Desalination and Water Treatment
www.deswater.com

doi:10.5004/dwt.2018.22210

115 (2018) 64–73
May

Characteristics of activated carbon from sludge and peanut shell and its application for phenol adsorption

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Received 2 July 2017; Accepted 10 March 2018

ABSTRACT

In this study, activated carbon (AC) was prepared from municipal sludge with peanut shell as the carbon additive and the ferric chloride as activator. Then the properties of AC activated by ferric chloride (FAC) and non-activated (NAC) were analyzed. Main results show that FAC has extremely higher BET surface area (781.5 m^2/g) and pore volume (0.859 m^3/g) than NAC, which implies that ferric chloride can effectively promote the formation of developed pore structures of AC. Moreover, the obtained ACs were used as adsorbent to remove phenol in aqueous solution. Results indicate that FAC and NAC have a higher adsorption capacity in the range pH of 7–9 and 1 g/L is the most suitable dosage of AC, considering of the removal rate and adsorption capacity. The adsorption kinetic data and adsorption isotherm data are well described by the pseudo-second order model and Freundlich isotherm model. The maximum equilibrium adsorption capacity is 109.58 mg/g (1 g/L, pH 8.0, 298K). In summary, it is an effective method to prepare AC from sludge and a small amount of peanut shell by using ferric chloride as activator and to remove phenol. This provides a potential option for sludge resource recovery.

Keywords: Activated carbon; Ferric chloride; Phenol absorption; Sludge resource recovery

1. Introduction

Sludge is the by-product of the sewage treatment plant and it contains a certain amount of organic compounds, heavy metals and pathogenic microorganisms. The sludge treatment and disposition have received much attention due to large amount of yields and the potential harm to the environment. Common sewage sludge treatment methods, including landfill, incineration, farmland utilization and so on, have many disadvantages. With the increasingly stringent environmental requirements of the public, our society urgently need some low cost and environment-friendly sludge treatment methods.

Research shows that the preparation of activated carbon (AC) from the pyrolysis of sludge rich in organic matter can avoid the disadvantages of the general sludge disposal method. Furthermore, the obtained AC can be used as a kind of adsorbent to remove the pollutants in aqueous solution [1]. Many researchers [2–4] have studied the preparation

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of sludge-based activated carbon (SAC), and the obtained products have been widely used in the adsorption of heavy metals, organic pollutants and even gaseous contaminants. However, due to the limited content of organic matter in the sludge, SAC usually has lower BET surface area and less developed pore structure which limits its application in practice [5–7]. In order to improve the quality of SAC, researchers found that the addition of carbonaceous matters in sludge can effectively increase the content of organic matter in the precursor, thus improving the BET surface area and pore volume of SAC. Additionally, some researchers added the coconut shell to sludge, which significantly increased the BET surface area of final products [6]. In other researches, pistachio almond, hazelnut, walnut and apricot were added to the sludge respectively, and the BET surface area of prepared ACs reached a higher range (635 m^2/g–1208 m^2/g) [7].

The preparation methods of AC are usually divided into direct carbonation, physical activation and chemical activation. Chemical activation is the most widely used, which has the advantages of low activation temperature, better porous structure and higher BET surface area of final product [8–11]. Chemical activation involves the impregnation of raw materials and chemicals. The common chemical agents are sulfuric acid, phosphoric acid, potassium hydroxide, zinc chloride and so on, among which zinc chloride is the most widely used [12]. Researches showed that ferric chloride and zinc chloride have similar activation effect. Due to the fact that the iron particle size is relatively small [13], the AC activated by ferric chloride always have a smaller pore size, which leads to a higher efficiency in removing phenol and other small molecule pollutants [14]. There have been studies using ferric chloride to activate the ligno cellulosic matter [15] like Arundodonax, Linn [16], rayon [17] and other matters for the preparation of AC. However, fewer reports are found to prepare AC from sludge by using ferric chloride as activator.

In this study, with peanut shell as carbon source additive, the ferric chloride as activator, municipal sludge was used as the main raw material to prepare AC. Further, the produced product was used as an adsorbent to remove phenol in aqueous solution. The purpose is to explore a new method of preparing AC with high quality from sludge and to verify the feasibility of removing phenol.

2. Materials and methods

2.1. Materials and chemicals

Sludge was from an urban treatment plant in the north of Shenyang city (Shenyang, China). The peanut shell, as the additional raw material, was from Shandong ZhengDa Food Processing Plants (Weifang, China). $FeCl_3·6H_2O$ and phenol were purchased from Sino-pharm Chemical Regent Co., Ltd (Shanghai, China). All chemical reagents used in this research were of analytic grade and the solutions were prepared with deionized water.

2.2. Preparation of AC

The process of preparing AC was as follows: (I) sludge sample and peanut shell were pre-dried to constant weight at 105°C in a dry oven, then grinded to sizes ranging from 1 to 2 mm; (II) mixed the sludge with peanut in a 3:1 mass proportion (sludge: peanut shell) to generate the precursor; (III) 5 g of precursor obtained was well mixed with 50 ml ferric chloride solution which prepared by dissolving 4.13 g $FeCl_3·6H_2O$ into 50 ml deionized water (the mass ratio of ferric chloride to precursor is 0.5) and another 5 g precursor was merely mixed with 50 ml deionized water, then two kinds of mixture were placed in thermostatic stirrers to fully impregnate for 12 h at 298 K respectively; (IV) two kinds of mixture were dried at 105°C for 24 h in a dry oven; (V) dried samples were packed into a stainless steel boat and then placed in a tube furnace (TCGC-1400, Hefei Kejing Material Technology Co. Ltd., Hefei, China.) under a continuous flow of nitrogen (100 cm^3/min) heated at a constant rate 10°C / min and held at 700°C for 1.5 h; (VI) took out the products and washed them with 0.5 mol/L hydrochloric acid solution, then repeatedly rinsed them with hot deionized water until the pH of the washed solution became stable ; (VII) dried the washing products at 105°C for 24 h and grounded them into powder with a size of 0.074 mm–0.15 mm; (VIII) named the final product activated with ferric chloride “ferric-activated carbon (FAC)” and that not activated with activator“non-activated carbon (NAC)”.

2.3. Characteristics of AC

The pyrolysis process of activator and precursor were analyzed by the Thermo gravimetric analyzer (TG-DTA-50, Shimadzu corporation, Japan). The concentration of the heavy metal in leaching liquor of raw sludge and FAC was measured by Flame atomic spectrophotometer (AA-700, Shimadzu, Japan). Specific surface area and pore size distribution analyzer (ASAP2020, Micro meritics Instrument Co. Ltd., USA) was applied to investigate the BET surface area, pore volume and the pore-size distribution of AC. The surface functional groups of AC were investigated by Fourier transform infrared (FTIR) spectrometer spectroscope (FT-08, Lumex Instruments, Canada).

2.4. Phenol adsorption test

2.4.1. AC dosage on phenol adsorption

The ability of FAC and NAC for adsorbing phenolin aqueous solution at different dosages (0.5–2.5 mg/L) was examined in batch adsorption test. AC was added to iodine flasks of 40 ml phenol solution (100 mg/L) and then shook them in a shaking incubator (TS-1210B-2, Suzhou Flying Test Instrument Co. Ltd., Suzhou, China) on 160 rpm at 298 K for 24 h. The residual phenol solution was filtered through 0.45 μm micro porous membranes, and then analyzed by high performance liquid chromatography analyzer (1260, Agilent Technology Co. Ltd., USA) with C18 as the stationary phase (Micro sorb MW-100-5) and a mixture of methyl alcohol and deionized water (80:20, vol.) as the mobile phase. In this process, the flow rate was maintained at 1 ml/min and the wavelength was set at 270 nm. The adsorption capacity of phenol at equilibrium, (mg/g), is calculated by:

$$q_e = (C_0 - C_e)V/W \quad (1)$$

where C_0 and C_e (mg/L) are the initial concentration and equilibrium concentration of phenol. V (L) is the volume of phenol solution, and W (g) is the mass of AC.

2.4.2. Initial pH on phenol adsorption

40 ml phenol solution was poured into iodine volumetric flasks with pH value in the range of 4–11. The initial concentration of phenol is 100 mg/L. 0.04 g AC was added in the flacks. Then those flacks were put in a shaking incubator (TS-1210B-2,Suzhou Flying Test Instrument Co., Ltd., Suzhou, China) and shaken at a constant speed of 160 rpm at 298 K. The equilibrium concentration (C_e) was measured after 24 h. The pH was measured with a pH-measuring instrument (PHS-3, Shanghai Precision Instrument Technology, Shanghai, China).

2.4.3. Adsorption kinetics

0.04 g AC was filled in iodine volumetric flasks of 40 ml phenol solution. The initial concentration is 100 mg/L and the pH is 8.0. Then those flasks were shaken on 160 rpm and at 298 K. Pseudo-first order model,Pseudo-second order model [18] and Intra-particle diffusion model [19] were used to fit the kinetic data. The Pseudo-first order model described mono-nuclear adsorption process, the Pseudo-second order model described binuclear adsorption process [20], and the Intra-particle diffusion model was used to predict the rate-limiting step of the adsorption process [21]. These models are expressed as:

Pseudo-first order mode $\ln(q_t - q_e) = \ln q_e - k_1 t$ (2)

Pseudo-second order model $t/q_t = 1/k_2 q_e^2 + t/q_t$ (3)

Intra-particle diffusion model $\ln q = K_{int} t^{0.5} + C$ (4)

where q_t (mg/g) is the adsorption capacity of phenol at time t (min), K_1 (1/min) is the adsorption rate constant, K_2 (g/(mg·min)) is the rate constant of second-order equation, K_{int} (mg/g·min$^{1/2}$) is the intra-particle diffusion rate constant and C (mg/g) is the film thickness of intra-particle diffusion adsorption.

2.4.4 Adsorption isotherms

Equilibrium isotherm experiments were carried out at 298, 308, and 318K over initial phenol concentrations of 25–250 mg/L. 0.04 g AC was added to 40 ml phenol solution with the pH 8.0. Two of most common adsorption isotherm equations, Langmuir [22] and Freundlich, were applied to fit the experimental data. These equations can be written as:

The Langmuir model $q_e = q_m K_L C_e / (1 + K_L C_e)$ (5)

The Freundlich model $q_e = K_F C_e^{1/n''}$ (6)

$R_L = 1/(1 + k_L C_0)$ (7)

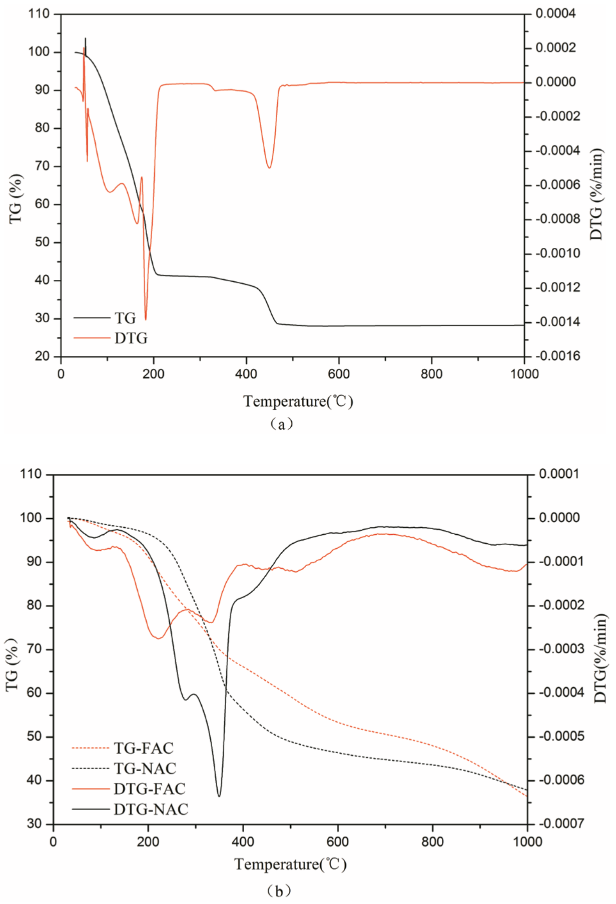
where q_m (mg/g) and K_L (L/mg) are Langmuir constants related to adsorption capacity and the energy of sorption. K_F (mg/g) and $1/n$ are Freundlich constants related to the adsorption capacity and sorption strength respectively. R_L is a dimensionless equilibrium constant reflecting the essential characteristic of the Langmuir equation, which indicates the type of isotherm to be irreversible ($R_L = 0$), favorable ($0 < R_L < 1$), linear ($R_L = 1$) or unfavorable ($R_L > 1$) [23].

3. Result and discussion

3.1. Pyrolysis behaviors

Thermal analysis of activator and the precursors of FAC and NAC are shown in Fig. 1. From the TG analysis of activator (Fig. 1a), it can be seen that the first mass loss was observed during 50–200°C , because the release of the crystallization water from activator. The second mass loss occurred in the temperature range of 400–500°C, which may be due to the decomposition of the ferric chloride.

TG curves of the precursor of FAC (Fig. 1b), can be divided into four periods in the range of 100–1000°C . The first large mass loss at 100–200°C is due to the evaporation of the surface water and the physically bound water in precursor. The second mass loss in the temperature range of 200–500°C is caused by the organic matter (protein and carbohydrate) in sludge and the decomposition of organic sub-



(a)

(b)

Fig. 1. The TG-DTG analysis curves, (a) Activator, (b) the precursor of FAC and NAC.

stances such as lignin, cellulose in peanut shell. The mass loss of third stage at 500–700°C is slight. In this stage, the refractory material and the residual organic matter in second stage are decomposed. The fourth stage (800–1000°C) mainly involves the decomposition of inorganic matter and the formation of charcoal from carbonization [23]. Observing the TG curves of the precursor of NAC (Fig. 1b), no obvious mass loss occurred in the temperature range of 500–700°C and 800–1000°C . Besides, the second phase of the maximum mass loss is delayed by about 80°C, comparing with that of FAC. These results suggest that ferric chloride can promote the deep decomposition of organic matter in precursor, which contributes to further formation of pore structures [23].

3.2. Textural properties and surface morphology analysis

The N_2 adsorption/desorption isotherms and the pore-size distribution curves of FAC and NAC are shown in Fig. 2. BET surface area, total pore volume (V_t), micro porous volume (V_m), average pore diameter (D_p), and the ratio of micro porous volume to total volume (V_m/V_t) are presented in Table 1.

In Fig. 2a, the isotherm of FAC and NAC have the characteristics of the IV type isothermal. N_2 adsorption curve has a rapid growth under low pressure due to the pore filling process and then the isotherms approaches horizontal, which indicates that the micro porous have been completely filled with gas. There appears a hysteresis loop in the high pressure region, which is caused by the capillary condensation.

The pore-size distribution curves of AC are shown in Fig. 2b. FAC and NAC are mainly mesoporous, which is due to the fact that sludge also cantains a large amount of inorganic substances. However, the pore size of almost all holes is less than 10 nm. In addition, there is still a considerable number of micro porous in FAC and the accounted for 34.06% of the (Table 1).

In Table 1, the BET surface area of FAC is 781.51 m^2/g, which is almost 2.6 times of NAC in the same preparation conditions, the and of FAC are much larger than NAC. In addition, it is worth noting that the (0.2926 cm^3/g) of FAC is close to a kind of powdered coconut shell activated carbon (MAC) (0.356 cm^3/g). The above results indicate that the ferric chloride promoted the formation of the pore structure in the process of sludge pyrolysis, thus significantly increase the BET surface area and total pore volume.

The surface morphology of FAC and NAC can be observed clearly by scanning electron microscope (SEM). The observed results of FAC and NAC from SEM are shown in Fig. 3. The surface of NAC (b) is uneven, and exists clastic structure. There are a small number of holes, but no regular pore structure. It can be clearly observed that the surface of FAC (a) have dense holes of different sizes, comparing with NAC, which indicates that more developed pores formed after the activation of the ferric chloride.

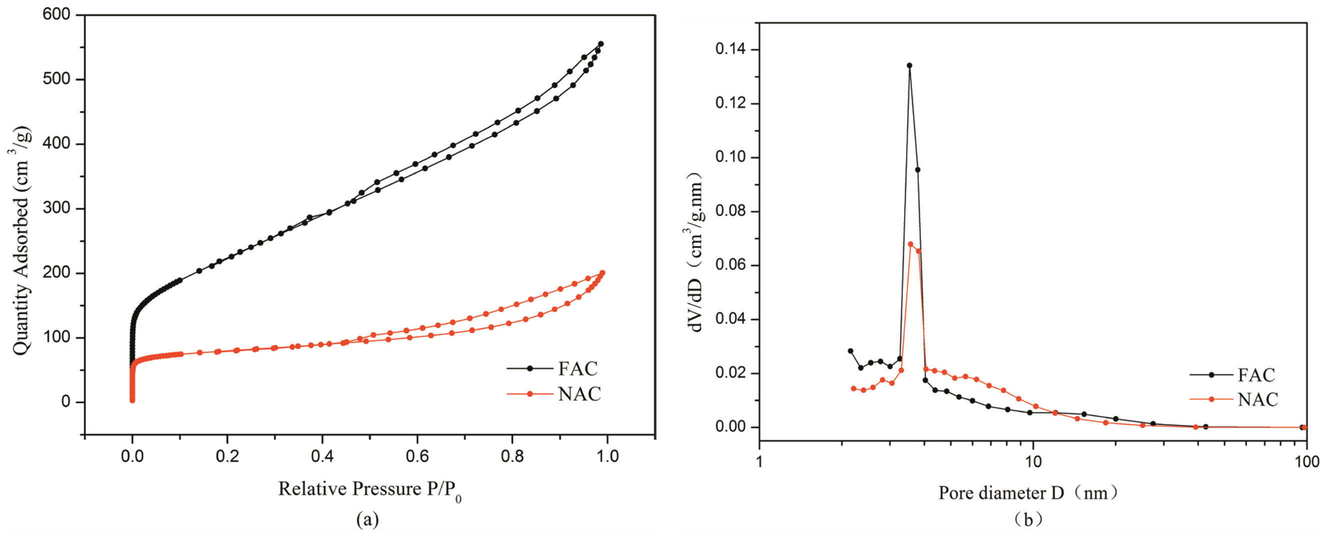


Fig. 2. (a) N_2 adsorption/desorption isotherms, (b) pore size distributions of the FAC and NAC.

Table 1
BET surface area and porosity of different kinds of AC

Sample	BET surface area (m^2/g)	V_t (cm^3/g)	V_m (cm^3/g)	V_m/V_t	D_p (nm)
MAC	894.26	0.587	0.356	60.64%	2.625
FAC	781.51	0.859	0.293	34.06%	3.493
NAC	294.95	0.272	0.087	31.94%	4.199

V_t: total pore volume; V_m: microporous volume; D_p: average pore size.

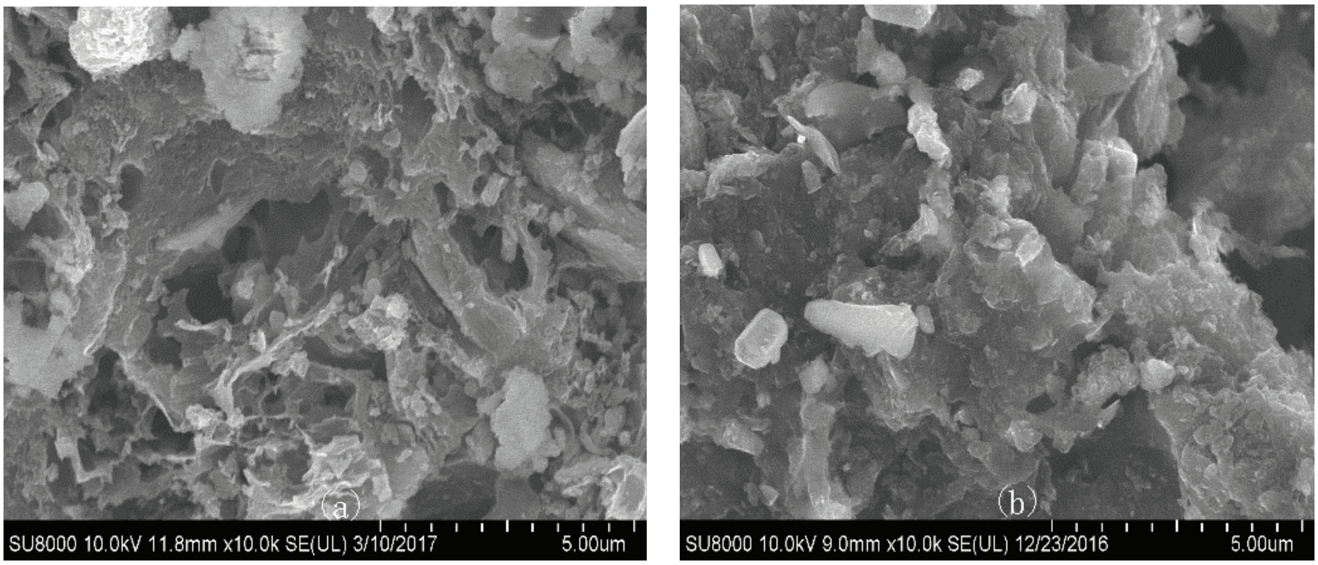


Fig. 3. SEM micro graphs of ACs, (a) FAC, (b) NAC.

3.3. Surface groups analysis

The surface groups of AC were characterized qualitatively by FTIR spectrometer. The obtained infrared spectra (400 cm^{-1}–4000 cm^{-1}) of the FAC and NAC are presented in Fig. 4. The short sharp peak at about 683 cm^{-1} indicates the existence of bending vibration of the alcohol hydroxyl or phenolic hydroxyl group. The broad peak located at around 1033 cm^{-1} is assigned to stretching vibration and the intensity of FAC is stronger than the NAC, which suggests that the content of corresponding functional groups in FAC is higher than that of NAC. The bands appearing around 1570 cm^{-1} is corresponded to the vibration in the aromatic rings [25]. The peak in 1853 cm^{-1} is stretching vibration of carbonyl groups. The bands at 2108 cm^{-1} and 2270 cm^{-1} can be attributed to the presence of groups and groups respectively.

3.4. Leaching properties of sludge and FAC

Sludge contains a certain amount of heavy metals, which pose a potential threat to environment. The content of heavy metal in leaching liquor of raw sludge and FAC were measured respectively. The leaching liquor was extracted according to horizontal vibration method (HJ 557-2010). This study just investigated the leaching properties of the FAC, because FAC and NAC were prepared from the same raw materials. As shown in Table 2, the content of various heavy metals in the leach liquor are lower than the *Standard for Pollution Control on the Landfill Site of Municipal Solid Waste* (GB16889-2008) and *Identification Standards for Hazardous Wastes Identification for Extraction Toxicity* (GB5085.3-2007), thus its leaching toxicity remains within a safe level. In addition to Pb and Cd, the concentration of heavy metals in FAC leaching solution is lower than that of sludge, which indicates that the heavy metals in the sludge were cured, and many soluble heavy metal compounds have changed into insoluble substances in the process of sludge carbonization.

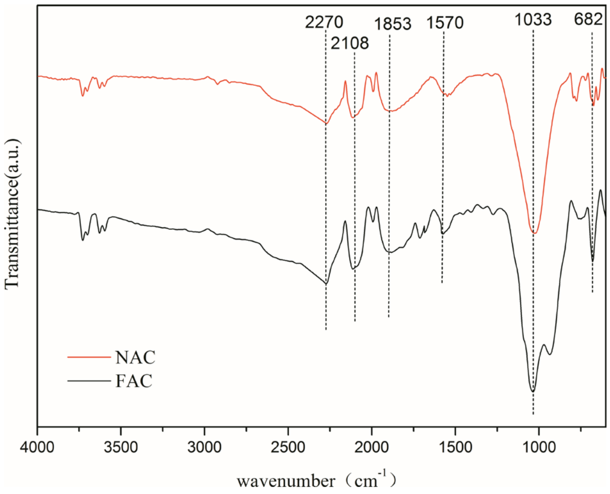


Fig. 4. FTIR analysis of FAC and NAC.

3.5. Adsorption of phenol

3.5.1. Effect of dosage on phenol adsorption

As shown in Fig. 5, with the increase of the dosage of AC, the removal rate of phenol is gradually increasing, but the adsorption capacity is negatively correlated with the increasing dosage. The increased removal rate is due to more adsorption sites provided by AC. The reason for the decrease of adsorption capacity is mainly that the interaction of electrostatic adsorption sites or over-lapping of the adsorption sites, which decreases the overall surface area of the AC and increases the diffusion path length [26].

3.5.2. Effect of initial pH on phenol adsorption

PH in solution is an important factor affecting phenol adsorption, because the pH value can affect the surface

Table 2
Results of extraction toxicity (mg/L)

Heavy metal	Cu	A_S	Zn	Pb	Cd	Cr	Ni	Hg
Leaching liquor from sewage sludge	0.310	0.006	0.0135	$<10^{-3}$	$<10^{-3}$	0.012	0.171	0.012
Leaching liquor from FAC	0.005	$<10^{-3}$	$<10^{-3}$	$<10^{-3}$	$<10^{-3}$	<0.003	0.022	0.003
Standard for Pollution Control on the Landfill Site of Municipal Solid Waste (GB16889-2008)	40	0.3	100	0.25	0.15	4.5	0.5	0.05
Identification standards for hazardous wastes Identification for extraction toxicity (GB5085.3-2007)	100	5	100	5	1	15	5	0.1

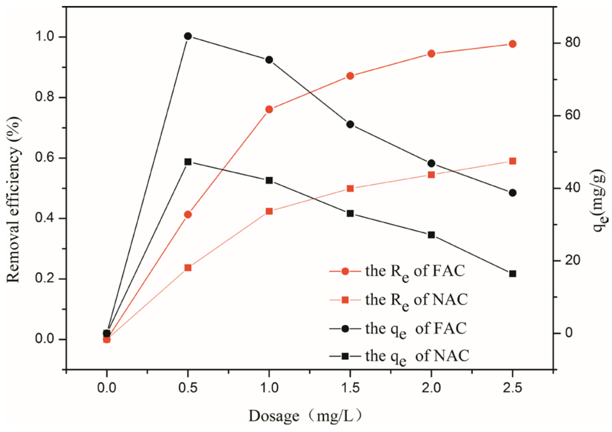


Fig. 5. Effect of AC dosage on phenol adsorption.

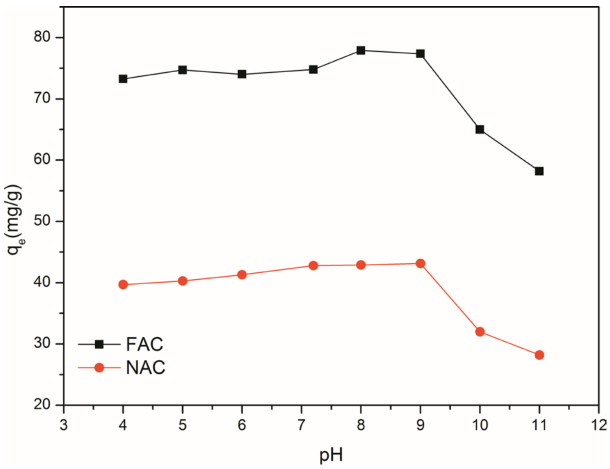


Fig. 6. Effect of solution pH on phenol adsorption.

charge of the adsorbent and the ionization degree of the adsorbate [27]. The adsorption capacity of phenol increases slightly with the range from 4 to 9, then sharply decreases when pH > 9 (Fig. 6). Phenol is a kind of weak acid compounds with $pK_a \approx 9.89$, and it exists mainly in the form of ions when $pH > pK_a$ [28].The adsorption of phenol on FAC and NAC reduces at high pH values is due to the electrostatic repulsion between the negative charge of the surface and the phenolate-phenolate anions in solution [29]. Similar results also been observed in the adsorption of phenol on commercial activated carbon [30].

3.5.3 Analysis of adsorption kinetics

From Fig. 7a, the adsorption capacity of phenol on FAC and NAC versus contact time at a constant initial concentration (100 mg/L). With the increase of contact time, the adsorption capacity is gradually increasing until adsorption reaches equilibrium. The adsorption capacity almost reaches 90% of the equilibrium adsorption capacity in the first hour, then the adsorption rate gradually decreases. The fitting results of three kinetic models are shown in Fig. 7b, Fig. 7c and Fig. 7d respectively. The calculated kinetics parameters are presented in Table 3. For both NAC and FAC, Pseudo-second order model is more suitable to describe the adsorption kinetics than Pseudo-first order model, because the correlation coefficient (R^2) is higher than Pseudo-second order model and the values of $q_{e,cal}$ (calculated equilibrium adsorption capacity) close to the $q_{e,exp}$ (experimental equilibrium adsorption capacity). These results indicate that the adsorption process proceeded by chemisorption. However, the correlation coefficients of the Pseudo-first order are also reasonably well fitted, so it can be inferred that adsorption system also exists physical adsorption.

As shown in Fig. 7d, the intra-particle multi-linearity plot of phenol adsorption on FAC and NAC present three stages. The first stage shows a sharp increase, which is due to the exist of external surface adsorption. The second shows slowly increase by boundary layer (film) diffusion, which indicates this portion is the rate-limiting step. The third stage usually is the final equilibrium stage. The lower correlation constant suggests that the inner-diffusion can be neglected during the process of adsorption [31].

Because the plot does not pass through the origin and K_{int2} is greater than those of K_{int3} (Table 3), the adsorption process is mainly controlled by boundary layer (film) diffusion.

3.5.4. Analysis of adsorption isotherms

The adsorption isotherms of phenol on the NAC and FAC are presented in Fig. 8 and the fitting parameters are summarized in Table 4. The adsorption capacity of FAC is higher than NAC (Fig. 8a and Fig. 8b), which is mainly due to the fact that FAC have a larger BET and a more developed pore structure including the total pore volume and

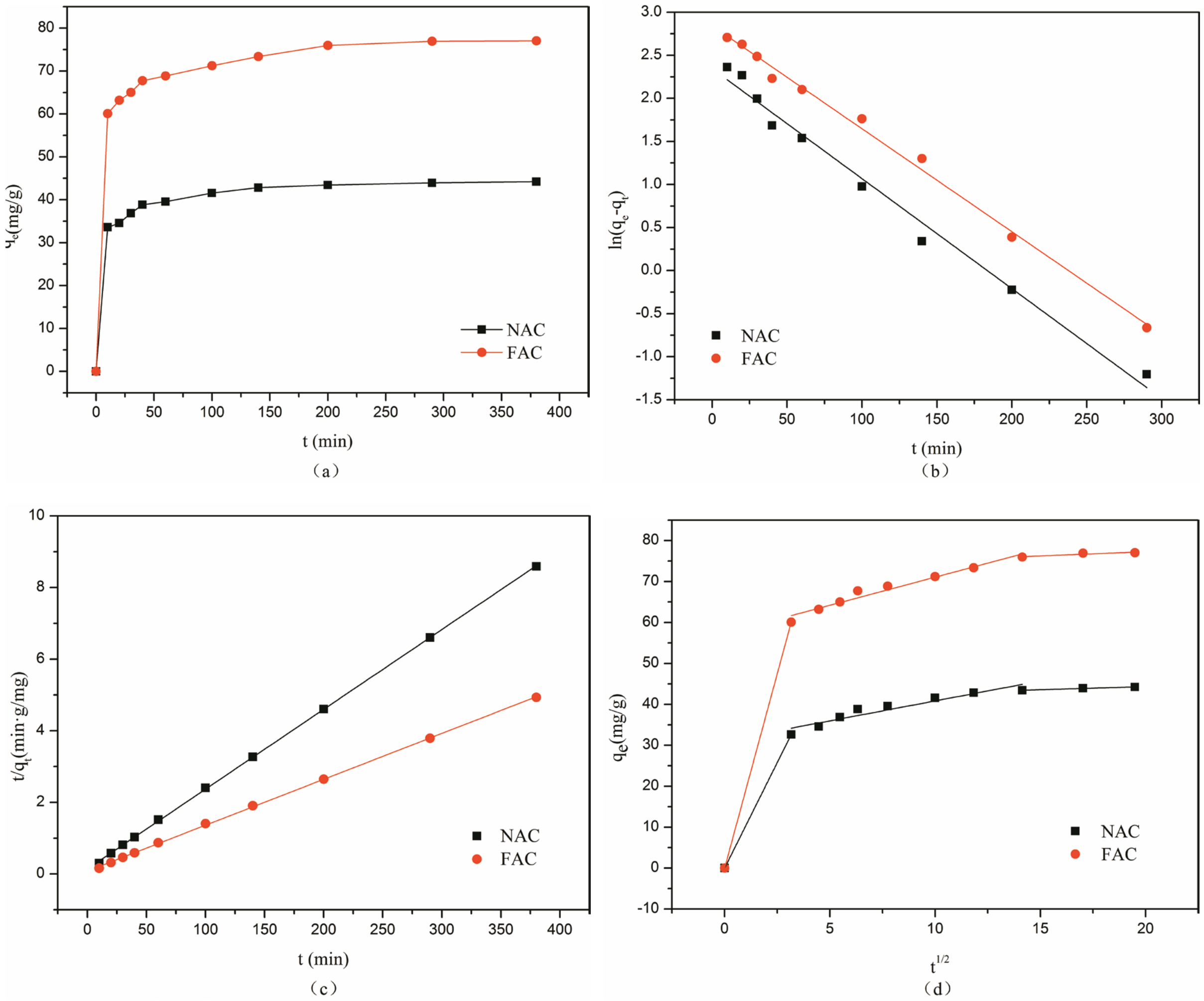


Fig. 7. Effect of contact time on adsorption of phenol (a), (b) Pseudo-first-order kinetics model, (c) Pseudo-second-order kinetics model, (d) the intra-particle diffusion model.

Table 3
Kinetics parameters of the Pseudo-first order, Pseudo-second and Intra-particles diffusion model

Samples	Pseudo-first order model			
	$q_{e,exp}$ (mg/g)	$q_{e,cal}$ (mg/g)	K_1 (1/min)	R^2
NAC	44.23	10.43	0.013	0.986
FAC	77.05	17.17	0.012	0.994
	Pseudo-second order model			
	$q_{e,exp}$ (mg/g)	$q_{e,cal}$ (mg/g)	K_2 (1/min)	R^2
NAC	44.23	43.48	0.159	0.997
FAC	77.05	78.13	0.160	0.999

Intra-particle diffusion model									
	K_{int}			C			R^2		
	K_{int1}	K_{int2}	K_{int3}	C_1	C_2	C_3	R_1^2	R_2^2	R_3^2
NAC	1.770	0.975	0.150	0	31.09	41.33	1	0.89	0.98
FAC	3.227	1.373	0.205	0	57.34	73.19	1	0.95	0.72

$q_{e,exp}$: experimental equilibrium adsorption capacity. $q_{e,cal}$: calculated equilibrium adsorption capacity

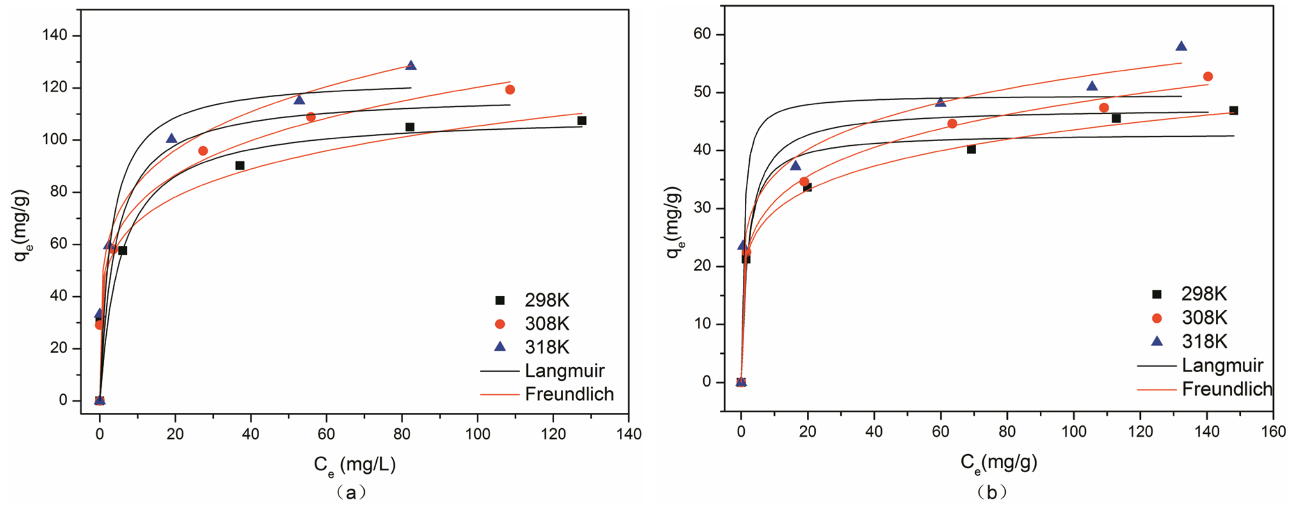


Fig. 8. Adsorption isotherms of phenol on AC, (a) FAC, (b) NAC.

Table 4
Adsorption isotherm parameters for FAC and NAC

Sample	Temperature(K)	Langmuir				Freundlich		
		q_m (mg/g)	K_L (L/mg)	R_L	R^2	K_F (mg/g)	$1/n$	R^2
NAC	298	43.03	0.571	0.008	0.949	20.01	0.169	0.999
	308	47.36	0.459	0.011	0.930	20.23	0.189	0.987
	318	49.63	1.41	0.004	0.900	24.38	0.193	0.997
FAC	298	109.58	0.183	0.023	0.882	44.89	0.186	0.980
	308	117.65	0.25	0.018	0.897	47.04	0.204	0.982
	318	124.21	0.349	0.013	0.877	52.04	0.205	0.979

Table 5
Comparison of phenol adsorption onto different sludge activated carbon and some commercial activated carbon

Raw material	Activator	BET surface area (m^2/g)	Adsorption capacity (mg/g)	Reference
Municipal sludge (25% peanut shell)	Ferric chloride	781.51	109.58	This study
Sewage sludge	Nitric acid	121	96.15	[32]
Sewage sludge	Citric acid and Zinc chloride	792.4	188.94	[33]
Sewage sludge	Zinc chloride	195.28	45.12	[34]
Anaerobic sludge	Sulfuric acid	390	42,04	[35]
Surplus biological sludge	Sulfuric acid	253	26.7	[36]
Sewage sludge	None	60	9.8	[37]
CCM200 (carbon cryogel)	/	/	140	[38]
Chemical-PAC (from coal)	/	/	98	[39]

micro volume (Table 1). Besides, different surface chemical properties on FAC and NAC also have different effects on the adsorption of phenol.

The equilibrium data fits the Freundlich model well (Fig. 8a) and the correlation coefficients (R^2) are also higher than Langmuir model in different temperatures (Table 4), which implies the phenol covered by the adsorbent is heterogeneous. The parameters of Freundlich model, $1/n$ values, are all between 0.1 and 0.5 in different temperature (Table 4), which denotes that the adsorption of phenol by FAC and NAC is favorable. In this study, the maximum equilibrium adsorption capacity of NAC on phenol is 109.58 mg/g at 298 K, which is higher than many reported in literatures (Table 5).

It is observed that, with the increase of temperature, the adsorption capacity of FAC and NAC increases (Table 4), which indicates that the chemical reaction between adsorbent and phenol belongs to endothermic reaction and the increasing temperature is in favor of the adsorption of phenol.

4. Conclusion

A kind of activated carbon of high quality has been successfully prepared by ferric chloride activation, and it can effectively remove the phenol in solution, thus effectively realizing the resource utilization of sludge. The ferric chloride can promote the formation of more developed pore structure in the process of pyrolysis, and contributes to the increase of BET surface area and pore volume. The concentration of heavy metals in FAC leaching liquor is within the safe range of the relevant standards. FAC has a higher adsorption capacity for adsorbing phenol and can be classified as a kind of effective adsorbent.

Acknowledgements

We gratefully acknowledge the financial support by the Major Science and Technology Program for Water Pollution Control and Treatment of China (2012ZX07202-005,2014ZX07510-001).

Symbols

C (mg/g) — The film thickness of intra-particle diffusion adsorption
C_0 and C_e — The initial concentration and equilibrium concentration of phenol
K_1, K_2 and K_{int} — The adsorption rate constant of first-order equation, the rate constant of second-order equation and the intra-particle diffusion rate constant
K_F and $1/n$ — Freundlich constants related to the adsorption capacity and sorption strength
q_e — The adsorption capacity of phenol at equilibrium
q_m and K_F — Langmuir constants related to adsorption capacity and the energy of absorption
q_t — The adsorption capacity of phenol at time t (min)
R_L — A dimensionless equilibrium constant reflecting the essential characteristic of the Langmuir equation
V (L) — The volume of phenol solution
W (g) — The mass of AC

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