

Michael Owor
Tina Hartwig
Andrew Muwanga
Dieter Zachmann
Walter Pohl

Impact of tailings from the Kilembe copper mining district on Lake George, Uganda

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M. Owor (✉) · A. Muwanga
Department of Geology,
Makerere University,
P.O. Box 7062, Kampala, Uganda
E-mail: mowor@sci.mak.ac.ug
Tel.: +256-772-625149
Fax: +256-41-531061

T. Hartwig · D. Zachmann · W. Pohl
Institute of Environmental Geology,
Technical University, P.O. Box 3329,
38023 Braunschweig, Germany

Abstract The abandoned Kilembe copper mine in western Uganda is a source of contaminants, mobilised from mine tailings into R. Rukoki flowing through a belt of wetlands into Lake George. Water and sediments were investigated on the lakeshore and the lakebed. Metal associations in the sediments reflect the Kilembe sulphide mineralisation. Enrichment of metals was compared between lakebed sediments, both for wet and dry seasons. Total C in a lakebed core shows a general increment, while Cu and Co decrease with depth. The contaminants are predominant (> 65%) in

the $\leq 63 \mu\text{m}$ sediment size range with elevated Cu and Zn (> 28%), while Ni, Pb and Co are low (< 18%) in all the fractions. Sequential extraction of Fe for lakeshore sediment samples reveals low Fe mobility. Relatively higher mobility and biological availability is seen for Co, Cu and S. Heavy metal contents in lake waters are not an immediate risk to the aquatic environment.

Keywords Copper mining · Heavy metals · Aqueous geochemistry · Lake George · Uganda

Introduction

Environmental problems arising from mining activities are receiving increasing attention particularly in the industrialised nations but less so in the developing countries which, however, continue to be the world's major producers of raw materials. In Kilembe, the mining of copper left a legacy of metalliferous material (tailings, rockfill, rockwaste) dumped within a mountain river valley. Up to 15 million tonnes of waste was generated during the processing of the copper-cobaltiferous pyrite ores. The exposure of the sulphidic components in the wastes to an oxic environment (especially under tropical weathering conditions) leads to complex oxidation processes resulting in a marked increase of acidity (AMD—acid mine drainage, ARD—acid rock drainage) and mobilisation of sulphates and metallic elements towards the surrounding

river basins and soils (Singer and Stumm 1970; Sengupta 1993; Neumann-Mahlkau 1993; de Anta et al. 1994). Heavy metals released in the process, which in normal concentrations are essential components of biochemical functions (e.g. Cu, Zn, Co, Cr, Ni), are toxic when present in elevated concentrations. Their particular threat as pollutants lies in the fact that they are not decomposed by natural processes but tend to be dispersed in mineral and organic substances (Förstner and Whitmann 1979).

Mobilisation of heavy metals and of sulphate from mines, tailings and waste rock from mines, as a consequence of AMD or ARD has been thoroughly investigated (Förstner and Wittmann 1979; Förstner 1981; Sengupta 1993; Nordstrom and Southam 1997; Plumlee and Logsdon 1999). Other than the complex processes of mobilisation, the fate of heavy metals also depends on local climatic-hydrologic conditions (Wohrlab et al.

1992; Dyck and Peschke 1995), and on the geological and physical environment (Hemond and Fechner 1994). Geochemical and sedimentological factors control transport in surface waters, including retardation in exchange with soil and sediment (Flemming and Trevors 1988; Alloway 1990; Fergusson 1990; Müller et al. 1994; Miall 1996; Nickson et al. 1998; Guderian and Gunkel 1999; Iskandar and Selim 1999). Transient or permanent immobilisation may occur in contact with Mn-Fe-hydroxides (Jenne 1968) or humic substances (Rashid and Leonard 1973; Hessen and Tranvik 1998). Chemical speciation in soil and sediment that determines mobility is investigated by sequential leaching (Tessier et al. 1979; Zachmann and Block 1994). The copper concentrations in many aquatic environments are kept naturally under control by prevailing redox conditions, but seems to be almost entirely contained in the sediments. Abiotic factors (hardness and pH), iron hydroxides and organic complexing humic substances were also found to lower the toxicity of a given value of copper concentrations (Bugenyi 1982). Pollutant concentrations in particulate matter (sediment) often provide a more stable and convenient means of obtaining an indication of the state of the associated waters (Förstner and Wittmann 1979).

In this study, the fate contaminants from the Kilembe copper mining area transported through R. Rukoki into Lake George, mainly based on geochemical data from lake water and lakebed sediments was evaluated. The main objective was to determine the contaminant distribution (grain size fraction, spatial, depth, seasonal) and speciation (mobility and bioavailability) in the lakebed sediments, and finally estimate the present environmental danger, impact and remedial measures that should be taken.

Study area and geological setting

The study area is in Kasese District, western Uganda (Fig. 1) largely covered by part of the Queen Elizabeth National Park (ca. 2,000 km²) with a rich ecosystem that sustains the tourism industry. Precipitation occurs mainly from March to May, and September to October with an annual mean of 1,000 mm. Lake George encompasses an area of about 270 km² and has its main catchment area in the Ruwenzori range. The largest inflow is R. Rukoki with a monthly flow rate of ~7 m³/s (gauged from 1954 to 1983, Hartwig et al. 2005). Discharge from Lake George is only through the Kazinga Channel that flows southwestwards into Lake Edward. Because of high evaporation and very little outflow, water exchange in Lake George is very limited which renders the lake basic (pH 9–10). It is shallow and extremely eutrophic with high biological productivity.

Lake George lies in the western branch of the East African Rift System formed from late Oligocene to late

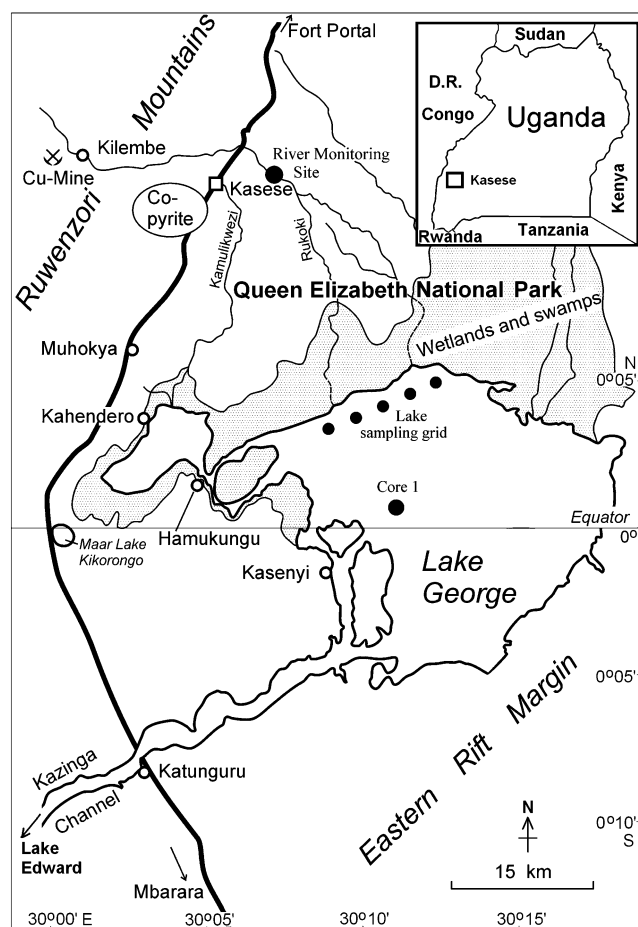


Fig. 1 The study area showing the northern basin of Lake George in Kasese, Uganda

Miocene (Ebinger 1989). In the centre of the rift rises the horst of the Ruwenzori Mountains to Margherita peak (5,109 m), built from Precambrian rocks. The Precambrian Basement is composed of gneisses, quartzites, schists and varying amounts of mafic igneous rocks. Lacking significant volcanic fill, the rift floor west of the Ruwenzori Mountains is characterised by narrow, deep and stratified lakes that have accumulated organic rich sediments (> 4,000 m). The Kasese region lies on the eastern margin of the Ruwenzori Mountains. North of Lake George a plain is developed on Holocene fluvio-lacustrine sediments (silt, clay, sand, locally with travertine-limestone).

Kilembe copper mineralisation occurs within an amphibolite unit of the Kilembe Series rocks that are part of the Ruwenzori fold belt (2.5–1.85 Ga, Cahen et al. 1984). Primary sulphides at Kilembe are essentially pyrite, chalcopyrite and pyrrhotite in the approximate ratio of 12:7:1. There are rare amounts of linnaeite, sphalerite, siegenite, pentlandite and molybdenite (Bird 1968; Davis 1969). Exsolution pentlandite

extends from cracks in the pyrite mineral and contains ca. 2% Co, 2% Ni with traces of Cr, V, Ag, Zn and Zr (Barnes et al. 1959). The oxidised zone contains minerals like cuprite, malachite, chrysocolla and azurite, which were mined by open cast methods in the northern deposit. Ore grade mineralisation (averaging 1.95% Cu with Cu:Co ratio of about 3:1) was worked (1956–1978) from a cluster of five separate deposits (Warden 1985) resulting in 270,000 tonnes of blister copper and 1,113,000 tonnes of cobaltiferous pyrite by-product. An estimated 7 million tonnes of tailings from the sulphide flotation mining process were disposed of in the Kilembe valley on alluvium and lower slopes bordering R. Rukoki. The pyrite concentrates (ca. 1.35% Co) were stockpiled at Kasese, since by then there were no economically viable methods to extract the cobalt (BRGM 1994). These wastes have been exposed to tropical weathering. Residual sulphides were oxidised, resulting in the mobilisation and migration of heavy metals via the old seasonal Kamulikwezi that flows through a belt of wetlands into Lake George.

Materials and methods

Physico-chemical water parameters at water and sediment sampling sites were determined in situ (procedures certified by the International Organisation of Standardisation). Turbidity was measured using a WTW Turb 350 IR turbidimeter (ISO 7027: 1999); dissolved oxygen, the WTW Oxi 315i meter (ISO 5814: 1990); pH, the WTW pH 315i meter (ISO 10390: 2005); and electrical conductivity (EC), the WTW Cond 315i meter (ISO 284: 2003).

Two repeat sampling campaigns along a sampling grid were carried out for both dry and wet season hydrological regimes (2001–2003). Water (50 samples) from the lakeshore was collected into disposable 500 ml polyethylene bags with a 1-m long PVC bailer, from about 1 m below the water surface. Duplicates were taken after every five collected samples. Field blanks were also taken to assess the quality of the field-sampling program. Samples were filtered through 0.2 µm pore size with the sub-sample for cations acidified using a couple of drops of $\approx 65\%$ HNO_3 , and cooled (about 4°C) while on transit to the laboratories for analyses.

Relatively undisturbed sediment sampling (68 samples) was done with a Kajak gravity corer and bucket auger (5–15 cm depths). Sediments were sampled from below the water column at the exact water sample sites of the lakeshores into polyethylene bags. Samples were collected into 500 ml polyethylene bags and cooled (about 4°C) while on transit to the laboratories for analyses. Three sediment cores were obtained with one

being analysed from the bottom of Lake George in August 2003 from a small, single-engine speedboat. Upper, unconsolidated sediments including the sediment-water interface were collected using a 9-cm diameter HON-Kajak gravity corer (Renberg 1991). One Kajak core 1 (47 cm deep) was preserved, and sampled at 1-cm intervals (45 samples) into polyethylene bags on the boat, then cooled on transit to the laboratories for chemical analyses.

Laboratory analyses were done at the Institute of Environmental Geology, Technical University, Braunschweig, Germany. Water was analysed for major (Ca, K, Mg, Na) and trace (Cd, Co, Cu, Fe, Mn, Cr, Ni, Pb, Zn) elements with both the Fisons Maxim III ICP-OES, and ARL 3520 ICP-MS (Cd, Pb). During the determinations, blank and standard solutions (with certified reference materials) were measured at regular intervals (after every eight samples).

Sediments were sieved with nylon sieves (< 0.63 , < 0.20 , < 0.125 and < 0.063 mm ranges), and oven dried at 60°C until dry weight for moisture determination. Total carbon contents were measured with a LECO C&S 144 instrument. Reference standards were initially analysed and subsequently every ten measurements. The samples were added to 2 g of tungsten and analysed. Sediments were dry-sieved (< 0.063 mm), digested (7 ml HF, $\approx 40\%$; 2 ml HClO_4 , $\approx 100\%$; 7 ml HNO_3 , $\approx 65\%$), filtered (0.2 µm cellulose acetate) and stabilised with conc. HNO_3 (100 µl in 10 ml sample). Major (Al, Ca, Fe, K, Mg, Mn, Na, S) and trace (Cd, Co, Cr, Cu, Ni, Pb, Zn) elements were then analysed using both the ICP-OES and ICP-MS.

Sequential extractions involve the selective extraction of trace metals from operationally defined sediment solid fractions (Tessier et al. 1979; Jacob et al. 1990). Sediment samples were subjected to a series of increasingly aggressive, phase-specific reagents under certain conditions. The six-step sequential extraction procedure (Jacob et al. 1990) of sediments (Fe, S, Co, Cu) is summarised in Table 1. One particular important limitation of sequential extraction techniques is the potential for reagents to be non-phase specific and for metal ions solubilised by a reagent to readsorb onto a different sediment phase (Nirel and Morel 1990). Nevertheless, this method does provide useful information on trace metal mobility and reactivity in a particular environmental context.

Results

Water chemistry

Water pH ranges from about 5.0 to 10.2 with the inflow river being slightly acidic to neutral while the lake is

Table 1 Summary of the sequential extraction procedure (Jacob et al. 1990)

Step	Phase	Elution reagents	pH	Reaction time (h)	Solid/solution ratio
I	Exchangeable	Ammonium acetate (1 M)	7	2	1:20
II	Carbonate	Sodium acetate (1 M)	5	5	1:20
III	Easily reducible (Fe- & Mn- hydroxides)	Hydroxylammonium chloride (0.1 M)	2	12	1:100
IV	Moderately reducible (Fe- & Mn oxides)	Ammonium acetate/oxalic acid (0.2 M)	3	24	1:100
V	Sulphidic and organic	Hydrogen peroxide (30%) + ammonium acetate (1 M)	7	12	1:100
VI	Residual (stable silicates & oxides)	Aqua regia (HCl 37% & HNO ₃ 65%)		2	1:100

Table 2 Average and variation in physico-chemical characteristics of the Lake George shore water samples

	pH	EC turbidity		DO	Ca	K	Mg	Na	Mn	Fe	Pb	Cd	Co	Cr	Cu	Ni	Zn
		$\mu\text{S}/\text{cm}$	NTU	(mg/l)	(mg/l)	(mg/l)	(mg/l)	(mg/l)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)	($\mu\text{g}/\text{l}$)
Mean	8.8	236.8	40.9	7.0	32.1	3.2	10.2	18.6	9.5	115.8	2.8	0.05	0.5	0.6	100.3	144.7	39.6
SD	1.2	17.6	35.4	3.8	21.4	1.1	1.7	1.5	15.3	121.9	6.4	0.1	0.6	0.8	123.7	218.4	49.2
n	117	117	117	110	32	45	39	24	43	48	24	24	24	24	50	50	42

SD Standard deviation

more alkaline (Table 2). Dissolved oxygen is generally low in the lake and higher in the river (0.1–21 mg/l). EC ranges from 30 to 280 $\mu\text{S}/\text{cm}$ with the inflow river waters mostly below 200 $\mu\text{S}/\text{cm}$, while the lakeshore samples are above that due to their relatively higher total dissolved ions. High turbidity levels are observed in the lakeshore waters owing to the elevated biological activity and suspended organic matter (up to 300 NTU). Across the mouth of R. Rukoki, peaks of metal load (in excess 150 $\mu\text{g}/\text{l}$ of Fe, Cu, Zn) are evident both at the

eastern and western mouths (Fig. 2). Copper and Fe have the highest values at these two sites.

Sediment chemistry

Aluminium and Fe reach 80,000 ppm with Fe getting in excess of this (Table 3). Sulphur, K, Mg and Ca are mostly below 40,000 ppm while Mn reach 2,000 ppm, respectively. The Cu and Co range up to 1,200 ppm with

Fig. 2 Plot of distance (west to east) of Cu, Fe and Zn in lakeshore water samples across both mouths of R. Rukoki (arrows indicate R. Rukoki inflow)

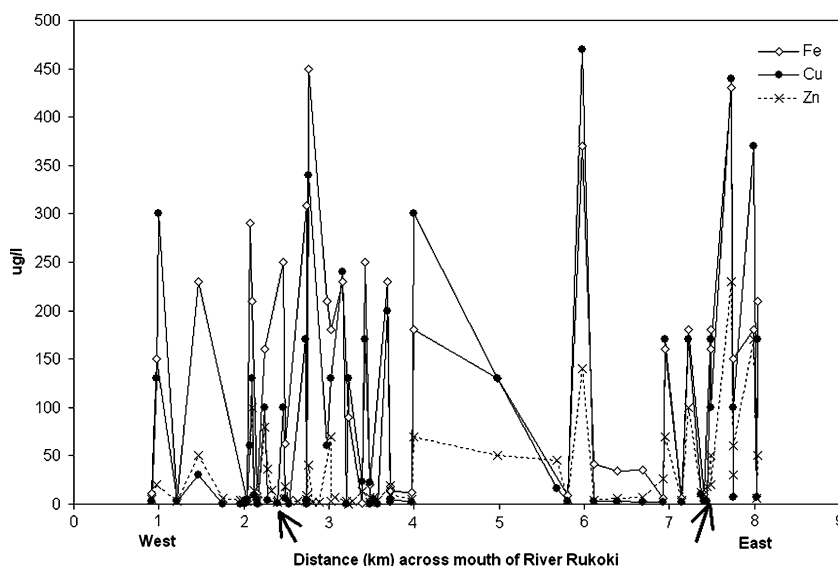


Table 3 Average and variation in the chemical composition of Lake George sediment and core samples

	Total C (%)	Mn (ppm)	Fe (ppm)	Mg (ppm)	Al (ppm)	Ca (ppm)	Na (ppm)	K (ppm)	S (ppm)	Co (ppm)	Cr (ppm)	Cu (ppm)	Ni (ppm)	Pb (ppm)	Zn (ppm)	Cd (ppm)
Mean	12.0	521	38,728	7,228	27,103	9,029	2,945	6,414	22,412	118	52.7	253	60.9	20.4	60.0	0.4
SD	9.0	228	14,422	3,961	18,300	5,357	5,812	6,233	17,244	174	26.4	268	53.6	21.4	19.2	0.2
N	134	134	134	134	134	134	134	134	134	134	134	134	118	134	134	113

SD Standard deviation

most of the Co being below 200 ppm. Lead, Cr, Ni and Zn have concentrations below 200 ppm, with Ni having a few extremes in excess, whereas Pb reveals the least levels. At the river inflow sites, sediments show elevated metal concentrations at both the western and eastern mouths (Figs. 3, 4). Copper and Co (> 200 ppm) as well as Fe and S (> 2,000 ppm) have the highest levels with the western side showing higher contents. There is clearly passage of metals through the wetlands barrier, being more pronounced in the western zone.

Within the different grain size ranges heavy metals were analysed for selected sample points. At the lakeshores sediments are dominated by the ≤ 63 and

200 μm ranges (15–50%) (Table 4). More than half of the metals are found in the $\leq 63 \mu\text{m}$ grain size fraction, with Cu and Zn most prevalent (37–80 ppm) as well as substantial amounts of Cr (up to 57 ppm). In this fraction Ni, Pb and Co have 14–45 ppm. Coarser fractions all have low metal contents (< 5 ppm), only Zn and geogenic Cr reaching 9–24 ppm.

Lakebed core 1 reached a depth of 47 cm beneath a water column of 2.8 m (Fig. 1). The sediment is composed of very dark greyish brown, fine sandy to silty clayey sapropel getting more uniformly sorted and firm as the water content decreases with depth.

Fig. 3 Plot of distance (west to east) of Fe and S in lakeshore sediment samples across both mouths of R. Rukoki (arrows indicate R. Rukoki inflow)

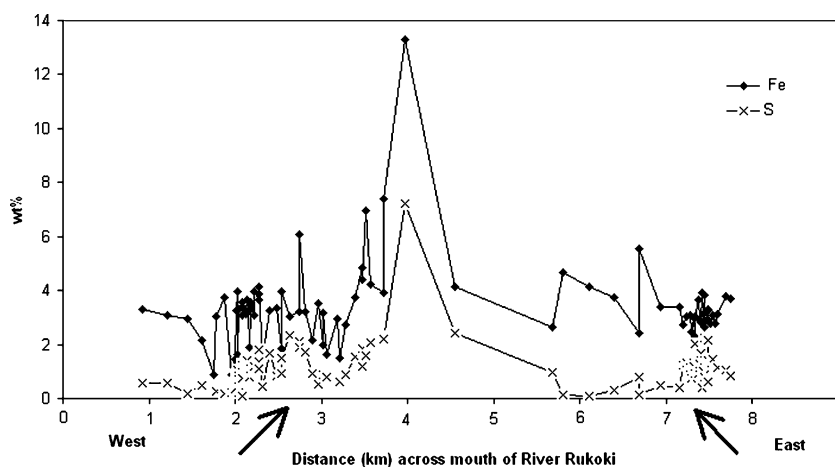


Fig. 4 Plot of distance (west to east) of heavy metals in lakeshore sediment samples across both mouths of R. Rukoki (arrows indicate R. Rukoki inflow)

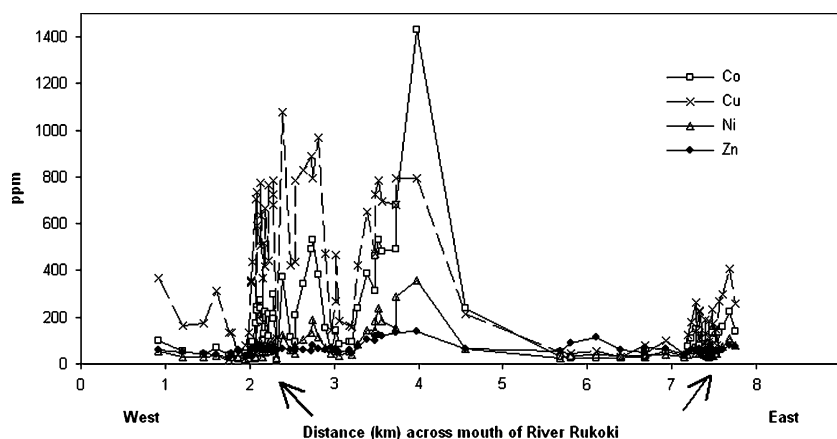


Table 4 Average heavy metal grain size class distribution in the Lake George shore sediment samples

Grain size class	wt (%)	Co (ppm)	Cr (ppm)	Cu (ppm)	Ni (ppm)	Pb (ppm)	Zn (ppm)
< 63	25	31	38	56	32	11	58
63	20	6	16	8	6	2	10
125	17	5	10	8	6	3	8
200	35	3	4	2	3	3	5
630	4	4	6	2	4	4	6

Scatter plots of core 1 intended to capture trends in metal behaviour did not yield significant relationships. Exceptions are the positive correlation between Cu and Co, and the negative correlation between total C with Cu and Co (Fig. 5). Obviously, there is a low influence of organic matter on the sorption of these metals. The strong correlation between Cu and Co is attributed to a comparable behaviour of both metals in this environment.

Depth plots of selected elements are presented for core 1 (Fig. 6). Total carbon contents in the sediments range from 22 to 32% showing a slight increment with depth. Iron ranges from 3.6 to 5.4% without showing significant change with depth. Sulphur contents are between 2.3 and 4.9% and the data seem to indicate a small increase with depth. Trace elements decrease downwards, with Cu from 25 to 107 ppm and Co 11 to 28 ppm. Both Co and Cu show a pronounced decrease to about 35 cm depth indicating a higher recent rate of deposition.

Several plots in Fig. 6 show depth-dependant slopes that may represent different sedimentation rates prior to and after mining started in 1950. The response of the Cu–Co input does not seem to agree with the inference that after 1950 the sedimentation rates in Lake George

doubled to about 0.5 cm/year (Russell and Schnurrenberger, Michigan University Lakes Research Group, USA, written communication 2004).

The enrichment factor (EF) was calculated as the ratio of elemental concentrations normalised to an immobile element, which is here aluminium, that is:

$$EF = \frac{(E/Al)_{\text{site1}}}{(E/Al)_{\text{site2}}}, \quad (1)$$

where, E is the element analysed and the reference material is Aluminium.

A comparison was made between contamination in lakeshore sediments against core 1. It is evident that the lakeshore samples are highly enriched in the mining-derived metals Cu and Co with a much smaller difference concerning the elements Pb, Zn and Cr.

Seasonal heavy metal inputs into Lake George from the wetlands and R. Rukoki were also compared from results included in Table 3. March 2002 was wet with a total monthly rainfall of over 154 mm. While March and August 2003 were dry months with monthly rainfall totals of about 44 and 37 mm, respectively. Results show that there is a significant difference in the concentrations of Fe, Pb and Cr during the wet compared to dry periods. Sulphur and metals have no discernable concentration differences between the wet and dry periods, although lower metal contents would be expected during the wet season (because of dilution).

The sequential extraction results of sediment samples are presented for Fe, S and the two main heavy metals (Cu, Co). Highest total S amounts are found on the western side of the R. Rukoki mouth (10,000–12,000 ppm) (Fig. 7). Most of the S is contained within the organic/sulphidic phase (70–95%). Some S is contained in the exchangeable (2–23%) and the residual (1–7%) phases. Other phases have negligible amounts

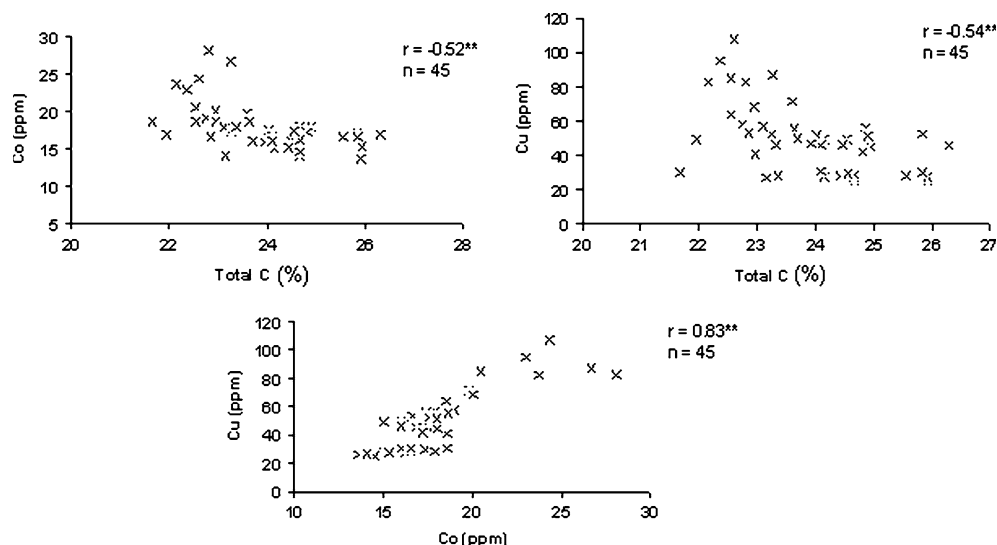
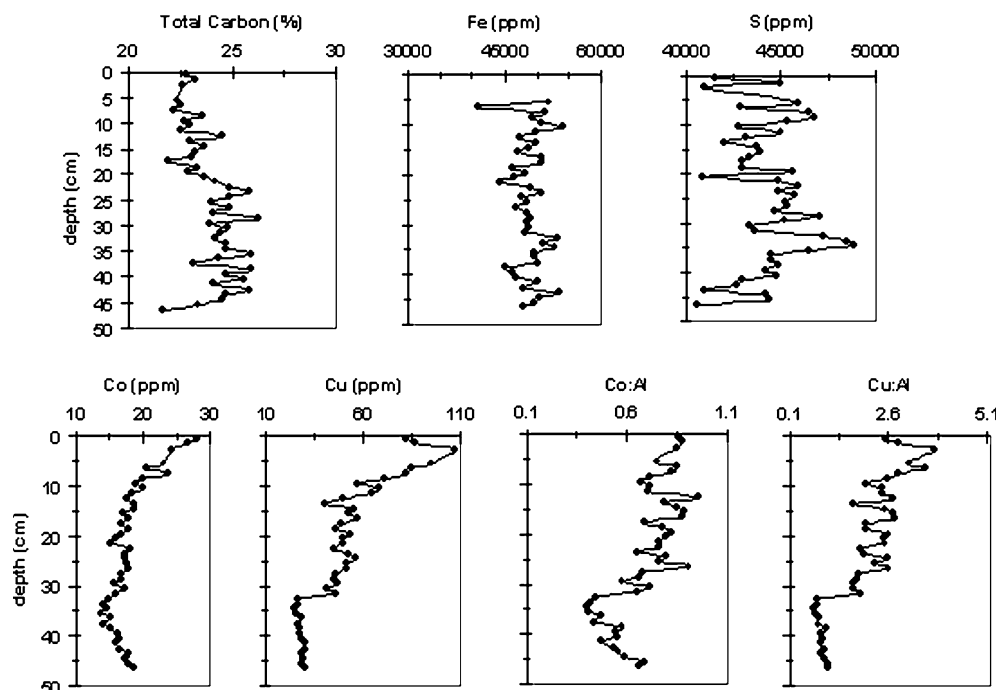
Fig. 5 Scatter plots of Lake George bottom core 1

Fig. 6 Depth plots of selected elements in the Lake George bed core 1



< 1% S and are randomly distributed in different samples.

Total Fe in sediment amounts to 20,000–31,000 ppm at the lakeshores. Most of the Fe is contained within the residual phase (70–93%) with highest contents near the

western river mouth (Fig. 7). Most of the remaining Fe occurs in the moderately reducible (6–27%) and the easily reducible (1–7%) phases. The moderately reducible phase is elevated on the western side of the R. Rukoki mouth. The easily reducible phase is highest in the west and minimal amidst the R. Rukoki mouth. The other three phases have contents below the instrumental detection limits.

Highest total Co amounts are found on the western side (365 ppm) of the R. Rukoki mouth decreasing towards the east (100–170 ppm) (Fig. 8). Co occurs mainly within the residual (3–83%) and organic/sulphidic (5–75%) phases. The residual phase is highest on the eastern R. Rukoki mouth and decreases on the western side. The organic/sulphidic phase is elevated on the western side. Most of the remaining Co occurs in the easily reducible (1–36%), moderately reducible (9–33%) and exchangeable phases (0.4–32%). Contents are uniformly distributed in both reducible phases at the R. Rukoki mouth. The exchangeable and carbonate (1–10%) phases are highest on the western side decreasing towards the east.

Highest total Cu amounts are found towards the western side (650–700 ppm) with the lowest amidst the R. Rukoki mouth (180 ppm) (Fig. 8). The Eastern side is also relatively high at 320 ppm. More of the Cu is contained within the residual (2–77%) and moderately reducible (19–62%) phases. Significant amounts occur in the organic/sulphidic (3–47%) phase. The residual phase is highest amidst the R. Rukoki mouth, and lowest on the western side. Moderately reducible and organic/sulphidic phases are highest on the western

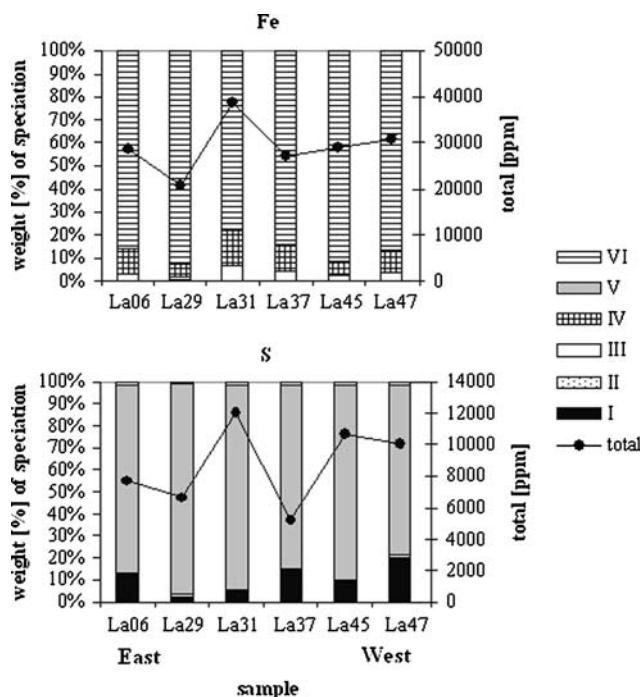


Fig. 7 Speciation of major elements (Fe and S) in the lakeshore sediment samples

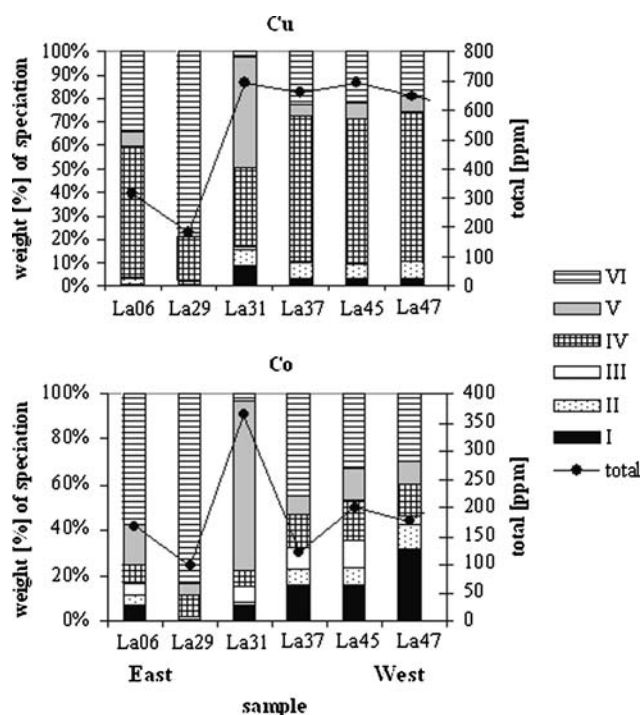


Fig. 8 Speciation of trace elements (Cu and Co) in the lakeshore sediment samples

and eastern sides of the R. Rukoki and least amidst the R. Rukoki mouth. The rest of the Cu is contained in the easily reducible (0.1–7%), carbonate (1–9%) and exchangeable phases (1–11%). Contents are highest in the exchangeable, carbonate and easily reducible phases on the western side of the R. Rukoki mouth and lowest towards the eastern side.

The mobility of metals in sediment samples may be evaluated on the basis of absolute and relative content of fractions weakly bound to sediment components. The relative index of metal mobility was calculated as a 'mobility factor' (MF) (Salbu et al. 1988; Narwal and Singh 1998) on the basis of the following equation:

$$MF = \frac{(F1 + F2 + F3)}{(F1 + F2 + F3 + F4 + F5 + F6)}, \quad (2)$$

where, $F(1-6)$ is analogous to the sequential extractions steps (I–VI) measured in %.

The MF index describes the potential mobility (Salbu et al. 1988), since it is a ratio of proportion of mobile ($F1 + F2 + F3$) to the total sum of fractions. The MF reveals values in the range 0.3–47% for all heavy metals in sediment samples analysed. Indices of mobility are lowest for Fe (2–7%). Sulphur has moderate indices of mobility at the lakeshores (10–21%). The R. Rukoki mouth heavy metal mobility factors are generally low (0.3–7%). Cobalt has the highest MF of up to 47% at

the western side of the R. Rukoki mouth. The indices are higher for Co and Cu at the western side (10–47%) compared to the eastern side (1–16%).

Discussion

Elevated Cu, Co, Fe and SO_4 concentrations at the R. Rukoki mouths with higher loadings to the west indicate a more frequent inflow of R. Rukoki water and sediment from within the wetlands barrier through the western zone. During the rainy season the river floods the papyrus belt that does then not function as an effective geochemical barrier. The close correlations of Cu, Co, Cd, Ni and to a lesser degree Pb and Zn, point to the influence of the Kilembe sulphide mineralisation on the contaminant concentrations in the sediments.

Lake George bottom sediments are quite variable, including organic mud, clay, silt and some sand. An inflexion in core 1 (similarly noted by Viner 1977) showed a small organic carbon peak followed by a decrease near the sediment surface (Fig. 6). This is thought to reflect mechanical disturbance of the uppermost sediment column. Disturbance may induce higher decomposition of organic matter near the surface due to oxygenation from the top downwards, with accumulation of readily decomposable material at about 18–20 cm. Haworth (1977) showed that at least the upper 10 cm of the present Lake George sediment is liable to continuous re-suspension due to wave action. Also, it is possible that material has been eroded from recent sediments. The assumption is that there had been some re-suspension throughout the lake's history, the amount being determined by wind-induced turbulence.

The iron distribution with depth is probably related to the degree of mechanical disturbance. A small amount of discontinuous oxygenation could be sufficient to create insoluble Fe(III) compounds. Under reducing conditions Fe and S precipitate FeS-phases, but there is no consistent proportion between the two elements. It is obvious that below that near-surface zone reducing conditions have existed in the lake sediments with a high organic enrichment throughout most of the lake's history.

Sediments not only reflect the current quality of surface waters, but also provide vital information on the transportation and fate of pollutants (Santschi 1984; Finney and Huh 1989). Sequential extraction analyses were used to provide insight into the geochemical mode of sediment elemental retention.

Trace metals retained in the sediments by electrostatic attraction to negatively charged surfaces must compete with major cations for outer-sphere sorption sites. Given the relative abundance of Ca^{2+} , Mg^{2+} , Na^+ and K^+ in the lake waters, these take up the available sites more easily. This reflects the importance of low ionic strength of the aqueous matrix, which in freshwater systems

allows aqueous trace metals to occupy cation exchange sites.

The acetic acid extraction fraction is assumed to be bioavailable to sediment ingesting biota (Turner and Olsen 2000). Oxyhydroxide (oxide) minerals, along with organic matter, have long been recognised as the predominant metal sorbents in aquatic systems. Metals associated with oxide minerals are likely to be released if these minerals were to dissolve.

The organic fraction reflects metals strongly bound to sediment organic matter. The trends in associations can be attributed to the abundance of the organic matter at the lakeshores. Organically bound Cu (and other metals) is more phytoavailable than other fractions retained in primary minerals (residual) and inorganic precipitates (e.g. sulphides and carbonates) (Stevenson 1991).

The residual fraction represents metals occluded within the crystal structure of recalcitrant minerals. This fraction is not available to biological or diagenetic processes except over very long time scales (Tessier et al. 1979). It has been suggested that this fraction is important for non-contaminated sediments. In uncontaminated settings, the residual fraction is usually the most important geochemical phase for trace metal retention.

These findings suggest that the average order of potential element mobility in the Lake George sediments is: $\text{Co} > \text{S} > \text{Cu} > \text{Fe}$. Stumm and Morgan (1996) have shown that chemisorption of trace metals to mineral and organic surfaces is related to the hydrolysis of metal cations in solution. Overall, the trend in potential mobility of the metals based on geochemical fractionation is in accordance with that expected on the basis of hydrolysis trends reported previously (Stumm and Morgan 1996). It is generally assumed that the mobility and potential bioavailability of heavy metals for plant uptake decreases approximately in the order of a sequential extraction sequence (Lottermoser et al. 1999). Elements held in absorptive sites of stream sediments are most easily available to fluvial organisms. Low mobility indices in the river and lake sediments point to high stability of Fe. Relatively higher mobility values are considered to be indications of moderately high lability and biological availability of Co, Cu and S in these sediments.

Conclusions

Contaminant levels in waters at the lakeshores are not an immediate danger to the general environment except for Cu that at the R. Rukoki mouth may cause stress to aquatic life (above 15 $\mu\text{g/l}$).

Lakeshore sediments have high Cu and Co contents reaching 1,078 and 1,429 ppm, respectively. Elevated Cu, Co, Fe and SO_4 occur near the R. Rukoki mouths with higher loadings to the west. These contaminants are clearly derived from the abandoned Kilembe mine.

The lakebed core 1 has total C (22–32%) that shows a general increment with depth. Main heavy metals, Cu (25–107 ppm) and Co (11–28 ppm) show a decrease with depth.

Associations in the lakebed sediments are realised among the sulphide mineralisation metals (Cu, Co, Cd, Ni, Pb and Zn). Higher enrichments of Cu and Co as well as lesser amounts of Pb, Zn and Cr occur near the northern lakeshore compared to the central lakebed. Significant decreases of mainly Fe, Pb and Cr arise during the dry compared to wet periods at the lakeshores.

The contaminants (mostly Cu and Zn) are predominantly distributed (> 28%) in the $\leq 63 \mu\text{m}$ sediment grain size range. Zinc and Cu are also significant in the coarser grain size fractions (63–630 grain size class), with Ni, Pb and Co being low (< 18%) in all the fractions.

Sequential extraction of contaminants in lakeshore sediments reveals low mobility indices (2–7%) that point to high stability of Fe. Relatively higher mobility associated with moderately high lability and biological availability is indicated for Co, Cu and S (10–47%). These contaminants may pose long-term risks for bioaccumulation in the Lake George aquatic system.

There are significant contaminant loads efficiently immobilised in the lake sediments. River Rukoki floods the wetland on the northern lakeshores during the rainy seasons, distributing contaminants several 100 m into the lake (Hartwig et al. 2005). Although the high alkalinity of the lake impedes contaminant transfer to the water column, concomitant seasonal river flooding may possibly resuspend buried reduced sediments.

It is therefore essential to continue and expound on the monitoring network that gives guidelines to trigger preventive action and protect the lake environment. Over the long term, remediation of the old Kilembe copper mine will be crucial to curb further contaminant input that continually augments metals trapped in reduced sediments near the lake's northern shores.

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References

- Alloway BJ (ed) (1990) Heavy metals in soils. Halsted Press, Blackie, p 339
- Barnes JW, Barbour EA, Smit JS (1959) Symposium on the geology of copper—Africa, Kilembe copper mine—Uganda. Unpubl report JWB/77 Geol Surv Uganda
- Bird HH (1968) Falconbridge's copper operations in Uganda. *Can Min Metall Bull* 61:1075–1082
- BRGM (1994) Environmental impact study (Kasese, Uganda). Bureau de Recherches Géologique et Minières (BRGM), France report R 37058
- Bugenyi FWB (1982) Copper pollution studies in Lakes George and Edward, Uganda: the distribution of Cu, Cd, and Fe in the water and sediments. *Environ Pollut B3*:129–138
- Cahen L, Snelling NJ, Delhal J, Vail JR (1984) The geochronology and evolution of Africa. Clarendon Press, Oxford, p 195
- Davis GR (1969) Aspects of the metamorphosed sulphide ores at Kilembe, Uganda. In: James CH (ed) *Sedimentary ores ancient and modern* (revised): proceedings of the 15th inter-University geol Congr, Leicester 1967, pp 273–295
- de Anta RC, Otero AP (1994) Soils affected by acid mine waters in Galicia (NW Spain). *Water Air Soil Pollut* 73:241–263
- Dyck S, Peschke G (1995) *Grundlagen der Hydrologie* 3 Aufl. Verlag für Bauwesen, Berlin, p 536
- Ebinger CJ (1989) Tectonic development of the western branch of the East African Rift System. In: Morley CK, Ngenoh DK, Ego JK (1999) *Introduction to the East African Rift System*. In: Morley CK (ed) *Geoscience of Rift Systems—evolution of East Africa: AAPG Studies in Geology*, vol 44, pp 1–18
- Fergusson JE (1990) The heavy elements: chemistry, environmental impact and health effects. Pergamon, Oxford, p 614
- Finney BP, Huh CA (1989) High resolution sedimentary records of heavy metals from the Santa Monica and San Pedro Basins, California. *Mar Pollut Bull* 20:181–187
- Flemming CA, Trevors JT (1988) Copper toxicity and chemistry in the environment: a review. *Water Air Soil Pollut* 44:143–158
- Förstner U (1981) Trace metals in fresh waters (with particular reference to mine effluents). In: Wolf KH (ed) *Handbook of strata-bound and stratiform ore deposits*, vol 9. Elsevier, Amsterdam, pp 271–303
- Förstner U, Wittmann GTW (1979) *Metal pollution in the aquatic environment*. Springer, Berlin Heidelberg New York, p 486
- Guderian R, Gunkel G (1999) *Handbuch der Umweltveränderungen und Ökotoxikologie, Band 3a: Aquatische Systeme: Grundlagen—Physikalische Belastungsfaktoren—Anorganische Stoffeinträge*. Springer, Berlin Heidelberg New York, p 330
- Hartwig T, Owor M, Muwanga A, Zachmann D, Pohl W (2005) Lake George as a sink for contaminants derived from Kilembe copper mining area, western Uganda. *Min Water Environ* 24(3):114–123
- Haworth EY (1977) The sediments of Lake George (Uganda), V: the diatom assemblages in relation to the ecological history. *Arch Hydrobiol* 80(2):200–215
- Hemond HF, Fechner EJ (1994) *Chemical fate and transport in the environment*. Academic, New York
- Hessen DO, Tranvik LJ (1998) *Aquatic humic substances*. Springer, Berlin Heidelberg New York, p 346
- Iskandar IK, Selim MH (1999) Fate and transport of heavy metals in the Vadose zone. Lewis-CRC-Press, Boca Raton, p 345
- Jacob G, Dunemann L, Zachmann D, Brasser T (1990) Untersuchungen zur Bindungsform von Schwermetallen in ausgewählten Abfällen. *Abfallwirt J* 2(7/8):451–457
- Jenne EA (1968) Controls on Mn, Fe, Co, Ni, Cu and Zn concentrations in soils and water; the significant role of hydrous Mn and Fe oxides. *Am Chem Soc Adv Chem Ser* 73:337–387
- Lottermoser BG, Ashley PM, Lawie DC (1999) Environmental geochemistry of the Gulf Creek copper mine area, north-eastern New South Wales, Australia. *J Environ Geol* 39(1):61–74
- Miall AD (1996) *The geology of fluvial deposits*. Springer, Berlin Heidelberg New York, p 586
- Müller HW, Schwaighofer B, Kalman W (1994) Heavy metal contents in river sediments. *Water Air Soil Pollut* 72:191–203
- Narwal RP, Singh BR (1998) Effect of organic materials on partitioning, extractability and plant uptake of metals in an alum shale soil. *Water Air Soil Pollut* 103:405–421
- Nickson R, McArthur J, Burgess W (1998) Arsenic poisoning of Bangladesh groundwater. *Nature* 395:338
- Nirel PMV, Morel FMM (1990) Pitfalls of sequential extractions. *Water Res* 24:1055–1056
- Nordstrom DK, Southam G (1997) Geomicrobiology of sulfide mineral oxidation. In: Banfield JF, Nealson KH (eds) *Geomicrobiological review in mineralogy*, vol 35. Mineralogical Society of America, Washington, pp 361–390
- Plumlee GS, Logsdon MJ (eds) (1999) *The environmental geochemistry of mineral deposits. Part A: processes, techniques and health issues*. *Rev Econ Geol* 6A:371
- Renberg I (1991) The HON-Kajak sediment corer. *J Palaeolimn* 6:167–170
- Neumann-Mahlkau P (1993) Acidification by pyrite weathering on mine waste stockpiles, Ruhr District, Germany. *Eng Geol* 34:125–134
- Rashid MA, Leonard JD (1973) Modifications in the solubility and precipitation behaviour of various metals as a result of their interaction with sedimentary humic acid. *Chem Geol* 11:89–97
- Salbu B, Krekling T, Oughton DH (1988) Characterization of radioactive particles in the environment. *Analyst* 123:843–849
- Santschi PH (1984) Particle flux and trace metal residence times in natural waters. *Limnol Oceanogr* 29(5):1100–1108
- Sengupta M (1993) *Environmental impacts of mining—monitoring, restoration, and control*. Lewis, London, p 494
- Singer PC, Stumm W (1970) Acidic mine drainage: the rate determining step. *Science* 167:1121–1123
- Stevenson FJ (1991) Organic matter–micronutrient reactions in soil. In: Mortvedt JJ (ed) *Micronutrients in agriculture*. SSSA, Madison, pp 145–186
- Stumm M, Morgan JJ (1996) *Aquatic chemistry*. Wiley-Science, New York, p 1022
- Tessier A, Campbell PGC, Bisson M (1979) Sequential extraction procedure for the speciation of particulate trace metals. *Anal Chem* 51:844–851
- Turner A, Olsen YS (2000) Bioavailability of trace metals in contaminated estuarine sediments: a comparison of chemical and enzymatic extractants. *Estuar Coast Shelf Sci* 51:717–728

-
- Viner AB (1977) The sediments of Lake George (Uganda), IV: vertical distribution of chemical features in relation to ecological history and nutrient recycling. *Arch Hydrobiol* 80(1):40–69
- Warden AJ (1985) Reappraisal of geological setting and potential of Kilembe copper mine, Uganda. *Trans Inst Min Metall B94*:94–105
- Wohlrab B, Ernstberger H, Meuser A, Sokollek V (1992) *Landschaftswasserhaushalt*. Verlag Paul Parey, Hamburg, p 352
- Zachmann D, Block R (1994) Studies on the availability of toxic heavy elements in soils and sediments in the vicinity of a lead smelting site (Germany). *Water Air Soil Pollut* 78:317–334