BEHAVIOUR OF SOME HEAVY METALS IN WATER-SEDIMENT SYSTEM OF EL-SALAM CANAL AS AFFECTED BY ADDED MnO_2

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ABSTRACT: Mn and Fe oxides are one of the most soil and sediment components affecting various chemical redox reactions under anaerobic conditions. The effect of MnO_2 on the behaviour of some heavy metals was studied in sediment samples taken from El-Salam Canal. It is shown that all sediment suspensions reach reducing conditions after a few days of submergence. Addition of MnO_2 importantly enhanced the removal of As, Cu, Ni and Pb from the overlaying water particularly at the first period of waterlogging. This effect indicates the possible role of some native sediment components such as MnO_2 on the removal of some heavy metals through some redox chemical reactions which can convert such metals to insoluble or immobile forms which reduce their environmental hazard effect.

INTRODUCTION

El-Salam Canal is one of the national promising projects involves the reuse of drainage waters. Bahr Hadous drain (1.905 billion m^3) and El-Serw drain (0.435 billion m^3) are the main sources for the water of the canal in addition to Nile water (2.11 billion m^3) (DRI, 1993).
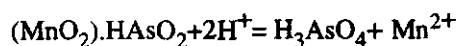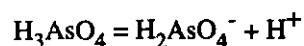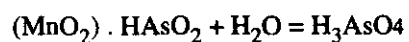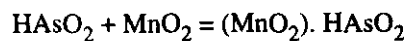
The canal extends through deposits of varying modes of formation including fluviatile, marine, lacustrine and aeolion

(Said, 1990). Its total length is about 242 km divided into two parts viz., 87 and 155 km in western and eastern banks of Suez Canal, respectively.

About 400.000 feddans in North Sinai will be reclamined using El-Salam Canal water, (JICA, 1996). Heavy metal content in the water could be a limiting factor for the reuse of drainage water for agriculture. In this respect, Shawky et al., (1994) found that the water of Bahr Hadous drain contains high levels of Cu, Pb, Cd and Co. However, Farag and Mehana (2000) found that the concentration of heavy metals (Fe, Mn, Zn, Cu, Pb, Cd, Co, Ni) in El-Salam Canal water are much lower than their maximum permissible limits for irrigation water. The role of transition metal oxide/ hydroxide minerals such as Mn oxide, in rodex reactions in soils and aqueous sediments is pronounced (Stumm and Morgan, 1970 ; Oscarson et al., 1981). Mn oxide, occurs as suspended particles, in surface water and as coatings on soil and sediments. Mn (III/IV) can play an important role in soils and sediments via oxidation of inorganic cations. Both advantageous and deleterious effects to environmental quality can be involved. The positive side, involves oxidation of arsenite (As III) to arsenate (As V) by Mn (III/IV) oxides (Oscarson et al., 1980). The negative side is oxidation of chromic (Cr III) to chromyl (Cr VI) which is very mobile in soils and sediments (Amacher and Baker, 1982). As (V) can be reduced to As (III) by bacteria and phytoplankton (Andreae and Klumpp, 1979).. Johnson and Pilson (1975) concluded that As (III) could be oxided to As (V) under sea water conditions. Mn (III/IV) oxides play an important role in the abiotic oxidation of As (III) to As (V) (Oscarson et al., 1980 & 1981). This is very vital effect because As (III) is very toxic pollutant and more soluble

and mobile (Deuel and Swoboda, 1972) than As (V). When As (III) added to MnO_2, it can be oxidized to As (V) (Oscarson et al., 1981) or sorbed on the oxide surface. The oxidation of As (III) by MnO_2 can be shown by Oscarson et al., (1983) as follows :

$$HAsO_2 + MnO_2 = (MnO_2). HAsO_2$$

$$(MnO_2) . HAsO_2 + H_2O = H_3AsO4$$

$$H_3AsO_4 = H_2AsO_4^- + H^+$$

$$H_2AsO_4^- = HAsO_4^{2-} + H^+$$

$$(MnO_2).HAsO_2+2H^+= H_3AsO_4+ Mn^{2+}$$

Each mole of As (III) oxidized releases about 1.5 moles H^+ into the system. pH will be lowered but around ~ 7.0. Each mole of As (III) oxidized to As (V) results in a mole of Mn (IV) to be reduced to Mn (II) which can be partially dissolved in solution (Oscarson et al., 1981).

On the other hand, manganese oxide, and hydroxide, may also catalyze the oxidation of other heavy metals such as Cu^{2+}, Ni^{2+}, and Pb^{2+} by disproportionation to Mn^{2+} and MnO_2 (Hem, 1978). The disproportionation results in vacancies in the Mn oxide structure. Since the Mn^{2+} and Mn^{3+} in the oxides, have similar physical size, as Cu^{2+}, Ni^{2+} and Pb^{2+}, these metals can occupy the vacancies in the Mn oxide and became part of the structure, with disproportianation or with other redox processes involving Mn oxide, the solubility of the metals can be affected.

The aim of the current study is to investigate the effect of Mn oxide on the behavior of As (III), Cu (II), Ni (II) and Pb (II) in the sediments of El-Salam Canal.

MATERIALS AND METHODS

Five sediment samples and five water samples were collected from Bahr Hadous Drain intersection point with El-Salam Canal. The sampling was repeated five Km far along. The sediment samples were air dried and crushed to pass a 2 mm sieve before use. The samples were analysed for total form of Fe,

Mn, Cu, Pb, Ni and As after digested in acid mixtures (Chapman and Pratt, 1961). pH of the samples was determined using a standard method. Some physical and chemical properties of the sediment samples are presented in Table 1. Some chemical characteristics of the water samples are shown in Table 2.

All experiments were carried out on soil suspensions made up by putting 50 g of sediment and 140 ml of deionized water in 250 ml conical flasks. All flasks were received 10 ml solutions in order to add 5 μgl^{-1} from As^{3+}, Cu^{2+}, Ni^{2+}, Pb^{2+} using $NaAsO_2$, $CuSO_4$. $5H_2O$, $NiSO_4$. $6H_2O$ and Pb $(CH_3 COO)_2$. As such sediment to water suspensions of 1/3 was obtained.

The experiment carried out in ten flasks, half of them treated with 0.2% freshly prepared MnO_2 (birnessite) (Mckenzie, 1971) while the rest were left without treating. Each treatment was repeated 3 times. All flasks were flushed with nitrogen gas and continuously stirred for half an hour at the begining of the experiment. After that, all flasks were incubated under closed system conditions at 20°C $\pm$ 1.0°C. There were a set of flasks for every period of determination.

After 0, 2, 4, 6, 8, 10, 12 and 14 days, redox potential (Eh) and pH were measured using platinum and glass electrodes, respectively while a calomel electrode was used as a reference. Suspensions were centrifugated and filtered and the filtrates were analysed for Fe, Mn, As, Cu, Ni and Pb according to standard methods.

RESULTS AND DISCUSSION

Redox potential evolution

The rodex potential (Eh) is one of the most important sediment physico-chemical parameters expressing its reducing power.

The evolution of the rodex potential measured at intervals following submergance is given

Table (1) : Some physical and chemical properties of the sediments used

No.	Texture	CaCO₃ (g kg⁻¹)	Organic Matter (g kg⁻¹)	Total N (g kg⁻¹)	CEC (c mol kg⁻¹)	pH (1 : 25)	Fe₂O₃ (g kg⁻¹)	Total Fe %	Total Mn (mgkg⁻¹)	Total heavy metals(mg kg⁻¹)			
										As	Cu	Ni	Pb
1	Clay	29.6	1.9	1.4	35.2	7.78	14.6	6.62	1095	2.90	32.1	18.2	28.2
2	Clay	30.1	1.3	1.1	35.7	7.76	13.9	6.58	1036	2.62	30.6	18.1	28.3
3	Clay	27.8	1.3	1.0	35.4	7.79	13.1	6.67	1012	2.33	29.2	17.6	27.7
4	Clay	31.0	1.0	1.2	34.9	7.72	12.8	6.66	1003	2.21	29.7	17.8	27.9
5	Clay	21.9	1.8	1.3	33.8	7.71	11.9	6.41	1006	2.06	29.9	17.7	26.3

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Table (2): EC (dSm^{-1}), pH and concentrations of total heavy metals (μgl^{-1}) in water samples at different studied locations along El-Salam Canal.

Sample No.	EC dSm^{-1}	pH	Heavy metals concentration μg l^{-1}					
			Fe	Mn	As	Cu	Ni	Pb
1	1.22	7.83	1832	165.2	7.6	19.7	33.7	22.4
2	1.20	7.79	1522	152.1	6.2	15.1	27.6	23.12
3	1.26	7.95	1493	150.3	5.8	13.1	28.2	17.60
4	1.18	7.86	1309	132.1	4.1	11.4	25.1	10.90
5	1.23	7.82	1220	121.7	3.8	11.8	19.6	8.60

in Table (3). Data reveal that redox potential rapidly decreased, reaching reducing conditions within a few days. A rapid decline was associated with the non MnO$_2$ treated samples reaching Eh values of 0.0 to -27mV comparing to 10 to -12 mV for sediment samples, treated with MnO$_2$ after 2 days of flooding. Atta and Cleemput (1981) found that a rapid decline in redox potential is characteristic of soils with a low content of reducible iron and manganese and a high content of easily oxidizable organic matter. It is note worthy that although the relatively high content of the native manganese in the five sediment samples (Table 1), but it could be mostly subjected to conversion to well crystalline form via aging processes, hence its reduction under such conditions could be lower than the freshly precipitated form. The former is characterized by its

Table (3): Evolution of sediment suspension Eh (mV) as affected by MnO2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	300	90	0.0	-72	-123	-153	-205	-220	-230
	0.2	315	100	10.0	-63	-100	-142	-182	-193	-197
2	0.0	305	82	-21	-83	-127	-162	-209	-230	-234
	0.2	310	93	-12	-61	-105	-143	-178	-196	-199
3	0.0	302	90	-27	-92	-121	-167	-212	-235	-239
	0.2	315	83	-8	-71	-103	-148	-187	-189	-191
4	0.0	305	100	-22	-93	-112	-170	-215	-241	-245
	0.2	312	82	-9	-76	-105	-141	-189	-196	-198
5	0.0	307	102	-23	-95	-117	-179	-220	-236	-239
	0.2	310	87	-10	-71	-102	-146	-167	-202	-206

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hard reducibility comparing to the freshly prepared one (Ponnamperuma, 1972). The total changes in Eh between the intial and final values were 512, 509, 506, 510 and 516 mV for MnO_2 treated sediment samples, respectively. The corresponding values for the non treated sediment were 530, 539, 541, 549 and 546 mV. This indicate that MnO_2 addition maintains the redox potential of the system at higher Eh values.

pH

Data in Table (4) show that the pH of the sediment suspensions changes towards the neutral side after submergence. This effect was found true with all sediment samples under study. Patrick and Mikkelsen (1971) reported that the pH of acid soils increases upon water logging, and the opposite was that for alkaline soils. This tendency indicats that the pH of suspensions is buffered around neutrality by some substances produced as a result of reduction processes.

Iron and manganese compounds in the form of hydroxides and carbonates and carbonic acid are likely to involve as buffering pH materials under such conditions (Ponnamperuma, 1972).

Water soluble Fe

The effect of MnO_2 addition on the water soluble Fe is presented in Table (5). It is clear that in the case of absence of MnO_2 which posses a relatively low Eh values (Table 3) the considerable increase in water soluble Fe occurred after 4 days of incubation. This delaying period, can be due to the well known fact that manganese compounds are reduced before iron ones. The relatively low amounts of water soluble Fe in all sediment samples treated with MnO_2 may be due to the higher Eh of the system.

MnO_2 addition poisoned the redox system at a relatively high oxidized state (Patrick and Mikkelsen, 1971). Under such conditions, relatively few

Table (4): Evolution of sediment suspension pH as affected by MnO2 addition during the term of incubation

Sediment No.	MnO₂ %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	7.79	7.53	7.50	7.43	7.33	7.26	7.20	7.13	7.11
	0.2	7.78	7.56	7.49	7.33	7.21	7.18	7.14	7.12	7.10
2	0.0	7.78	7.56	7.43	7.38	7.20	7.14	7.11	7.10	7.10
	0.2	7.79	7.62	7.41	7.36	7.21	7.15	7.10	7.08	7.06
3	0.0	7.79	7.71	7.43	7.30	7.26	7.20	7.13	7.11	7.11
	0.2	7.79	7.61	7.45	7.32	7.24	7.20	7.12	7.10	7.10
4	0.0	7.73	7.63	7.41	7.30	7.21	7.17	7.15	7.11	7.10
	0.2	7.74	7.60	7.42	7.32	7.23	7.16	7.11	7.10	7.10
5.	0.0	7.26	7.63	7.40	7.31	7.19	7.16	7.10	7.08	7.05
	0.2	7.25	7.65	7.42	7.36	7.18	7.15	7.13	7.11	7.03

Table (5): Water soluble Fe^{2+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	$MnO_2 \%$	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	0.01	0.13	0.62	0.66	0.70	0.75	0.32	0.22	0.12
	0.2	0.01	0.09	0.21	0.27	0.35	0.42	0.20	0.13	0.10
2	0.0	0.02	0.26	0.76	0.81	0.92	0.76	0.37	0.26	0.14
	0.2	0.02	0.08	0.26	0.32	0.33	0.40	0.25	0.12	0.12
3	0.0	0.11	0.29	0.68	0.71	0.81	0.89	0.42	0.29	0.16
	0.2	0.11	0.09	0.22	0.30	0.36	0.41	0.27	0.13	0.10
4	0.0	0.01	0.23	0.76	0.88	0.93	0.95	0.43	0.23	0.17
	0.2	0.01	0.07	0.23	0.37	0.40	0.43	0.23	0.14	0.12
5	0.0	0.02	0.29	0.60	0.72	0.81	0.87	0.51	0.26	0.16
	0.2	0.02	0.09	0.20	0.23	0.32	0.35	0.27	0.14	0.14

amounts of Fe are converted to water soluble Fe via reduction and hence remain in a solid form (Rabenhorst and James, 1992 ; Leonard et al., 1995 ; Rabenhorst, 1997 and Hussein and Rabenhorst, 1999).

Table (5) also reveal that water soluble Fe continued to increase in all sediment samples from 4 to 10 days then sharply declined by the end of the incubation course. This effect may be due to the precipitation of water soluble Fe by sulfide ions formed as a result of reduction of SO_4^{2-} under such extremely reduced conditions as shown in Table 3 which indicates that all sediment samples have Eh values between -191 and -254 mV which satisfy the reduction of SO_4^{2-} to S^{2-} ions (Jakoben et al., 1981).

Water soluble Mn

Data in Table 6 depict that flooding of sediment samples resulted in an increase in water soluble-Mn in all studied treatments particularly when

MnO_2 was added. It is worthwhile to note that water soluble Fe started to increase only after 4 days (Table 5), while water soluble Mn started to increase almost immediately after submergence. The differences among the two elements may be due to the high positive redox potential required for Mn reduction comparing to Fe. Saoud et al., (1986) attributed these effects to the differences in solubility between Fe and Mn compounds.

The final lowest concentration values of water soluble Mn may be due to its precipitation as Mn sulfide (Ponnamperuma, 1972).

Water soluble As

Data in Table 7 manifest the effect of the oxidizing power of the MnO_2 treated sediment samples with respect to As (III). It is shown that MnO_2 addition caused a sharp decrease in the water soluble As (III) content in all sediment samples during the initial period of incubation. This suggests as was found by

Table (6): Water soluble Mn^{2+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	1.01	1.79	1.81	1.21	1.01	0.80	0.61	0.32	0.21
	0.2	1.21	3.86	3.92	3.12	2.96	1.21	1.11	0.71	0.32
2	0.0	1.10	1.62	1.68	1.12	1.01	0.62	0.52	0.21	0.18
	0.2	1.20	3.13	3.52	3.02	2.81	1.10	0.96	0.62	0.29
3	0.0	1.20	1.52	1.62	1.10	0.92	0.53	0.42	0.24	0.17
	0.2	1.30	3.21	3.70	3.00	2.65	1.21	0.86	0.50	0.23
4	0.0	1.22	1.43	1.52	1.13	0.86	0.46	0.39	0.26	0.18
	0.2	1.35	3.51	3.81	3.12	5.25	1.31	0.76	0.43	0.25
5	0.0	1.20	1.36	1.42	1.11	0.93	0.52	0.22	0.29	0.17
	0.2	1.30	3.62	3.94	3.01	2.46	1.03	0.71	0.46	0.28

Table (7): Water soluble As^{3+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	3.64	3.01	2.12	1.13	0.87	0.36	0.22	0.17	0.11
	0.2	3.65	2.02	1.32	0.91	0.36	0.20	0.13	0.10	0.02
2	0.0	3.62	3.12	2.13	1.19	0.96	0.42	0.26	0.16	0.10
	0.2	3.62	2.11	1.34	0.82	0.38	0.23	0.16	0.08	0.02
3	0.0	3.60	3.31	2.16	1.22	0.91	0.51	0.29	0.18	0.11
	0.2	3.61	2.02	1.37	0.65	0.41	0.21	0.16	0.09	0.01
4	0.0	3.43	3.06	2.19	1.23	0.98	0.59	0.31	0.19	0.11
	0.2	3.42	2.11	1.42	0.56	0.45	0.26	0.17	0.08	0.06
5	0.0	3.42	3.11	2.22	1.36	0.86	0.66	0.33	0.18	0.10
	0.2	3.41	2.01	1.51	0.69	0.51	0.21	0.12	0.07	0.02

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Oscarson et al. (1981) that Mn (IV) oxides were very effective abiotic oxidant with respect to As (III). This behavior means that at the initial period of submergence, as MnO_2 is subjected to reduction, it acts as As (III) oxidant, hence water soluble As decreases. The coversion of As (III) to As (IV) through the oxidation of As (III) by MnO_2 leveled off with elapsing time. Luh et al., (1973) and Gohda, (1974) stated that manganese is the main sediment component responsible for the oxidation of As (III). Deuel and Swoboda, (1972) cleared the difference in solubility between As (III) and As (V). As (V) has very low solubility comparing to As (III).

The oxidation of As (III) to As (V) by Mn (IV) is a thermo-dynamically favorable (Oscarson et al., 1980 ; Oscarson et al., 1981). It was found that synthetic Mn (IV) oxide was very effective on lowering the concentration of added As (III) to the sediment samples. This was found true under all sediment samples.

Since the reduced state As (III) (arsenite) is much more toxic (Webb, 1966 and Penrose, 1974) and more soluble and mobile than the oxidized state As (V), consequently, the obtained data is of great importance with respect to aquatic environmental quality.

After 12 days of submergence, data in Table (7) reveal that the As (III) concentration reaches the lowest values which may be due to its precipitation as sulfide under the extremely low redox potential conditions (Table 3).

The MnO_2 treated sediment samples had a higher Eh than the untreated sediments (Table 3). The lower the redox potential of the system, the more energy was required to affect the transfer of electrons from As (III) to the sediment because the electron activity is greater (Sparks, 1995).

Cu, Ni and Pb

Data in Tables (8, 9,10) reveal that Mn (IV) oxide was very

Table (8): Water soluble Cu^{2+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	4.64	4.01	2.10	1.81	1.61	1.21	1.02	0.76	0.21
	0.2	4.65	2.22	1.72	1.02	1.03	0.72	0.61	0.21	0.11
2	0.0	4.82	4.10	2.21	1.71	1.51	1.36	1.12	0.65	0.11
	0.2	4.81	2.36	1.65	1.02	0.92	0.61	0.73	0.21	0.10
3	0.0	4.92	4.32	2.36	1.82	1.63	1.30	1.16	0.93	0.19
	0.2	4.92	2.13	1.51	1.23	0.82	0.71	0.52	0.36	0.12
4	0.0	4.82	4.13	2.11	1.96	1.52	1.21	1.11	0.82	0.21
	0.2	4.81	2.10	1.46	1.32	0.73	0.61	0.43	0.21	0.11
5	0.0	4.63	4.01	2.22	1.81	1.71	1.13	1.11	0.96	0.20
	0.2	4.61	2.13	1.78	1.11	0.70	0.72	0.37	0.12	0.11

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Table (9): Water soluble Ni^{2+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	4.82	4.21	2.01	1.96	1.73	1.38	1.12	0.82	0.11
	0.2	4.83	2.01	1.63	1.43	1.11	0.82	0.51	0.36	0.10
2	0.0	4.85	4.32	2.11	1.81	1.65	1.22	1.02	0.76	0.09
	0.2	4.84	2.10	1.53	1.36	1.22	1.03	0.41	0.21	0.08
3	0.0	4.83	4.13	2.23	1.62	1.53	1.45	1.02	0.65	0.12
	0.2	4.82	2.32	2.02	1.26	1.41	1.11	0.51	0.31	0.09
4	0.0	4.76	4.22	1.96	1.71	1.46	1.36	1.06	0.71	0.13
	0.2	4.75	2.04	1.52	1.12	1.00	0.89	0.62	0.22	0.08
5	0.0	4.75	4.36	1.82	1.82	1.23	1.21	1.07	0.63	0.11
	0.2	4.74	2.11	1.33	1.12	1.03	0.83	0.73	0.21	0.11

Table (10): Water soluble Pb^{2+} concentration (mgl^{-1}) in the sediment suspensions as affected by MnO_2 addition during the term of incubation

Sediment No.	MnO_2 %	Time, days								
		0	1	2	4	6	8	10	12	14
1	0.0	4.70	2.2	1.70	1.50	1.21	1.10	0.93	0.21	0.11
	0.2	4.68	1.5	1.00	0.92	0.30	0.20	0.10	0.01	0.01
2	0.0	4.62	2.1	1.60	1.4	1.22	1.12	0.96	0.23	0.13
	0.2	4.60	1.4	1.02	0.91	0.36	0.22	0.11	0.02	0.01
3	0.0	4.60	2.2	1.72	1.5	1.29	1.14	0.82	0.26	0.16
	0.2	4.61	1.3	1.23	0.82	0.37	0.25	0.13	0.01	0.01
4	0.0	4.58	2.1	1.63	1.32	1.20	1.16	0.87	0.29	0.13
	0.2	4.58	1.3	1.10	0.82	0.30	0.27	0.12	0.01	0.01
5	0.0	4.53	2.2	1.51	1.33	1.27	1.18	0.81	0.27	0.16
	0.2	4.52	1.4	1.12	0.96	0.36	0.24	0.11	0.02	0.02

effective in removing Cu, Ni and Pb from the overlying water of the studied sediment suspensions.

MnO_2 treated sediment samples exhibited a sharp decrease in water soluble Cu, Ni and Pb particularly during the initial period of incubation, where after, a gradual decrease was noted in all sediment samples by then, a sharp decline was noted by the end of incubation period. The first decrease in water soluble Cu, Ni and Pb particularly in MnO_2 treated sediment samples can be due to catalyzing the oxidation of these heavy metals by disproportionation to Mn^{2+} and MnO_2 (Hem, 1978, Sparks, 1995). The decrease in water soluble forms of these elements thereafter may be due to destruction of MnO_2 via reduction reactions which can be noted from the increase in the concentration of water soluble Mn (Table 6). The net result is the libiration of heavy metals which adsorped on the surface of MnO_2.

The final decrease in the concentration of water soluble forms of these elements may be due to the precipitation of these metals as sulfides under the extremely low redox potential (Table 3). Stumm and Morgan (1970) pointed out that on the basis of solubility product constants of sulfide compounds, elements such as Pb, Co, Ni, Hg, Ag, Cu and Zn remain essentially fixed in reducing sidements of the pore water with a sulfide concentration of 10^{-9} mol per litre. The solubility product of metal sulfides are 6.3×10^{-36}, 3.2×10^{-8}, 1.4×10^{-15}, 3.2×10^{-19} and 3.2×10^{-27} for Cu S, Fe S, Mn S, Ni S and Pb S, respectively.

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تأثير إضافة ثاني أكسيد المنجنيز على سلوك بعض المعادن الثقيلة في نظام راسب – ماء لترعة السلام

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تمت دراسة تأثير إضافة ثاني أكسيد المنجنيز على سلوك بعض المعادن الثقيلة في نظام راسب – ماء في رواسب جمعت من القاع على مسار ترعة السلام إبتداء من إلتقائها مع مصرف بحر حادوس ثم بعد كل ٥ كم علي ذلك المسار.

أوضحت النتائج أن العينات تحت الدراسة قد وصلت إلي مرحلة الإختزال بعد أيام قليلة من الغمر وإستمر الإنخفاض فى جهد الأكسدة والإختزال حتى نهاية التجربة.

إتضح أن ثاني أكسيد المنجنيز المضاف قد شجع على إزالة المعادن الثقيلة As ، Cu ، Ni ، Pb من الماء حيث تحولت إلي صورة غير ذائبة خاصة في الفترات الأولي من الغمر بالماء. أما في الفترات المتأخرة ومع إنخفاض جهد الأكسدة والإختزال إلي قيم شديدة السالبية فإن إنخفاض تركيز الصورة الذائبة في الماء من تلك العناصر يرجع إلي تحولها إلي كبريتيدات تلك المعادن شحيحة الذوبان.

نتائج التجربة توضح أهمية بعض مكونات الرواسب مثل ثاني أكسيد المنجنيز فى إزالة بعض المعادن الثقيلة من الماء وتحويلها إلي صورة غير ذائبة مما يقلل من تأثيرها الضار على البيئة.