



Adsorption of mercury on laterite from Guizhou Province, China

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Abstract

The adsorption behaviors of Hg(II) on laterite from Guizhou Province, China, were studied and the adsorption mechanism was discussed. The results showed that different mineral compositions in the laterite will cause differences in the adsorption capacity of laterite to Hg(II). Illite and non-crystalloids are the main contributors to enhancing the adsorption capacity of laterite to Hg(II). The pH of the solution is an important factor affecting the adsorption of Hg(II) on laterite. The alkaline environment (pH 7–9) is favorable to the adsorption of Hg(II). The amount of adsorbed Hg(II) increases with increasing pH. When the pH reaches a certain value, the amount of the adsorbed Hg(II) will reach the maximum level. The amount of adsorbed Hg(II) decreases with increasing pH. The optimal pHs of laterite and kaolinite are 9 and 8, respectively. The optimal initial concentrations of Hg(II) on laterite and kaolinite are 250 and 200 µg/ml, respectively. The adsorption isotherms were described by the Langmuir model. The adsorption of Hg(II) on laterite is a quick process while that of Hg(II) on kaolinite is a slow reaction. Laterite from Guizhou Province is a promising environmental material which can be used in the removal of Hg(II) from wastewater.

Key words: adsorption; laterite; Hg(II); pH; isotherm; kinetics

Introduction

Mercury (Hg) is a remarkably toxic metal. It is accumulated in the ecosystem via the food chain and can be regenerated by several sources, resulting in contamination of atmospheric and aqueous systems (Arica *et al.*, 2004; Genc *et al.*, 2002; Morel *et al.*, 1998). The main sources of mercury emissions to land, water and air are the processes of ores mining and smelting (in particular Cu and Zn smelting), burning of fossil fuels (mainly coal), industrial production processes (mercury cells and chlor-alkali processes), and consumption-related discharges (including waste incineration) (Haidouti, 1997).

The mercury content of coal in Guizhou is relatively high (Feng *et al.*, 2002), and the wastewater discharged from industrial plants or factories is responsible for some mercury pollution case in Guizhou, China (Yan *et al.*, 2005).

Since clay minerals have capabilities of adsorption, ion exchange, and expansion, they are highly capable of adsorbing all kinds of pollutants in waters, thus making soils possess self-purification functions. Clay minerals are promising for controlling environmental pollution (Tang *et al.*, 2002).

At present, several studies are focused on the adsorption

of heavy metals on monominerals (e.g., illite, goethite, etc.) (Chen *et al.*, 2003; Dzombak and Morel, 1999; Quigley *et al.*, 1996; Zhao *et al.*, 2001; Zheng *et al.*, 2001). Li and Zheng (2002) studied the adsorption of organic molecules, metal ions, and dyes on palygorskite. Recently, considerable research has also been done on the adsorption of mercury on clay minerals (Li *et al.*, 2001; Shao *et al.*, 2004; Wang *et al.*, 2001, 2005) and different kinds of soils (Ding *et al.*, 2001; Feng *et al.*, 1996; Jin *et al.*, 1999; Li and Qian, 2006; Lin *et al.*, 2007; Luo and You, 1996; Wang *et al.*, 2003a, 2003b; You and Luo, 1996; Yuan *et al.*, 2005; Zhang *et al.*, 2006; Zhao *et al.*, 2005). However, little research has been conducted on the adsorption of mercury on the special geological material—laterite, which is derived from carbonate rocks in Guizhou.

Guizhou is one of the important provinces, where laterite derived from carbonate rocks is widespread, in China. It is also one of the typical subtropical karst regions in the world (Zhu and Lin, 1996). There are many kinds of clay minerals in laterite derived from carbonate rocks. Thereby, study on the adsorption of mercury on laterite is of geological and environmental significance.

Different adsorption behaviors of soils toward heavy metals depend mainly on their mineral compositions and external conditions (Chen, 1996; Rouquerol and Sing, 1982). In this study, the authors studied the effect of

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mineral composition, pH, initial concentration of Hg(II), and contact time on the adsorption of Hg(II) on laterite from Guizhou, and discussed the mechanism of adsorption.

1 Materials and methods

1.1 Laterite samples and Hg(II) solutions

Laterite samples were collected from the Anshun dolomite-derived laterite profile, which is characterized by typical subtropical karst geomorphological, hydrological and eco-environmental features in the karst regions of Guizhou Province. A laterite profile usually exhibits obvious vertical zoning. In accordance with the intensity of weathering, four zones can be divided from the top to the bottom: the surface soil zone (I); the completely weathered zone (II); the moderately weathered zone (III); and the bedrock zone (IV). The surface soil zone is located in the uppermost part of the laterite profile and experiences the whole process of laterite development. It is dark-yellow in color (Sample No. 9 was collected from this zone). In a well-developed laterite profile, the completely weathered zone may account for over 90% of the total thickness. It can be divided into the red weathered bed and the yellow weathered bed according to the weathering degree and color. The red weathered bed is located in the upper part of the weathering zone. It is usually 3–5 m thick, named after its special dark-red or reddish-yellow color (Sample No. 8, 6, 5 and 4 were collected from this weathering zone). The red weathered bed is usually interbedded with ferrous concretion bands (Sample No. 7). In a well-developed laterite profile, the yellow weathered bed lies at the middle or bottom of the completely weathered zone and is usually interbedded with several dark-red or mottled ferromanganese concretion bands or a hard iron crust (Sample No. 2 and 3 were collected from this weathered zone). The moderately weathered zone is a transitional zone from laterite to carbonate rocks. The zone is usually 0.1–0.2 m thick. At the bottom of the zone, weathered dolostone is silty and loose in texture, and at the top, light purple or purplish-red clay layers are developed (Sample No. 1 was collected from this weathered zone). Sample No. 10 was the purer kaolinite, as a candidate for contrast study.

The samples were air-dried naturally, crushed as fine as possible, and sieved with the griddle which is 0.149 mm in mesh diameter. Clay minerals in the samples were determined by X-ray diffraction XRD (D'MAX-2200, Science Company, Japan).

Mercuric chloride (HgCl_2) has been chosen in this study because chloride ions are ubiquitous in natural water and industrial wastewater, and among the inorganic mercury complexes, mercuric chloride is one of the most abundant and soluble chemical in aqueous solutions. Moreover, according to the theory of ions adsorption in aqueous solutions (Yiacumi and Tien, 1995), the dominant presence of non-ionic species as HgCl_2 leads to a decrease of sorption capacity (Manohar *et al.*, 2002). Thus, the use of mercuric chloride provides a more realistic and conservative estimation of the mercury removal efficiency

of the sorbents.

The standard stock solutions of Hg(II) were realized by dissolving an appropriate mass of mercuric chloride (HgCl_2 , analytical grade, HG3-1068-77, Tongren Chemical Reagent Factory, Guizhou, China) in high purity water (18.2 M Ω cm). The concentrations of Hg(II) were estimated to be 1,000 $\mu\text{g/ml}$. The solutions used in the experiment were obtained by diluting the standard stock solutions of Hg(II).

1.2 Adsorption test

Experiments on the adsorption of Hg(II) on laterite were carried out under batch conditions. The laterite samples (1 g) were mixed with aqueous Hg(II) solutions (20 ml) in the measuring cylinders (25 ml). The Hg(II) concentrations were all 150 $\mu\text{g/ml}$. The measuring cylinders were shaken in a temperature regulated water-bath shaker (SHA-B, Fuhua Apparatus Corporation Ltd., Jintan, China) for 2 h, and put aside for additional 22 h. The temperature of the water bath was kept at $(20 \pm 1)^\circ\text{C}$. At the end of every adsorption test, the suspensions were centrifuged by centrifugal 80-2 (Shanghai Operation Instruments Factory, China) at 3,000 r/min for 5 min. After that, the supernatant was obtained and analyzed for the Hg(II) concentrations by the plasma emission spectrograph (ZCP-AES, Jobin Yvon Company, France).

In the following experiments, we selected some representative samples (No. 1, 7 and 10) as the candidates for contrast study.

The effect of pH on the adsorption capacity was investigated in the pH range of 3.5–10. Dilute HNO_3 or NaOH was used for pH adjustment. The pH of the suspensions was measured by a pH meter (Multi 340i, WTW Company, Germany). After determining the optimum time for Hg(II) adsorption, the kinetic study was performed. Samples were collected at appropriate time intervals to analyze the mercury concentrations. The Hg(II) concentrations were also 150 $\mu\text{g/ml}$. The adsorption isotherms of Hg(II) on laterite were investigated in the Hg(II) concentration range of 100–300 $\mu\text{g/ml}$. A blank experiment (without adsorbent) was performed to check that no Hg(II) consumption occurred other than by adsorption on laterite from Guizhou (i.e., not on the vessel walls) and it was also checked the mass balance in the quantitative analysis of Hg(II) in the solid phase.

2 Results and discussion

2.1 Effect of mineral compositions

The relative contents of minerals in the laterite samples from Guizhou are listed in Table 1. Table 2 presents the details of the adsorption of Hg(II) on laterite from Guizhou. Different mineral compositions in the laterite cause differences in the adsorption capacity of laterite to Hg(II). Sample No. 1 shows the best adsorption capability, with an adsorption ratio of 67.88%. The reason is that the relative contents of illite and non-crystalloids in Sample No. 1 are high, up to 17.99% and 27.79%, respectively.

Table 1 Relative contents of minerals in laterite from Guizhou (%)

Sample No.	1	2	3	4	5	6	7	8	9	10
Sampling site	Transitional zone from laterite to carbonate rock	3.3 m	2.3 m	1.8 m	1.3 m	0.8 m	Ferrous concretion band	0.4 m	Surface soil (0 m)	Kaolinite band
Quartz	2.48	2.22	5.64	3.16	9.68	3.70	1.11	34.89	40.80	25.37
Kaolinite	3.98	13.94	8.62	9.08	13.17	12.02	6.34	5.36	–	60.54
Chlorite	0.78	4.48	4.50	6.36	4.52	5.62	2.07	3.87	6.48	–
Montmorillonite	1.03	1.32	1.18	3.09	3.29	2.24	1.03	Y	–	–
Illite	17.99	7.42	4.15	7.95	6.79	4.04	3.23	10.29	–	14.08
Hornblende	0.93	–	–	–	–	–	1.56	–	–	–
Gibbsite	1.43	5.44	7.81	7.25	4.11	9.53	1.68	Y	3.83	–
Anatase	1.20	2.09	2.92	3.09	4.42	2.58	1.61	3.73	4.58	–
Feldspar	–	–	5.33	2.13	2.90	5.73	Y	6.72	8.30	–
Plagioclase	2.08	2.06	3.30	2.49	2.88	4.65	0.86	–	–	–
Calcite	1.57	1.58	2.05	Y	2.13	3.05	1.16	–	–	–
Dolomite	10.94	1.23	1.56	1.36	1.85	2.69	0.90	–	–	–
Iron mineral	14.62	9.73	12.64	13.39	9.04	7.27	49.72	13.64	14.37	–
Non-crystalloid	27.79	36.91	28.87	31.37	18.59	21.75	19.19	11.45	11.30	–

Y: minor amount of the mineral.

Table 2 Details of adsorption of Hg(II) on laterite from Guizhou

Sample No.	1	2	3	4	5	6	7	8	9	10
Adsorption ratio (%)	67.88	12.76	10.77	9.39	15.20	15.92	16.67	17.30	63.43	7.17
Adsorption amount ($\mu\text{g/g}$)	2,037	383	323	282	456	478	500	519	1,903	215

Illite and non-crystalloids are the main contributors to the adsorption of Hg(II) on laterite. Illite is good at adsorbing Hg(II). Sample No. 10 shows the worst adsorption capability. Its adsorption ratio is only 7.17%. The reason is that the relative contents of kaolinite in Sample No. 10 are as high as up to 60.54%, and kaolinite is not good at adsorbing Hg(II). This is consistent with that reported in the literature (Li *et al.*, 2001).

2.2 Effect of pH on the adsorption capacity

As seen in Fig.1, the alkalescent environment is favorable to the adsorption of Hg(II) on laterite from Guizhou. The amount of adsorbed Hg(II) increases slowly in the range of lower pH, and increases sharply in the range of higher pH, and then reaches the maximum at a certain pH. Then, the adsorption capacity reduces slowly with increasing pH. For Sample No. 1, a slow but gradual increase of Hg(II) adsorption in the pH range of 3.5 to 7.17 and a sharp increase in the pH range of 7–9 were observed. When the pH is 9, the amount of adsorbed Hg(II) will reach the maximum. Then, the amount of adsorbed Hg(II) reduces slowly in the pH range of 9–10. The amount of adsorbed Hg(II) on Sample No. 10 increases slowly

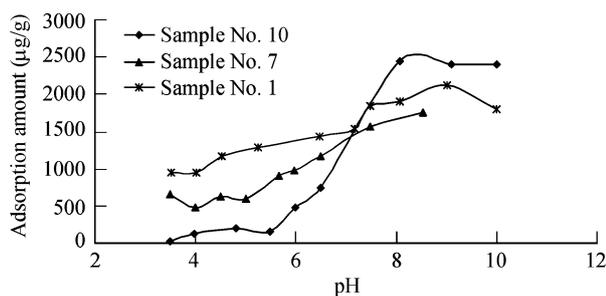


Fig. 1 Effect of pH on the adsorption of mercury on laterite from Guizhou.

with pH from 3.5 to 5.5, increases sharply with pH from 5.5 to 8, and reaches the maximum at pH 8. Then, the adsorbed capacity reduces slowly in the pH range of 8–10. The pH shows a similar influence on the adsorption of Hg(II) on Sample No. 7. However, when the pH was about 10, the solution was present in colloid state and could not be centrifuged. The reason needs to be further studied.

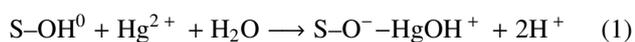
The pH can not only affect the existing state of the adsorbate, but also the characteristics of the external charges of the sorbent. Thus, pH is the main factor affecting the adsorption of heavy metals.

First, the reason why the higher pH is favorable to the adsorption of Hg(II) on laterite is explained from the viewpoint of Hg(II) speciation. The Hg(II) speciation depends mainly on the pH of the solution. At lower pH (pH 3–4), Hg(OH)⁺ and especially Hg²⁺ are dominating species in the solution and the concentrations of H⁺ ions are high. There is competitive adsorption between H⁺ ions and Hg²⁺ ions at the binding sites of laterite. This will affect the exchange adsorption of Hg(II). Therefore, the amount of adsorbed Hg(II) is reduced with decreasing pH. As seen in Hg(II) speciation diagram reported in literature (Manohar *et al.*, 2002), in the presence of Cl⁻, the dominant Hg(II) species at pH > 4.0 is Hg(OH)₂. The species such as HgCl₂ or Hg(Cl₂)₂ and Hg(OH)Cl are also present in small concentrations. For the pH ranging from 4 to 8, the concentrations of mercury hydroxides increase exponentially. Hg(II) is mainly hydrolyzed in the form of Hg(OH)₂, which is easier to be adsorbed than HgCl₂ (Liu *et al.*, 2007; Walcarius and Delacote, 2005; Wang *et al.*, 2003b). The effect of H⁺ ions will be reduced gradually at this moment. The main adsorption function of Hg(II) on laterite is still exchangeable adsorption. Hg(II) ions in the solutions form hydroxides that are hard to dissolve when the pH value is above 5. The laterite that acts as crystalline seeds can accelerate the precipitation of

hydroxides. During the precipitation, co-precipitation will occur. Complexation adsorption may also take place on the surface of laterite. These factors act cooperatively to promote the adsorption of heavy metal ions on the surface of laterite (Chen *et al.*, 2003; Huang *et al.*, 2002; Mishra *et al.*, 2004; Wuhan University, 1995; Zhao *et al.*, 2001). Therefore, higher pH values are favorable to the adsorption of Hg(II) on laterite. When the pH reaches a certain value, the concentration of OH⁻ in the solution will increase with increasing pH (pH > 8). This makes Hg(OH)Cl become more active than Hg(OH)₂ (Wang *et al.*, 2003b), and at the same time, the cooperation effects of exchangeable adsorption, complexation adsorption, precipitation, and co-precipitation reach the maximum. Thus, the amount of adsorbed Hg(II) decreases slowly.

As viewed from the surface structure of soil, the reason why the pH can affect the adsorption of Hg(II) is described as follows. The soil colloid surface is amphoteric. The fact that the surface of soil colloids adsorbs H⁺ makes the soil colloids positively charged, and the adsorption of OH⁻ on the surface of soil colloids makes the colloids negatively charged (Li, 2001). Significant adsorption does not occur at lower pH because the net surface charge of the laterite is positive in this pH range and the adsorption of cationic Hg(II) species such as Hg²⁺ and Hg(OH)⁺ is unfavorable owing to electrostatic repulsion (Manohar *et al.*, 2002). When the pH of the solution is higher, the electropositive Hg(II) will be adsorbed on the negatively charged soil colloid surface. Therefore, the amount of adsorbed Hg(II) will increase with increasing pH.

The mechanism of the adsorption of Hg(II) on laterite can be illustrated by the following surface complexation reactions:



where, S-OH⁰ indicates the amphoteric surface hydroxyl groups (Si-OH or Al-OH) (Benhammou *et al.*, 2005). From the two reactions, we can see that the pH is a decisive factor controlling the direction of the reaction. When pH is low, the concentrations of H⁺ in the solution will increase. The reaction proceeds towards the left side, namely, the reaction proceeds towards the desorption. Hg(II) is not easily adsorbed. On the contrary, when pH is high, the reaction proceeds towards the right side. This is favorable to the adsorption of Hg(II) on the laterite.

2.3 Isothermal adsorption

As seen in Fig.2, the influence of the initial concentrations of Hg(II) on the adsorption capacity of laterite shows the same trend. However, the extent of the influence is different. The amount of adsorbed Hg(II) of Sample No. 1 increases gradually with the increasing of initial concentrations of Hg(II), then reaches the maximum, and finally, reaches the equilibrium. The optimal adsorption concentration of Hg(II) is 250 µg/ml. However, the influence of the initial concentrations of Hg(II) on the adsorption capacity of Sample No. 7 is insignificant. The amount of adsorbed Hg(II) on Sample No. 7 increases rather slowly, while for

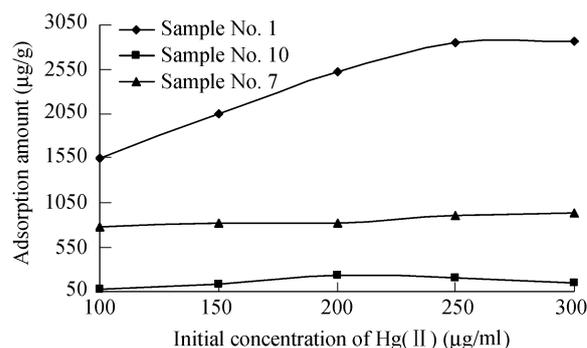


Fig. 2 Hg(II) adsorption isotherms on laterite from Guizhou.

Sample No. 10, the amount of adsorbed Hg(II) tends to increase first and then decrease. The optimal adsorption concentration of Sample No. 10 (purer kaolinite) is 200 µg/ml.

The reason is that at the first stage of the reaction, Hg(II) occupied the low energy adsorption sites where affinity is relatively weak. At that moment, Hg(II) can be adsorbed on more sites of laterite. Therefore, at lower initial concentrations, the adsorption rate of Hg(II) on laterite is high. With Hg(II) initial concentrations increasing, the low energy adsorption sites on laterite reach saturation gradually. At the same time, Hg(II) occupies the high energy adsorption sites. The speed of adsorption on the interface decreases, therefore, the adsorption rate decreases.

The Langmuir model is presented as follows:

$$\frac{1}{S} = \frac{1}{S_m} + \frac{1}{S_m k} \times \frac{1}{C} \quad (3)$$

where, S (µg/g) is the amount of adsorbed adsorbate by unit mass of soil at equilibrium, S_m (µg/g) is the maximum amount of adsorbed adsorbate by unit mass of soil, and k is the constant related to the adsorption energy, C (µg/g) stands for the equilibrium concentration of adsorbate in the solution.

The adsorption isothermal curves of Sample No. 1, 7, and 10 can be described using the Langmuir model. In this way, we obtain the adsorption isotherm equation $y = 0.1625x + 0.0003$ with correlation coefficient $r = 0.9922$ for Sample No. 1. The adsorption isotherm equation for Sample No. 7 is $y = 0.6237x + 0.001$ with correlation coefficient $r = 0.9408$, and the adsorption isotherm equation for Sample No. 10 is $y = 21.9380x + 0.0008$ with correlation coefficient $r = 0.8523$. From the above analysis, it is found that the adsorption behavior of Hg(II) on laterite can be accurately described by the Langmuir model with a high relativity.

From the slope of the three equations, we can get the maximum amount of adsorbed Hg(II) on laterite and the adsorption constant, i.e., S_m (Sample No. 1) = 3,333 µg/g, $k = 1.85 \times 10^{-3}$; S_m (Sample No. 7) = 1,000 µg/g, $k = 1.60 \times 10^{-3}$; S_m (Sample No. 10) = 1,250 µg/g, and $k = 3.65 \times 10^{-5}$. k represents the binding energy of soil colloid surface to ions. The larger the k value, the larger the binding energy of the soil colloid surface to ions, the stronger the adsorption capacity of soil colloid to ions

(Wu *et al.*, 2003), and the larger the maximum amount of adsorbed Hg(II) on the soil. The sequence, k (Sample No. 1) $> k$ (Sample No. 7) $\geq k$ (Sample No. 10) suggested that the adsorption capacity of Sample No. 1 is better than that of Sample No. 7, and is far better than that of Sample No. 10. The gradation of the maximum amount of adsorbed Hg(II) on laterite samples is similar to that of k values. This coincides with the above experimental results.

The fact that the adsorption isotherm data of Sample No. 1, 7, and 10 can be described with the Langmuir model also suggested that the adsorption of Hg(II) on the laterite is a monolayer binding process (Fu, 1990). Physical adsorption and chemical adsorption take place on the surface of laterite together. Physical adsorption refers to the adsorption at low energy site, where the adsorption heat is low and the adsorption rate is high. The adsorption could reach equilibrium easily and also desorption could occur easily. Chemical adsorption refers to the adsorption at high energy site, where the adsorption heat is high and the adsorption speed is slow. The adsorption can not reach equilibrium easily and desorption is hard also. Therefore, the amount of adsorbed Hg(II) increases rapidly in the case of low Hg(II) concentrations and increases slowly in the case of high Hg(II) concentrations (Sample No. 1 and 10).

2.4 Adsorption kinetics

As listed in Table 3, the adsorption of Hg(II) on laterite is a rapid reaction. For Sample No. 1, about 80% Hg(II) was adsorbed within 15 min. The speed of adsorption decreased gradually after 15 min and reached equilibrium at 22th hour. For Sample No. 7 (ferrous concretion band), about 80% Hg(II) was adsorbed within 2 h. The speed of adsorption decreased gradually after 2 h and reached equilibrium at 20th hour. The adsorption of Hg(II) on kaolinite is a slow reaction. For Sample No. 10 (purer kaolinite), about 20% Hg(II) was adsorbed within 2 h. The amount of adsorbed Hg(II) increased slowly and did not reach equilibrium at 24 h. At the moment, the adsorption ratio was only 5.34%.

Diffusion coefficient (D_i) is an important parameter, which reflects the distribution of solutes in the liquid and solid phases. It also reflects the adsorption trend on the

soil surface. D_i is calculated from the Boyd formula (Yang, 1986):

$$\sqrt{D_i} = \frac{R\sqrt{\pi}}{6} \times \frac{A_0 - A_t}{A_0 - A_\infty} \times \frac{1}{\sqrt{t}} \quad (4)$$

where, R is the particle diameter (cm), A_0 ($\mu\text{g/ml}$) is the initial concentration of adsorbate, A_t ($\mu\text{g/ml}$) is the adsorbate concentration at time t , A_∞ ($\mu\text{g/ml}$) is the equilibrium concentration of the adsorbate, and $\frac{A_0 - A_t}{A_0 - A_\infty}$ is the ratio indicative of the approaching equilibrium.

The influence of contact time on the diffusion coefficient of mercury during the adsorption reaction is shown in Fig.3. The D_i value decreased sharply during first 2 h and then decreased slowly. In the first stage of the reaction, since the saturation degree of mercury on the laterite surface is low, there will be many adsorption sites in laterite, which can adsorb Hg(II) in the solution. Thus, the adsorption rate was high and the D_i value was high as well. For instance, D_i (Sample No. 1, $t = 0.25$ h) = 4.98×10^{-5} cm^2/h when $R = 0.0149$ cm, and $A_0 = 150$ $\mu\text{g/ml}$. As the amount of adsorbed Hg(II) increased rapidly with increasing contact time, the adsorption sites on laterite surface were saturated by mercury ions and thus, D_i decreased. For example, D_i (Sample No. 1, $t = 2$ h) = 0.66×10^{-5} cm^2/h when $R = 0.0149$ cm and $A_0 = 150$ $\mu\text{g/ml}$. It decreased as much as 86.7%. Therefore, the equilibrium state can be judged in terms of the ratio of the diffusion coefficient D_i .

The reason why D_i decreased with the prolongation of contact time is described as follows. Adsorbed Hg(II) on the non-exchangeable sites prevented Hg(II) in the solutions from being adsorbed on the exchangeable sites. Therefore, the diffusion rate of Hg(II) was restrained on the surface of laterite particles (Tiller *et al.*, 1979). According to the theory of double electric layer of soil colloids (Chen, 1990), in the laterite samples of this study, illite is the main contributor to the adsorption of Hg(II). The colloid surface of illite charges permanently negative charges. The charge of Hg(II) is opposite to that on the colloid surface. Hg(II) passes through the double electric layer and reaches the colloid surface rapidly. This will reduce the potential difference between the potential ions layer and the diffusion layer. When the potential difference between

Table 3 Effect of contact time on the adsorption of Hg(II) on laterite from Guizhou

Contact time (h)	Adsorption amount ($\mu\text{g/g}$)		
	No.1	No.7	No.10
0.25	1,696	435	1.5
0.50	1,705	449	3.3
0.75	1,706	456	5.4
1.00	1,711	506	6.6
1.50	1,736	511	24.6
2.00	1,753	602	39.3
2.50	1,800	617	43.5
3.00	1,852	629	46.8
5.00	1,923	632	51.9
10.00	1,948	664	82.5
15.00	1,956	688	103.8
20.00	2,078	714	121.2
22.00	2,117	716	134.7
24.00	2,116	716	160.2

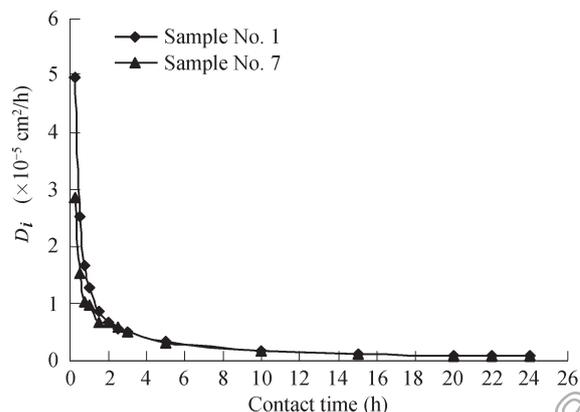


Fig. 3 Influence of contact time on the diffusion coefficient of mercury.

the two layers reverses, an antiparticle layer will come into being, which will prevent Hg(II) from diffusing further into the colloid surface. Thereby, at the start of the adsorption reaction, the D_i value and the adsorption ratio are large and tend to decrease with increasing contact time. The amount of adsorbed Hg(II) will reach equilibrium.

2.5 Implications

Laterite from Guizhou Province possesses a better capacity of adsorbing HgCl₂. The amount of adsorbed HgCl₂ on laterite is 2,037 µg/g and the maximum amount of adsorbed HgCl₂ on laterite is 3,333 µg/g. Laterite derived from carbonate rocks is widespread in Guizhou Province. The laterite is abundant and is also available without any cost. This suggests that the laterite is a promising environmental material which can be used in the removal of Hg(II) from wastewater. In addition, the laterite can be possibly used for the protection or remediation of natural water bodies.

To avoid the secondary contamination of Hg(II), a desorption experiment can be performed for spent laterite. The reagents of the desorption could be NaCl, HCl, KCl, and H₂O. By adjusting the pH of the desorption solution to acidic, and by stirring the solution the desorption ratio can be increased (Walcarius and Delacote, 2005; Zhang *et al.*, 2006). The regenerated laterite can be reused and the desorbed Hg(II) can be purified and reused as well. The experiment on the desorption of Hg(II) from laterite will be conducted in the future study.

3 Conclusions

The adsorption capacity of laterite to Hg(II) depends mainly on the mineral compositions, pH, and the initial concentration. Among the minerals in laterite from Guizhou Province, illite and non-crystalloids are the main contributors to the adsorption of Hg(II). pH is an important factor affecting the adsorption of Hg(II) on laterite. The alkaline environment (pH 7–9) is favorable to the adsorption of Hg(II) on laterite. The amount of adsorbed Hg(II) increases with increasing pH. When the pH of the solution reaches a certain value, the amount of adsorbed Hg(II) will reach the maximum. Then, the amount of adsorbed Hg(II) decreases with increasing pH of the solution. The optimal pH values of laterite and kaolinite are 9 and 8, respectively. The optimal initial concentrations of Hg(II) on laterite and kaolinite are 250 and 200 µg/ml, respectively. The adsorption isotherms are described by the Langmuir model. The adsorption of Hg(II) on laterite is a quick process. For Sample No. 1, about 80% Hg(II) was adsorbed within 15 min, and for Sample No. 7, about 80% Hg(II) was adsorbed within 2 h. The adsorption of Hg(II) on kaolinite is a slow reaction. After twenty-four hours, the adsorption ratio of Hg(II) on kaolinite was only 5.34%. The laterite from Guizhou Province is a promising environmental material which can be used in the removal of Hg(II) from wastewater.

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