

Effect of Surfactant on Adsorption and Mobility of Lead and Cadmium in Soils

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THE PRESENT work investigates the effects of anionic sodium dodecyl sulphate (SDS) surfactant at critical micelles concentration (CMC) value on the sorption of Pb^{2+} and Cd^{2+} at different initial concentrations on four selected soil samples with different clay content through batch equilibrium experiments. Langmuir and Freundlich models were used to describe the sorption processes in the absence and presence of the aqueous solution of the surfactant. The sorption data fitted very well with both Langmuir and Freundlich isotherm model which gave high correlation coefficients. The K_L and Q_m ranged from 0.0900.140- Lmg^{-1} and 300010000- $mgkg^{-1}$ for adsorption of Pb^{2+} in absence of SDS, respectively. While, K_L and Q_m ranged from 0.030 -0.060 Lmg^{-1} and 6002500- $mgkg^{-1}$ and 0.950.0.99- for adsorption of Cd^{2+} in absence of SDS, respectively. The presence of anionic surfactant SDS in adsorption of Pb^{2+} , K_L and Q_m ranged from 0.0080.030- Lmg^{-1} and 20008000- $mgkg^{-1}$, respectively. While, K_L and Q_m ranged from 0.0100.022- Lmg^{-1} and 5001200- $mgkg^{-1}$ for adsorption of Cd^{2+} , respectively. The K_f and n ranged from 90140- Lkg^{-1} and 2.02.98- for adsorption of Pb^{2+} in absence of SDS, respectively. While, K_f and n ranged from 58120- Lkg^{-1} and 1.52.08- for adsorption of Cd^{2+} in absence of SDS, respectively. The presence of anionic surfactant SDS in adsorption of Pb^{2+} , K_f and n ranged from 74115- Lkg^{-1} and 1.642.59-, respectively. While K_f and n ranged from 22.8796.18- Lkg^{-1} and 1.331.37- for adsorption of Cd^{2+} , respectively. The results showed that both of Pb^{2+} and Cd^{2+} showed a reduction in their adsorption to soil particles when surfactant was present.

Keywords: Adsorption Isotherms, Heavy metals, (SDS) surfactant.

Heavy metal contamination of soil is one of the most important environmental problems throughout the world (Doumett *et al.*, 2008). The ability of heavy metals to accumulate and cause toxicity in biological systems-humans, animals, microorganisms and plants has been reported (D'amore *et al.*, 2005). As chemical hazards, heavy metals are non-biodegradable and can remain almost indefinitely in the soil environment. However, their availability to biota can change considerably depending on their chemical speciation in the soil. The adequate protection and restoration of the soil ecosystems, therefore, require the characterization and remediation of soils that are contaminated with heavy metals (Nwachukwu *et al.*, 2010). Studies on the sorption of heavy metals show that, in general, it was a function mainly of pH, amount of carbonates, cation exchange capacity (CEC), organic matter content, clay minerals and specific surface area. It was also suggested that

Fe and Mn oxides are major species involved in heavy metal sorption. Adsorption characteristics can be depicted by an adsorption isotherm. An adsorption isotherm is the presentation of the amount of solute adsorbed per unit weight of adsorbent as a function of the equilibrium concentration in the bulk solution at constant temperature. Langmuir and Freundlich adsorption isotherms are commonly used for the description of adsorption data (Valairat, 2004).

Application of surfactants for heavy metal remediation has gained popularity in recent years. Surfactants (surface active agents) possess a hydrophilic functionality, increasing solubility in aqueous media, and a hydrophobic moiety, enabling them to adsorb to hydrophobic organic compounds. At concentrations in excess of the critical micelle formation (CMC) in aqueous solution, the surfactant molecules aggregate to

form clusters called micelles that enhance the remediation capability of surfactant suspensions. Since micelles are composed of an inner hydrophobic core and a hydrophilic exterior surface, the surfactant micelles can solubilize both ionic and non-ionic compounds, such as heavy metals and Polychlorinated biphenyls (PCBs). Many studies directed to the extraction of hydrophobic contaminants from soils using surfactant micelles (Huang *et al.*, 1997; Noordman *et al.*, 1998) have demonstrated that solubilization can be attributed to the incorporation of hydrophobic contaminants within the hydrophobic core of the surfactant micelles in solution. The objective of this research was to investigate adsorption isotherm of Pb^{2+} and Cd^{2+} , in the aqueous solution as well as in aqueous surfactant solution at CMC value on four selected soil samples with different clay content.

Materials and Methods

Soils

Four soils with different clay contents (10%, 20%, 40% and 50%, referred to S_{10} , S_{20} , S_{40} and S_{50} , respectively) were collected from 030-cm depth. The soil samples were pooled, air dried, passed through a 2 mm sieve and homogenized. Soil particle size was determined by the hydrometer method (Ashworth *et al.*, 2001). Soil pH was determined in 1:2.5 soil/water extracts with a combined pH electrode (Melero *et al.*, 2007). Cation exchange capacity was determined with sodium acetate and ammonium acetate for Na extraction (Rhoades, 1982). Calcium carbonate was determined using the Collin's calcimeter methods (Nelson and Sommers, 1982). Organic matter determinations conducted using Walkely-Black method as described by Nelson and Sommers (1982). Oxide contents were determined by the dithionite-citrate method (USSCS, 1972; Sherdrick and McKeague, 1975) and the Fe, Al and Mn contents of the extract were determined by atomic absorption spectrophotometer (AAS), Perkin-Elmer model 1100B. Specific surface area of soil was determined using O-phenanthroline method (Sparks, 1998).

Metals

The heavy metals (Pb^{2+} and Cd^{2+}) were selected for adsorption studies. The anionic sodium dodecyl sulphate (SDS) surfactant, critical micelles concentration (CMC) = 2.4 gL^{-1} . All chemicals used were of analytical grade reagents and used without pre-treatments. Standard stock solutions of the metals were prepared in deionised Egypt. *J. Soil Sci.* **57**, No. 2 (2017)

water.

Adsorption experiment

Batch equilibrium adsorption isotherms studies of (Pb^{2+} and Cd^{2+}) were conducted in the soil-metal-water-surfactant system, in the absence and presence of aqueous solutions of anionic surfactants (SDS), at one critical micelles concentration (CMC), according to Gomez *et al.*, 2004; Iman *et al.*, 2008. Competitive sorption measurements by Duplicate soil samples of 1.0g were shaken for 48 h in polypropylene centrifuge tubes at room temperature with 50 ml of a mixed solution containing concentrations of 0, 10, 20, 30, 40, 50, 100 and 200 $mg L^{-1}$ of nitrate salts of Pb^{2+} and Cd^{2+} in a background of 0.01 M $NaNO_3$. The samples plus blanks (no metal) and control (no soil), were shaken continuously at temperature controlled (25 °C) water bath shaker (1500 rpm). The tubes were centrifuged for 20 min at 3500 rpm. The clear supernatant was removed and analyzed for the metal ion of Pb^{2+} and Cd^{2+} solution with by atomic absorption spectrophotometer (AAS). The total amount of metal adsorbed in the adsorption processes was calculated from the difference between the amount added initially and that remaining in solution after equilibration. The measured liquid phase concentrations were then used to calculate the adsorption capacity. The sorbet amount was calculated by mass balance equation:

$$q_e = (C_i - C_e)V/M \quad (1)$$

where: q_e , C_i , C_e , M and V are the sorbet amounts of the metal ions ($mgkg^{-1}$), initial metal concentration in solution (mgL^{-1}), equilibrium metal concentration (mgL^{-1}), mass of sorbent (g), and the volume of the solute solution (L), respectively. The adsorption isotherms were calculated for each soil using the Langmuir and Freundlich equations:

$$C_e/C_s = 1/K_L Q + C_e/Q \quad (2)$$

$$\text{Log } C_s = 1/n \text{ Log } C_e + \text{Log } K_f \quad (3)$$

where: C_e = equilibrium solution phase concentration (mgL^{-1}), C_s = equilibrium solid phase concentration ($mgkg^{-1}$), Q = Langmuir isotherm adsorption capacity ($mgkg^{-1}$), K_L = binding energy coefficient (Lmg^{-1}), n = adsorption intensity constant (kgL^{-1}) and K_f = adsorption capacity constant (partition coefficient) (Lkg^{-1}). The amount of each metal that had been released from the soil sample was measured. The

distribution coefficient, K_d was calculated as:

$$K_d = \Sigma (C_s \cdot C_e) / \Sigma (C_e)^2 \quad (4)$$

to measure the adsorption extent. The adsorption constants (K) normalized to soil organic carbon content, K_{OC} and soil clay content, K_C . These are important parameters that play a significant role in environmental fate assessment of heavy metals and were evaluated by using the following equation (Grestl, 1984; Singh, 1996).

$$K_C = (K_d \times 100) / \text{Clay}\% \quad (5)$$

$$K_{OC} = (K_d \times 100) / \text{OC}\% \quad (6)$$

Results and Discussion

TABLE 1. Characteristics of the studied soils

Soil	pH	Clay OM TAM CaCO ₃				CEC (Cmol _c kg ⁻¹)	SA (m ² g ⁻¹)
		%					
S ₅₀	7.9	50	1.5	5.5	3.8	52.5	230
S ₄₀	7.7	40	1.2	4.2	3.4	42.4	217
S ₂₀	7.5	20	0.8	3.8	3.0	30.2	120
S ₁₀	7.4	10	0.4	3.2	2.4	15.5	80

OM: organic matter; CEC: cation exchange capacity; TAM: total amorphous material; SA: surface area.

Adsorption study of Pb²⁺ and Cd²⁺ in soils

The adsorption isotherms for Pb²⁺ and Cd²⁺ on soils in the absence and presence of the aqueous solutions of the surfactants are depicted in Figures (12-). All these isotherms indicate the amount of Pb²⁺ and Cd²⁺ adsorbed per unit mass of solid adsorbent (C_s , mgkg⁻¹) versus the equilibrium concentration (C_e , mgL⁻¹). From these isotherms, it is clear that adsorption followed the order as S₅₀ > S₄₀ > S₂₀ > S₁₀ in the aqueous solution as well as in aqueous surfactant solution at CMC value studied. The higher adsorption onto S₅₀ soil may be due to the greater amount of organic matter, clay and calcium carbonate content (Singh, 1996), the higher surface area of S₅₀ soil followed by S₄₀, S₂₀ and S₁₀ soils (Table 1).

Adsorption isotherm studies showed that the sorption rates could be described by both Langmuir and frendlich models. The values of K_L , Q_m and R^2 demonstrated in Table 2 and for adsorption of Pb²⁺ and Cd²⁺ on selected soil samples. The K_L , Q_m and R^2 ranged from 0.090 - 0.140 Lmg⁻¹, 300010000-mgkg⁻¹ and 0.991.00- for adsorption of Pb²⁺ in absence of SDS, respectively. While, K_L , Q_m and R^2 ranged from 0.030 -0.060 Lmg⁻¹, 6002500-

Soil characteristics

The analyzed physico-chemical properties of soils are presented in Table 1. The values of soil pH varied from 7.4 to 7.9 indicating neutral in nature, respectively. The content of organic matter and clay were different from 0.4 to 1.5 and 10 to 50% for S₁₀ and S₅₀, respectively. However, the average CaCO₃ percentage ranged between 2.5 to 3.8% for S₁₀ and S₅₀, respectively. The quantity of organic matter and clay attributed a major concern to permit the downward movement of Pb²⁺ and Cd²⁺ in the lower depth. Soil of S₅₀ reflected high CEC (52.5 Cmol_ckg⁻¹) and surface area (230 m²g⁻¹) due to rich organic matter content which influenced the adsorption pattern of heavy metals.

mgkg⁻¹ and 0.950.099- for adsorption of Cd²⁺ in absence of SDS, respectively. The presence of anionic surfactant SDS in adsorption of Pb²⁺, K_L , Q_m and R^2 ranged from 0.0080.030- Lmg⁻¹, 2000-8000 mgkg⁻¹ and 0.840.98-, respectively. While, K_L , Q_m and R^2 ranged from 0.0100.022- Lmg⁻¹, 5001200- mgkg⁻¹ and 0.9470.99- for adsorption of Cd²⁺, respectively (Table 3).

The Langmuir sorption model served to estimate the maximum metal adsorption values Q_m . The constant K_L represents the affinity between the sorbate and the sorbent and it indicate the binding capacity (Maather *et al.*, 2009).

Data demonstrated in Tables 2 & 3 represent the values of partition coefficient K_f for adsorption of Pb²⁺ and Cd²⁺ on selected soil sample. The values of K_p , n , and R^2 demonstrated in Table 2 for adsorption of Pb²⁺ and Cd²⁺ on selected soil samples. The K_p , n and R^2 ranged from 90140-Lkg⁻¹, 2.02.98- and 0.810.93- for adsorption of Pb²⁺ in absence of SDS, respectively. While K_p , n and R^2 ranged from 58.0120- Lkg⁻¹, 1.52.08- and 0.800.88- for adsorption of Cd²⁺ in absence of SDS, respectively. The presence of anionic

surfactant SDS in adsorption of Pb^{2+} K_p , n and R^2 ranged from 74115- Lkg^{-1} , 1.642.59- and 0.92-0.97, respectively. While K_p , n and R^2 ranged from 22.87- 96.18 Lkg^{-1} , 1.331.37- and 0.910.99- for adsorption of Cd^{2+} , respectively (Table 3), these results agreed with literature. The results reveal

that the model parameters are largely dependent on the initial sorbate concentration value. The K_f indicates the binding affinity between the sorbate and sorbent (Martina *et al.*, 2005).

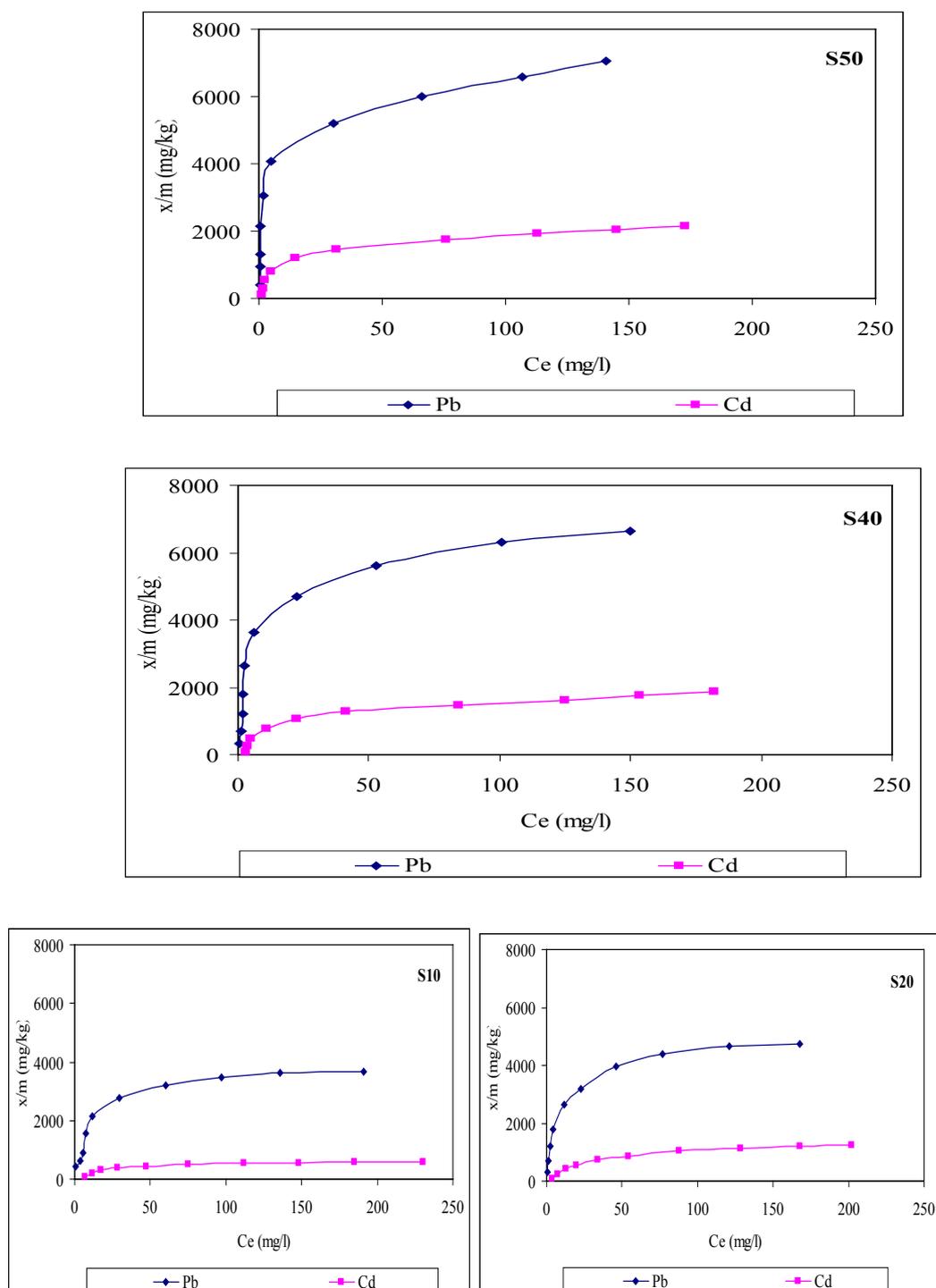


Fig. 1. Adsorption isotherms of Pb and Cd on soils in absence of SDS solution at CMC value

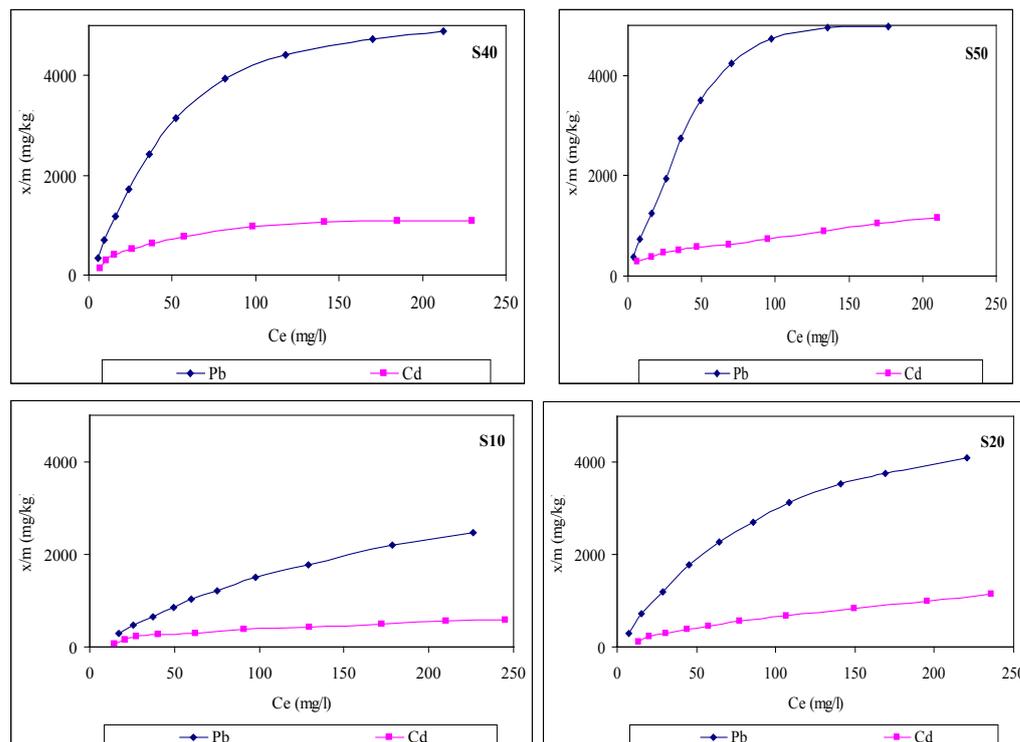


Fig. 2. Adsorption isotherms of Pb and Cd on soils in presence of SDS solution at CMC value

TABLE 2. The characteristic parameters of Langmuir and Freundlich models isotherms for adsorption process of Pb and Cd on the selected soil samples in absence of SDS solution

metals	Langmuir			Freundlich		
	Q_m (mg/kg)	K_L (l/mg)	R^2	n (kgL^{-1})	K_f (l/kg)	R^2
S_{50}						
Pb	10000	0.140	0.995	2.98	140	0.81
Cd	2500	0.060	0.991	2.08	120	0.83
S_{40}						
Pb	8000	0.080	0.997	2.43	130	0.83
Cd	2000	0.030	0.947	1.78	100	0.80
S_{20}						
Pb	5000	0.110	0.999	2.29	110	0.93
Cd	1400	0.020	0.979	1.65	66	0.88
S_{10}						
Pb	3000	0.090	0.996	2.0	90	0.91
Cd	600	0.030	0.986	1.5	58	0.80

Q_m : maximum adsorption; K_L : binding energy coefficient in absence of SDS solution; K_f : partition coefficient in absence of SDS solution; n : sorption intensity constant; R^2 : coefficient of determination.

TABLE 3. The characteristic parameters of Langmuir and Freundlich models isotherms for Adsorption process of Pb and Cd on the selected soil samples in presence of SDS solution

Langmuir				Freundlich		
metals	Q_m (mg/kg)	K_L^* (l/mg)	R^2	n (kgL ⁻¹)	K_f^* (l/kg)	R^2
S_{50}						
Pb	8000	0.030	0.968	2.59	115	0.969
Cd	1200	0.022	0.935	1.37	96.20	0.978
S_{40}						
Pb	5000	0.030	0.969	1.92	100	0.945
Cd	1000	0.026	0.992	1.42	81.30	0.906
S_{20}						
Pb	4000	0.020	0.979	1.47	85.33	0.971
Cd	1000	0.006	0.971	1.32	28.37	0.997
S_{10}						
Pb	2000	0.008	0.839	1.64	74.03	0.918
Cd	500	0.010	0.941	1.33	22.87	0.908

Q_m : maximum adsorption; K_L^* : binding energy coefficient in presence of SDS solution; K_f^* : partition coefficient in presence of SDS solution; n : sorption intensity constant; R^2 : coefficient of determination.

The distribution coefficient, K_d is defined as the ratio of insecticide concentration in the soils and Pb^{2+} and Cd^{2+} concentration in solution. The value obtained for the constant, mentioned as given in Table 4. The K_d , ranged from 26.7977-, and 3.5315.31- LKg^{-1} for adsorption of Pb^{2+} and Cd^{2+} in absence of SDS respectively. While K_d , ranged from 12.68 -38.71, and 2.896.63- LKg^{-1} for adsorption of Pb^{2+} and Cd^{2+} in presence of SDS respectively. A smaller K_d value indicates that a smaller amount of the soil-borne element is needed to produce $1Lkg^{-1}$ of the element in solution phase thus potentially higher exposure risks (Gueu *et al.*, 2007). The value of K_f and K_d considered as a prime approximation in characterizing adsorption capacity of Pb^{2+} and Cd^{2+} in both sets of soil. It is observed that high K_d value for Pb^{2+} than Cd^{2+} and in both the soils due to significant amount of organic matter that exhibited higher adsorption capacity.

The Freundlich constant and distribution coefficient in surfactant free systems are denoted as K_f and K_d whereas in surfactant-soil-water systems are denoted as K_f^* and K_d^* respectively. On examination of Figures (12-) and the K_p , K_f^* and K_d , K_d^* values given in Table 4, it is clear that adsorption followed the order $S_{50} > S_{40} > S_{20} > S_{10}$ in the aqueous solution as well as in aqueous surfactant solutions at CMC value studied. The higher adsorption onto S_{50} soil may be due to its

containing greater amount of organic carbon, clay and calcium carbonate and its higher surface.

The ratios of K_d^*/K_d values were used to evaluate the remediation efficiency of the surfactant. If K_d^*/K_d ratio exceeds 1, then the remediation of a Pb^{2+} and Cd^{2+} contaminated soil is considered to be inefficient, while if K_d^*/K_d is less than 1, the soil remediation is considered to be efficient (Lee *et al.*, 2004). The K_d^*/K_d ratios at the CMC value of SDS are less than one (Table 4) and this obvious reduction in the K_d^*/K_d values is due to the solubility enhancement of Pb^{2+} and Cd^{2+} in the presence of surfactant. The obtained results of $K_d^*/K_d < 1$ demonstrate that SDS is efficient in remediating the soils contaminated with Pb^{2+} and Cd^{2+} .

The affinity of Pb^{2+} and Cd^{2+} towards organic matter and clay content of the soils was evaluated by calculating K_{OC} and K_C values which are summarized in Table 5. K_{OC} values of both the soils were much higher than K_C , indicated that contribution of the surface of organic matter in Pb^{2+} and Cd^{2+} adsorption was much higher than the clay surface in soils. In the present study, S_{50} soil exhibited higher K_{OC} values than S_{40} , S_{20} and S_{10} soils. This is common for the soils with low organic matter content. Similar report was made by (Kumar and Philip, 2006) the soil separates is primarily responsible for adsorption of heavy metals.

TABLE 4. The characteristic parameters of Linear model isotherm for Adsorption process of Pb and Cd on the selected soil samples in absence (K_d) and presence (K_d^*) of SDS solution

Soils	K_d (L/kg) without SDS		K_d^* (L/kg) with SDS		K_d^*/K_d	
	Pb	Cd	Pb	Cd	Pb	Cd
S ₅₀	77.00	15.31	38.71	6.63	0.50	0.43
S ₄₀	57.41	12.55	30.13	6.45	0.52	0.51
S ₂₀	38.78	7.93	23.36	5.45	0.60	0.69
S ₁₀	26.79	3.53	12.68	2.89	0.47	0.82

K_d : Distribution coefficient in absence of SDS solution; K_d^* : Distribution coefficient in presence of SDS solution.

TABLE 5. The adsorptive capacity of organic carbon and clay content for studied soils in absence and presence of SDS solution

Soils	without surfactant			
	K_{oc}		K_c	
	Pb	Cd	Pb	Cd
S ₅₀	8829.33	1755.55	154.00	30.62
S ₄₀	8228.77	1798.83	143.53	31.38
S ₂₀	8337.70	1704.95	193.90	39.65
S ₁₀	11519.70	1517.90	267.90	35.30
	with surfactant			
S ₅₀	4438.75	760.24	77.42	0.87
S ₄₀	4318.63	924.50	75.33	1.28
S ₂₀	5022.40	1171.75	116.80	3.44
S ₁₀	5452.40	1242.70	126.80	8.19

K_{oc} : adsorptive capacity of organic carbon; K_c : adsorptive capacity of clay.

The statistical analysis (Tables 6 & 7) showed highly significant positive correlation between Pb^{2+} or Cd^{2+} maximum adsorption capacity Q_m (surfactant-free) and (surfactant–soil–water systems) and clay, OM, CEC, TAM, SA and pH. This could be attributed to the increase of clay content which is associated with increases in the adsorption sites expressed by high CEC, OM and total amorphous material, which in agreement with the findings of (Vega et al., 2009). Comparing the experimental results for Pb^{2+} and Cd^{2+} , the general trend is that the maximum adsorption capacity Q_m increases when the pH increases. This is not only because of the hydrogen ion competition at low pH, but also because of the weakly acidic nature of the active sites on the adsorbent whose deprotonation with increasing pH, favors the metal adsorption (Pagnanellia et al., 2003).

With a view of determining, by means of a statistical approach, the effect of soil properties on the adsorption of Pb^{2+} and Cd^{2+} , Pearson's correlations between the K_p , K_f^* and K_d , K_d^* adsorption constants and the characteristics of the soils were calculated. The results point to the existence of a positive correlation between the K_p , K_d (surfactant-free) and K_f^* , K_d^* (surfactant–soil–water systems) and the organic matter, clay content, TAM, SA, pH and CEC (Tables 6 & 7). Thus, it is confirmed that the organic carbon content of the soils is determining factor for the adsorption of Pb^{2+} and Cd^{2+} , on the studied soils and indicates that adsorption of Pb^{2+} and Cd^{2+} , occurs primarily through hydrophobic processes and partitioning into soil organic matter.

TABLE 6. Correlation coefficient between parameters of Langmuir (Q_m and K_L), Freundlich (K_f and n) and Linear (K_d) models isotherms and soil properties in absence of SDS solution

property	Q_m for Pb	Q_m for Cd		K_L for Pb	K_L for Cd
pH	0.93**	0.97**		0.60	0.82*
OM	0.95**	1.00**		0.54	0.70
TAM	0.83*	0.95**		0.76	0.86*
CEC	0.95**	1.00**		0.55	0.69
Clay	0.97**	0.98**		0.48	0.74
SA	1.00**	0.96**		0.35	0.65

Continued

K_f for Pb sor.	K_f for Cd	n for Pb	n for Cd	K_d for Pb	K_d for Cd
0.99**	0.98**	0.97**	0.99**	1.00**	0.98**
0.98**	0.92**	0.95**	0.96**	0.99**	1.00**
0.99**	0.98**	1.00**	1.00**	0.98**	0.94**
0.97**	0.91**	0.95**	0.96**	0.98**	1.00**
0.96**	0.94**	0.93**	0.96**	0.99**	0.99**
0.91**	0.88*	0.87*	0.90**	0.96**	0.98**

Q_m : maximum adsorption; K_L : binding energy coefficient; K_f : partition coefficient; n: sorption intensity constant; K_d calc.: Distribution coefficient.

TABLE 7. Correlation coefficient between parameters of Langmuir (Q_m and K_L), Freundlich (K_f and n) and Linear (K_d) models isotherms and soil properties in presence of SDS solution

property	Q_m for Pb	Q_m for Cd		K_L^* for Pb	K_L^* for Cd
pH	0.96**	0.94**		0.89*	0.79
OM	0.99**	0.94**		0.96**	0.78
TAM	0.90**	0.98**		0.82*	0.64
CEC	0.98**	0.95**		0.96**	0.75
Clay	0.99**	0.91**		0.94**	0.84*
SA	1.00**	0.84*		0.96**	0.89*

Continued

K_f^* for Pb	K_f^* for Cd	n for Pb	n for Cd	K_d^* for Pb	K_d^* for Cd
0.96**	0.97**	0.93**	0.93**	0.97**	0.86*
0.96**	0.95**	0.86*	0.84*	1.00**	0.94**
0.88*	0.89*	0.88*	0.92**	0.96**	0.81*
0.95**	0.94**	0.84*	0.82*	1.00**	0.95**
0.99**	0.98**	0.91**	0.87*	0.98**	0.90*
0.99**	0.98**	0.87*	0.81*	0.95**	0.91**

Q_m : maximum adsorption; K_L^* : binding energy coefficient in presence of SDS solution; K_f^* : partition coefficient in presence of SDS solution; n: sorption intensity constant; K_d^* : Distribution coefficient in presence of SDS solution.

However, positive correlations with other soil properties suggest that in addition to soil organic matter other soil components also contribute significantly to adsorption.

Comparison to both the soils, the higher Pb^{2+} and Cd^{2+} adsorption was found in S_{50} at each concentration might be due to rather high clay and organic matter content than S_{10} soil which retained the chemical on soil surface (Table 3).

Effect of the anionic surfactant (SDS) on the adsorption of Pb^{2+} and Cd^{2+}

Figure 2 depicts the effect of the anionic surfactant solution at concentration equal CMC on the adsorption of Pb^{2+} and Cd^{2+} on various soils studied. At concentration CMC, values a marked decrease in both K_L^* , K_f^* and K_d^* values was observed (Tables 3 & 4). This decrease may be attributed to several mechanisms such as (i) competition for active hydrophobic adsorption sites on the soil surface between the Pb^{2+} and Cd^{2+} and SDS molecules (ii) SDS equilibria involving monomers and micelles (iii) partitioning of Pb^{2+} and Cd^{2+} amongst hydrophobic sites and SDS micelles and (iv) interaction between Pb^{2+} and Cd^{2+} molecules and SDS monomers. The decrease in adsorption of Pb^{2+} and Cd^{2+} with increasing SDS concentration with respect to the surfactant-free system for the soils studied suggests that SDS enhanced the solubilization of Pb^{2+} and Cd^{2+} in the suspensions studied, with affinity of Pb^{2+} and Cd^{2+} for SDS micelles being higher than that for monomers for soils concerned (Ziqing et al., 1995). The micelles reduce the adsorption of Pb^{2+} and Cd^{2+} by increasing the apparent solubility and reducing the interfacial tension that hinders mass transfer.

Important aspect to be considered is the interaction of surfactant with soil, since it may, on one hand, alter the surfactant concentration in solution, thereby decreasing its efficiency for desorption, and on the other, alter the soil surface, where surfactant molecules may be adsorbed in the form of monomer or forming hemicelles or admicelles. Thus surfactant adsorption increases the organic C content of the soil and increased hydrophobic surfaces, which may contribute to decrease in the organic compound desorption. of all the above processes, the study of surfactant-enhanced desorption for organic pollutants adsorbed on soil has been addressed by many investigators in recent years although such information can only be considered a beneficial

effect in the context of major engineered remediation processes. In the study of surfactant-enhanced desorption, enhanced solubility of pollutants has been clearly indicated by several authors at surfactant concentrations higher than the CMC. However, at surfactant concentrations below the CMC competitive adsorption of organic compound by soil and/or by a surfactant in solution may occur, and hence an increase or decrease in desorption of compound from soil, depending on the characteristics of soil and organic compound (Liu, 2001). Many factors affect the adsorption-desorption of metal ion type; soil properties, organic matter, clay content, soil pH and environmental conditions (Mah et al., 2005). The main effect of surfactant at concentrations close to CMC is to increase the affinity of metal ion for the soil with, except for soils high in clay content where the surfactant effect is to enhance the affinity of metal ion for aqueous phase.

Results for SDS test indicate that the anionic surfactant demonstrated a noticeable increase of metal removal at or near CMC concentration. Above the CMC, the amount of metal removal does not show any significant variations. Surfactant can enhance the removal of heavy metals by mechanisms such as ion exchange, counter-ion binding, and dissolution precipitation. Depending on the surfactant type and the metal ion involved, one of these procedures is dominant. In the removal of Pb^{2+} , and Cd^{2+} using SDS, counter ion binding and precipitation can play an important role. With increasing concentrations beyond the CMC, the micelles which trapped the metal cations appear to precipitate the metals and allow them to attach to soil particles (Amruthur and Hesham, 2013).

Present tests were conducted with well-defined soil characteristics. In field applications, it may be necessary to repeat some tests on samples formed to resemble

closely site specific soils. Field soils contain other trace pollutants. Temperature, pH, hardness, and soil additives influence the observed CMC (Meyers, 2006). Surfactants are less toxic compared to the chelating agents. As such, surfactants are preferred to chelates heavy metals in site remediation. However, chelating agents are not very biodegradable and may induce metal mobility that leads to ground water contamination.

Conclusions

A surfactant work can be a remediation tool by lowering the contaminant-water interfacial tension and thereby causing a degree of contaminant mobility, and enhances contaminant solubility in water, so it is responsible for increasing the solubility. The results showed that a reduction in both of Pb^{2+} and Cd^{2+} adsorption to soil particles was observed in the presence of surfactant.

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تأثير مسيلات السطوح علي إدمصاص وحركة الرصاص والكاديوم في التربة

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الفلزات الثقيلة مثل الرصاص والكاديوم من أهم ملوثات البيئة، وعملية معالجة الأراضي الملوثة بهذه الفلزات تصاحبها مشاكل كثيرة، لذا فهي تحتاج إلي التحسين والتطوير حتي تكون أكثر كفاءة وسهولة وشمولية. ويهدف هذا البحث إلي دراسة كفاءة تأثير مسيلات السطوح ((SDS عند قيمة التركيز الحرج ((CMC علي إدمصاص وحركة الرصاص والكاديوم علي أربعة أنواع من الأراضي الرسوبية مختلفة القوام بنسب طين (10 , 20, 40, 50%)، تم تطبيق نموذج معادلة لانجمير وفروندليش. أظهرت النتائج أن كمية الرصاص المدمصة أعلى من الكمية المدمصة من الكاديوم في جميع عينات التربة وذلك في وجود وغياب مسيلات السطوح. كما أظهرت النتائج أن مسيلات السطوح أثبتت فعاليتها في إختزال الكمية المدمصة من الفلزات الثقيلة علي أسطح مواقع التبادل مما يساعد في معالجة الأراضي والمياه الملوثة.