small compared to the amount of filling water from other sources such as river water or mine water from other mines.

In the Lusatian district, mine dewatering water has to be treated with lime because of its acidity before it is used for filling pit lakes. For example, the water treated in the treatment plant Rainitza is used for filling Lake IIse (V:  $135 \times 10^6$  m<sup>3</sup>). The general water scarcity in the Lusatian district requires the use of all sources.

#### 3.3 Flushing with river water

The motivation for intentional flushing of pit lakes was to achieve stabilisation of the water level, therefore avoiding erosion and landslides, and the management of the water quality. Inflow of acidic groundwater continues often after filling and primary neutralisation of pit lakes. This import of acidity causes a decrease in pH if the rate of import of alkalinity is smaller than that of acidity. Such a new acidification is referred to "re-acidification" since an initially neutralised pit lake becomes acidic again. The main sources of alkalinity import to be considered are: surface inflows, groundwater, biogeochemical alkalinity generation inside the pit lake including its sediment and elution of side walls and shore material. The major sources of acidity are the same ones as relevant for the filling of pit lakes. The re-acidification and because of the time because of the stock of alkalinity of the lake water from primary neutralisation and because of the time required for full groundwater rebound. In general, substantial changes in the hydrological regime of a pit lake may cause re-acidification if relevant sources of acidity are present in the catchment of the pit lake.



Figure 5 Development of water level and pH in Lake Senftenberg (Lusatian lignite mining district, Germany; adapted from Werner, 1999)

Figure 5 shows the development of the pH in the two basins of Lake Senftenberg. This lake is the first example in Germany for neutralisation by diversion of river water and also for lake flushing as the main measure for sustaining neutral conditions (Werner et al., 2001). Lake Senftenberg consists of two basins separated by an island which results from an overburden dump. The diversion of water from the Schwarze Elster River neutralised only the main basin since shallow areas at the ends of the island with dense macrophyte stands hindered water exchange between the two basins. In 1995, the inflow pipeline had to be closed for inspection and repair. This temporary absence of river water inflow caused an exceptionally long lake water level reduction and intermediate re-acidification (signed by the arrow in Figure 5). This episode underlines the importance of the flushing for the water quality of Lake Senftenberg.

For the future, the flushing of a chain of currently filled pit lakes is planned in the Lusatian lignite mining district. This chain of pit lakes comprises from east to west the lakes Spreetal (V:  $97 \times 10^6 \text{ m}^3$ ), Sabrodt

(V:  $27 \times 10^6 \text{ m}^3$ ), Bergen (V:  $3 \times 10^6 \text{ m}^3$ ), Bluno (V:  $64 \times 10^6 \text{ m}^3$ ), Neuwiese (V:  $56 \times 10^6 \text{ m}^3$ ), Partwitz (V:  $133 \times 10^6 \text{ m}^3$ ), and Sedlitz (V:  $212 \times 10^6 \text{ m}^3$ ). The water for flushing will be diverted from the Spree River in the east. The Schwarze Elster River will receive the outflow in the west.

Beyond acidification, other aspects of water quality may be managed by flushing of pit lakes with river water, e.g. the concentration of sulphate in the pit lakes or in the receiving downstream waters. High sulphate concentrations cause corrosion of concrete along contaminated rivers and leads to expenses for purification of water for industrial purposes and water supply (drinking water threshold for sulphate in the European Union: 240 mg L<sup>1</sup>). Additionally, sensitive aquatic species may be detrimentally affected even to death in waters downstream of pit lakes. Due to these diverse reasons and the expected risks of reacidification, a sophisticated, model-based water management system was implemented for the Lusatian lignite mining district (Schlaeger et al., 2003; Koch et al., 2005). This system combines currently filling of pit lakes with river water and mine water and lake flushing with river water. Flushing will be the future main strategy to sustain acceptable water quality in the pit lakes as well as in the rivers. The management system includes the water exchange among the rivers and their catchments: about  $30 \times 10^6$  m<sup>3</sup> yr<sup>-1</sup> between Neiße and Spree and about  $60 \times 10^6$  m<sup>3</sup> yr<sup>-1</sup> between Spree and Schwarze Elster (Luckner, 2006). In order to manage potential future phases of water scarcity in the mentioned rivers caused by climate change, diversion of water from the larger rivers Elbe and Oder is under discussion (Koch et al., 2009).

Flushing also occurs where the lakes are used for flood protection or as reservoirs. The lakes Zwenkau, Stöhna, Witznitz and Borna (all Central German district, see Figure 4) are used for flood protection. They replace the retention capacity of mined areas in the former flood plains. Substantial stretches of Weiße Elster and Pleiße Rivers have artificial river beds where mining occurred in the former flood plains. Because of the natural water scarcity in Lusatia and the importance of the Spree River for the water supply for Berlin, construction of reservoirs and the use of pit lakes as reservoirs has been an important issue for many decades (Kaden et al., 1985; Grünewald, 2001; Koch et al., 2009). The pit lakes used as reservoirs in the catchment area of the Spree River are: Bärwalde (V:  $173 \times 10^6 \text{ m}^3$ ), Burghammer (V:  $35 \times 10^6 \text{ m}^3$ ), Lohsa I (V:  $5.8 \times 10^6 \text{ m}^3$ ) and Lohsa II (V:  $97 \times 10^6 \text{ m}^3$ ). Additionally, the pit lakes Senftenberg and Knappenrode (V:  $7.8 \times 10^6 \text{ m}^3$ ) are used as reservoirs in the catchment of the Schwarze Elster River.

As mentioned above, the entire Mulde River permanently flows through the Muldereservoir. The Muldereservoir has beneficial consequences for Elbe River and the North Sea (Zerling et al., 2001; Klemm et al., 2005). The upper catchment of Mulde River was subject of metal mining and metallurgy for about 800 years. Substantial fractions of the resulting load of heavy metals and arsenic of Mulde River are trapped in the Muldereservoir. The comparison between the total load of Elbe River into the North Sea and the load trapped in the sediment of the Muldereservoir shows the importance of this pit lake for the river system downstream (average values from 1993 to 1997, trapped load as t yr<sup>-1</sup> and as percent (%) of the total load of Elbe River into the North Sea; Zerling et al., 2001): As 21.6 t yr<sup>-1</sup> – 27.0%, Cr 14.6 t yr<sup>-1</sup> – 20.6%, Cd 5 t yr<sup>-1</sup> – 90.3%, Pb 43 t yr<sup>-1</sup> – 50.8%, Zn 243 t yr<sup>-1</sup> – 15.8%, Cu 26.4 t yr<sup>-1</sup> – 22.8%.

## 4 Application in other countries

Filling of pit lakes with river water is known from the Czech Republic (Sixta, 1998; Svoboda et al., 2008), from Spain (Arnold et al., 2002; Delgado, 2005) and from Australia (Lund and McCullough, 2008; Salmon et al., 2008). In Spain and in the Czech Republic, the main goal of the diversion of river water for the filling of pit lakes is the stabilisation of the side walls and the early use of the pit lakes. Due to the geological conditions, acidification is not likely in almost all Czech pit lakes in former lignite mines in North Bohemia (Svoboda et al., 2008). However, the available amount of water is limited requiring an integrated river basin management for the two main rivers of the region, rivers Ohre and Bilina (Svoboda et al., 2008). The results of the water quality development of the pit lakes in the Czech Republic appear satisfying and have met the expectations so far (Prikryl, 2010). Filling of pit lakes in former coal mines in north-west Spain is underway yet. In the case of the Lake Meirama, the mine was operating at the bottom of a river valley. The river was

diverted around the active mine. After filling the pit lake, the river will be connected to the lake (Delgado, 2005).

Lake Kepwari (V:  $24 \times 10^{6}$  m<sup>3</sup>, Collie region of Western Australia) was filled with river water over three winters until 2005. The pH increased from 4.3 (2002) to 4.8 (2005), but decreased afterwards to 4.5 in 2009 (Salmon et al., 2008; McCullough et al., 2010). A second goal was the protection of a downstream reservoir from high salt concentrations in the seasonal river from first-flush events at the beginning of the wet season (McCullough et al., 2010). The high salt content in the first flush of rivers after the dry season results from agricultural land use and occurs in many regions of Australia (e.g. Jolly et al. 2001). Saline river water was diverted three times into Lake Chicken Creek 4 (surface area (A): 21.6 ha, maximum depth ( $z_{max}$ ): 41 m, Collie region) increasing the pH from 2.6 to 5.7 and chloride concentration from 980 mg L<sup>-1</sup> to 2,540 mg L<sup>-1</sup>. Without river water, the pH and chloride concentration decreased to 4.0 and 1,410 mg L<sup>-1</sup> in 2009, respectively (McCullough et al., 2010). Lake Stockton (A: 15.4 ha,  $z_{max}$ : 47 m, Collie region) was temporarily flushed with mine water resulting in a temporal reduction of acidity and increase of pH (McCullough et al., 2010). These are examples for transferability of experiences among mining regions. The described development is comparable to the findings in Germany, regarding both the beneficial use of river and mine water as well as the phenomenon of re-acidification.

Kalin et al. (2001) reported the filling of the B-Zone pit lake (A: 24 ha,  $z_{max}$ : 54 m) with water from the neighbouring natural Lake Wollastone in winter 1991/1992. The excellent water quality of the pristine Lake Wollaston resulted in good water quality in the B-Zone pit lake too. The pit lake water quality even improved over time (1992–1998) as indicated by changes in the plankton community (Kalin et al., 2001).

A very special case of filling a pit lake with surface water is the Island Copper Mine pit lake on Vancouver Island, Canada (V:  $241 \times 10^6$  m<sup>3</sup>, A: 1.73 km<sup>2</sup>,  $z_{max}$ : 350 m; Pelletier et al., 2009). The lake was filled mainly with sea water. Only the top layer was filled with freshwater making the lake meromictic. The intention was to establish a system for storage and treatment of Acid Rock Drainage (ARD). Low level contaminated ARD was diverted into the top layer whereas the more heavily contaminated ARD was diverted to a depth of 220 m below the lake surface. Flushing of the top layer by diversion of fresh and brackish water is essential for the stability of the meromixis. Otherwise, the density of the top layer would increase gradually. For more details see Fisher and Lawrence (2006) and Pelletier et al. (2009).

The South Pit Lake in Tennessee (550 m long, 146 m wide, 61 m deep; Wyatt et al., 2006) is another special case. The pit lake is flushed permanently by the North Potato Creek (mean flow 31 m<sup>3</sup> min<sup>-1</sup>, mean pH 5, 0.5 meq L<sup>-1</sup> acidity, 10 mg L<sup>-1</sup> iron at the treatment site) as part of the treatment strategy for the stream water. The North Potato Creek is contaminated with heavy metals since it drains a substantial part of Tennessee Copper Basin where mining took place in the past. The intention was to use the lake as disposal site for sludge resulting from the lime based treatment. Because the lake is meromictic, its monimolimnion also serves as source for iron rich water used to improve the flocculation in the treatment system. Although not the focus of the treatment system, the flushed surface layer of the South Pit Lake became pH neutral within a few weeks after implementation of the treatment system (Wyatt et al., 2006).

Probably, not all cases of filling and flushing of pit lakes with river water or mine water are known to the authors. However, the mentioned examples demonstrate that these strategies for filling and management of pit lakes are not an exclusively German issue.

## 5 Conclusions

Fast filling of pit lakes with river water and mine water and flushing with river water have proved to be very useful strategies of pit lake creation and management in Germany. Pit walls were stabilised and acidification could be prevented, or, at least, reduced. These strategies were also applied in other countries.

The basic prerequisite for the use of river water and mine water for filling and management of pit lakes is the availability of water. Water scarcity may be a limiting factor. That is, the applicability of filling and

flushing of pit lakes with river water and mine water strongly depends on the climate and the intensity of the use of water downstream the pit lakes. In the case of limited water availability, the use of floods may be evaluated different from the practices in Germany. Floods may be the only options for the filling of pit lakes under certain conditions. However, the ecological needs of the river system downstream the pit lakes have to be kept in mind, including the variability of the flow rate under such conditions.

The water quality of the used river water and mine water has to suit the requirements of the planned use of the pit lakes. Otherwise, treatment of the river water, the mine water or the pit lake is necessary. The pit lakes can also be used as reactors under certain conditions. For example, the removal of moderate concentrations of metals from water may be cheaper by frequent fertilisation of a pit lake than in typical mine water treatment facilities. The concentrations of copper and zinc have been kept below the permitted limits in the top layer of the Island Copper Mine pit lake by year-round fertilisation with nitrogen and phosphorus (Pelletier et al., 2009). Although the main goal in this case is the pit lake, the example indicates that this approach may be used as a treatment for contaminated river water or mine water at other sites. The Island Copper Mine pit lake is also a good example for the maintenance of meromixis by the superficial inflow of fresh water. Sustaining meromixis is an important aim in pit lakes which have been used as disposal sites for waste materials from mining or other sources.

Any case of using pit lakes for flood protection and water storage is essentially flushing the lakes with river water; although the management of the water quality of the pit lake is usually not the main goal. An optimum may result if all potential aspects are considered when planning and managing such pit lakes.

## References

- Abel, A., Michael, A., Zartl, A. and Werner, F. (2000) Impact of erosion-transported overburden dump materials on water quality in Lake Cospuden evolved from a former open cast lignite mine south of Leipzig, Germany, Environmental Geology, Vol. 39, pp. 683–688.
- Arnold, I., Rivas Rodenas, F., Rolland, W. and Strempel, P. (2002) Flooding-project of the open cast mine As Pontes, Spain, Surface Mining, Braunkohle and Other Minerals, Vol. 54, pp. 321–331.
- Baborowski, M., von Tümpling, W. and Friese, K. (2004) Behaviour of suspended particulate matter (SPM) and selected trace metals during the 2002 summer flood in the River Elbe (Germany) at Magdeburg monitoring station, Hydrology and Earth System Sciences, Vol. 8, pp. 135–150.
- Banks, D., Younger, P.L., Arnesen, R-T., Iversen, E.R. and Banks, S.B. (1997) Mine-water chemistry: the good, the bad and the ugly, Environment Geological, Vol. 32, pp. 157–174.
- Boehrer, B., Schultze, M., Liefold, S., Behlau, G., Rahn, K., Frimel, S., Kiwel, U., Kuehn, B., Brookland, I. and Büttner, O. (2003) Stratification of mining Lake Goitsche during flooding with river water, Tailings and Mine Waste '03, Swets and Zeitlinger, Lisse, pp. 223–231.
- Briechle, D., Holl-Hagemeier, C., Lögters, C., Mayers-Beecks, E., Milbert, G., Paas, W. and Prüfert, J. (1998) Raw material extraction in the Rhenish Lignite District: the effect of deep open-cast mining on the population and environment, Chronique de la Recherche Miniere, Vol. 533, pp. 13–29.
- Delgado, J.L. (2005) Rehabilitation of Meirama pit lake, in Proceedings 9th IMWA Congress, J. Loredo and F. Pendas (eds), 5–7 September, Oviedo, Spain, Departemento de Explotacion y Prospeccion de Minas, University of Oviedo, Oviedo, pp. 267–272.
- Duffek, A. and Langner, C. (2002) Pretention in an acidic mining lake under the influence of flooding with river water, Verhandlungen Internationale Vereiningung Limnologie, Vol. 28, pp. 1717–1720.
- Eissmann, L. (2002) Tertiary geology of the Saale-Elbe Region, Quaternary Science Reviews, Vol. 21, pp. 1245–1274.
- Fisher, T.S.R. and Lawrence, G.A. (2006) Treatment of Acid Rock Drainage in a meromictic mine pit lake, Journal of Environmental Engineering, Vol. 132, pp. 515–526.
- Gröschke, A., Uhlmann, W., Rolland, W. and Grünewald, U. (2002) Hydrochemische Entwicklung Lausitzer Tagebauseen während der Flutung Beispiel Gräbendorfer See, Hydrologie und Wasserbewirtschaftung, Vol. 46, pp. 256–267.
- Grünewald, U. (2001) Water resources management in river catchments influenced by lignite mining, Ecological Engineering, Vol. 17, pp. 143–152.
- Grützmacher, G., Hindel, R., Kantor, W. and Wimmer, R. (2001) Chemical investigations of aquifers affected by pyrite oxidation in the Bitterfeld lignite district, Waste Management, Vol. 21, pp. 127–137.
- Herzsprung, P., Schultze, M., Hupfer, M., Boehrer, B., von Tümpling, W., Duffek, A., van der Veen, A. and Friese, K. (2010) Flood effects on phosphorus immobilisation in a river water filled pit lake – Case study Lake Goitsche (Germany), Limnologica, Vol. 40, pp. 182–190.
- Hoth, N., Feldmann, H., Rinker, A., Glombitza, F. and Häfner, F. (2005) Reductive processes within lignite dumps chance of a long-term natural attenuation process, Geoderma, Vol. 129, pp. 19–31.

- Jolly, I.D., Williamson, D.R., Gilfedder, M., Walker, G.R., Morton, R., Robinson, G., Jones, H., Zang, L., Dowling, T.I., Dyce, P., Nathan, N., Nandakumar, N., Clarke, R. and McNeill, V. (2001) Historical stream salinity trends and catchment salt balances in the Murray–Darling Basin, Australia, Marine and Freshwater Research, Vol. 52, pp. 53–63.
- Kaden, S., Luckner, L., Peukert, D. and Tiemer, K. (1985) Decision support model systems for regional water policies in open-pit lignite mining areas, International Journal of Mine Water, Vol. 4, pp. 1–16.
- Kalin, M., Cao, Y., Smith, M. and Olaveson, M.M. (2001) Development of the phytoplankton community in a pit-lake in relation to water quality changes, Water Research, Vol. 35, pp. 3215–3225.
- Kleeberg, A. and Grüneberg, B. (2005) Phosphorus mobility in sediments of acid mining lakes, Lusatia, Germany, Ecological Engineering, Vol. 24, pp. 89–100.
- Kleeberg, A., Schapp, A. and Biemelt, D. (2008) Phosphorus and iron erosion from non-vegetated sites in a post-mining landscape, Lusatia, Germany: Impact on aborning mining lakes, CATENA, Vol. 72, pp. 315–324.
- Klemm, W., Greif, A., Broeckaert, J.A.C., Siemens, V., Junge, F.W., van der Veen, A., Schultze, M. and Duffek, A. (2005) A study on arsenic and the heavy metals in the Mulde River system, Acta hydrochimica et hydrobiologica, Vol. 33, pp. 475–491.
- Koch, H., Grünewald, U., Kaltofen, M. and Kaden, S. (2009) Anpassungsstrategien für die Wasserbewirtschaftung auf den globalen Wandel im Einzugsgebiet der Spree, Korrespondenz Wasserwirtschaft, Vol. 2, pp. 600–605.
- Koch, H., Kaltofen, M., Grünewald, U., Messner, F., Karkuschke, M., Zwirner, O. and Schramm, M. (2005) Scenarios of water resources management in the Lower Lusatian mining district, Germany, Ecological Engineering, Vol. 24, pp. 49–57.
- Koschorreck, M. and Tittel, J. (2007) Natural alkalinity generation in neutral lakes affected by acid mine drainage, Journal Environmental Quality, Vol. 36, pp. 1163–1171.
- Krüger, B., Kadler, A. and Fischer, M. (2002) The creation of post-mining landscapes of lignite mining in the New Federal States, Surface Mining – Braunkohle and other minerals, Vol. 54, pp. 161–169.
- Lessmann, D., Uhlmann, W., Grünewald, U. and Nixdorf, B. (2003) Sustainability of the flooding of lignite mining lakes as a remediation technique against acidification in the Lusatian mining district, Germany, in Proceedings 6th ICARD, T. Farell and G. Taylor (eds), 14–17 July, Cairns, Australia, The Australian Institute of Mining and Metallurgy, Carlton Victoria, Publication Series No 3/2003, pp. 521–527.
- Luckner, L. (2006) Sanierung einer Bergbaufolgelandschaft: Teil 1: Die Lösung des Wassermengen-Problems, Wasserwirtschaft-Wassertechnik, Vol. 3/2006, pp. 33–37.
- Lund, M.A. and McCullough, C.D. (2008) Limnology and Ecology of Low Sulphate, Poorly-buffered, Acidic Coal Pit Lakes in Collie, Western Australia, in Proceedings 10th IMWA Congress, N. Rapantova and Z. Hrkal (eds), 2–5 June, Karlovy Vary, Czech Republic, VSB-Technical University of Ostrava, Faculty of Mining and Geology, Ostrava, pp. 591–594.
- McCullough, C.D., Zhao, L.Y.L. and Lund, M.A. (2010) Mine voids management strategy (I): pit lake resources of the Collie Basin, MiWER/Centre for Ecosystem Management Report 2009–10, Edith Cowan University, Perth, Australia, unpublished report to Department of Water, Western Australia.
- Meybeck, M. (2005) Global occurrence of major elements in rivers, in Surface and ground water, weathering, and soil (volume 5 of Treatise on geochemistry), J.I. Drever (ed), Elsevier, Amsterdam, pp. 207-223.
- Nixdorf, B., Hemm, M., Schlundt, A., Kapfer, M. and Krumbeck, H. (2001) Braunkohlentagebauseen in Deutschland: Gegenwärtiger Kenntnisstand über wasserwirtschaftliche Belange von Braunkohlentagebaurestlöchern, UBA-Texte, 35–01, Umweltbun desamt, 519 p.
- Nordstrom, D.K. and Alpers, C.N. (1999) Geochemistry of acid mine waters, Reviews in Economic Geology, Vol. 6A, pp. 133–160.
- 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 and L.E. Eary (eds), Society for Mining, Metallurgy, and Exploration SME, pp. 187–202.
- Prikryl, I. (2010) Kvalita vody jezera Chabarovice na konci napousteni, in Proceedings 14th Magdeburger Gewässerschutzseminar, Povodi Ohre (ed), 4–6 October, Teplice, Czech Republic, Povodi Ohre, Chomutov, pp. 97–100.
- Pusch, D., Oh, D-Y., Wolf, S., Dumke, R., Schröter-Bosin, U., Höhne, M., Röske, I. and Schreier, E. (2005) Detection of enteric viruses and bacterial indicators in German environmental waters, Archives of Virology, Vol. 150, pp. 929–947.
- Salmon, S.U., Oldham, C.E. and Ivey, G.N. (2008) Assessing internal and external controls on lake water quality: Limitations on organic carbon-driven alkalinity generation in acidic pit lakes, Water Resources Research, Vol. 44, W10414, pp. 1–15.
- Schlaeger, F., Schonlau, H. and Köngeter, J. (2003) An integrated water resources management approach for the River Spree and its catchment, Water Science and Technology, Vol. 47, pp. 191–198.
- Schultze, M. and Boehrer, B. (2008) Development of Two Meromictic Pit Lakes a Case Study from the Former Lignite Mine Merseburg-Ost, Germany, in Proceedings 10th IMWA Congress, N. Rapantova and Z. Hrkal (eds), 2–5 June, Karlovy Vary, Czech Republic, VSB-Technical University of Ostrava, Faculty of Mining and Geology, Ostrava, pp. 611–614.
- Schultze, M., Boehrer, B., Büttner, O., Herzsprung, P. and Geller, W. (2011a) Processes during filling of pit lakes with river water as indicated by water quality changes in Lake Goitsche, Germany, Science of the Total Environment (submitted).
- Schultze, M., Boehrer, B., Duffek, A., Herzsprung, P. and Geller, W. (2005) Introduction of river water as a tool to manage water quality in pit lakes, in Proceedings 9th IMWA Congress, J. Loredo and F. Pendas (eds), 5–7 September, Oviedo, Spain, Departemento de Explotacion y Prospeccion de Minas, University of Oviedo, Oviedo, pp. 273–279.
- Schultze, M., Boehrer, B., Kuehn, B. and Büttner, O. (2002) Neutralisation of acidic mining lakes with river water, Verhandlungen Internationale Vereiningung Limnologie, Vol. 28, pp. 936–939.
- Schultze, M., Pokrandt, K-H., Scholz, E. and Jolas, P. (2011b) Use of mine water for filling and remediation of pit lakes, in Proceedings 11th IMWA Congress, T. Rüde and A. Freund (eds), 4–11 September, Aachen, Germany, Institute of Hydrogeology of the RWTH Aachen University, Aachen, (in press).

Filling and management of pit lakes with diverted river water and with mine water — German experiences

- Seifert, A., Brause, H. and Rascher, J. (1993) Geology of the Niederlausitz Lignite District, Germany, International Journal of Coal Geology, Vol. 23, pp. 263–289.
- Sixta, J. (1998) Gestaltung der Bergbaufolgelandschaft im ehemaligen Tagebau Chabaoovice, BRAUNKOHLE, Surface Mining, Vol. 50, pp. 581–584.
- Statistisches Bundesamt (2010) Statistisches Jahrbuch 2010 für die Bundesrepublik Deutschland, Hanseatische Gesellschaft für Verlagsservice, p. 396.
- Storch, A., Tröger, K., Hoth, N. and Knöller, K. (2007) First Evidence of Active Sulphate Reduction in a Lignite Mine Dump Site at Low pH Values the Plessa Site, Advanced Materials Research, Vol. 20–21, pp. 201–204.
- Svoboda, I., Pecherova, E., Prikryl, I. and Kasparova, I. (2008) The development of future lakes in opencast mine residual pits in the Krušné Mountain region of the Czech Republic, in Proceedings 10th IMWA Congress, N. Rapantova and Z. Hrkal (eds), 2–5 June, Karlovy Vary, Czech Republic, VSB-Technical University of Ostrava, Faculty of Mining and Geology, Ostrava, pp. 619–622.
- Totsche, O., Fyson, A. and Steinberg, C.E.W. (2006) Microbial alkalinity production to prevent reacidification of neutralized mining lakes, Mine Water and the Environment, Vol. 25, pp. 204–213.
- Trettin, R., Glaser, H.R., Schultze, M. and Strauch, G. (2007) Sulfur isotope studies to quantify sulfate components in water of flooded lignite open pits Lake Goitsche, Germany, Applied Geochemistry, Vol. 22, pp. 69–89.
- Werner, F. (1999) Modellgestützte Untersuchungen zur Genese der Wasserbeschaffenheit in Folgelandschaften des Braunkohlebergbaus, Dissertation, University of Leipzig, 124 p.
- Werner, F., Bilek, F. and Luckner, L. (2001) Impact of regional groundwater flow on the water quality of an old post-mining lake, Ecological Engineering, Vol. 17, pp. 133–142.
- Wolf, S. (2005) Evaluierung der hygienischen Wasserqualität unter besonderer Berücksichtigung von Bakteriophagen am Beispiel eines Tagebausees, Dissertation, Technische Universität Dresden, 131 p.
- 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), 26–30 March, St. Louise, USA, American Society of Mining and Reclamation (ASMR), Lexington, pp. 2529–2539.
- Zerling, L., Müller, A., Jendryschik, K., Hanisch, C. and Arnold, A. (2001) Der Bitterfelder Muldestausee als Schadstoffsenke, Verlag der Sächsischen Akademie der Wissenschaften zu Leipzig, 69 p.



#### Martin Schultze

Helmholtz-Centre for Environmental Research – UFZ, Department Lake Research

Martin is a chemist and has researched and made management recommendations for water quality predictions and limnology of pit lakes since 1984. Martin has mainly worked with pit lakes in former lignite mines in Germany but also has experience with other types of mineralisation. Martin is co-editor of the Springer book "Acidic Mining Lakes" (second edition).



#### Professor Walter Geller

#### Helmholtz-Centre for Environmental Research – UFZ, Department Lake Research

Dr. Walter Geller is a professor of limnology, initially working on the ecophysiology of Daphnia and on zooplankton in lakes. From 1992 to 2008, he was head of the inland water research at the Helmholtz-Centre for Environmental Research in Magdeburg. His research comprised the limnology of opencast lignite-mining lakes in East-Germany and of Patagonian lakes. Dr. Geller is senior-editor of the Springer book "Acidic Mining Lakes" (first and second edition).



#### Dr Friedrich-Carl Benthaus

#### Lausitzer und Mitteldeutsche Bergbau-Verwaltungsgesellschaft LMBV, Engineering Department Remediation

Graduated in Mining engineering at RWTH Aachen and TU Berlin, he received his Ph.D. on developing knowledge based systems on mine planning. As head of geotechniques and water management department he works with LMBV on planning and preparation for slope stability as well as on solutions on water management and acid mine water drainage treatment.



#### **Dr Peter Jolas**

Mitteldeutsche Braunkohlengesellschaft MIBRAG

Dr. Peter Jolas is a mining engineer. After doing his PhD in 1988 on water management issues in mining, he was working on projects for water management in mines of the Central German lignite mining district. Since 1990, he is the head of the department for geotechnology of MIBRAG. He is responsible for all issues related to geology, water and soil mechanics in this mining company.