

**INTERNAL METAL DISTRIBUTION IN SEDIMENT –
PORE WATER – WATER SYSTEMS OF BIGHTS
AT NASSER LAKE, EGYPT**

M. E. SOLTAN*

S. M. N. MOALLA

M. N. RASHED

E. M. FAWZY

Aswan, South Valley University, Egypt

ABSTRACT

Understanding and controlling the fate of pollutant metals is important to the long-term health of freshwater. This study tracks the distribution of some heavy and major metals (Fe, Mn, Zn, Pb, Cd, Co, Ni, Cr, Cu, Ca, Mg, Na, and K) between sediment – pore water – water system of some bights of Nasser Lake. Sediment and water (surface and bottom) samples were collected from the beginning and the end of three bights at Nasser Lake, Egypt. In all studies, metals were measured using atomic absorption technique, except Na and K which were analyzed by atomic emission spectrophotometry. Water quality variables were monitored at surface and bottom water. Chemical partitioning patterns of metal show decrease in concentrations of measured metals in sediment (in exchangeable fraction), reflect the decrease of pore water contents, wherefore the water body characterize by low metal concentrations. The alkaline pH values of water bights minimize the reflux of metal ions from lake sediment and increase of metals sedimentation rate. The application of Pearson correlation approaches for the interpretation of large data matrix obtained was performed using SPSS statistical package program. Positive

*Current address: Faculty of Science and Arts, Qassim University, KSA.

correlations arise between different metals in pore waters and lake water suggesting that sediment upper layer served as a metals reservoir from water, but not as an ultimate mechanism to control metal concentrations in the adjacent water. Also these correlations between components give chance to follow the distribution of measured metals between sediment – pore water – water lake system.

Today, metallic pollutants have a great ecological significance due to their toxicity and cumulative behavior (Purves, 1985). They are not biodegradable, contrary to most pollutants, and undergo a global eco-biological cycle in which natural waters are the main pathways (Nürnberg, 1984). Knowledge of the metallic pollutant concentrations in sediments and of their evolution is of interest because while they act as traps, catching pollutants, they are also considered to be reservoirs capable of solubilizing some of their constituents under certain physicochemical conditions (variation in pH, redox potential, early diagenesis, etc.). The total metal concentrations provide important knowledge about the pollution level, if the background or geochemical composition is known.

Calmano, Hong, and Forster (1993) have pointed out that metals in the surface of bottom sediments would be released into the water phase by physicochemical processes; this means any change in environmental conditions in the system may render the remobilization of metals from sediments. Nowadays, heavy metals are frequently detected in the environment, including both bed sediments and water columns, and have gradually become a major concern worldwide due especially to industrial and urban wastes discharged into water bodies (Boughriet, Ouddane, Fischer, Wartel, & Leman, 1992; Pardo, Barrado, Perez, & Vega, 1990; Warran & Zimmerman, 1993).

Bottom sediments in natural systems are formed by sedimentation of particles from the overlying water column. During this process, water will be trapped and entrained in the sediment forming the interstitial or pore water (Van Ryssen, Learmakers, & Baeyens, 1999). Sediment serves as a valuable sink for pollutants eliminating them from the water column and hence reducing potential toxicity to aquatic organisms (Salomons, De Rooij, Kerdijk, & Bril; Van Ryssen et al., 1999).

The so-called “pore water hypothesis” states that exposure to contaminants occurs mainly through the solution phase, or indirectly by phases in equilibrium with the pore water (Vijver, Jager, Posthuma, & Peijnenburg, 2003). The importance of the pore water for uptake of chemicals has been demonstrated for oligochaete species (Lokke & Van Gestel, 1998; Peijnenburg, Baerselman, De Groot, Jager, Posthuma, & Van Veen, 1999a, 1999b) and soil microbes (Mc Grath, 2002; Plette, Nederhof, Temminghoff, & Van Riemsdijk, 1999; Vijver et al., 2003). Within the sediment system, chemical species will tend toward equilibrium between pore water and solid phase according to the prevailing conditions of redox, pH, salinity, DOC, etc. Knowledge of metals concentrations

in pore water and of the metal speciation in the solid part help us, together with the results of all other investigations, to classify the sediment with respect to the mobility of metals and hence the potential harm to ecosystem (Van Ryssen et al., 1999).

Because the metals can be either adsorbed onto sediment or accumulated by benthic organisms to toxic levels, the bioavailability and subsequent toxicity of the metals have become the major research topic associated with sediments (Ankley, Leonard, & Mattson, 1994; Berry, 1996; Berry, Hansen, Mahony, Robson, Di Toro, Shipley, et al., 1996; Dewitt, Swartz, Hanzen, McGovern, & Berry; Pesch, Hansen, Boothman, Berry, & Mahony, 1995). These studies have also indicated that the metal concentrations in sediments were not predictive of bioavailability, and toxicity of metals in sediments was correlated to the concentrations of dissolved heavy metals in the sediment pore water (Yu, Tsai, Chen, & Ho, 2001). The mobility and bioavailability of metals bound to sediments depend on multiple factors, with sediment characteristics and the physicochemical form of the metal being the key factors. Generally, free metal ions are the most mobile and the most bio-available form (Prokop, Vangheluwe, Van Spring, Janssen, & Holoubek, 2003). Oxidation of organic matter increases under aerobic conditions (Fu, Allen, & Cao, 1992).

New insights into the issue of metal bioavailability show that dissolved metal fractions in the pore-water are of major importance for uptake by various biota (Allen, 2001). Complex ions, polymers and micro particulates, as well as sorption on solid surfaces and biological surfaces, reduce the activity of the free ionic form of the metals and hence the potential for exerting toxicity (Roy & Campbell, 1997).

Sediment in rivers not only plays an important role influencing the pollution of river water but also can be used to record the history of river pollution. Thus, sediment acts as both carriers and potential sources of contaminants in an aquatic environment (Yu et al., 2001). Many authors studied the quality of Nasser Lake sediment; Awadallah, Ismail, Arifien, Moalla, and Grass (1994) investigated the main and trace elements in Nile mud and sediments of the Aswan High Dam Lake using an atomic absorption spectrophotometer (AAS). This study showed that the composition of the Nile mud sediments changes in accordance with the large gradients in the flow rate with elution and absorption processes. Moalla, Awadallah, Rashed, and Soltan (1998) studied the distribution and phases of Cd, Cr, Cu, Fe, Mn, Ni, Pb, and Zn in Nile sediments taken from Nasser Lake (Aswan, Egypt) and showed that with the exception of Fe and Cd, all the studied metals have total concentrations which are within or close to the geochemical background levels in stream sediments. Soltan, Rashed, and Taha (2000) studied the ability of sediments to adsorb metal ions even in acidic conditions and reported that the sediments of the river Nile have greater potential for the removal of heavy metals in acidic and basic medium by adsorption and sorption mechanisms. Fawzy (2000) studied the speciation of heavy elements in the river Nile sediments,

and concluded that for all elements studied, the average total concentrations in aquatic sediments, associated waters may exceed the highest desirable levels values for the particular element in water. The study has also shown the importance of different sediment characteristics in spatial distribution of heavy elements in the surface sediments of the River Nile.

The main study objectives aim to:

1. assess pore water metal concentration levels that could be encountered in the lake;
2. investigate the mechanism of metals distribution between pore-water, sediment, and water lake; and
3. estimate the potential mobilization of heavy metals from sediment surfaces, as the key for assessing the extent of available metals in the lake.

METHODOLOGY

Study Site

Lake Nasser is about 500 km long, 350 km of which are in Egypt and 150 km in the Sudan. In southern Egypt and in northern Sudan, major rock formations are the Precambrian basement complex (granites, schists, gneisses, and the Nubian sand stones).

The lake has around 100 side branches (bights) that constitute about 76% of its total area, 58 of them being located on the eastern side and 42 on the western side (Figure 1). The map of Nasser Lake shows the study area and sampling locations. Of three bights, two lie at the western side, El-Matar and El-Caria (3 and 1 Km far from Dam wall, respectively), and one at the eastern side (El-Sage) 4 Km far from Dam wall, El-Matar and El-Caria (3 and 1 Km far from Dam wall, respectively).

Samples Collection and Handling

Lake Water

Water samples were taken from the beginning and end of the three bights under study. Samples were collected from the surface and bottom of the bights during the period from July to September 2003, using Nansen bottle samples (Goodwin & Goodard, 1974). For elemental analysis, water samples were filtered immediately using 0.45 μm filter paper mounted on a Pyrex filter holder with filtrates acidified to pH2 with A.R nitric acid in order to prevent adsorption of metals on the bottle's surface.

Lake Sediment

Sediment samples were collected with a Peterson grab sampler (made of stainless steel) from the beginning and end of each bight at the same sites as the

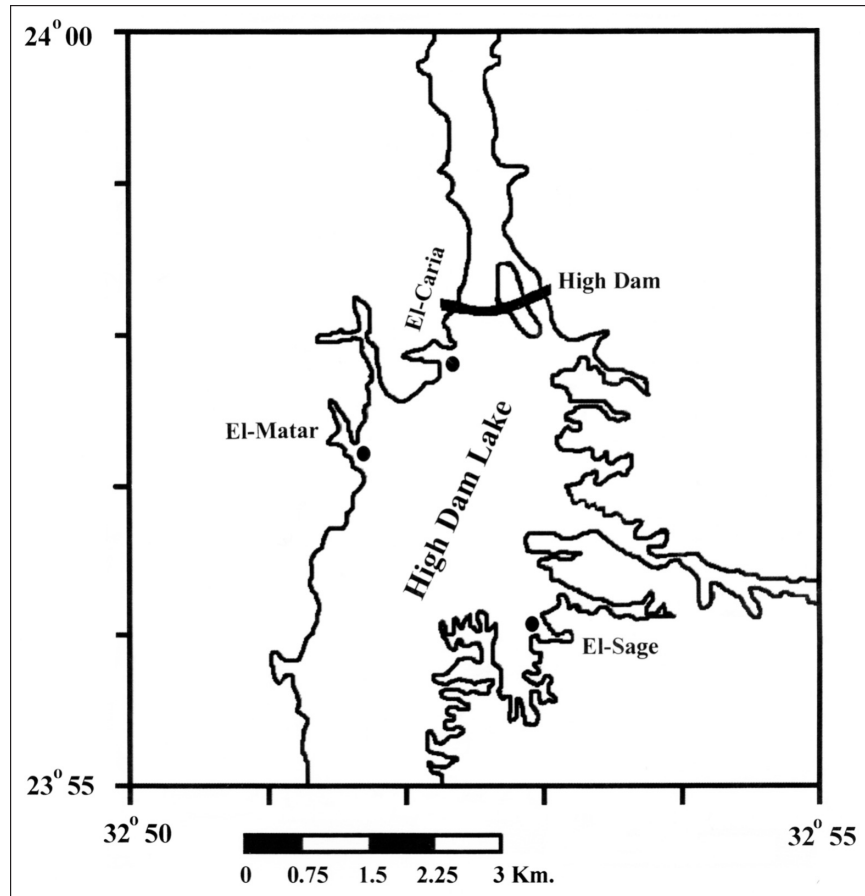


Figure 1. Sampling sites in Nasser Lake (•).

water samples. The samples were dried at 105°C in an electric furnace for 48 hours, powdered in an agate mortar and sieved to under 63 μm (Förstner & Salomons, 1980), then kept in polyethylene bottles.

Measurements and Analysis

The chemical and physical parameters of water samples were measured using standard procedures (APHA, 1992); chemical parameters in sediment samples were determined using a textbook of soil analysis (Baruah & Barthakur, 1997).

Analytical Procedures

Pore Water Samples

Sediment pore water samples were extracted by centrifuging sediments at 8×10^4 rpm for 30 minutes. The remaining pore water was vacuum-filtered through precleaned 0.45 μm membrane Millipore filters for the analysis of metals.

Sediment Samples

The pH values of sediment samples were measured in 1:2.5 sediment: bi-distilled water and electrical conductivity was measured in sediment suspension obtained in the pH determination using conductivity meter. Organic matter in sediment samples was determined according to Allen (1989) and carbonate was determined titrimetrically (indirect method) in presence of phenolphthalein as indicator.

SELECTION OF EXTRACTANTS

In the choice of extracting reagents, particular emphasis was placed on the selectivity and extracting efficiency of each leaching solution. The extraction methods that are the most informative for environmental purposes are the total element, moderately and easily extractable element extraction techniques. Sediment samples (1 g dry wt.) were subjected to single step leaching procedures designed to partition the particulate heavy metals into the following fractions (Salomons & Förstner, 1980; Tessier, Campbell, & Bisson, 1980; Sacki et al., 1993; Moalla et al., 1998):

1M Ammonium Acetate pH 7

One gram of each sediment sample shaken with 10 ml 1M ammonium acetate for 3 hours (exchangeable phases).

0.04M Hydroxylamine HCl

One gram of each sediment sample was extracted for 6 hours with 25 ml of 0.04M HONH_2Cl in 4M HOAc (fraction bond to iron manganese oxides).

Total Metal Concentrations

One gram of each sediment sample was digested with a mixture of HF , HNO_3 , and HCl in a Teflon beaker. Digestion was repeated three times to ensure complete dissolution; the residue was dissolved in 5 ml 2M HNO_3 and rewarmed at 80°C for 20 minutes (residual phase). All extracted solutions were filtered off through Whatman No. 42 filter paper and then used for analysis.

Water Samples

The pH (pH meter, Orion Research, Model SA 520, U.S.A.), the conductivity and salinity (conductivity meter, HANNA Instruments, HI 8033, Italy), dissolved oxygen (Winkler's method) were measured *in situ* for water samples according to standard methods (APHA, 1992). Temperature was measured *in situ* by thermometer. CO_3 and HCO_3 (titrimetrically), Cl (Mohr's method), SO_4^{2-} (turbidimetry), PO_4^{3-} (molybdenum blue method), SiO_2 (molybdo silicate), NO_2^- (modified Griess-Ilosvay method), NO_3^- (chromotropic acid method), Ca and Mg (complexometric titration), TSS and TDS and total organic matter were determined in water samples according to standard methods (APHA, 1992).

Elemental Analysis and Instrumental Technique

Heavy metal concentrations in lake water, different leachates of sediment, and pore-waters were determined by atomic absorption (Fe, Mn, Ca, Mg, Zn, Co, Ni, Cu, Cd, Pb, and Cr) and atomic emission (K and Na) spectrophotometry.

A flame atomic absorption spectrophotometer (Model Solaar 969, Ati Unicam Comp.) equipped with a digital and direct concentration read-out and an air-acetylene burner was used. Single-element hollow cathode lamps and standard instrumental conditions were used for each element (see Table 1). Suitable precautions were taken to minimize interference when necessary. Element concentrations in the different leachates were quantified by appropriate calibration curves of standard element concentrations prepared in the extraction solution.

All standard solutions for AAS or flame emission were prepared from the certified stock solutions (Spectrosol, BDH Chemical L.t.d., Poole, England).

Quality Control and Assurance

Accurate analysis of heavy and toxic elements is dependent upon the prevention of element contamination. All glass and plastic wares should be soaked in diluted acid, e.g., 10% (v/v) nitric acid, and rinsed thoroughly with distilled and deionized water before use to ensure that there is no contamination of the laboratory accessories.

During sampling and laboratory analysis of heavy metal, care should be devoted to prevent sample contamination and to ensure the reliability and quality of analytical results. First of all the use of metallic tools should be avoided whenever possible. Sediment samples should be collected using tools made of stainless steel and stored in nonmetallic containers, such as glass bottles or polyethylene bags, at 4°C prior to laboratory treatment.

In order to provide valid and reliable data in a timely manner, a quality control system must be implemented throughout the analytical process. Quality control (QC) is defined as a system of procedures and practices that result in an increase in precision and a decrease in bias. The use of duplicate analysis, spiked samples,

Table 1. Atomic Absorption Spectra Technique Applicable to the Analysis of Ecological Materials

Element	Wave length (nm)	Flame	Suitable ranges (ppm)	Potential interferences	Control of interferences
Mn	279.5	Air – C ₂ H ₂	0–2 0–20	None significant	
Fe	248.3	Air – C ₂ H ₂	0–20	Si	Add CaCl ₂
Cu	324.8	Air – C ₂ H ₂	0–5	None significant	
Co	240.7	Air – C ₂ H ₂	0–50	Fe if high	Extract
Pb	217.0	Air – C ₂ H ₂	0–10	Cl possible	
Zn	213.9	Air – C ₂ H ₂	0–1	None significant	
Mg	285.2	Air – C ₂ H ₂	0–3	Ca, Al, P, Si	Add releasing agent (Sr or La salt)
Ca	422.7	Air – C ₂ H ₂	0–10 0–40	Al, P, Si, S	As above

Source: Allen, 1989.

standard reference materials, and QC check samples are all mechanisms used to demonstrate the control of quality (Wong, Li, Zhang, Qi, & Min, 2002). In general, to detect contamination and evaluate the reproducibility and effectiveness of the analytical procedures, procedural blanks, duplicates, and certified standard reference material should be included in the analytical program (Wong et al., 2002).

Accuracy of atomic absorption spectrophotometer and validity of the processes were tested with a reference material.

The detection limit is defined as the concentration corresponding to three times the standard deviation of 10 blanks.

Statistical Analysis and Graphical Presentation

In order to investigate elemental associations among the heavy metals and major elements in sediment, and to examine the relationships between the concentrations of elements in water, surface sediment, and sediment Pore water, Pearson correlation, minimum, maximum, standard deviation, standard error, and mean values were performed using SPSS statistical package program version 9.0.

Grapher 2.03.43 description Grapher 2.03 and Micro Word 2000 were used for graphical presentation.

RESULTS AND DISCUSSION

Mean concentrations ($\mu\text{g g}^{-1}$, dry wt.), extractability (percentage of metal extraction “%E”), standard deviation (*SD*), and standard error (*SE*) of both hydroxylamine hydrochloride and ammonium acetate extractants are summarized in Table 2. Physical and chemical characteristics of sediment samples at different study sites are recorded in Table 3. Minimum, maximum, and mean concentrations ($\mu\text{g g}^{-1}$, *SD*, and *SE* of surface and bottom water samples are listed in Table 4. Distribution of metal concentrations ($\mu\text{g g}^{-1}$, dry wt.) in three different extracts of sediment samples ($< 63 \mu\text{m}$) are represented graphically in Figure 2. Percentage of major, minor, and trace metals in pore water are represented graphically in Figure 3. Metal contents (mean) in water, pore water, and in sediment (available) samples is represented graphically (as a percentage) in Figure 4.

Comparison of Chemical Partitioning of Trace, Minor, and Major Metals in Different Extracts of Some Sediment Bights at Nasser Lake

Chemical extraction methods are commonly used in studies of toxic trace element speciation, to elucidate their distribution among the various sediment constituents. The main accumulation mechanisms of heavy metals in sediment imply the existence of three main speciation categories), namely ammonium acetate (pH 7), which extracts the available exchangeable fraction, hydroxylamine hydrochloride, which leaches the metals' bond to Fe/Mn oxide, and the acids attack system, able to release the metals in residual form (lattice metals).

Comparison of the monitoring results of different samples reveals that, in all sediment samples at the study sites, the highest metal concentrations were generally found in residual form (Figure 2). The mean values for the measured metal concentrations of sediment samples in residual form, reflect high iron content ($34348.7 \mu\text{g g}^{-1}$). The comparative increase in Fe level leached by concentrated acids mixture found at El-Matar bight ($36750 \mu\text{g g}^{-1}$), indicates that most of the iron exists as well crystalline oxides, which is probably consistent with the high abundance of magnetite (Fe_2O_3) and ilmenite (FeTiO_3) minerals in Nasser Lake sediments (Moalla et al., 1998). Iron oxides may result from the physical erosion of the drainage area already containing iron oxides. These results agree reasonably well with those reported by Fawzy (2000) for River Nile, Moalla et al. (1998) for Nasser Lake, Tessier et al. (1980) for Yamaska and St. Francois (Quebec, Canada), and Gibbs (1977) for Amazon (Brazil) and Yukon (Alaska) Rivers.

Table 2. Mean Concentrations ($\mu\text{g g}^{-1}$, dry wt.), Extractability (E%), Standard Deviation (SD), and Standard Error (SE) for Nasser Lake Sediment

Items	Fraction bond to Fe/Mn oxide				Exchangeable phase				Pore water phase			
	Mean	E%	SD	SE	Mean	E%	SD	SE	Mean	E%	SD	SE
Fe	4316	12.6	364	210	34.43	0.1	5.06	2.92	2.6	0.2	2.92	1.7
Mn	247	40.4	59.6	34.4	51	8.3	26	15	0.40	60.2	0.3	0.2
Zn	137.1	24.6	31.7	18.3	13.61	2.44	6.67	3.85	1.7	0.01	0.24	0.14
Pb	5.63	28.14	2.73	1.58	1.04	5.2	0.26	0.15	0.04	0.1	0.03	0.02
Cd	0.001	87.5	0.00023	0.00013	0.003	75	0.004	0.0022	0.001	0.02	0.0	0.0
Co	1.4	27.5	0.696	0.402	0.36	7.1	0.56	0.32	0.001	0.10	0.0	0.0
Ni	9	32.4	0.964	0.557	0.32	1.2	0.08	0.04	0.023	0.30	0.01	0.0033
Cr	5.4	5.2	0.85	0.5	2.9	2.8	0.37	0.2	0.10	0.01	0.012	0.01
Cu	1.94	8.0	0.604	0.349	0.001	0.004	0.0003	0.0002	0.004	0.10	0.01	0.003
Ca	6252	42.6	4635	2676	6383	29.4	3062	1768	110.5	0.50	74	43
Mg	4873	67.2	2221	1283	1372	18.9	533	308	36.32	0.50	10.2	6.0
Na	90	12.7	39.1	22.5	51.6	7.3	13.44	7.76	37.23	5.3	2.4	1.4
K	20.13	11	4.8	2.72	14	7.6	2.72	1.57	10.43	5.7	3.8	2.2

Table 3. Physicochemical Characteristic of Lake Nasser Sediment

Item	Location		
	El-Matar	El-Sage	El-Caria
pH	7.57	7.00	7.40
cond. μ mhos	64.50	167.50	279.50
O.M. mg g^{-1}	3.30	5.10	5.34
CaCO_3 mg g^{-1}	378.80	395.00	373.80

Distribution of total Mn concentrations shows great variation against studied sites ($423.15 - 794.4 \mu\text{g g}^{-1}$). This may indicate most sediment Mn present in silicate minerals. A considerable amount of Mn was extracted by hydroxylamine hydrochloride, (40.4% of total Mn sediment) comparing to the low available Mn concentration (8.3% of total Mn sediment).

Mn oxides play a significant role in the fixation for Co in soils owing to the strong affinity of Co for Mn oxides (Kabata-Pendias & Pendias, 1992). The results, illustrated in Figure 2, shows a relative increase of Co-concentration in fractions bound to Fe/Mn oxides ($E = 27.3\%$ of total Co sediment). This was anticipated since Co is usually strongly associated with Mn oxides (Wong et al., 2002). Total extractable Zn concentration in sediments ranged between $491.4 \mu\text{g g}^{-1}$ to $629.4 \mu\text{g g}^{-1}$, while total extractable Cu content varied between $22.5 \mu\text{g g}^{-1}$ to $27.6 \mu\text{g g}^{-1}$ and total extractable Ni concentrations ranged between $26.25 \mu\text{g g}^{-1}$ to $29 \mu\text{g g}^{-1}$. However total extractable Cd exhibited slight concentrations ($0.0007 \mu\text{g g}^{-1}$ to $0.001 \mu\text{g g}^{-1}$).

A comparison of the extraction results for Cu, Pb, and Co shows that high extractability was obtained by total acids attack against studied sediment samples. This is possibly attributable to the enrichment of these metals in the residual phases, which consists essentially of detrital silicate minerals. Na and K show the highest concentrations compared to the other metals present in the studied bights, consistent with the fact that some silicate minerals, e.g., alkali feldspar, can be completely dissolved by weathering, releasing sodium and potassium into solution (Elwa, 1985). This also can be due to the geomorphology of these regions. The dominant rocks of western side are Nubian sandstone, limestone, dolomite, and kaolin, while the main rock types at the eastern side are granite basalt, marble, and Nubian sandstone (Soltan & Rashed, 2003). Similar results have been reported by other workers and are usually explained in terms of the theory mentioned by Mackereth (1966). He reported that the sediment concentrations of Na, K, and associated elements such as Mg must be directly proportional to the intensity of erosion. A slower rate of erosion allows more of these elements to be leached

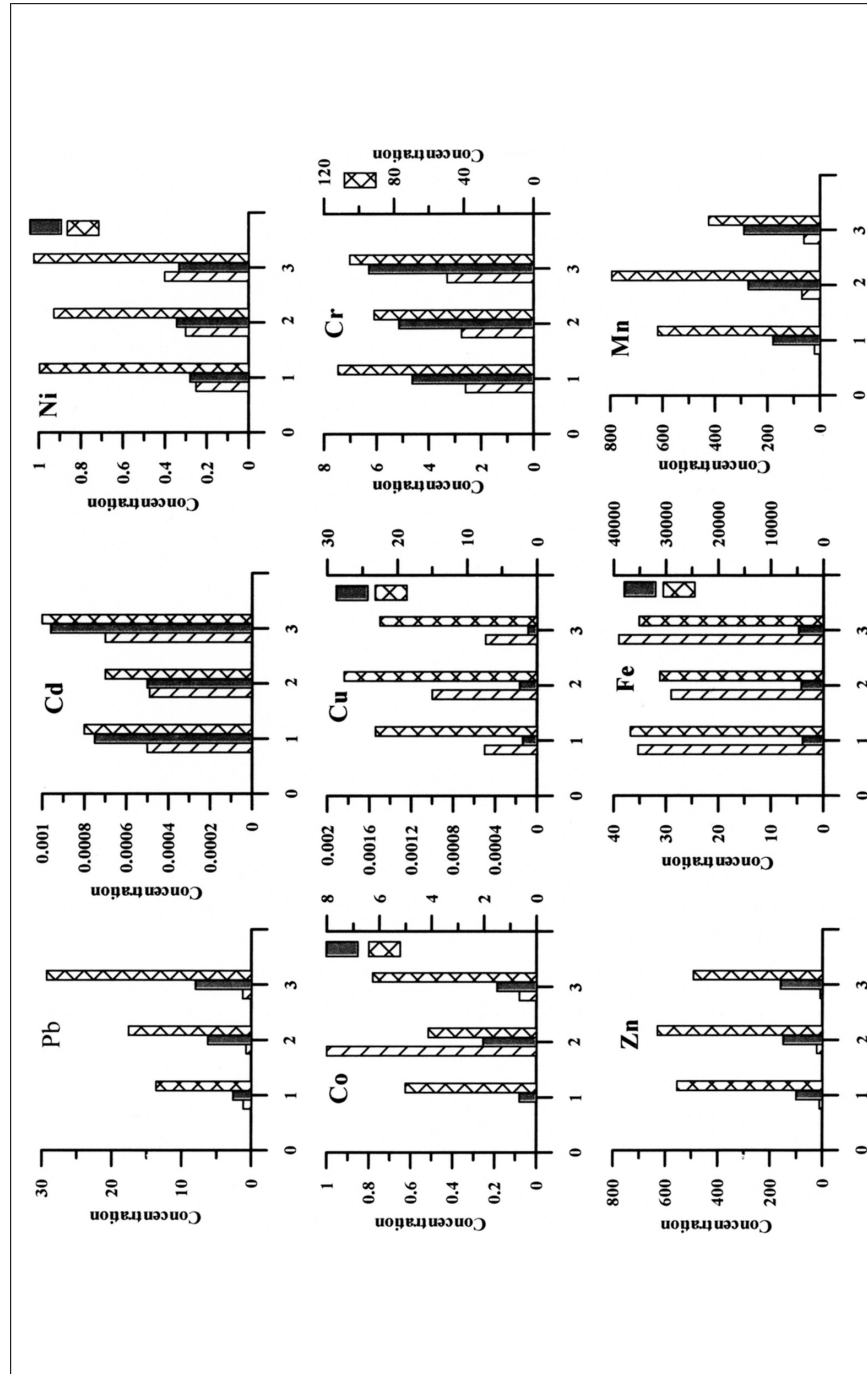
Table 4. Summary Basic Statistics of Heavy, Trace Elements, and Different Parameters of Nasser Lake Water

Items	Surface water					Bottom water				
	Mean	Minimum	Maximum	SD	SE	Mean	Minimum	Maximum	SD	SE
pH	8.4	8.32	8.4	0.057	0.0328	8.4	8.4	8.42	0.012	0.01
Cond., μScm^{-1}	207.7	198	218	10.02	5.78	209	196	220	12.12	7.0
Salinity, unit	0.09	0.08	0.10	0.01	0.01	0.1	0.09	0.10	0.006	0.003
Temperature	29.3	28.5	29.5	0.29	0.17	29.5	29	30	0.5	0.29
HCO_3^- , mg/l ^a	96.77	94.1	100.1	3.06	1.76	104.1	98.1	112.1	7.21	4.16
CO_3^{2-} , mg/l ^a	10.67	8	16	4.6	2.67	9.33	8	12	2.31	1.33
Cl^- , mg/l	9.93	9.5	10.8	0.75	0.433	10.37	9.5	10.8	0.75	0.43
SO_4^{2-} , mg/l	8.37	5.4	14.1	4.97	2.87	5.47	5.3	5.8	0.29	0.17
PO_4^{3-} , mg/l	0.31	0.15	0.5	0.18	0.1	0.22	0.21	0.23	0.012	0.01
SiO_2 , mg/l	5.740	5.4	6.32	0.505	0.291	6.0	5.7	6.3	0.3	0.173
NO_3^- , mg/l	0.81	0.8	0.83	0.02	0.01	1.9	0.5	4.4	2.2	1.27
NO_2^- , mg/l	nil	nil	nil	0.0	0.0	nil	nil	nil	0.0	0.0
COD, mg/l	6.4	5.4	8	1.4	0.81	5.6	4.9	6.2	0.66	0.38
BOD, mg/l	1.7	1.0	2.4	0.7	0.4	1.6	0.5	3.1	1.4	0.8

Ca, $\mu\text{g g}^{-1}$	18.22	15.4	19.75	2.44	1.41	13.25	8.3	16.7	4.40	2.54
Mg, $\mu\text{g g}^{-1}$	8.03	6.80	8.70	1.07	0.62	6.10	5.40	6.75	0.68	0.39
TDS, mg/l	132.87	126.9	139.5	6.33	3.65	133.6	125.44	140.5	7.6	4.39
TSS, mg/l	1.87	0.70	3.00	1.15	0.66	2.70	1.90	4.10	1.24	0.72
O.M, mg/l	1.18	0.25	1.90	0.85	0.49	1.30	0.25	2.13	0.95	0.55
Fe, $\mu\text{g g}^{-1}$	0.09	0.07	0.10	0.017	0.01	0.08	0.07	0.10	0.017	0.01
Mn, $\mu\text{g g}^{-1}$	0.13	0.02	0.34	0.19	0.11	0.01	0.01	0.02	0.01	0.003
Cu, $\mu\text{g g}^{-1}$	0.00243	0.0013	0.004	0.0014	0.00081	0.002	0.002	0.003	0.00058	0.00033
Zn, $\mu\text{g g}^{-1}$	0.1	0.01	0.20	0.095	0.055	0.087	0.01	0.14	0.068	0.039
Cd, $\mu\text{g g}^{-1}$	0.001	0.00049	0.0005	0.00001	0.0	0.0005	0.0005	0.0005	0.0	0.0
Co, $\mu\text{g g}^{-1}$	0.001	0.0005	0.001	0.00025	0.00015	0.0006	0.0005	0.0007	0.0001	0.00006
Ni, $\mu\text{g g}^{-1}$	0.001	0.001	0.002	0.00058	0.00033	0.0017	0.001	0.002	0.0006	0.0003
Pb, $\mu\text{g g}^{-1}$	0.0003	0.0003	0.0003	0.0	0.0	0.004	0.003	0.004	0.001	0.00033
Cr, $\mu\text{g g}^{-1}$	0.003	0.002	0.004	0.001	0.00058	0.01	0.01	0.01	0.0	0.0
Na, $\mu\text{g g}^{-1}$	28.97	26.70	30.80	2.08	1.20	27.40	24.80	29.1	2.31	1.33
K, $\mu\text{g g}^{-1}$	3.80	3.50	4.20	0.36	0.21	3.72	3.70	3.75	0.029	0.017

^aas CaCO₃

Note: SD = Standard Deviation, SE = Standard Error.



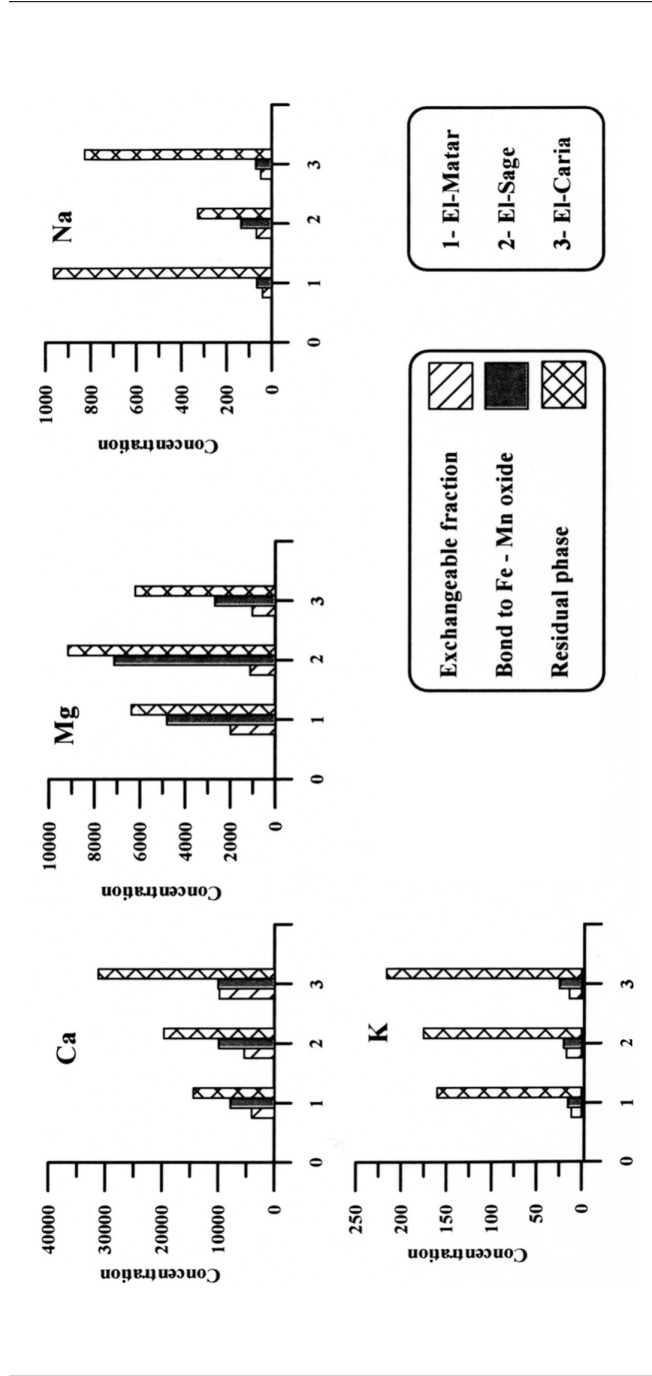


Figure 2. Distribution of Mean Metal Concentrations (Different Phases) in Some Bights Sediment, Nasser Lake Sediment.

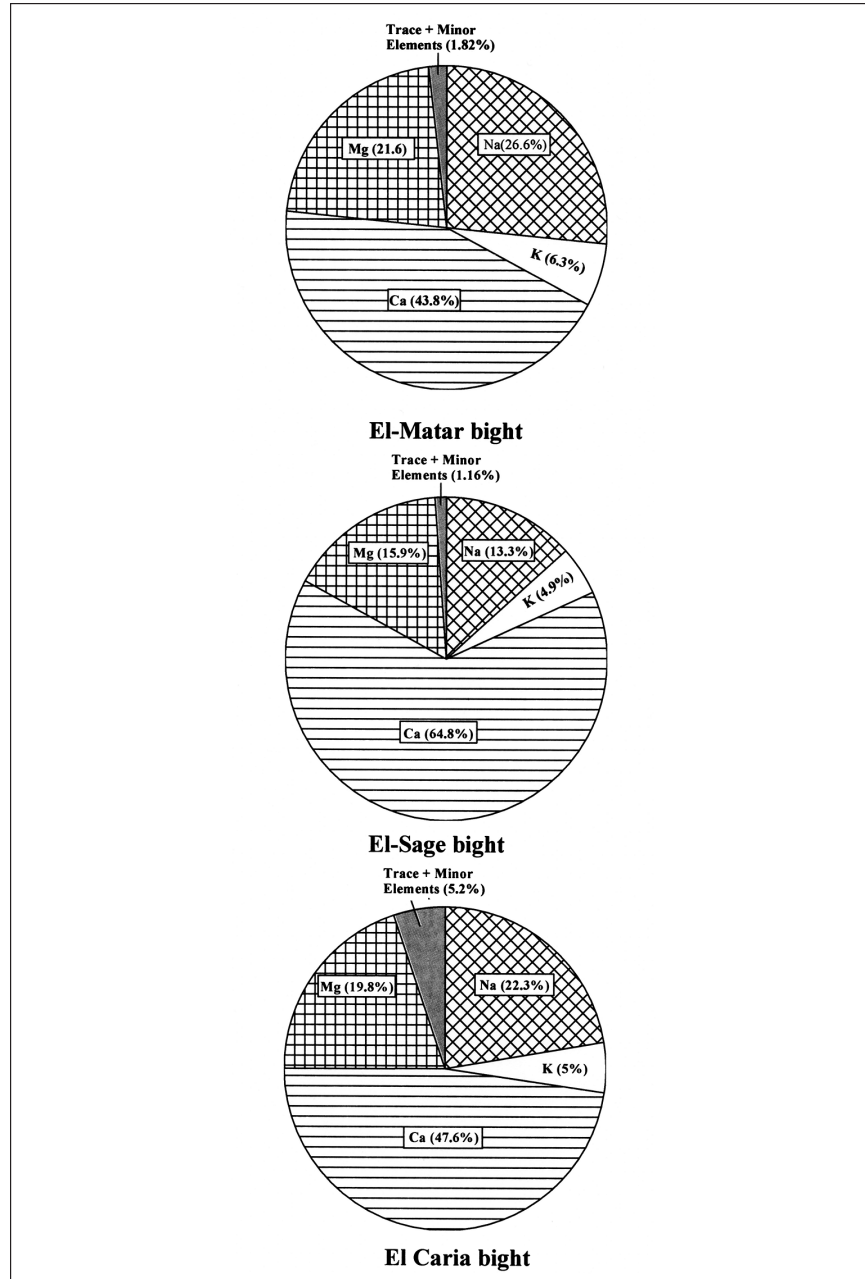


Figure 3. Distribution of major, minor, and trace metals in pore water of some bights sediment, Nasser Lake.

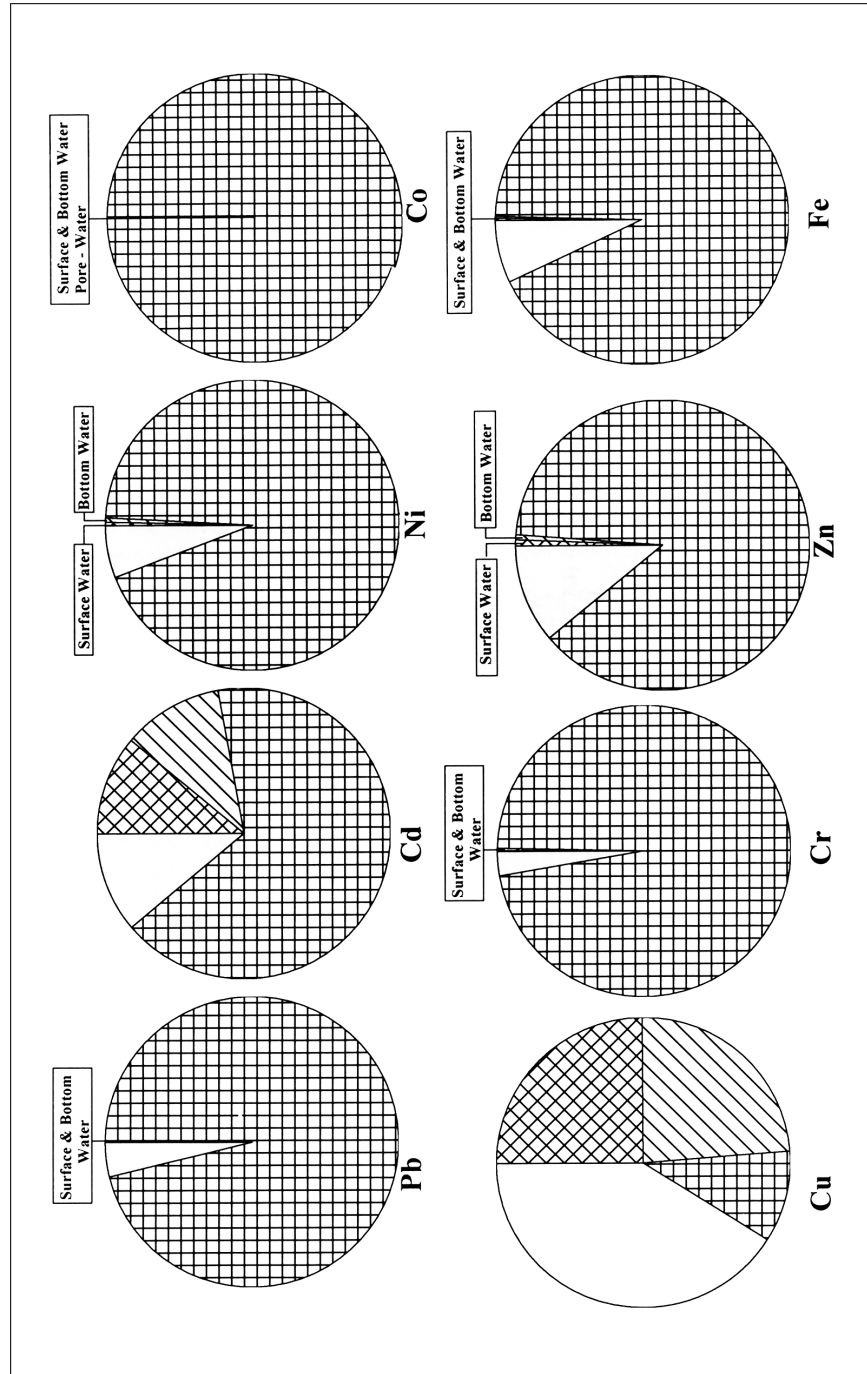
from soil as soluble components and so their resultant concentration in the sediment is reduced, because these elements are deficient in the accumulating minerals (Fawzy, 2000). The study shows that the complete digestion using a mixture of concentrated acids is the most suitable and powerful digestion system for total metal analysis.

Amounts of measured metals contents in the samples of studied sites were extracted by ammonium acetate (pH 7), representing the exchangeable form (Förstner & Salomons, 1980). These available metals, in the course of changing the physicochemical properties of the water sediment system, can be released into overlying water masses and equilibrate easily with solid phases. Average iron and manganese contents, extracted by ammonium acetate were 29.0-39 $\mu\text{g g}^{-1}$ and 21.3-70.1 $\mu\text{g g}^{-1}$ and represented 0.1% and 8.3% of the total iron and manganese sediment concentrations, respectively.

Numerous studies have shown that trace metals will readily adsorb to both iron and manganese oxides or hydroxides (Förstner & Wittman, 1981; Stumm & Morgan, 1981), since much of this would originally be expected to exist in colloidal forms, subsequent coagulation, and sedimentation would lead to sediment enrichment. Hetero-coagulation on existing layer particles could also lead to sediment enrichment. Another mechanism that could increase the trace metal concentration of sediment is the dissolution of iron and manganese oxide under anaerobic conditions (or from inputs of industrial effluents) with subsequent reprecipitation in more oxic waters. This reprecipitation of colloidal iron and manganese oxide particles would scavenge trace metals present in the water column by adsorption and co-precipitation. The exchangeable Pb and Cd in the sediment were generally low, accounting for 5.2% and 75% respectively, of total soil Pb and Cd. The relatively high exchangeable fraction of Zn in surficial sediment could be attributed to lower sediment pH (Wong et al., 2002).

Iron (III), Mn (IV), and Mn (VII) are present as oxides which have very strong capacities to adsorb heavy metals (Okazaki, 1987). Therefore, it's expected that the changes in chemical phases of Fe and Mn affect considerably the behavior of other trace metal elements (Saeki, Okazaki, & Matsumoto, 1993). The samples of all bights sediments exhibited moderate concentrations of the metals bound to Fe/Mn oxides (Cu, 1.94 $\mu\text{g g}^{-1}$; Zn, 137.1 $\mu\text{g g}^{-1}$; Cr, 4.7 $\mu\text{g g}^{-1}$; Co, 1.4 $\mu\text{g g}^{-1}$; Ni, 9 $\mu\text{g g}^{-1}$; and Pb, 5.63 $\mu\text{g g}^{-1}$) in comparison with the low concentrations in the exchangeable fraction (Figure 2). This high level of Pb content in this fraction may indicate its association with Fe/Mn oxides or may be attributed to adsorption of Pb on potassium-rich minerals. This is consistent with the fact that Pb^{2+} can be incorporated into potassium-rich minerals by replacement of potassium having a similar ionic radius of about 1.3 Å (Moalla, 1990). This is also in agreement with results reported by Gupta and Chen (1975). They reported out that 16-48% of the Pb in coastal marine sediments was bound to the Fe/Mn oxides phases.

The high percentage of the total zinc concentration extracted by hydroxylamine hydrochloride solution may reflect the strong association with amorphous Fe/Mn



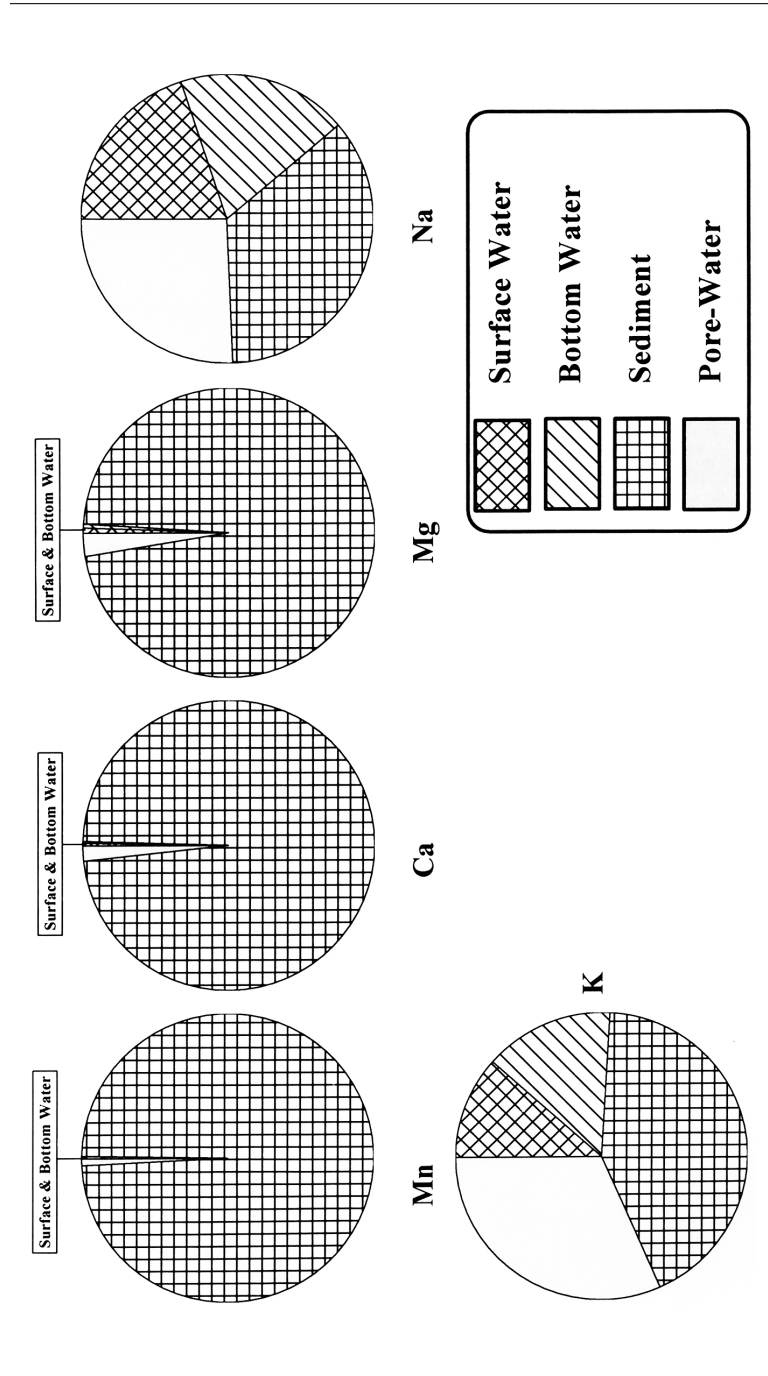


Figure 4. Distribution of Metal Concentrations (%) in Sediment, Pore water, and surface and bottom water of some bights, Nasser Lake.

oxides. These findings are in agreement with the known ability of amorphous Fe/Mn oxides to scavenge Zn from solution (Fawzy, 2000; Jenne, 1968). Oxides of Fe and Mn effectively adsorb or occlude most trace and toxic metal cations. Dissolution of these elements should, therefore, be accompanied by a release of trace metals, such a release was only weakly reflected in the behavior of these trace metals. The results of analysis indicate the above interpretation, where higher percentages of total concentrations of Zn (24.6%) and Co (27.3%) were extracted by hydroxylamine hydrochloride than ammonium acetate (Zn, 2.44%; Co, 7.02%, of total Zn and Co sediment, respectively).

Hydroxylamine hydrochloride extractable of Na (mean = $90 \mu\text{g g}^{-1}$) is generally higher than the corresponding amount extracted by ammonium acetate (mean = $51.6 \mu\text{g g}^{-1}$). This is possibly attributed to the enrichment of these metals in ferromanganesian (biotite, amphibole, pyroxene, olivine, and opaques) minerals (Balkis & Cagatay, 2001).

Mean Cu concentration in fraction bond to Fe/Mn oxides is about $1.94 \mu\text{g g}^{-1}$, with average about $1.3 \mu\text{g g}^{-1}$ – $2.5 \mu\text{g g}^{-1}$. Shuman (1985) and Shata, Deek, and Okbah, (1993) showed that copper is associated mainly with the Fe/Mn oxide fraction in sediment samples. There is evidence from electron microprobe analysis of surface soil that Pb and Ca are concentrated and distributed approximately equally between the Fe and Mn oxides (da Silva, Abate, Lichtig, & Masini, 2002). Cadmium remarkably changed in the chemical phase distributions with oxidation of sediments compared with other heavy metals (Saeki et al., 1993). Although the concentrations of Cd (0.003 – $0.001 \mu\text{g g}^{-1}$) in exchangeable form and in fraction bond to Fe/Mn oxide, respectively were lower than those of the other measured metals, but its effect on the environment and human beings is very strict even in low concentration. It is necessary to pay attention to the potential mobility of Cd induced by the oxidation of sediment. Contrary to Cd, Pb does not play a major role in aquatic food chains and natural soils in the Pearl River Delta and shows that the chemical partitioning pattern of Pb indicated that Pb was largely associated with Fe/Mn oxide and residual fractions.

The follow-up of different metal concentrations in lake sediment exhibited that all measured metals are characterized by relatively high concentrations in fractions bound to Fe/Mn oxides (Table 2).

All available metal concentrations are below the maximum permissible value set by Bowen (1979), Turekian and Wedepohl (1961), and Wedepohl (1968).

Chemical Composition of Pore Water Samples

Pore water extracted from sediment of studied sites was analyzed for heavy metal levels Figure 3. Dissolved metal concentrations in pore water samples were much higher than those of surface waters (Figure 4).

Concentrations of Fe ($2.6 \mu\text{g g}^{-1}$), Mn ($0.4 \mu\text{g g}^{-1}$), and Zn ($1.7 \mu\text{g g}^{-1}$) were more than an order of magnitude greater than those in surface waters

(Fe, $0.09 \mu\text{g g}^{-1}$; Mn, $0.13 \mu\text{g g}^{-1}$; Zn, $0.1 \mu\text{g g}^{-1}$ as a means of studied bights), indicating that the rapid oxidation of ions and subsequent sedimentation on the bottom sediment lead to a dramatic decrease in the ion concentrations in the water bodies (Chabbi, 2003). Similar patterns are also observed for Ca ($110.5 \mu\text{g g}^{-1}$, $18.22 \mu\text{g g}^{-1}$); Mg ($36.3 \mu\text{g g}^{-1}$, $8.0 \mu\text{g g}^{-1}$); Na ($37.2 \mu\text{g g}^{-1}$, $29.0 \mu\text{g g}^{-1}$); and K ($10.4 \mu\text{g g}^{-1}$, $3.8 \mu\text{g g}^{-1}$) in pore and surface water, respectively.

Tack, Singh, and Verloo (1998) studied heavy metal concentrations in consecutive saturation extracts of dredged sediment-derived surface soils and concluded that metal concentrations in sediment extracts may provide indications about concentration levels that could be encountered in pore water in the field. Comparing the trace metal concentrations in pore waters, it is clear that they are much higher than in surface and bottom water (Figure 4). For soil that remains in an oxidized state, metal concentrations in the pore water would remain relatively high during that period and show a decreasing trend (Tack et al., 1998).

Chabbi (2003) studied metal concentrations in pore water of the Lusatian Lignite mining sediments and internal metal distribution in *Juncus Bulbosus* and indicated that iron in the pore waters was very likely present as iron oxhydroxides or bound to phosphates. Mn and Cu were probably in oxidized form.

Monitoring results of sampling locations in the lake shows that the concentrations of Co and Cd were significantly low in pore water of study sites (Figure 3). It could be assumed that some other binding phases in sediment might be associated with the remaining Cu (Yu et al., 2001). Similar assumptions has been reported elsewhere (Ankley, Di Toro, Hansen, & Berry, 1996; Leonard, Ankley, & Hoke, 1996; Van den Berg, Loch, Van der Heijdt, & Zwolsman, 1998). Prokop et al. (2003) conducted studies to investigate the effect of land deposition of contaminated sediments on the bioavailability and mobility of metals; these indicate that the major factors controlling the release of metals are pH, organic matter content, major element chemistry, and biological activity. As metals are usually bonded to the small-size fraction of sediment particles or dissolved organic matter in pore water, faster leaching through facilitated transport of these particles without desorption may be expected (Prokop et al., 2003).

Pore water metal contents are an order of magnitude higher than generally recorded in surface and bottom water of study bights. In sulphidic systems, the production of sulphide via bacterial sulphate reduction would be expected to buffer pore water metals to low levels, as metal sulphide minerals are thermodynamically favored. The presence of high pore water metals levels (Figure 3) either indicates reduced levels of pore water sulphate due to freshwater mixing (supported by pore water anion concentrations) or a presence of metals at concentrations in excess of that which can be buffered by bacterial sulphide production (Ankley et al., 1996).

Moderately high concentrations were found for Pb ($0.04 \mu\text{g g}^{-1}$), Ni ($0.02 \mu\text{g g}^{-1}$), Cr ($0.1 \mu\text{g g}^{-1}$), Cu ($0.004 \mu\text{g g}^{-1}$), Fe ($2.6 \mu\text{g g}^{-1}$), Na ($37.2 \mu\text{g g}^{-1}$), K ($10.4 \mu\text{g g}^{-1}$), Ca ($110.5 \mu\text{g g}^{-1}$), and Mg ($36.3 \mu\text{g g}^{-1}$) compared to those

of surface and bottom waters, respectively (Pb, 0.0003 $\mu\text{g g}^{-1}$; 0.0004 $\mu\text{g g}^{-1}$; Ni, 0.0013 $\mu\text{g g}^{-1}$; 0.0017 $\mu\text{g g}^{-1}$; Cr, 0.003 $\mu\text{g g}^{-1}$, 0.01 $\mu\text{g g}^{-1}$; Cu, 0.002 $\mu\text{g g}^{-1}$, 0.002 $\mu\text{g g}^{-1}$; Fe, 0.09 $\mu\text{g g}^{-1}$, 0.08 $\mu\text{g g}^{-1}$; Na, 28.97 $\mu\text{g g}^{-1}$, 27.43 $\mu\text{g g}^{-1}$; K, 3.8 $\mu\text{g g}^{-1}$, 4.72 $\mu\text{g g}^{-1}$; Ca, 18.22 $\mu\text{g g}^{-1}$, 13.3 $\mu\text{g g}^{-1}$; Mg, 8.0 $\mu\text{g g}^{-1}$, 6.1 $\mu\text{g g}^{-1}$). The high levels of pore water metals will lead to the flux of metals *via* molecular diffusion from the pore water to the surface of sediments. Fe (II) and Mn (II) produced in the pore water *via* bacterial iron and manganese reduction will be oxidized at the sediment surface, thereby scavenging pore water trace metals. As a result, sediment will become enriched in metals (Sullivan & Taylor, 2003).

The presence of the exchangeable fraction and elevated pore water metal levels (Figure 4) could indicate freshwater diagenesis, or reflect the high levels of metals in the sediment (Sullivan & Taylor, 2003).

The Water Quality Parameters

The monitoring of chemical properties of lake water shows moderate pH value between 8.32-8.43 and 8.40-8.42 for surface and bottom water, respectively. pH of the aquatic systems is an important indicator of the water quality. Unpolluted streams normally show a near-neutral or slightly alkaline pH. The relatively high pH value observed on surface water of El-Matar bight (8.43) could be due to abundance of high density aquatic plants (*Myrriophyllum spicatum*). Conductance qualitatively reflects the status of inorganic pollution and is a measure of total dissolved solids and ionized species in the water. The sampling site, surface, and bottom water of El-Matar bight recorded the lowest conductance values (198 $\mu\text{S cm}^{-1}$ and 196 $\mu\text{S cm}^{-1}$, respectively) against study bights. El-Caria bight recorded a slight increase in values of surface and bottom water (218 $\mu\text{S cm}^{-1}$ and 220 $\mu\text{S cm}^{-1}$). No great variation in conductance value was measured against all study bights. This could be ascribed to relatively continuous mixing of water column of Nasser Lake.

Salinity concentrations distributed between 0.08-0.10 unit and 0.09-0.10 unit in surface and bottom water, respectively. El-Matar bight recorded the highest carbonates values on surface and bottom water (16 mg L^{-1} , 12 mg L^{-1} , respectively) while El-Sage bight recorded the highest values of bicarbonates (100 mg L^{-1} , 112.1 mg L^{-1}) of surface and bottom water, respectively. Variation in carbonate and bicarbonate concentrations shows that studied bights recorded higher concentration of bicarbonate contents compared to carbonate concentrations.

Uptake of CO_2 in HCO_3^- by phytoplankton populations and the CO_2 - H_2O interaction during the photosynthetic processes decrease bicarbonates content in lake water (Table 4). Chloride concentration is found to vary between 9.93-10.37 mg L^{-1} as a mean of surface and bottom water, respectively. Increased rate of evaporation caused by high air temperature (time of sampling) increased chloride content in some study bights.

The oxygen content in the water samples depends on a number of physical, chemical, biological, and microbiological processes. The analysis of dissolved oxygen (DO) is a key test in water pollution control (Zayed, Nour El-Dien, & Rabie, 1995). The DO levels fluctuated between 5.4 to 8 mg L⁻¹ and 4.9 to 6.2 mg L⁻¹ of surface and bottom water respectively. The highest DO value (8 mg L⁻¹) measured in surface water of El-Matar bight was caused by photosynthetic activities ($\text{CO}_2 + \text{H}_2\text{O} \leftrightarrow \text{O}_2 + \text{Energy}$) of aquatic plants (Iskaros, 1993) and water circulation. Bottom water of El-Caria bight recorded the lowest DO level (4.9 mg L⁻¹). This due to the decomposition of organic matter resulting from the effluents of tourist hotel and ships, which consumed the dissolved oxygen in this process.

BOD values indicate the extent of organic pollution in the aquatic systems, which adversely affect the water quality. BOD concentrations ranged between 1.70 mg L⁻¹ and 1.57 mg L⁻¹ as measured in surface and bottom water, respectively (Table 4). BOD levels are low and less than 3 mg L⁻¹, so that pollution is at a tolerable level (Roy, Sarbadhikary, Guha, Mandal, & Bhallachryya, 1996).

Sulphate is of interest mainly where control of total dissolved solids is a major consideration. Concentration of sulphates is in the range 14.1-5.4 mg L⁻¹ and 5.8-5.3 mg L⁻¹ of surface and bottom water, respectively. The highest sulphate content 14.1 found in well oxygenated water (surface water of El-Matar bight).

Silicate contents in water samples fluctuated between 5.40 and 6.32 mg L⁻¹ and 5.70 and 6.30 mg L⁻¹ as minimum and maximum values of surface and bottom water, respectively.

Phosphates and nitrates are important parameters to assess the water quality. Total phosphates (PO_4^{3-}) in studied bights water are not high. Lowest levels were observed at El-Matar bight, but the levels increase with high abundance of aquatic plants (*Myrriophyllum spicatum*) at El-Caria bight and increase the rate of its decay (Awadallah, Ismail, Abd El Aal, & Soltan, 1993). Little high nitrate concentration was found in the lake. Interestingly, low nitrate levels were found at El-Caria and El-Matar bights and the highest values at El-Sage bight. The source of higher levels of nitrate could be the wastes of trader ships traveling across the main stream of the lake and the discharges from some fish and agricultural farms, which were constructed since 1987 on some of the lake shores (Rashed, 2001). Lake water is nitrite-free.

The chemical oxygen demand (COD) is used as a measure of the oxygen equivalent to the oxidizable organic matter content of a sample. The COD measurements (Table 4) varied between 0.25-1.90 mg L⁻¹ and 0.25-2.13 mg L⁻¹ of surface and bottom water of lakes studied. El-Caria bight exhibited the highest COD concentration (2.13 mg L⁻¹ of bottom water) due to the increase of discharge resulting from the tourist village found at this bight and tourism ships' wastes. TDS influences the quality of water parameters. High levels of dissolved substances in the water systems increase the biological and chemical oxygen demand

which deplete the dissolved oxygen levels in the aquatic systems (Jonnalagadda & Mhere, 2001). The levels in a broad sense reflect the pollution burden of the aquatic system. The concentrations of total dissolved solids were ranged between 132.9 and 133.6 mg L⁻¹ as a mean of surface and bottom water, respectively. TDS levels of studied bights were found to be lower than 500 mg L⁻¹ (WHO, 1993), so the water is a suitable source for drinking and irrigation. The total TSS values include all material like silt, clay, organic matter, plankton, and microscopic organisms. It is observed that TSS values against all studied bights varied between 0.7-3.0 mg L⁻¹ and 1.9-4.1 mg L⁻¹ of surface and bottom water. The highest value at El-Caria bight and the lowest value at El-Matar demonstrate self-purification of the lake.

Ca and Mg are considered as major ions in lake water. There was no great variation in Ca concentrations between studied bights. The highest Ca (27.30 mg L⁻¹) and Mg (13.44 mg L⁻¹) concentrations at bottom water of El-Sage bight and surface water of El-Caria bight could be caused by the increase of CaCO₃ solubility and re-dissolution of the microorganisms containing Mg (Awadallah et al., 1993).

Assessment of lake water quality—From the obtained data, it can be concluded that there are pronounced changes in the physical and chemical composition of the lake water, a phenomenon which is due to regional environmental and physical changes (water masses movement, turbulence), hydrological, biological, geochemical, and biogeochemical variation. All these processes have great influence on the distribution of physical and chemical components, on populations (bacteria, plankton, benthic fauna and fish) living in the lake, on sedimentation of trace elements and on the water quality. However, all measured environmental parameters are within the desired safety baseline levels of water (Egyptian Chemical Standards (ECS), 1994; EPA, 1992; WHO, 1993; Alaska, 1994).

Trace metals in lake water—Vertical distribution of trace metals shows relative increases in their contents at bottom water (Table 4). The investigated water samples reflect low limits for trace metals. Cd and Pb as toxic and non-essential metals in lake water varied between 0.00049-0.0005 µg g⁻¹; 0.0003-0.0003 µg g⁻¹; and 0.0005-0.0005 µg g⁻¹; 0.003-0.004 µg g⁻¹ on surface and bottom water, respectively. The gasoline leakage from fishery boats increases lead and cadmium levels in lake water.

Detection limits of Zn and Cu are 0.1 µg g⁻¹, 0.087 µg g⁻¹, and 0.0024 µg g⁻¹, 0.002 µg g⁻¹ as mean concentrations of these metals in surface and bottom water, respectively. Zn and Cd are chemically similar and may compete for binding sites in the sediments (Grant, Buckley, Baily, & Selles, 1998), so the Zn/Cd ratio was used as an additional index to evaluate water quality. Simeonov et al. (2003) assessed the surface water quality in Northern Greece and concluded, using a PCA (a powerful pattern recognition technique that attempts to explain the variances of a large dataset of inter-correlated variables with a smaller set of

independent variables), that the total concentration of each element is made up of the sum of elemental concentrations from each identified polluting or natural source component. Occurrence of iron (mean = $0.09 \mu\text{g g}^{-1}$; $0.08 \mu\text{g g}^{-1}$ of surface and bottom water, respectively) may originate from surrounding rocks (Soltan, 1988). Cr, Co, Zn, and Mn play essential and biochemical roles in life processes in the aquatic environment. Distribution of such elements show increase in their contents at bottom water (Table 4), possibly attributable to the fact that metals are significantly soluble at low pH (Soltan, 1991).

The heavy metals move with the water current with suspended solids and sediment as the main carriers. They also partly settle in the sediment due to the slight alkalinity of water and due to the presence of magnetite (Fe_3O_4) as the primary Fe minerals in lake bed, where it can easily hydrolyze to form negative charge species (Haiyan & Stuanes, 2003; Zhang, Zhang, & Dong, 1986). This caused an inconsistent change in the heavy metal concentrations. In addition, hydrological parameters such as water flow and velocity also affected the removal and transformation of heavy metals in the water and sediments (Inkonnikov, 1993).

According to the values established for some trace metals, e.g., Zn, $0.2 \mu\text{g g}^{-1}$, $0.14 \mu\text{g g}^{-1}$, and Cu, $0.004 \mu\text{g g}^{-1}$, $0.003 \mu\text{g g}^{-1}$ (as maximum values of surface and bottom water, respectively), indicating that the abnormal concentrations of these elements may be attributed to discharges of agriculture or fish farms at the different sides of study bights or, it can be induced by wastes of tourism or trade ships that travel across the main body of the lake. The main natural source of heavy metals in water is weathering of minerals (Klavins, Briede, Rodinov, Kokorite, Parele, & Klavina, 2000; Kotas & Stasicka, 2000).

To diminish or eliminate the relative increase in heavy metals concentrations, first priority must be given to pollution sources, not only to control their discharge concentration, but also their discharge amount, with zero discharge as the aim.

In terms of mean concentrations in water the sequence was:

Na > Ca > Mg > K > Mn > Zn > Fe > Cr > Cu > Ni > Co > Cd > Pb for surface water

Na > Ca > Mg > K > Zn > Fe > Mn = Cr > Pb > Cu > Ni > Co > Cd for bottom water

Therefore, Cd is one of the less accumulated elements.

Statistical Analysis

Horizontal variation of element concentrations present in a complex picture as it is affected by many factors, including variations in the tourist, industry effluents influx, and intensity of erosion during periods of high water floods and agitation of the bottom sediments by the traffic of lake and navigation.

Iron is considered a conservative element, i.e., iron in the exchangeable form is comparatively low. Figure 2 shows pronounced spatial variations in the three different extractant solutions. This may be attributed to the great sensitivity of iron to pH and redox variations (Fawzy, 2000). Iron is a mobile element and subjected to diagenetic remobilization (Förstner & Wittman, 1979). Obtained data

reveals good positive correlation between iron and lead in exchangeable form ($r = 0.990$). This is obviously consistent with known sorptive properties of Fe/Mn oxides (Moalla, 1990). Mn and Fe are present at much higher concentrations in different extractant solutions (Figure 2), and have strong capacities to adsorb heavy metals (Okazaki, 1987). Therefore, it is expected that the changes in chemical phases of Fe and Mn affect considerably the behavior of other trace metal elements (Saeki et al., 1993).

Excellent positive correlation arises between Zn and Mn ($r = 1.00$) in fractions bonded to Fe/Mn oxide, consistent with the fact that Zn could be associated with Mn co-precipitation or adsorption on a manganese oxide or hydroxide (Bertin & Boury, 1995). Good positive correlation coefficient between Mn and Mg ($r = 0.998$) may be attributed to its accommodation with Mg-bearing minerals, especially olivine and ferromagnesium minerals (Moalla, 1990). A statistical evaluation of the data shows excellent correlations between Fe and Cr ($r = 1.00$), Na ($r = 0.997$), and K ($r = 0.997$) in residual form. This suggests that these metals are associated with amorphous iron oxide.

Comparing the analytical results, the high Cu concentrations exist in fractions bound to Fe/Mn oxide (Figure 2). This suggests its association with amorphous iron oxides. High significant correlation arises between carbonates in one hand and available Zn ($r = 1.00$) and total Cu ($r = 0.994$) on the other hand. A possible explanation of the association of these metals with carbonate are that metals are preferentially bound to carbonate (Waller & Pickering, 1992). The pKsp values for Zn and Cu carbonate are 10.84 and 9.60, respectively. Available Cu shows high significant correlation with Mg in different phases, i.e., in fractions bonded to Fe/Mn oxide ($r = 0.990$) and residual form ($r = 1.00$), consistent with the fact that some metals such as Cu are strongly enriched with Mg in ultramajic rocks (e.g., proxene and olivine). Copper is an essential micronutrient for plants and animals and is involved with many enzyme systems (Bowen, 1979). Chester, Thomas, Lin, Basahan, and Jacinto (1988) reported that 50% of the total copper in the surface water particulates is held in organic association and deposited at the sediment surface.

Chromium is widely distributed in sediment although its concentrations are generally low. Horizontal and vertical distribution of Cr shows that it is relatively equally distributed between available form and fraction bound to Fe/Mn oxide ($E = 3.5, 4.6\%$, respectively of total Cr sediment). Chromium is one of several heavy metals that causes serious environmental contamination in soil, sediments, and ground water (Mei, Puryear, & Newton, 2002). Chemical speciation plays an influential role in solubility and potential bioavailability of heavy metal in soils (Wong et al., 2002). Cr (III) and Cr (VI) are the major stable chemical forms in the environment (Cary, Allaway, & Olson, 1977b) with Cr (VI) causing the greatest concern because of its move, toxic, carcinogenic properties (Wolterbeek & Van der Meer, 2002). High significant correlation is observed between total Cr and Fe ($r = 1.00$). This suggests association of Cr with amorphous iron

oxides (Moalla, 1990), also reflecting the occurrence of most sediment Cr in highly resistant minerals.

Ni and Co concentrations are significantly high in fractions bound to Fe/Mn oxide. This consistent with their known ability to be present in high concentrations in ferromanganese nodules. Cobalt is an essential micro-nutrient for animals.

Concerning distribution of Co and Ni in the studied bights it is obvious that the relatively higher concentrations are detected in eastern bight (El-Sage). A statistical evaluation of the data shows high significant correlation between available Mn content and Ni bound to Fe/Mn oxide fraction. This may be attributed to association of Ni with Fe/Mn oxides. Excellent positive correlations are observed between available Ni concentration and iron ($r = 1.00$). This can be consistent to the co-precipitation of Ni and Fe sulphides (Moalla et al., 1998).

Horizontal contribution of Co reflects a very weak relationship with Mn, and means that the manganese is not the only geochemical support phase of Co.

Spatial variation of measured metal along the study area reflects remarkable increases in Pb and Cd concentrations at west bights. This may be caused by discharges of tourist ships and waste from the tourism village built at the western bank of the lake (El-Caria bight).

Some heavy metals are required for life; but excessive accumulation in living organisms is always toxic. The danger of heavy metals is aggravated by their almost indefinite persistence in the environment. For example, lead, which is one of the more persistent metals, was estimated to have a soil retention time 150 to 5000 years (Kumar, Dushenkov, Motto, & Raskin, 1995).

Cd and Pb may cause serious problems through food chains (Chen, Lee, & Liu, 2000). The potential bioavailability fraction of Pb is generally accounted for 5.2% of total sediment Pb. A positive correlation coefficient is found between total lead content and iron bound to Fe/Mn oxide fraction ($r = 0.990$), supported by the fact that Pb is strongly associated with ferromanganese minerals (Moalla et al., 1998). Cadmium is known as more mobile and soluble than other metals in sediment samples, but Pb is well known to be relatively immobile (Chen et al., 2000).

Data analyzed focused on Zn concentration as highest trace metal contents in sediment samples against all studied sites. Zinc is one of the most abundant of the essential elements required by the human body and approximately 100 times as abundant as copper. Total Zn concentration in analyzed samples ranged from $491.4 \mu\text{g g}^{-1}$ to $629.4 \mu\text{g g}^{-1}$ dry wt.

Heavy metals reaching the lakes are derived by biological or physicochemical process. Shata et al. (1993) found that adsorption of dissolved zinc onto suspended sediment was achieved within 1 hour. This incorporation of trace elements into bottom sediments of Nasser Lake undoubtedly involves high organic matter and considerable amount of carbonate. The present study agrees with these results, evidenced by the good positive correlation of Zn on one hand, and organic

matter ($r = 0.999$), and highly significant correlation coefficient with carbonates ($r = 1.000$) on the other hand.

The average of available Zn content and that bound to Fe/Mn oxide fraction ranged between 8.4-21.13 $\mu\text{g g}^{-1}$ and 100.9-160 $\mu\text{g g}^{-1}$, respectively. The percent Zn distribution in Fe/Mn oxide fraction (24.6% of total Zn sediment) is generally significant. This is supported by highly significant correlation between Zn and Mn in this fraction ($r = 1.000$), and this relationship can be explained by association of Zn with Mn (co-precipitation or adsorption on a Mn oxide or hydroxide). Positive correlation is found between total Zn and Mn ($r = 0.990$). This means Mn is the only geochemical support phase of Zn (Bertin & Bourg, 1995).

The highest Zn value is recorded in sediment samples of eastern bight (Figure 2) which is characterized by relatively high organic matter considerable carbonates content (Figure 3). These results fall in line with those reported by Neumann, Leipe, and Shimmiel (1998) for sediments of the Oder River. They pointed out that heavy metals with high enrichments in the surficial sediments, such as Zn, Cu, and Pb show strong correlations with organic matter throughout the study area. A statistical evaluation of the data shows excellent correlations between total Zn content and Mg bond to Fe/Mn fraction ($r = 1.000$) and carbonate ($r = 1.000$). This may reflect the strong association with Mg carbonate minerals. No observed correlations between pH and measured trace metals (Co, Cu, Pb, Cr, Zn) indicate that the adsorption of these metals is weakly pH dependent (Soltan et al., 2000). The analytical results of sediment samples were compared and showed that levels of available Na and K never exceed 66.5, 16.9 $\mu\text{g g}^{-1}$ or around 7.3% and 7.6 of total Na and K sediment, respectively, while amounts that bond to Fe/Mn oxide fractions recorded about 12.7% and 11% of total Na and K sediment. High concentrations of Na and K in this fraction may be consistent with enrichment of these metals in ferromanganesian minerals (Fawzy, 2000).

A statistical analysis of the database shows negative significant correlation values between available Na and K and pH ($r = -1.000$, $r = -0.995$, respectively). This reflects similar adsorption behavior for these metals, whereas high adsorption capacity was obtained at pH <6; when pH value increase, it causes hydrolysis and sedimenting of metals (Soltan et al., 2000). Negative correlations arise in the different extracts among the extractable Na and the corresponding metals (available Na and total Cr content $r = -0.999$ and total Na-concentration and available Co $r = -0.991$). This is possibly attributable to the presence of Na-bearing silicate minerals with low metal contents.

High significant correlation coefficient between total K and Pb ($r = 1.000$). This is considered with adsorption of Pb on potassium rich minerals. This is consistent with the fact that Pb^{2+} can be incorporated into potassium-rich minerals by replacement of potassium having a similar ionic radius of about 1.3Å^o (Moalla, 1990).

As a general trend, Ca and Mg concentrations show significantly high amounts of these metals present in fractions bound to Fe/Mn oxide (Figure 2). The

mean values of available Ca and Mg in surficial sediment samples were recorded 29.4% and 18.9% of total Ca and Mg sediment. Considerable amounts of Ca and Mg were leaching by $\text{NH}_2\text{OH}\cdot\text{HCl}$ (42.6% and 67.2% of total Ca and Mg sediment, respectively). This suggests that a high abundance of these two elements are present in Fe/Mn oxide fraction, which may explain the perfect positive correlation between Ca in different extracts and available iron ($r = 0.994$) and total iron content ($r = 1.000$), also that arise between Mg and total Mn concentration ($r = 0.998$).

Perfect positive correlations are observed between available Ca concentration on one hand and Ni ($r = 0.994$), K ($r = 0.999$), and Pb ($r = 0.999$) in different extracts on the other hand. Studies on bottom sediments have indicated that appreciable quantities of trace metals may be associated with sedimentary carbonates, either as separate phases or in structural positions within calcium or magnesium carbonates (Moalla, 1990). Statistical evaluation revealing positive correlation between total Mg and Cu ($r = 0.990$) indicates that accumulation of these metals has the same rate. Mackereth (1966) considered that most of the Mg in the aquatic sediment exists in the clay minerals and is transported from the catchments to the lake in soiled form.

A good correlation is observed between total Mg content and available Co concentration ($r = 0.993$). This is consistent with the fact that some metals such as Co are strongly enriched with Mg in ultramafic rocks (e.g., pyroxene and olivine).

Positive correlations are recorded between Mg in fraction bound to Fe/Mn oxide and total Mn and Zn ($r = 0.998, 1.000$, respectively). This may be interpreted in terms of accommodation of these elements with Mg-bearing minerals. Statistical analysis of trace metals gave significant positive correlations between iron pore water content and Mn concentration in surface water ($r = 0.993$) and bottom water Cu content ($r = 0.993$). This is consistent in that iron in pore waters was very likely present as iron oxhydroxides, and the physicochemical properties of iron oxyhydroxides are known to control the solubility of Cu in many aquatic environments (Chabbi, 2003). A similar pattern is also observed between Mn content in pore water samples and surface water Zn concentration ($r = 1.000$), which can be explained by the fact that the release of trace metals is clearly connected with Fe and Mn peaks in the pore water (Van Ryssen et al., 1999).

The free metal in the pore water is generally thought to show the greatest toxicity, so it is important to understand binding dynamics, controlling conditions and sorption/desorption properties (Van Ryssen et al., 1999). According to the concentration gradients at the sediment water, interface release of metals from the pore water to the surface water can occur. This is supported by highly significant positive correlations between pore water Pb content and surface water iron concentration ($r = 1.000$), also between surface water Ni concentration and Ni, Ca, Mg, Na, and K concentration extractable of pore water ($r = 1.000, 0.992, 0.989, 0.989, 0.998, 0.998$, respectively).

Positive correlations arise between sulphate content of bottom water and some pore water metals, such as Ni ($r = 1.000$), Ca ($r = 0.992$), Mg ($r = 0.989$), Na ($r = 0.998$), and K ($r = 0.988$). Dissolution of iron and Mn oxide under anaerobic conditions occurs with subsequent re-precipitation in more toxic water. This re-precipitation of colloidal iron and manganese oxide particles would scavenge trace metals present in the water column by adsorption and co-precipitation (Förstner & Wittman, 1981). This is in agreement with the negative correlation between iron concentration extractable of pore water and concentration of Ni in surface water ($r = -0.993$).

Negative correlations arise between water pH value and different metal concentrations Co ($r = -0.990$), Cd ($r = -1.000$). This relation indicates similar adsorption behavior for these metals whereas high adsorption capacity was obtained at $\text{pH} < 6$. According to Prokop et al. (2003), Cd and Zn tend to pass into solution at pH values lower than 4, but when the pH value increases, it causes hydrolysis and sedimenting of metals (Soltan et al., 2000). Analyzed data show a highly positive correlation between phosphate contents and TDS concentration ($r = 1.000$), which suggested that the intensity of light enhances growth of plants and reduces the concentrations of phosphate in water column (Soltan, 1991).

Positive correlation exists between Ca and Mg concentrations ($r = 0.995$) of surface water, suggesting that the two elements have similar transport mechanisms (Fawzy, 2000).

The statistical analysis shows negative correlations (up to $r = -0.997$) between trace metals and DO, pH, and CO_3^{2-} . This is due to the trace metals having low solubility under oxidizing conditions (Soltan, 1995). Increased biological activity of aquatic plants and consumption of silicate by growth of phytoplankton blooms and diatoms removes greatest amounts of Ca and Mg. This is supported by highly negative correlation coefficient ($r = -0.989, -0.995$) between SiO_2 and both Mg and Cu, respectively.

CONCLUSION

The study has provided a comparison of metal concentrations in sediment pore water water samples of some bights of Nasser Lake. In general it can be concluded that:

- Significant positive correlations between most measured metals in water, sediment, and pore water point to the possibility of transfer of some metals to associated water.
- Low exchangeable metal concentrations of sediment samples reflect the decrease of pore water contents, wherefore, the water body is characterized by low metal concentrations. The alkaline pH values of bight water minimizes

the reflux of metal ions from lake sediment and the increase of metal sedimentation rate.

- Overall data suggest that Cd and Pb are less available metals.
- Chemical partitioning of heavy and major metals in pore water samples is much higher than in the surface and bottom water of the lake. This suggests that water flow through the sediment can reduce the actual toxicity of the upper layer of deposited sediment but at the same time can intensify the risk of fresh water contamination.
- The study examined the distribution of different metals in three extractant solutions of sediment and reported that single sequential extraction analysis reveals a strong association of measured metals to Fe/Mn oxide fractions.
- Cadmium and chromium show relative ability to distribute between exchangeable form and fractions bound to Fe/Mn oxides.
- Results of trace and toxic metal concentrations measured in this study for all components are within the allowed and desired safety baseline levels and consistent with local and international requirements.
- Due to the locality of Nasser Lake as the important source for Egyptian life, further studies are needed for the evaluation of water quality.

REFERENCES

- Alaska Drinking Water Standards. (1994). *Alaska Department of Environmental Conservation drinking water regulations*. State of Alaska, Department of Environmental Conservation Report 18-ACC-80.
- Allen, H. E. (2001). Introduction. In H. E. Allen (Ed.), *Bioavailability of metals in terrestrial ecosystems: Importance of partitioning for bioavailability to invertebrates, microbes and plants*. Pensacola, FL: SETAC Press.
- Allen, S. E. (1989). *Chemical analysis of ecological materials*. Oxford: Blackwell Scientific Publications.
- Ankley, G. T., Leonard, E. N., & Mattson, V. R. (1994). Prediction of bioaccumulation of metals from contaminated sediments by the oligochaete, *lumbricus variegatus*. *Water Research*, 28, 1071-1076.
- Ankley, G. T., Di Toro, D. M., Hansen, D. I., & Berry, W. I. (1996). Technical basis and proposal for deriving sediment quality criteria for metals. *Environmental Toxicology Chemistry*, 15, 2056-2066.
- American Public Health Association (APHA). (1992). *Standard methods for the examination of water and wastewater* (18th ed.). Washington, DC: APHA.
- Awadallah, R. M., Ismail, S. S., Abd El Aal, M. T., & Soltan, M. E. (1993). *Investigation of Drinking and Nile Water Samples of Upper Egypt*, 19(3), 217-230.
- Awadallah, R. M., Ismail, S. S., Arifien, A. E., Moalla, S. M., & Grass, F. (1994). Distribution of trace elements in mud sediments of the Aswan High Dame Lake. *Chemistry Erde*, 54, 67-82.
- Balkis, N., & Cagatay, M. N. (2001). Factors controlling metal distributions in the surface sediments of the Erdek Bay, Sea of Marmara, Turkey. *Environment International*, 27, 1-13.

- Baruah, T. C., & Barthakur, H. P. (1997). *A textbook of soil analysis*. Delhi: Vikas Publishing House.
- Berry, W. J., Hansen, D. J., Mahony, J. D., Robson, D. L., Di Toro, D. M., Shipley, B. P., Rogers, B., Corbin, J. M., & Boothman, W. S. (1996). Predicting the toxicity of metalspiked laboratory sediments using acid volatile sulphide and interstitial water normalizations. *Environmental Toxicology Chemistry*, *15*, 2067-2079.
- Bertin, C., & Bourg, A. C. M. (1995). Trends in the heavy metal content (Cd, Pb, Zn) of river sediments in the drainage basin of smelting activities. *Water Research*, *29*, 1729-1736.
- Boughriet, A., Ouddanc, B., Fischer, J. C., Wartel, M., & Leman, G. (1992). Variability of dissolved Mn and Zn in the Seine estuary and chemical speciation of these metals in suspended matter. *Water Research*, *26*, 1359-1378.
- Bowen, H. J. M. (1979). *Environmental chemistry of the elements*. New York: Academic Press.
- Calmano, W., Hong, J., & Förstner, U. (1993). Binding and mobilization of heavy metals in contaminated sediments affected by pH redox potential. *Water Science Technology*, *28*(8-9), 223-235.
- Cary, E. E., Allaway, W. H., & Olson, O. E. (1977). 1. Control of chromium concentrations in food plants. 2. Chemistry of chromium in soils and its availability to plants. *Journal of Agricultural Food Chemistry*, *25*(2), 305-309.
- Chabbi, A. (2003). Metal concentrations in pore water of the Lusatian lignite mining sediments and internal metal distribution in juncus bulbosus. *Water, Air, and Soil Pollution*, *3*, 105-117.
- Chen, Z. S., Lee, G. J., & Liu, J. C. (2000). The effects of chemical remediation treatments on the extractability and speciation of cadmium and lead in contaminated soils. *Chemosphere*, *41*, 235-242.
- Chester, R., Thomas, A., Lin, F. J., Basaham, A. S., & Jacinto, G. (1988). The solid state speciation of copper in surface water particulates and oceanic sediments. *Marine Chemistry*, *24*, 261-292.
- da Silva, I. S., Abate, G., Lichtig, J., & Masini, J. C. (2002). Heavy metal distribution in recent sediments of the Tietê-Pinheiros river system in São Paulo state, Brazil. *Applied Geochemistry*, *17*, 105-116.
- Dewitt, T. H., Swartz, R. C., Hansen, D. J., McGovern, D., & Berry, W. J. (1996). Bioavailability and chronic toxicity of cadmium in sediment to the estuarine amphipod leptoch eirus plumulosus. *Environmental Toxicology Chemistry*, *15*, 2095-2101.
- Egyptian Chemical Standard (ECS). (1994). Law No. 4, protection of the Nile River and water stream from pollution. Cairo, Egypt: Ministry of Irrigation.
- Elewa, A. S. (1985). *Bulletin NRC, Egypt*, *19*(3), 225.
- Environmental Protection Agency (EPA) Water Quality Standards. (1992). Establishment of numeric criteria for priority toxic pollutants; State's compliance, final rule. *Federal Register*, *40 CFR part 131*, *57*(246), 60847-60916.
- Fawzy, E. M. (2000). Physico-chemical studies on the speciation of heavy elements in the River Nile sediments. M.Sc. thesis Faculty of Science, South Valley University, Egypt.
- Förstner, U., & Salomons, W. (1980). Trace metal analysis on polluted sediments. 1. Assessment of sources and intensities. *Environmental Technology Letters*, *1*, 494-505.

- Förstner, U., & Wittmann, G. T. W. (1979). *Metal pollution in the aquatic environment* (486 pp.). Berlin: Springer-Verlag.
- Förstner, U., & Wittman, G. T. W. (1981). *Metal pollution in the aquatic environment*. Berlin: Springer.
- Fu, G., Allen, H. E., & Cao, Y. (1992). The importance of humic acids to proton and cadmium binding in sediments. *Environmental Toxicology Chemistry*, *11*, 1363-1372.
- Goodwin, M. H., & Goodard, C. I. (1974). An inexpensive multiple level of water sampler. *Canadian Journal of the Fisheries Research Board of Canada*, *31*, 1667-1668.
- Grant, C. A., Buckley, W. T., Bailey, L. D., & Selles, F. (1998). Cadmium accumulation in crops Canada. *Journal of Plant Science*, *78*, 1-17.
- Gupta, S. K., & Chen, K. Y. (1975). Partitioning of trace metals in selective chemical fractions of near-shore sediments. *Environmental Letters*, *10*, 129-158.
- Haiyan, W., & Stuanes, A. O. (2003). Heavy metal pollution in air-water-soil-plant system of Zhuzhou City, Hunan Province, China, *147*, 79-107.
- Hansen, D. J., Berry, W. J., Mahony, J. D. Bothman, W. S., Di Toro, D. M., Robson, D. L., Ankley, G. T., Ma, D., & Pesch, C. E. (1996). Predicting the toxicity of metal contaminated field sediments using interstitial concentration of metals and acid-volatile sulfide normalizations, environmental toxicology. *Chemistry*, *15*, 2080-2094.
- Inkonnikov, V. (1993). Pollution of sediments in the lake Ladoga catchment area and the Gulf of Finland influencing factors and processes. *Land Degradation and Rehabilitation*, *4*, 297-306.
- Jenne, E. A. (1968). Controls on Mn, Fe, Co, Ni, Cu and Zn concentrations in soils and water: The significant role of hydrous Mn and Fe oxides. In R. F. Gould (Ed.), *Trace inorganics in water*. Advances in Chemistry Series No. 73, American Chemistry Society, 337-387.
- Jonnalagadda, S. B., & Mhere, G. (2001). Water quality of the Odzi River in the eastern Highlands of Zimbabwe. *Water Research*, *35*(10), 2371-2376.
- Kabata-Pendias, A., & Pendias, H. (1992). *Trace elements in soils and plants* (2nd ed.). Boca Raton, FL: CRC Press.
- Klavins, M., Briede, A., Rodinov, V., Kokorite, I., Parele, E., & Klavina, I. (2000). Heavy metals in rivers of Latvia. *Science Total Environmental*, *262*, 175-183.
- Kotas, J., & Stasicka, Z. (2000). Chromium occurrence in the environment and methods of its speciation. *Environmental Pollution*, *107*, 263-283.
- Kumar, P. B. A. N., Dushenkov, V., Motto, H., & Raskin, I. (1995). Phytoextraction: The use of plants to remove heavy metals from soils. *Environmental Science & Technology*, *29*, 1232-1238.
- Leonard, E. N., Ankley, G. T., & Hoke, R. A. (1996). Evaluation of metals in marine and freshwater surficial sediments from the environmental monitoring and assessment program relative to proposed sediment quality criteria for metals. *Environmental Toxicology Chemistry*, *15*, 2221-2232.
- Lokke, H., & Van Gestel, C. A. M. (1998). *Handbook of soil invertebrate toxicity tests*. Chichester, England: Wiley.
- Mackereth, F. J. H. (1966). Some chemical observation on post-glacial lake sediments philos. *Transactions of the Royal Society, London Ser. B*, *250*, 165-213.
- Mei, B., Puryear, J. D., & Newton, R. J. (2002). Assessment of Cr tolerance and accumulation in selected plant species. *Plant and Soil*, *247*, 223-231.

- Moalla, S. M. N., Awadallah, R. M., Rashed, M. N., & Soltan, M. E. (1998). Distribution and chemical fractionation of some heavy metals in bottom sediments of Lake Nasser. *Hydrobiologia*, *364*, 31-40.
- Moalla, S. M. N. (1990). *Determination of some trace elements in the sediments of the High Dame Lake*. Ph.D. thesis, Faculty of Science, Assiut University, Egypt.
- Neumann, T., Leipe, T., & Shimmield, G. (1998). Heavy metal in reachment in surficial sediments in the Odder River discharge area: Source or sink for heavy metal. *Applied Geochemistry*, *13*, 329-337.
- Nürnberg, H. W. (1984). The Voltammetric approach in trace metal chemistry of natural waters and atmospheric precipitation. *Analytical Chemistry Acta*, *164*, 1-21.
- Okazaki, M. (1987). Behaviour of heavy metals in soils. *Japan Journal Water Pollution Research*, *10*, 407-412. (in Japanese)
- Pardo, R., Barrado, E., Perez, L., & Vega, M. (1990). Determination and speciation of heavy metal in sediments of the Pisuerga River. *Water Research*, *24*, 337-343.
- Peijnenburg, W. J. G. M., Posthuma, L., Zweers, P. G. R. C., Baerselman, R., De Groot, A. C., Van Veen, R. P. M., & Jager, T. (1999a). Prediction of metal bioavailability in Dutch field soils for the oligochaete enchytraeus crypticus. *Ecotoxicology Environment Safety*, *43*, 170-186.
- Peijnenburg, W. J. G. M., Baerselman, R., De Groot, A. C., Jager, T., Posthuma, L., & Van Veen, R. P. M. (1999b). Relating environmental availability to bioavailability: Soil-type dependent metal accumulation in the Oligochaete Eisenia Andrei. *Ecotoxicology Environment Safety*, *44*, 294-310.
- Pesch, C. E., Hansen, D. J., Boothman, W. S., Berry, W. J., & Mahony, J. D. (1995). The role of acid-volatile sulfide and interstitial water metal concentrations in determining bioavailability of cadmium and nickel from contaminated sediments to the marine polychaete neanthes arenaceodentata. *Environmental Toxicology Chemistry*, *14*, 129-141.
- Plette, A. C., Netherhof, M. M., Temminghoff, E. H. M., & Van Riemsdijk, W. H. (1999). Bioavailability of heavy metals in terrestrial and aquatic systems: A quantitative approach. *Environmental Toxicology Chemistry*, *18*, 882-890.
- Prokop, Z., Vangheluwe, M. L., Van Sprang, P. A., Janssen, C. R., & Holoubek, I. (2003). Mobility and toxicity of metals in sandy sediments deposited on land. *Ecotoxicology and Environmental Safety*, *54*, 65-73.
- Purves, D. (1985). *Trace element contamination of the environment*. Amsterdam: Elsevier.
- Rashed, M. N. (2001). Monitoring of environmental heavy metals in fish from Nasser Lake. *Environment International*, *27*, 27-33.
- Roy, T. T., Sarbadhikary, S. B., Guha, A. K., Mandal, B. M., & Bhattacharyya, S. N. (1996, September). On the estimation of biological oxygen demand of collected water samples with special reference to that of the Ganges Water. *Journal Indian Chemical Society*, *73*, 472-475.
- Roy, R. L., & Campbell, P. G. C. (1997). Decreased toxicity of Al to Juvenile Atlantic Salmon (*Salmo Salar*) in acidic soft water containing natural organic matter: A test of the free-ion model. *Environmental Toxicology Chemistry*, *16*, 1962-1969.
- Saeki, K., Okazaki, M. N., & Matsumoto, S. (1993). The chemical phase changes in heavy metals with drying and oxidation of the lake sediments. *Water Research*, *27*(7), 1243-1251.

- Salomons, W., & Förstner, U. (1980). Trace metal analysis on polluted sediments. II: Evaluation of environmental impact. *Environmental Technology Letters*, 1, 506-525.
- Salomons, W., De Rooij, N. M., Kerdijk, H., & Bril, J. (1987). Sediments as a source for contaminants. *Hydrobiologia*, 13, 30-38.
- Shata, M. A., El-Deek, M. S., & Okbah, M. A. (1993). Fractionation of Mn, Fe, Zn and Cu in sediments of Khor Kalabsha, Lake Nasser, Egypt. *Chemistry and Ecology*, 8, 89-103.
- Shuman, L. M. (1985). Fractionation method for soil microelement. *Soil Science*, 140, 11-22.
- Soltan, M. E., & Rashed, M. N. (2003). Laboratory study on the survival of water hyacinth under several conditions of heavy metal concentrations. *International Journal of Advances in Environmental Research*, 7, 321-334.
- Soltan, M. E. (1988). *Study of River Nile water pollution*. M.Sc. thesis, Faculty of Science Assiut University, Egypt.
- Soltan, M. E. (1991). *Study of River Nile water pollution*. Ph.D. thesis, Faculty of Science, Assiut University, Egypt.
- Soltan, M. E., Rashed, M. N., & Taha, G. M. (2000). Heavy metal levels and adsorption capacity of Nile River sediments. *International Journal of Environmental Analytical Chemistry*, 80(3), 167-186.
- Stumm, W., & Morgan, J. J. (1981). *Aquatic chemistry* (780 pp.). New York: Wiley.
- Sullivan, P., & Taylor, K. (2003). Sediment and pore water geochemistry in a metal contaminated estuary, Dulas Bay, Anglesey. *Environmental Geochemistry and Health*, 25, 115-122.
- Tack, F. M. G., Singh, S. P., & Verloo, M. G. (1998). Heavy metal concentrations in consecutive saturation extracts of dredged sediment derived surface soils. *Environmental Pollution*, 103, 109-115.
- Tessier, A., Campbell, P. G. C., & Bisson, M. (1980). Trace metal speciation in the Yamaska and St. Francois River (Quebec). *Canadian Journal of Earth Science*, 17, 90-105.
- Turekian, K. K., & Wedephol, K. H. (1961). *Bulletin of Geographic Society of America*, 72, 175-192.
- Van den Berg, G. A., Loch, J. P. G., Van der Heijdt, L. M., & Zwolsman, J. J. G. (1998). Vertical distribution of acid-volatile sulfide and simultaneously extracted metals in a recent sedimentation area of the River Meuse in the Netherlands. *Environmental Toxicology Chemistry*, 17, 758-763.
- Van Ryssen, R., Leermakers, M., & Baeyens, W. (1999). The mobilisation potential of trace metals in aquatic sediments as a tool for sediment quality classification. *Environmental Science & Policy*, 2, 75-86.
- Vijver, M., Jager, T., Posthuma, L., & Peijnenburg, W. (2003). Metal uptake from soils and soil-sediment mixtures by Larvae of *Tenebrio Molitor* (L) (Coleoptera). *Ecotoxicology and Environmental Safety*, 54, 277-289.
- Waller, P. A., & Pickering, W. F. (1992). Effect of time and pH on the lability of copper and zinc absorbed on humic acid particles. *Chemical Speciation and Bioavailability*, 4, 29-41.
- Warran, L. A., & Zimmerman, A. P. (1992). Trace metal suspended particulate matter associations in a fluvial system: Physical and chemical influences. In S. S. Rao (Ed.), *Particulate matter and aquatic contaminants* (pp. 127-155). Chelsea, MI: Lewis Publishers.

- Wedephol, K. H. (1968). *Origin and distribution of elements* (999 pp.). L. R. Ahrens (Ed.). Oxford: Pergamon Press.
- World Health Organization (WHO). (1993). *Guideline for drinking water quality* (2nd ed.), Geneva: WHO.
- Wolterbeek, H. Th., & Van der Meer, A. J. G. M. (2002). Transport rate of arsenic, cadmium, copper and zinc in *Potamogeton Pectinatus* L.: Radiotracer experiments with ^{76}As , $^{109, 115}\text{Cd}$, ^{64}Cu and $^{65, 69}\text{Zn}$. *The Science of The Total Environment*, 287, 13-30.
- Wong, S. C., Li, X. D., Zhang, G., Qi, S. H., & Min, Y. S. (2002). Heavy metals in agricultural soils of the Pearl River Delta, South China. *Environmental Pollution*, 119, 33-44.
- Yu, K., Tsai, L., Chen, S. H., & Ho, S. T. (2001). Chemical binding of heavy metals in Anoxic River sediments. *Water Research*, 35(17), 4086-4094.
- Zayed, M. A., Nour El-Dien, F. A., & Rabie, K. A. A. (1995). Seasonal variation of water quality of Nile River at Helwan area. *Journal Environmental Science Health*, A30(9), 1957-1972.
- Zhang, L., Zhang, S., & Dong, W. (1986). Main geochemical factors affecting heavy metal transformation in the Xiang River. *Journal of Science*, 6(4), 395-402. (in Chinese)

Direct reprint requests to:

M. E. Soltan
Chemistry Department
Faculty of Science
Aswan, South Valley University
Egypt
e-mail: mesoltan@hotmail.com