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Abstract: Bentonite was modified with cetyltrimethylammonium bromide (CTAB). The organically modified bentonite (OMB) was used to remove phenol from aqueous solution, the microstructural changes were characterized by X-ray diffraction (XRD) and scanning electronic microscopy (SEM), and phenol adsorption kinetic was obtained using batch adsorption test results. The results indicated that the rate of adsorption of phenol onto the OMB was positively correlated with the initial concentration, and the maximum adsorption capacity was found to be 10.1 mg/g at the initial concentration of 150 mg/L at 25 °C and pH 10. The investigations of adsorption kinetics models showed that the adsorption kinetic was better reflected by the pseudo-second-order kinetic model. Furthermore, the properties of the OMB samples with different adsorption times were obtained by SEM and XRD. The statistic analysis revealed that the pore diameter of the OMB samples decreased with the increasing adsorption time and gradually reached equilibrium.

Keywords: organically modified bentonite; adsorption; kinetics

1. Introduction

In order to purify wastewater, the research on efficient adsorbents for various pollutants has attracted the attention of researchers [1–5]. Bentonite, a non-metallic mineral with montmorillonite as the main mineral component, is a layered aluminosilicate clay mineral. It is one of the most widely used, non-metallic minerals [6]. Montmorillonite has a 2:1-type, layered, silicate structure, and its structural unit layer is composed of one layer of aluminum–oxygen octahedron sandwiched by two layers of silicon–oxygen tetrahedron. This special crystal structure enables bentonite to have obvious hydroexpansivity, as well as good adsorptivity and high ion-exchange capacity [6,7]. As bentonite is cheap and easy to obtain, it is widely used in industrial wastewater treatment, groundwater and soil pollution prevention and remediation, and nuclear waste treatment [8–10]. In geotechnical engineering, bentonite-doped materials have played an important role in the control of soil and water pollutants (such as being applied in cutoff walls for landfills) by virtue of the characteristics of high expansivity and low permeability of bentonite. These materials include geotechnical composite bentonite liners (GCLs), soil–bentonite cutoff walls, and cement–bentonite slurry walls [11–13].

Natural bentonite has its limitation in the field of environmental pollution treatment for these two reasons: its strong hydrophilicity and the hydrolysis of exchangeable cations...
between layers make it difficult to adsorb hydrophobic organic pollutants effectively; the negative charge on the surface of natural bentonite limits its adsorptivity for non-ionic and anionic pollutants [14]. Krol and Rowe [15] studied the barrier performance of soil–bentonite slurry walls against the organic pollutant trichloroethylene. The results showed that the adsorption coefficient of soil–bentonite slurry walls for trichloroethylene is 0 cm$^3$/g; that is, it is difficult for bentonite to adsorb trichloroethylene. Goodarzi et al. [16] investigated the effects of different organic pollutants on the macro- and microstructural response of sodium bentonite. They pointed out that the organic pollutants greatly reduce the compressibility of sodium bentonite and increase its permeability, causing a decline in the barrier property of bentonite. Liu et al. [17] explored the permeability of bentonite in an acidic environment. It was found that, with the increasing concentration of sulfuric acid solution, the expansivity of bentonite decreases, while its saturation permeability coefficient increases by an order of magnitude. In addition, it was pointed out by Ashmawy et al. [18] that, when the leachate is rich in calcium and magnesium ions, the permeability coefficient of the bentonite waterproof blanket increases due to ion exchanges, and its anti-seepage performance declines. Previous studies have shown that, due to the cation exchange phenomenon of bentonite, the permeability coefficient of bentonite in salt solution and acid increases significantly. Under conditions of a high concentration of salt or acid, bentonite will even crack and lose the expansivity and anti-seepage ability [19,20]. Thus, in the site environment with a high concentration of salt, acid, or organic pollutants, natural bentonite fails to show the expected performance in adsorptivity, expansivity, chemical compatibility, and anti-seepage property.

Based on the cation exchange between bentonite layers, natural bentonite can be modified into hydrophobic and oleophilic organic bentonite by exchanging inorganic cations between bentonite layers with organic cations or organic compounds. Thus, its chemical compatibility with organic pollutants can be improved so as to broaden its application in the field of environmental protection. Studies have shown that the structural characteristics of organically modified bentonite (OMB) are related to factors such as the type, concentration, dosage, and ratio of cationic surfactants [21]. By changing the dosage and ratio of surfactants and bentonite, the arrangement of organic molecules in the interlayer domain can be effectively controlled [22]. In the present research, the universally used organic cations for modification are quaternary ammonium salt cations with long alkyl chains, where cetyltrimethylammonium bromide (CTAB) surfactant is one of the most commonly used organic modifiers for bentonite [23–25]. After organic modification, the layer spacing of bentonite increases, which increases the storage space in the layer and, finally, increases the adsorption capacity for organic pollutants [26]. The experimental results show that, after organic modification of natural bentonite, the expansivity and compressibility of OMB are both reduced [27]. Furthermore, the length of carbon chain of surfactants is also an important factor affecting the structure and properties of OMB. Slaný et al. [28] investigated the effect of chain length on the arrangement of surfactants confined in montmorillonite interlayers using infrared spectroscopy in the mid-IR and near-IR regions and XRD and reported that the alkyl chain length has significant effect on the arrangement of the intercalated cations. The layer spacing, adsorption capacity, and thermal stability of OMB are directly proportional to the length of surfactant alkyl chain of surfactants [29–31]. Therefore, OMB with different structures and properties can be synthesized by using different surfactants according to various purposes and needs and is simpler and less expensive to prepare than anionic clays such as hydrotalcites [32].

Although there has been some research on phenol adsorption on OMB [33], no reports were found on the variation of the microscopic pore structure with adsorption time during the adsorption of phenol by OMB. To further explore the adsorption mechanism of OMB, it is necessary to investigate the microstructural changes of OMB under different adsorption conditions. In this paper, the organic modification of an artificial sodium bentonite was conducted with CTAB. Then, batch tests were carried out to investigate the adsorption process of OMB under different phenol solution concentrations, and the adsorption kinetic
was discussed. Furthermore, the changes of the microscopic characteristics of OMB in the phenol adsorption process were studied by scanning electron microscopy (SEM) and X-ray diffraction (XRD).

2. Materials and Methods

2.1. Test Materials

The artificial sodium bentonite used in this paper was purchased from Tianjin Tianyu Bentonite Technology Co., Ltd. (Tianjin, China), with an initial layer spacing of 1.493 nm. The main components of the artificial sodium bentonite were as follows: montmorillonite (78.3%), calcium feldspar (14.6%), quartz (6.4%), and orthoclase (0.7%). Moreover, the CTAB, phenol, 4-aminoantipyrine, potassium ferricyanide, ammonium chloride, and ammonia used in the test were chemically pure and purchased from Shanghai Aladdin Biochemical Technology Co., Ltd. (Shanghai, China).

2.2. OMB Preparation

In this experiment, CTAB was used as the surfactant and OMB was prepared by the wet method [34]. The preparation steps of OMB were as follows: Firstly, 100 g of sodium bentonite was weighed and put into a 1 L beaker, and 500 mL of deionized water was added to prepare 5% slurry. Secondly, 25 g of CTAB was weighed and added to the slurry, which was then stirred evenly. After shaking in a constant temperature shaker at 70 °C for 2 h, it was put into an electric air-drying oven at 70 °C for overnight aging until the cationic exchange between the surfactant and bentonite was completed. Thirdly, the prepared sample was taken out, filtered, and rinsed, and silver nitrate solution was used to detect bromide ions. When no bromide ion could be detected, the organic bentonite was put into an electric air-drying oven at 70 °C. After drying, the organic bentonite was taken out for grinding and passed through a 100-mesh sieve. Thus, the preparation of OMB was completed.

2.3. OMB Characterization

To study the changes in particle size, porosity, and layer spacing of montmorillonite, SEM and X-ray diffraction (XRD) experiments were performed on OMB samples before and after phenol adsorption with a scanning electron microscope (S-4800, Hitachi, Japan) and an X-ray diffractometer (D8 Advance, Bruker, Karlsruhe, Germany). A 0.075-mm sieve was used for crushing and preparing OMB powder samples with different adsorption times. Then, 10 mL of phenol solution was used to react with 0.25 g of soil samples at 25 °C. After centrifugation at 4000 r/min, the sludge at the bottom of the centrifuge tube was air dried and ground. XRD tests were then conducted after screening with a 0.075-mm sieve, and the measurements were conducted at 2–20° 2θ.

SEM was used to observe the changes of the microscopic characteristics of OMB before and after adsorption. The equilibrium adsorption time was determined by the phenol adsorption test of OMB. A third of the equilibrium adsorption time was selected as the time interval, and OMB samples with different adsorption times were scanned with a scanning electron microscope (S-4800, Hitachi Company, Tokyo, Japan) at 0 min, 20 min, 40 min, and 60 min, respectively, to observe the microstructural change characteristics of OMB. Then, functions, such as count, image processing, and dimensional measurement of Image Pro Plus software, were used to process the scanned images so as to obtain the particle diameter change of OMB and analyze its microstructural change rule in the process of adsorbing phenol.

2.4. Phenol Adsorption Test of OMB

2.4.1. Determination of the Calibration Curve of Phenol Solution

The 4-aminoantipyrine method was used to determine phenol in aqueous solutions, and the steps were as follows: Firstly, 0.00, 0.50, 1.00, 3.00, 5.00, 7.00, 10.00, and 12.50 mL of the prepared 0.01 mg/mL phenol solution were added into 8 cuvettes, respectively.
Secondly, 0.5 mL of ammonium chloride buffer was added to make the phenol solution pH 10 ± 0.2. Thirdly, 1 mL of 4-aminoantipyrine was added, and the solution was fully oscillated. After that, 1 mL of potassium ferricyanide solution was added, and the sample with the added detection agent was shaken evenly. After 10 min, the solution was taken out from cuvettes and placed into the T6 ultraviolet spectrophotometer (Beijing Persee General Instrument Co., Ltd., Beijing, China). The absorbance was measured at the wavelength of 510 nm, and the calibration curve was drawn, as shown in Figure 1.

![Figure 1. The calibration curve of phenol solution.](image)

2.4.2. Adsorption Test

The steps of adsorption tests are as follows: Firstly, eight different centrifuge tubes were prepared, 500 mg of OMB was weighed into each centrifuge tube, and 30 mL of phenol solution with a concentration of 50 mg/L was added, respectively. Secondly, the centrifuge tubes were put into a constant temperature shaker with the temperature set at 25 °C and then vibrated at a rotational speed of 180 rpm. Eight sets of centrifuge tubes were vibrated for 10 min, 20 min, 30 min, 40 min, 50 min, 60 min, 90 min, and 120 min, respectively. Thirdly, samples were taken out and put into the centrifuge to separate the sample at 4000 rpm. After centrifugation for 10 min, the supernatant was obtained, and the absorbance was determined using the 4-aminoantipyrine method. Thus, the concentration of the adsorbed phenol solution could be obtained according to the calibration curve.

The above steps were repeated to obtain the adsorption results when the concentration of phenol solution was set at 100 mg/L and 150 mg/L, respectively. According to Equations (1) and (2), the removal efficiency and the amount adsorbed of phenol onto OMB can be calculated:

$$ R = \frac{(C_0 - C_e)}{C_0} \times 100\% $$  \hspace{1cm} (1)

$$ Q_e = (C_0 - C_e) \times \frac{V}{m} $$  \hspace{1cm} (2)

where $C_0$ is the initial concentration of phenol solution, mg/L; $C_e$ is the residual concentration of phenol solution after adsorption, mg/L; $V$ is the solution volume, L; and $m$ is the adsorbent dosage, g.

2.4.3. Adsorption Kinetic Analysis of OMB

To analyze the adsorption kinetic of OMB, three kinds of adsorption kinetic model, including the pseudo-first-order equation, the pseudo-second-order equation, and the intraparticle diffusion equation, were used to fit the adsorption test results, the equations of which are shown in Equations (3)–(5) [35]:

Pseudo-first-order kinetic model: \[ \ln(q_e - q_t) = \ln q_e - k_1 t \]  \hspace{1cm} (3)
where $q_e$ is the equilibrium adsorption capacity, mg/g; $q_t$ is the adsorption capacity at time $t$ (min), mg/g; and $k_1$ is the adsorption rate constant for pseudo-first-order adsorption, min$^{-1}$.

Pseudo-second-order kinetic model: \[
\frac{t}{q_t} = \frac{1}{(k_2q_e^2)} + \frac{t}{q_e}
\]

where $k_2$ is the pseudo-second-order adsorption rate constant, g/(mg min).

Intraparticle diffusion model: \[
q_t = k_3t^{0.5} + C
\]

where $k_3$ is the intraparticle diffusion rate constant, mg/(g min$^{1/2}$); and $C$ is a constant for any experiment, mg/g.

3. Results and Discussion

3.1. Effect of Phenol Solution Concentration

The phenol adsorption curves of OMB at different phenol solution concentrations are shown in Figure 2. It can be seen from the figure that the pollutant concentration has an extremely obvious influence on the adsorption property of OMB. When the initial concentration of phenol solution was 50 mg/L, the adsorption rate of the whole adsorption process was small, the adsorption quantity increased slowly with time, and the adsorption peak was not obvious. When the initial concentration was 100 mg/L, the adsorption rate was high in the first 40 min, the adsorption quantity increased rapidly, and the adsorption reached equilibrium in 60 min. After that, desorption occupied a certain advantage, and the adsorption quantity decreased to a certain extent. When the initial concentration was 150 mg/L, the adsorption rate was high in the first 60 min, the adsorption quantity increased rapidly, and the adsorption reached equilibrium in 90 min. After that, desorption occupied a certain advantage, and the adsorption quantity decreased to a certain extent. The research results of Cheng [36] on the phenol adsorption of OMB showed that, with the increasing pollutant concentration, the removal rate and unit adsorption quantity also increase, which is consistent with the results of this study. This may be because, as the concentration increases, more contaminants come into contact with the OMB surface. With the increasing pollutant concentration, the phenol adsorption quantity of the OMB also increases. For phenol solution with initial concentrations of 50 mg/L, 100 mg/L, and 150 mg/L, the maximum adsorption quantities were 3.3 mg/g, 6.7 mg/g, and 10.1 mg/g, respectively. Diaz-Nava et al. [7] reported that the adsorption of phenol at pH 11 and 100 mg/L for surfactant-modified bentonite could be up to 6.0 mg/g, which is lower than the results measured here.
3.2. Adsorption Kinetic

Figures 3–5 show the kinetic fitting curves of the phenol adsorption of OMB, and Table 1 shows the adsorption kinetic parameters. Based on the fitting results, it can be seen that the pseudo-second-order kinetic model is in the best agreement with the adsorption test results. When initial concentrations of phenol solution were 50 mg/L, 100 mg/L, and 150 mg/L, the fitting values of the equilibrium adsorption quantity of the pseudo-second-order kinetic equation were 3.6 mg/g, 7.1 mg/g, and 10.5 mg/g, respectively. This result is the closest to the measured result. The adsorption rate constants of phenol solution were 22.4 g/(mg min), 19.6 g/(mg min), and 21.9 g/(mg min) under three initial concentrations. The study of Zhang et al. [33] showed that the adsorption results of OMB to organic dyes are best fitted with the pseudo-second-order kinetic model, which is similar to the conclusion in this paper. Comparing the adsorption results of three different initial concentrations of phenol solution with the results of the pseudo-second-order kinetic fitting models, it was found that, when the adsorption reached equilibrium, the removal rates of phenol with a concentration of 50 mg/L, 100 mg/L, and 150 mg/L were 65.695%, 66.434%, and 67.264%, respectively. The adsorption efficiency of the OMB increased with the increasing initial concentration, and similar results were observed by Zhang et al. [35].

Table 1. Phenol adsorption kinetic parameters of OMB.

| Adsorbent Concentration, mg/L | 50    | 100   | 150   |
|-------------------------------|-------|-------|-------|
| Pseudo-first-order kinetic equation |       |       |       |
| $q_e$                         | 2.1802| 2.9432| 2.6267|
| $k_1$                         | 0.0337| 0.0533| 0.0235|
| $R^2_1$                       | 0.8913| 0.6985| 0.7511|
| Pseudo-second-order kinetic equation |       |       |       |
| $q_e$                         | 3.6054| 7.1382| 10.5252|
| $k_2$                         | 0.0223| 0.0196| 0.0202|
| $R^2_2$                       | 0.9904| 0.9946| 0.9986|
| Intraparticle diffusion equation |       |       |       |
| C                             | 1.5720| 3.5595| 7.0880|
| $k_3$                         | 0.1660| 0.3398| 0.3075|
| $R^2_3$                       | 0.9753| 0.6378| 0.8386|

Figure 3. Adsorption fitting model for OMB using the pseudo-first-order kinetic equation.
3.3. Microscopic Characterization of OMB with Different Adsorption Times

Scanning results of the samples under the 100 mg/L phenol solution with different adsorption times are shown in Figure 6. When the adsorption time was 0 min, OMB was in a lamellar-stacked state with large pores, and the layer spacing of OMB was loose. After phenol adsorption for 20 min, the lamellar accumulation of OMB was relatively dense, the porosity was relatively reduced, and the lamella was relatively compact, indicating that phenol has been adsorbed by OMB. After phenol adsorption for 40 min, the soil layer structure spacing and the porosity decreased obviously, suggesting that OMB had adsorbed phenol to a relatively saturated state. After phenol adsorption for 60 min, the structure of OMB was more stable than that of 40 min, implying that OMB had completely adsorbed phenol to a fully saturated state and the equilibrium adsorption had been reached.
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Figure 6. Scanning images of soil samples with different adsorption times (10,000×): (a) 0 min; (b) 20 min; (c) 40 min; (d) 60 min.

The scanning images of OMB with different adsorption times were statistically analyzed for the apparent porosity and pore diameter in unit area, and the microstructural changes of pores in OMB were obtained. When the adsorption time was 0 min, the pore diameter was mainly distributed between 0.05–0.77 µm, and the apparent porosity was 0.362. After phenol adsorption for 20 min, the pore diameter was mainly distributed between 0.05–0.5 µm, and the apparent porosity was 0.152. After phenol adsorption for 40 min, the pore diameter was mainly distributed between 0.05–0.17 µm, and the apparent porosity was 0.064. After phenol adsorption for 60 min, the pore diameter was mainly distributed between 0.05–0.2 µm, and the apparent porosity was 0.034. It can be seen that the macropores of OMB give priority to phenol adsorption at the beginning, and the phenol adsorption quantity increases with the passage of time. With the increase of time, the macropores gradually decrease, and the pores within the grain gradually increase. The main reason is that the phenol pollutants in the macropores constantly move to the small pores, which leads to the increasing number of bentonite crystal layers that adsorb phenol. At this time, montmorillonite is hydrated, aggregate is decomposed, the pore structure is homogenized, and the adsorption reaches the equilibrium.

According to the XRD diffraction test results of soil samples with different adsorption times (Figure 7), the layer spacing of the OMB before the adsorption test was 1.867 nm. The structure of the OMB changed with the progress of the adsorption process. After phenol adsorption for 20 min, the layer spacing increased slightly, with a value of 1.892 nm. After phenol adsorption for 40 min, the increase in the layer spacing was obvious, with
a value of 1.924 nm, and the peak value was increased. This is because the degree of crystallization increases as the adsorption proceeds. After phenol adsorption for 60 min, the layer spacing became smaller, with a value of 1.900 nm. That is, the interlayer structure of montmorillonite no longer expanded, but began to fill the pores and gradually reached the adsorption peak.

![XRD diffraction results of soil samples with different adsorption times.](image)

**Figure 7.** XRD diffraction results of soil samples with different adsorption times.

### 4. Conclusions

In this work, OMB was prepared by using sodium bentonite and CTAB, and was used to remove phenol from aqueous solution. The rate of adsorption of phenol onto the OMB and equilibrium time were affected by the initial concentration. For a phenol solution with initial concentrations of 50 mg/L, 100 mg/L, and 150 mg/L, the corresponding maximum adsorption amounts were 3.3 mg/g, 6.7 mg/g, and 10.1 mg/g at pH 10, respectively. By comparing the fitting results of three kinetic models, the adsorption data of phenol onto OMB were well described by the pseudo-second-order kinetic model, while the deviation between the other two models and the measured results might increase with the increase of initial concentration. Analysis of SEM images and XRD revealed that the microstructure of bentonite changed after modification, and the adsorption of phenol filled the interlayer pores of OMB and partially expanded the layer spacing. This study is promising for designing on the basis of bentonite organoclay as adsorbents for purification of organic pollutants which are similar to phenol.

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References

1. Luo, Q.; Huang, X.; Luo, Y.; Yuan, H.; Ren, T.; Li, X.; Xu, D.; Guo, X.; Wu, Y. Fluorescent chitosan-based hydrogel incorporating titanate and cellulose nanofibers modified with carbon dots for adsorption and detection of Cr (VI). *Chem. Eng. J.* 2021, 407, 127050. [CrossRef]

2. Luo, Q.; Huang, Y.; Lei, Z.; Peng, J.; Xu, D.; Guo, X.; Wu, Y. Wood-derived nanocellulose hydrogel incorporating gold nanoclusters using in situ multistep reactions for efficient sorption and sensitive detection of mercury ion. *Ind. Crop. Prod.* 2021, 173, 114142. [CrossRef]

3. Luo, Q.; Ren, T.; Lei, Z.; Huang, Y.; Huang, Y.; Xu, D.; Wan, C.; Guo, X.; Wu, Y. Non-toxic chitosan-based hydrogel with strong adsorption and sensitive detection abilities for tetracycline. *Chem. Eng. J.* 2022, 427, 131738. [CrossRef]

4. Luo, Q.; Yuan, H.; Zhang, M.; Jiang, P.; Liu, M.; Xu, D.; Guo, X.; Wu, Y. A 3D porous fluorescent hydrogel based on amino-modified carbon dots with excellent sorption and sensing abilities for environmentally hazardous Cr (VI). *J. Hazard. Mater.* 2021, 401, 123432. [CrossRef]

5. Yuan, H.; Ren, T.; Luo, Q.; Huang, Y.; Huang, Y.; Xu, D.; Guo, X.; Li, X.; Wu, Y. Fluorescent wood with non-cytotoxicity for effective adsorption and sensitive detection of heavy metals. *J. Hazard. Mater.* 2021, 416, 126166. [CrossRef]

6. Annan, E.; Nyankson, E.; Agyei-Tuffour, B.; Armah, S.K.; Nkrumah-Buandoh, G.; Hodasi, J.A.M.; Oteng-Peprah, M. Synthesis and Characterization of Modified Kaolin-Bentonite Composites for Enhanced Fluoride Removal from Drinking Water. *Adv. Mater. Sci. Eng.* 2021, 2021, 6679422. [CrossRef]

7. Diaz-Nava, M.; Olguin, M.; Solache-Rios, M. Adsorption of phenol onto surfactants modified bentonite. *J. Incl. Phenom. Macro.* 2012, 74, 67–75. [CrossRef]

8. Uddin, F. Montmorillonite: An Introduction to Properties and Utilization; IntechOpen: London, UK, 2018.

9. Min, F.; Wang, X.; Li, M.; Ni, Y.; Al-qadhi, E.; Zhang, J. Preparation of High-Porosity and High-Strength Ceramistates from Municipal Sludge Using Starch and CaCO₃ as a Combined Pore-Forming Agent. *J. Mater. Civ. Eng.* 2021, 33, 04020502. [CrossRef]

10. Xu, H.; Shu, S.; Wang, S.; Zhou, A.; Jiang, P.; ZHU, W.; Fan, X.; Chen, L. Studies on the chemical compatibility of soil-bentonite cut-off walls for landfills. *J. Environ. Manag.* 2019, 237, 155–162. [CrossRef]

11. Li, Q.; Jia, Z.; Zhao, Y. Laboratory evaluation of hydraulic conductivity and chemical compatibility of bentonite slurry for grouting walls. *Environ. Earth. Sci.* 2021, 80, 569. [CrossRef]

12. Liu, G.; Wu, L.; Ye, C.; Liu, Y.; Huang, Q.; Wen, M.; Liao, B.; Lu, T.; He, T. Study on controlling of cadmium pollution with fly ash-bentonite blocking wall. *Chemosphere* 2019, 228, 656–667. [CrossRef] [PubMed]

13. Wan, Y.; Xue, Q.; Liu, L.; Wang, S. Relationship between the shrinkage crack characteristics and the water content gradient of compacted clay liner in a landfill final cover. *Soils Found.* 2018, 58, 1435–1445. [CrossRef]

14. Huang, Z.; Li, Y.; Chen, W.; Shi, J.; Zhang, N.; Wang, X.; Li, Z.; Gao, L.; Zhang, Y. Modified bentonite adsorption of organic pollutants of dye wastewater. *Mater. Chem. Phys.* 2017, 202, 266–276. [CrossRef]

15. Krol, M.M.; Rowe, R.K. Diffusion of TCE through soil-bentonite slurry walls. *Soil Sediment. Contam.* 2004, 13, 81–101. [CrossRef]

16. Goodarzi, A.; Fateh, S.N.; Shekary, H. Impact of organic pollutants on the macro and microstructure responses of Na-bentonite. *Appl. Clay Sci.* 2016, 121, 17–28. [CrossRef]

17. Liu, Y.; Bouazza, A.; Gates, W.; Rowe, R. Hydraulic performance of geosynthetic clay liners to sulfuric acid solutions. *Geotext. Geomembr.* 2015, 43, 14–23. [CrossRef]

18. Ashmawy, A.K.; El-Hajji, D.; Sotelo, N.; Muhammad, N. Hydraulic performance of untreated and polymer-treated bentonite in inorganic landfill leachates. *Clays Clay Miner.* 2002, 50, 546–552. [CrossRef]

19. Bai, J.; Zhang, H.; Xiao, L. Formation mechanism of carbide slag composite sustained-alkalinity-release particles for the source control of acid mine drainage. *Sci. Rep.* 2021, 11, 23793. [CrossRef]

20. Huang, X.; Li, J.; Xue, Q.; Chen, Z.; Du, Y.; Yan, W.; Liu, L.; Poon, C.S. Use of self-hardening slurry for trench cutoff wall: A review. *Constr. Build. Mater.* 2021, 286, 122959. [CrossRef]

21. Han, H.; Rafiq, M.K.; Zhou, T.; Xu, R.; Mašek, O.; Li, X. A critical review of clay-based composites with enhanced adsorption performance for metal and organic pollutants. *J. Hazard. Mater.* 2019, 369, 780–796. [CrossRef]

22. Guo, Y.K.; Liu, J.H.; Gates, W.P.; Zhou, C.H. Organo-modification of montmorillonite. *Clays Clay Miner.* 2020, 68, 601–622. [CrossRef]

23. Wang, Y.; Jiang, Q.; Cheng, J.; Pan, Y.; Yang, G.; Liu, Y.; Wang, L.; Leng, Y.; Tuo, X. Synthesis and characterization of CTAB-modified bentonite composites for the removal of Cs+. *J. Radioanal. Nucl. Chem.* 2021, 329, 451–461. [CrossRef]

24. Guo, M.; Yang, G.; Zhang, S.; Zhang, Y.; Gao, C.; Zhang, C.; Zhang, P. Co-modification of Bentonite by CTAB and Silane and its Performance in Oil-Based Drilling Mud. *Clays Clay Miner.* 2020, 68, 646–655. [CrossRef]

25. Al Rubai, H.F.; Hassan, A.K.; Sultan, M.S.; Abood, W.M. Kinetics of Adsorption of Reactive Red 120 Using Bentonite Modified by CTAB and Study the Effect of Salts. *Nat. Environ. Pollut. Technol.* 2021, 20, 281–289. [CrossRef]

26. Andrunik, M.; Bajda, T. Modification of bentonite with cationic and nonionic surfactants: Structural and textural features. *Materials* 2019, 12, 3772. [CrossRef]

27. Perelomov, L.; Mandzhieva, S.; Minkina, T.; Atroshchenko, Y.; Perelomova, I.; Bauer, T.; Pinsky, D.; Barakhov, A. The synthesis of organoclays based on clay minerals with different structural expansion capacities. *Minerals* 2021, 11, 707. [CrossRef]
28. Slaný, M.; Janković, L.; Madejová, J. Structural characterization of organo-montmorillonites prepared from a series of primary alkylamines salts: Mid-IR and near-IR study. *Appl. Clay Sci.* 2019, 176, 11–20. [CrossRef]

29. Zhao, M.; Wei, L.; Zheng, Y.; Liu, M.; Wang, J.; Qiu, Y. Structural effect of imidazolium-type ionic liquid adsorption to montmorillonite. *Sci. Total. Environ.* 2019, 666, 859–864. [CrossRef]

30. Rivas-Rojas, P.C.; Ollier, R.P.; Alvarez, V.A.; Huck-Iriart, C. Enhancing the integration of bentonite clay with polycaprolactone by intercalation with a cationic surfactant: Effects on clay orientation and composite tensile properties. *J. Mater. Sci.* 2021, 56, 5595–5608. [CrossRef]

31. Intachai, S.; Suppaso, C.; Khaorapapong, N. A Novel Process for Intercalating Alkylammonium Ions in a Thai Bentonite and its Effect on Adsorption Performance. *Clays Clay Miner.* 2021, 69, 477–488. [CrossRef]

32. Tang, Y.; Slaný, M.; Yang, Y.; Li, S.; Qin, F.; Zhao, Y.; Zhang, Z.; Zhang, L. Highly active Mg–Al hydrotalcite for efficient O-methylation of phenol with DMC based on soft colloidal templates. *J. Chem. Technol. Biot.* 2022, 97, 79–86. [CrossRef]

33. Zhang, N.; Li, Y.; Lu, M.; Li, Z.; Wang, X.; Meng, y.; Yuxin, Z. Adsorption properties of organic bentonite for phenols in wastewater. *J. Taiyuan Univ. Technol.* 2018, 05, 681–685.

34. Lin, T.; Wang, J.; Yin, X.; Wei, X. Modification of Bentonite and Its Application in Antimicrobial Material. *Bentonite Modif.* 2020, 5, 54–61.

35. Zhang, L.; Tu, L.-y.; Liang, Y.; Chen, Q.; Li, Z.-s.; Li, C.-h.; Wang, Z.-h.; Li, W. Coconut-based activated carbon fibers for efficient adsorption of various organic dyes. *RSC Adv.* 2018, 8, 42280–42291. [CrossRef]

36. Cheng, T. Research on the Characteristics of Organic Modified Bentonite for Phenol Removing. *Henan Chem. Ind.* 2016, 12, 21–23.