



ORIGINAL ARTICLE

Utilization of Modified Clay as a Low-cost Adsorbent for Landfill Leachate Treatment

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(Received: 2 December 2019

Accepted: 30 September 2020)

KEYWORDS

Leachate treatment;
Adsorption;
Surfactant;
Organoclay;
Bentonite

ABSTRACT: The generation of leachate is one of the main problems of municipal solid waste disposal, which can lead to a severe threat to surface and groundwater. In this research, landfill leachate was treated through modifying bentonite with HDTMA-Br (Hexadecyltrimethylammonium bromide). The effects of surfactant intercalating quantity, pH, contact time, and adsorbent dose was investigated in a batch experiment. Raw and modified clay was also characterized using different techniques including Fourier transformation infrared spectroscopy (FTIR), X-ray diffractometry (XRD), scanning electron microscopy (FE-SEM), Brunauer-Emmett-Teller (BET), and Thermogravimetric analysis (TGA). The optimization experiment revealed that 50 min of equilibration, 50g L⁻¹ of dosage and pH= 3, for the clay loaded with a capacity of 1.0 CEC (cation exchange capacity), were the optimum condition for COD removal of landfill leachate. The maximum turbidity reduction rate was 92.1%, at a pH of 11, adsorbent dosage of 40 g L⁻¹, and the contact time of 30 min. The findings of this research showed that this modified clay could be used as an affordable adsorbent for reducing COD and turbidity from landfill leachate.

INTRODUCTION

The environmental condition is no more in calmness or stability but rather becomes much unperceivable on account of the depredation unrestrained contamination in the 21st century [1]. Currently, municipal solid waste is a major globally challenge. By industrializing of developing countries, Solid waste is threatening the environmental conditions. [2].

Even though the landfilling of municipal solid waste is inevitable, leachate formation with high contamination is recognized as one of the major challenges of this kind of

solid waste managing. [3]. The leachate is formed through the percolating of water into landfill and its reaction with the waste in an anaerobic environment, resulting in the concentration of anthropogenic organic compounds, organic matter and inorganic elements. Accordingly, municipal solid waste and landfill leachate have a variety of organic compounds [4]. In the case of poor management, landfill leachate can become a source of hydrogeological contamination due to its permeation on the natural environment and groundwater [5].

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DOI: 10.22034/jchr.2020.1884351.1071

Therefore, to prevent their adverse environmental effects, landfill leachates should be appropriately collected, stored and maintained by optimal treatment processes, and, finally, the effluent from the treatment steps should be disposed of in a suitable place [6].

A variety of techniques, including oxidation, precipitation, evaporation, adsorption, reverse osmosis, and ion exchange have been utilized for the removal of various contaminants originated from landfill leachate. Among these, adsorption is one of the most effectual techniques for wide-ranging of wastewater [1].

This method is advantageous owing to its simple operation, affordability, high effectiveness, easy adsorbent's design and, above all, lack of secondary contaminants throughout the adsorption procedure, hence it is one of the suitable techniques for the removal of pollutants. [7].

A large number of adsorbents, such as natural materials, agricultural waste and residues, industrial by-products, and biomass materials have been used for water and wastewater treatment [8].

Among these, natural mineral clays are also found to be abundant and low cost precursors for the preparation of adsorbents with exclusive specifications, including, high specific surface area, micro and meso porosity, high cation exchange capacity, and swelling features [9].

Naturally, clay minerals have been of a great interest as adsorbents owing to their low costs, environmental compatibility, high absorbability, and ion exchange features. [10].

However, since the hydration of inorganic cations in exchangeable sites leads to hydrophilicity the surface of clay, so the natural clays are inefficient adsorbents for organic matters. By loading the cationic surfactants in the interlayer of clay, its surface specifications changes from hydrophilicity to hydrophobicity and also the basal spacing of the layers significantly increases. So the applications of the clay can be affected by these changes in surface properties. [11].

Recently many studies have been investigated to increase the adsorption capacity of clays for organic contaminations. Their reports indicate that the absorbability of organic matters can be increased through substituting the natural inorganic cations in interlayer of clays with organic cations, including quaternary

ammonium cations.

The modification of bentonite by Benzyl dimethyl hexadecyl ammonium bromide (BDHDA) was employed to remove polyphenols from olive oil industry wastewater. The results showed that, it could remove 88% of orthodiphenol from synthetic wastewater under optimized conditions, whereas raw bentonite could only remove 30% of this contamination. The results also showed that modified bentonite with 50% CEC loading capacity had the highest adsorption capacity of polyphenols [39].

Liu et al. used Na-Mt modified with 3 types of cationic surfactants (TMA⁺, DDTMA⁺ and HDTMA⁺) to remove tetracycline from a synthetic solution. The results revealed that the clay modified with all three types of cationic surfactants had higher adsorption capacity than Na-Mt. They also observed that, HDTMA had higher adsorption capacity than the others three modifiers and could optimally increase the adsorption capacity of tetracycline to 160% under optimum conditions [40].

In another investigation CTAB-modified bentonite was used to remove Congo red dye from aqueous solutions. It was detected that bentonite modification led to better absorbability for acidic dyes than natural bentonite. The research also revealed that adsorption capacity increased with rising the initial concentrations of CR, the adsorbent dosage, temperature and pH (in the range of 5 to 9) [24].

Park et al. examined the effect of modified montmorillonite using two kinds of cationic surfactants (DDDMA and DDTMA) on the removal of phenolic compounds from water. They observed that the size of the surfactant could significantly increase the interlayer distance and adsorption capacity of the composite. Moreover, adsorption of PNP to adsorbent was better than PCP due to its hydrophobic properties [10].

Jourvand and his colleagues (2015) utilized HDTMA-modified montmorillonite for the removal of methylene blue from a synthetic solution and reported that the methylene blue adsorption capacity increased and then decreased with rising the loading capacity up to 70% CEC. They also observed that the contact time and initial concentration of methylene blue could positively affect the removal, but the ionic strength and pH had not significantly effects on the elimination of methylene blue [38].

The purpose of this research was to examine the impact of bentonite modification with cationic surfactant on the reduction of organic pollution and turbidity of landfill leachate. As the crystallinity, structure, and nanomechanical features of the clay polymer composites are significantly affected by the length of the organic modifier chain [41], a long-chain cationic surfactant was chosen here as the modifying agent. So in this work, bentonite was modified with hexadecyltrimethylammonium bromide (HDTMA-Br). The concentration of this surfactant ranged from 0.25 to 3 times the cation exchange capacity (CEC), and the FE-SEM, XRD, TGA, FTIR, BET, techniques were employed to analyze the structural characteristics of the resulting materials.

Most previous studies focused on synthetic wastewater or diluted leachate. In our research however, the treatment of raw landfill leachate was studied using modified clay by absorption method. The effects of various factors including pH, contact time, and adsorption dosage, were examined on the adsorption capacity. Such a study is of

great importance for treating wastewater with high organic contamination and incorporating various compounds including industrial effluent and landfill leachate.

MATERIALS AND METHODS

Landfill leachate

The landfill leachate specimens here were procured from the evaporation lagoons in the sanitary landfill of Mohammad Abad located in south of Qazvin, Iran. The area encompasses an extent of 110 ha and daily receives approx. 750 tons of municipal solid waste. This sanitary landfill has been working as of May 2013 and is anticipated to have 20 years of life span. The leachate was sampled in clean containers and instantly transferred to the lab on ice (4°C) in darkness to avoid decomposition. The collection of leachate samples was done repeatedly each week and mean results were obtained. Table 1 shows the specifications of landfill leachate specimens.

Table 1. Characterisation of leachate specimens

Parameter	Unit	Mean Value	Parameter	Unit	Mean Value
COD	Mg L ⁻¹	185907	Pb	Mg L ⁻¹	0.5
BOD ₅	Mg L ⁻¹	40500	Zn	Mg L ⁻¹	<1.5
Turbidity	NTU	1673	As	Mg L ⁻¹	1.1
pH	-----	7.4	Cd	Mg L ⁻¹	<0.03
Al	Mg L ⁻¹	2.5	Hg	Mg L ⁻¹	<0.6
Co	Mg L ⁻¹	<0.5	Total Iron	Mg L ⁻¹	12.5
Cr	Mg L ⁻¹	<0.5	Ca	Mg.L ⁻¹	10047.6
Cu	Mg L ⁻¹	<0.2	K	Mg.L ⁻¹	3988.97
Mg	Mg L ⁻¹	1.9	Na	Mg.L ⁻¹	9918.4

Materials

Bentonite was obtained from the mine in Qazvin, Iran. The copper bisethylenediamine complex procedure was used to calculate the cation exchange capacity (CEC) of this clay (40 meq 100g⁻¹) [35]. Table 2 shows the composition of bentonite utilized here obtained with the X-ray fluorescence analysis (XRF- PHILIPS PW1410). The modification of clay was performed using the

HDTMA-Br (C₁₉H₄₂BrN) as surfactant with a purity of 98% (Merck Co.). To adjust the leachate pH, 1M HCl and NaOH, were used. The rest of utilized chemical substances were analytical grades and utilized without no extra purification. Distilled water was used to prepare the entire solutions.

Table 2. Composition of row bentonite.

Sample	Al ₂ O ₃	SiO ₂	MgO	CaO	K ₂ O	Fe ₂ O ₃	Na ₂ O	TiO ₂	P ₂ O ₅	MnO	LOI
percentage	11.72	67.046	3.391	3.253	3.028	1.737	0.95	0.138	0.031	0.047	8.6

Purification of Bentonite

The clay, was cleaned through dispersing a certain quantity of bentonite into deionized water, followed by agitating the mixture with an agitator at 250 rpm at room temperature for 24 h and then centrifuging at 4000 rpm for 20 min. To dry the purified bentonite, it was then placed in an oven at 105 °C for 24 h [38].

Synthesize of the modified clay

Modification of bentonite was performed according to the following procedure:

After dispersing 30g of bentonite in 300 ml of deionized water, it was blended with a magnetic stirrer for around 30 min. The stoichiometric quantity of the surfactant (HDTMA-Br) was also dissolved in deionized water and stirred at 60°C for 30min, followed by gradual addition of the surfactant solution to the clay suspension. Clay modification was achieved by stirring the mixture at 60°C for 4 h [10, 16]. The cation exchange capacity of bentonite was 40 meq 100g⁻¹, which indicates that the clay was potentially loaded with the cationic surfactant. The impact of cationic surfactant's loading level on the reduction of the landfill leachate contamination, was determined through the clay intercalation by various levels of HDTMA-Br from 0.25 to 3 CEC.

The resultant modified clay was rinsed repeatedly by deionized water to completely remove free bromide anions, which was confirmed with AgNO₃ [10, 11, 12, 14 and 24]. To dry the final composite, it was placed in an oven at 60°C for 48 h to fixed weight. A planetary ball mill (120-rpm for 15 h) was employed to convert the dried modified clay into nanoparticles. The particle size was estimated circa 27.8 nm using the Scherrer equation [36, 37]. The final product was stored a sealed container for subsequent uses.

Characterization and analysis

Surface morphology of the natural and modified clay were characterized by a scanning electron microscope (FESEM, MIRA3). To determine the chemical structure and functional groups of nano clay, the FTIR spectrum was used in the range of 400-4000 cm⁻¹ before and after modification. The IR is a procedure primarily used to

detect the location of surfactant molecules inserted in the interlayer of clay [11, 14, and 16]. Crystallinity, structural geometry and basal spacing (d_{001}) of natural and modified clay were identified with X-ray diffraction. Records of XRD patterns were taken at room temperature on a Philips, PW1730 model with Cuka radiation ($\lambda = 1.541874 \text{ \AA}$), at 40 kV and 150 mA ranging from 1 to 80° and a scan speed of 2° min⁻¹ [14, 17]. Thermogravimetric analysis of clay and the modified clay were obtained by a TA Instrument Inc. Q600 at a ramp of 10°C min⁻¹ from room temperature to 1000°C under a high-purity flowing nitrogen atmosphere. Surface area, porosity, and mean pore diameter were determined with the Brunauer-Emmett-Teller (BET) technique by the use of N₂. (BEISORP-MINIII, Japan). The degassing process of specimens was performed in advance for 12 h [14].

Adsorption experiments

The adsorbability of natural and modified clay was recognized through batch adsorption examinations. The impacts of parameters examined here included the surfactant's loading amount (0.25 -3 CEC), pH (2-11), contact time (20-100 min) and adsorbent dosage (10-40g L⁻¹) on COD and turbidity attenuation of landfill leachate. Adsorption tests were entirely done in a 100 ml beaker that contained 50 ml of landfill leachate and the intended dose of adsorbent at different range of pH and contact time, in room temperature, and with a rotating speed of 150 rpm. Centrifugation (3500 rpm, 15 min) of the suspension and then analysis of the supernatant was performed after the completion of determined time. A WTW turbidity meter (Turb550) was applied to measure turbidity. The chemical oxygen demand (COD) was determined by the standard open reflux procedure [34]. Results are reported as mean values obtained from the repetition of entire triplicate experiments, with a standard deviation < 3% for repeated tests in the whole experiments. The removal percentage of pollutants (R%) was measured from the equation below [14]:

$$R(\%) = \frac{C_0 - C_e}{C_0} * 100$$

In this formula, C₀ and C_e are the initial and final

concentration, respectively, and R is the removal rate of COD and turbidity.

Here, the pH of the leachate was attuned by one molar solution of HCl and NaOH.

RESULTS AND DISCUSSION

SEM

The SEM images of raw and modified bentonite are illustrated in Figure 1. There is a difference between the morphological features of bentonite and modified bentonite. A bulky and massive morphology with irregular flake are observed in raw bentonite. SEM of modified clay clearly indicates that the basal spacing

increases between clay layers, which represents the proper intercalating of cationic surfactant into the interlayer of clay. The occurrence of severe repulsion is expectable between the groups because of the hydrophilicity of the clay sheets and the hydrophobicity properties of the surfactant. The application of cationic surfactant leads to increased interlayer spacing and creating a more open morphology. Thus, the findings of SEM analysis correspond to those of XRD examination showing increased basal spacing in the modified clay in comparison to the natural clay [32]. Similar morphological images have been observed by previous researchers on the cationic surfactant-clay composite [12-14].

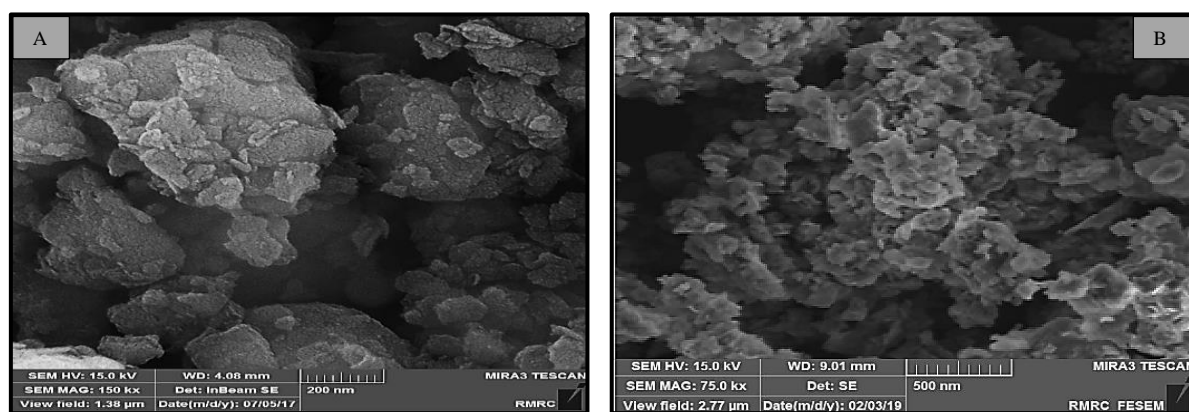


Figure 1. scanning electron microscopy of (a) bentonite, (b) HDTMA-BEN

XRD

The XRD pattern of clay is shown in Figure 2. This image indicates that the Avaj mine bentonite is rich in non-clay phases such as quartz, calcite, illite, and albite. XRD diffractogram of raw and modified clay shows that no new phase has been added to the modified clay in comparison with raw clay. This feature indicates that functionalizing does not change the structure of clay [13, 15].

The basal spacing of the original bentonite was 9.79 Å, which increased to 10.01 Å after modification with HDTMA-Br, which is similar to the previous findings [13,16]. Changing of the basal spacing often indicates that organic cation surfactant has been loaded in the

interlayer space of modified betonies [13, 17,18]. Even so, confirming the absorption of organic cations requires further proof due to incomplete information of XRD results on local conformation and phase state of the loaded surfactant [17]. Thus, the TGA and FTIR analysis are performed.

In theory, the reflection of pages d_{001} change to lower angles due to the intercalation mechanism [19], which is in agreement with the experimental findings. According to the findings, the peak of the d_{001} in natural bentonite can be seen at $2\theta = 9.01$, which emerges at $2\theta = 8.84$ when it was modified with HDTMA-Br.

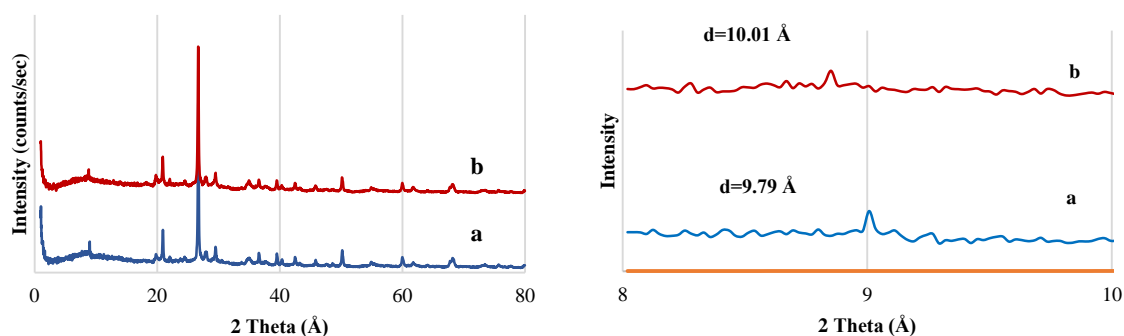


Figure 2 XRD pattern of (a) bentonite (b) HDTMA-BEN

FTIR

In the FTIR technique, the molecular structure of loaded surfactants within the interlayer space of clay is investigated [10]. Figure 3 displays the infrared spectrum of natural and modified bentonite at 4000–400 cm^{-1} wavelengths. There are several differences in the FTIR of natural and modified clay, which indicates the formation of the composite. As depicted in Figure 3, the bands of natural bentonite have appeared bands at 470, 520, 690, 793, 1028, 1439, 1643, 2372, 3401, and 3626 cm^{-1} . The sharp peak in 3626 cm^{-1} is related to the OH stretching vibration of the inner hydroxyl groups [14, 20]. The intensity of this band is reduced by loading the surfactant, and the peak is shifted to 3609 cm^{-1} , which is because of replacing water molecules by loading surfactants in the interlayer space of bentonite [10, 13], indicating the shift of clay properties to hydrophobicity [10]. On the other hand, broadband at 3401 cm^{-1} is attributed to the hydrogen bonds of water molecules adsorbed in the interlayer space of bentonite [14]. The band at 1643 cm^{-1} is attributed to the bending vibrations

of water molecules adsorbed on bentonite [13, 14] and the band at 1439 cm^{-1} is related to the C-H bending vibrations [14, 21]. For modified clay, however, the C-H bending vibration is transmitted to 1436 cm^{-1} .

The infrared spectrum of bentonite shows a band of 1028 cm^{-1} , as well, belonging to the Si-O-Si bending vibrations [13, 14, and 20]. Quartz bands (Si-O) emerge at about 790 cm^{-1} [7, 13, 14 and 22]. And the bands at 520 and 470 cm^{-1} belong to the O-Si-O bending vibrations [14]. For modified clay, the O-Si-O bending vibrations are shifted to a lower wavelength (from 520 and 470 cm^{-1} to 517 and 469 cm^{-1}), which corresponds to that of earlier studies [14]. However, the Si-O-Si bending vibration is shifted toward higher wavenumber (from 1028 to 1029 cm^{-1}). A band of 690 cm^{-1} is also attributed to the Al (Mg) -Si-O tetrahedron [13, 20 and 23]. In contrast, although the clay structure is maintained in the modification by HDTMA-Br, two peaks appear at 2852 and 2929 cm^{-1} which are ascribed to the asymmetric and symmetric stretching vibration of the C-H of the HDTMA-Br's aliphatic chain [9, 11, 14, 16 and 24].

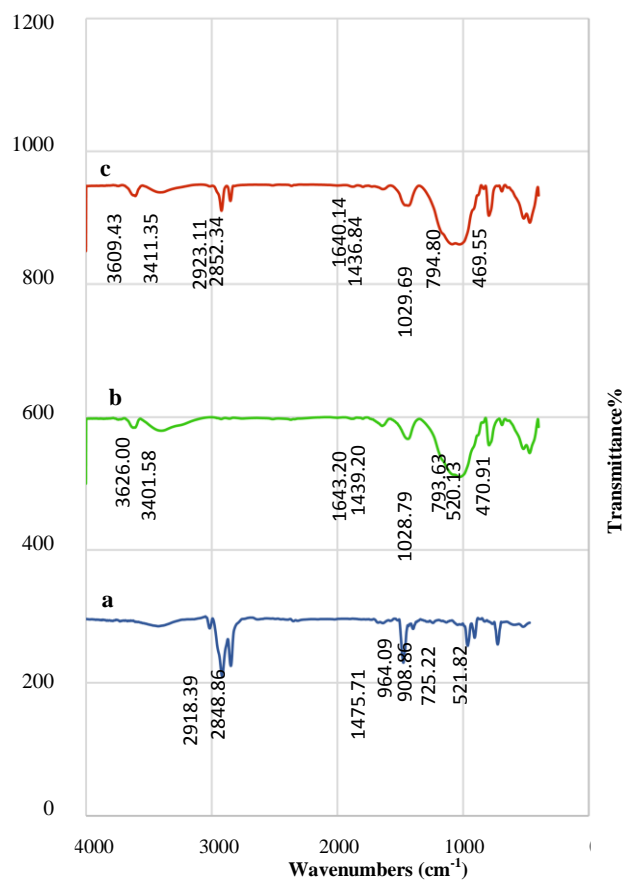


Figure 3. FTIR pattern of (a) HDTMA-Br (b) bentonite (c) HDTM-BEN.

TGA

Figure 4 depicts the thermogravimetric (TGA) curves of natural and the modified bentonite from room temperature to 1000°C under inert conditions. Accordingly, the thermal stability of the natural clay is greater than the modified bentonite. Two phases of mass loss are seen for unmodified clay. The first phase is from room temperature to around 200°C, resulting in slightly decreased weight of the clay. This mass reduction is attributable to the losing of humidity or the removal of interlayer water molecules, and its value is about 0.5% [13, 16, 22, 25, 26, 27, 28 and 29]. After this temperature to about 400°C, the weight remains nearly unchanged.

From 400°C to around 800°C, the weight undergoes a decrease anew because of the dehydroxylated of the clay or releasing exchangeable water corresponding to cations. (About 4.5%).

Three steps of mass loss could be noticed for modified clay. The first stage is similar to bentonite; it is attributed to the hydration process and corresponded to about 2.2% weight loss. The second stage of weight loss (from 200°C to about 700°C) is mainly attributed to organic modifier (surfactant) decomposition [12, 13 and 16]. And the final stage (above 700°C) is ascribed losing structural hydroxyl groups of bentonite [12, 16 and 30].

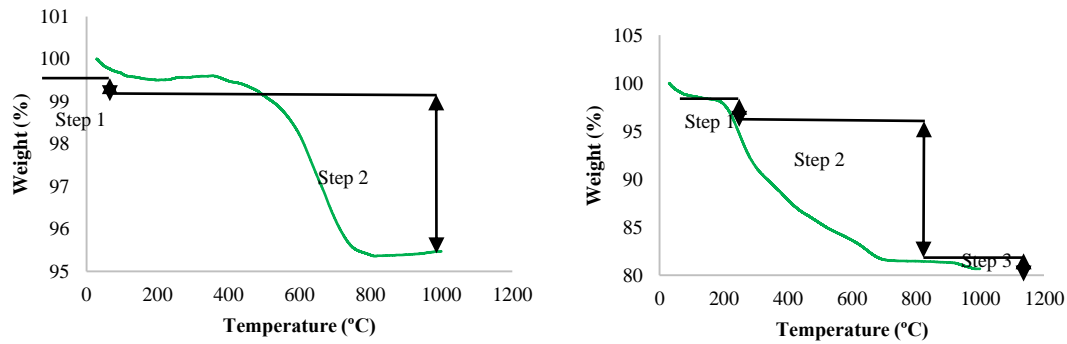


Figure 4. The TG pattern of (a) Bentonite (b) HDTMA-BEN.

BET

Effective and efficient adsorbents are proliferous structurally and have suitable surface chemistry [31]. The pore size and surface area were determined using a Brunauer-Emmett-Teller analyzer (BET). Table 3 lists the surface area, pore volume (porosity), and the mean pore diameter of modified and unmodified bentonites. The specific surface area and pore volume of functionalized bentonite decrease as compared with raw bentonite. This is probably due to the blockade of the interlayer space of the clay with surfactant molecules, leading to a drop in the active site numbers accessible for N₂ gas absorption [9, 13 and 14]. Besides, surfactant cations not only entered the interlayer space of the clay but were also mainly absorbed as a monolayer on the

outer surface [14]. In this case, the intercalated surfactant occupies the available pores between clay particles with the "house of cards" structure which results in reduced specific surface area and the pore volume [14].

On the other hand, there was a significant increase in pore diameter after the modification of clay with the cationic surfactant. From these results, it is estimated that the adsorption of organic contamination into the modified clay is not affected by surface area and pore volume, and these factors are not an essential factor for adsorption [14]. Moreover, these observations showed that the increasing basal space is not directly related to the growing surface area [13].

Table 3. Textural characteristics of natural and modified clay

	Surface area S_{BET} ($m^2 g^{-1}$)	Total pore volume ($C^3 g^{-1}$)	mean diameter (nm)	BJH(nm)
Bentonite	28.981	0.1158	15.982	1.29
Organoclay	11.416	0.09156	31.939	3.15

Determining the optimum quantity of surfactant

The optimum quantity of modifier was determined through loading bentonite by various levels of HDTMA-Br (0.25 to 3 CEC) followed by investigating the COD

content and turbidity elimination of landfill leachate. According to the findings, the two factors had the maximum removal efficiencies at 1.0 CEC (Fig. 5).

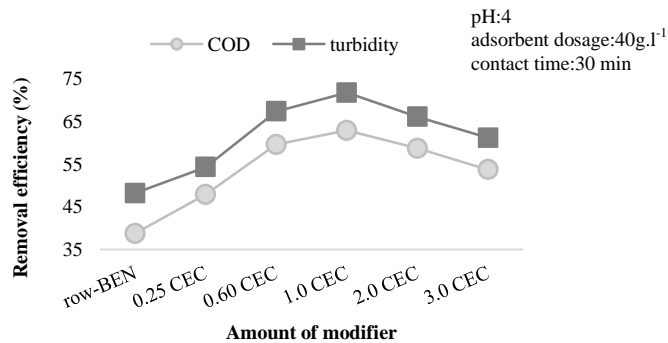


Figure 5. Optimal amount of modifier

Effect of pH

The adsorption can be affected by the hydrous solution pH which can influence the level of ionization and the production of different contaminants, eventually resulting in altered reaction kinetics and equilibrium of the adsorption procedure [22].

The adsorption mechanism, was studied in a pH ranging from 3 to 11 at 30 min, using an adsorbent dosage of 40 g L⁻¹. Figure 6 shows the results of the pH effect on COD and turbidity reduction with modified clay by the HDTMA-Br. As seen in Figure 6, COD and turbidity removal are significantly dependent upon the pH level. The elimination of COD improves under acidified conditions and adsorption reaches its maximum at pH: 3. Rising the pH over 3, leads to a gradual reduction in the

absorption and is then minimized at neutral pH. The surface charge of the clay can explain these findings mentioned above. The exterior surface of bentonite, contains weaker siloxane groups (Si-O) and tetrahedral sheets, which are then undergo conversion to Si-O bands, followed by turning into Si-OH by changing pH [21].

Rising the pH to an alkaline level, again leads to a gradual increase in COD elimination with the occurrence of precipitation. Concerning turbidity, the removal reached a maximum at pH: 11, which is attributable to concurrent adsorption and precipitation. Thus, pH values of 3 and 11 were selected for COD and turbidity, respectively for the subsequent examination.

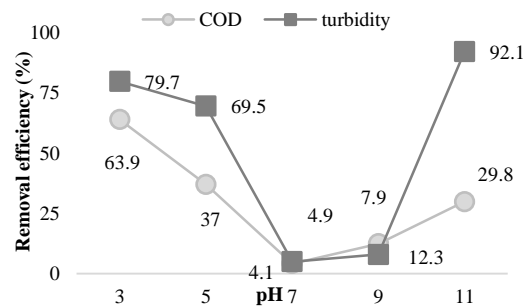


Figure 6. Impact of pH on COD and turbidity removal

Impact of adsorbent dose

Figure 7 displays the influence of adsorbent dose (variable from 10 to 70 g L⁻¹) for the reduction of COD and turbidity at an optimum pH. As seen in the figure, COD removal efficiency increases with increasing adsorbent dosage which can be ascribed to the increased surface area and adsorption sites by adding the adsorbent dosage [31]. The COD reduction rate reaches its maximum value at 50 g L⁻¹, after which it remains unchanged with raising the dose, followed by a decrease

in the COD removal rate with rising the adsorbent dose to above 60 g L⁻¹. Such a reduction may result from overlapping the adsorption sites due to over-crowding the adsorbent particles [22].

In the case of turbidity, the removal efficiency rises from 75.5% to 92.1%, and then the rate of adsorption gradually decreases by increasing the adsorbent dosage from 10 to 40 g L⁻¹ at pH: 11.

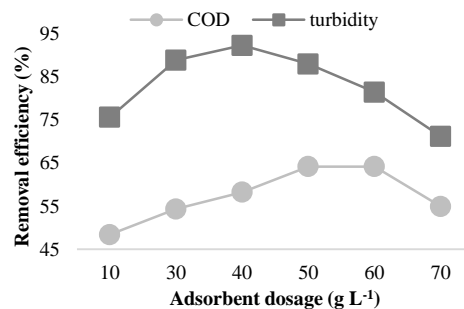


Figure 7. Influence of adsorbent dose on COD and turbidity removal

Influence of contact time

In all batch systems, contact time has a major contribution to COD attenuation [32]. Figure 8 shows gradual increases in the COD and turbidity removal efficiency with increasing the contact time till to a definite time. The COD removal rate was maximized in 50 min, with a 66.3% removal efficiency, and a maximal turbidity reduction rate was recorded at 30 min with a yield of 92.1%. A very higher adsorption in the initial phases is attributable to the accessibility of available adsorption sites throughout the early stage, and the rest of vacant sites were could hardly occupied because of the repulsive forces between dissolved molecules and solid phase once it reaches equilibrium [33].

The adsorbent was recovered after optimization. In this study, 0.1 M HCl was used to recover the adsorbent. In this case, changing the solution pH allows to replace contaminants with a cation or anion [42]. For this purpose, the leachate was first purified under optimum conditions. Then, the adsorbent was separated from the leachate and dried in an oven at 40°C for 24 h. The composite was then placed on a shaker in 50 ml of 0.1 M HCl solution until reaching equilibrium time [23]. The results showed that the regenerated sorbent had the same adsorption capacity as the original composite.

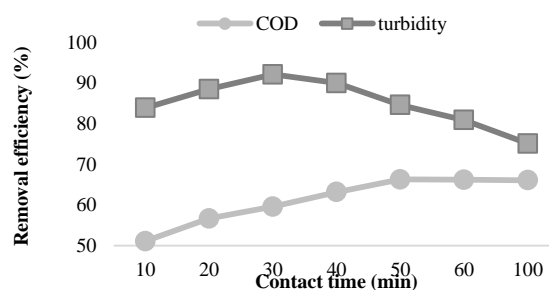


Figure 8. Influence of contact time on (a) COD (b) turbidity removal

CONCLUSIONS

In this study, modified Bentonite with HDTMA_Br was prepared and used for landfill leachate treatment in Mohammad Abad's landfill, Qazvin, Iran. SEM images showed increased basal spacing in the modified clay, and the XRD analysis and the FTIR spectrum represented the organic section loading of the surfactant between the bentonite layers. The findings of the batch experimentations revealed that the solution pH, the quantity of composite, the contact time, and the amount of surfactant loaded can affect the COD and turbidity removal of landfill leachate

The uppermost COD reduction efficiency of landfill leachate was 66.3%, which was obtained at pH = 3, an adsorbent dosage of 50 g L⁻¹, and a contact time of 50 min for the clay loaded with a capacity of 1.0 CEC. Also, the maximum turbidity reduction rate was 92.1%, which was attained at pH = 11, an adsorbent dose of 40 g L⁻¹, and a contact time of 30 min. Because of the low ion

exchange capacity (40meq 100g⁻¹) of the clay utilized here, it can be concluded that the adsorbent efficiency can be increased using the clay with high ion exchange capacity.

This study showed that modified bentonite with HDTMA-Br could be utilized as an effectual and inexpensive adsorbent for the pre-treatment of wastewater with high organic pollutants and turbidity, such as landfill leachate.

ACKNOWLEDGEMENTS

The authors thank the personnel at Alborz Industrial City and Alborz Wastewater Treatment Plant for their support of this study due to the provision of laboratory conditions.

Conflict of Interest

The authors declare that they have no conflict of interest.

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