One-pot alkanolamines-assisted synthesis of magnetic mesoporous silica for synthetic dye adsorption

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Abstract
Magnetic mesoporous silica (MMS) was synthesized in a one-pot system using various alkanolamines (triethanolamine, diethanolamine, tris (hydroxymethyl)aminomethane) as a basic catalyst. The characterization of the composites was conducted using scanning electron microscope, transmission electron microscope, X-ray diffractometer, surface area analyzer, and X-ray photoelectros spectroscopy. The MMS synthesized with tris(hydroxymethyl)aminomethane (MMSTRIS) showed the highest specific surface area, pore volume, and average pore diameter. However, when the composites were applied as adsorbents for brilliant green (BG) dye, MMS synthesized with diethanolamine (MMSDEA) showed the highest maximum adsorption capacity of 339.7 mg g⁻¹. The fastest adsorption rate constant of 1.57 × 10⁻² g mg⁻¹ min⁻¹ was obtained for MMSTRIS, which has the largest average pore size among all composites. The adsorption kinetic study suggested that the adsorption of BG onto the prepared MMS composites was mainly chemisorption process, which most likely involves electrostatic interaction and hydrogen bonding between BG molecule and the surface of the composites.

Keywords
One-pot synthesis · Magnetic mesoporous silica · Alkanolamine catalyst · Synthetic dye adsorption · Adsorption kinetic study

Introduction
Synthetic dyes have been extensively used for the dyeing process in various industries due to their low cost and easy production, wide spectrum of color, and stability of the color [1]. During the dyeing process, some parts of dyes are lost and discharged as wastewater effluent. The amount of this loss depends on the dye used and the type of dyed materials. Globally, approximately 700,000 metric tons of 100,000 commercial dyes are produced and up to 70,000 metric tons of these dyes are discharged into wastewater [2]. Most of the wastewater is previously treated to meet the suitable criteria for discharge water. However, the dye residues in wastewater still pose potential threats to the aquatic environment. The presence of dye in surface water, even in a very low concentration, may cause esthetic problems due to the change of water color. Furthermore, the light penetration into the water will be inhibited, preventing aquatic plant photosynthesis and eventually reducing the oxygen production in water [3, 4]. Some synthetic dyes are also capable of decreasing the photosynthesis rate of aquatic plants by inhibiting the growth rate, chlorophyll pigment, and metabolic activity [5, 6].

Various treatment technologies such as coagulation, chemical oxidation, biodegradation, membrane filtration, adsorption, photocatalysis, and others have been applied for removing dyes from wastewater [7–12]. Each technology has its advantages and limitations. Among those technologies, adsorption is still considered the most effective method due to being simple, inexpensive, and effective [13, 14]. Thus,
the effort in developing effective adsorbents is a continuing process.

Mesoporous silica, i.e., amorphous silica materials with pore size ranging from 2 to 50 nm is considered one of the best candidates as an effective adsorbent of dyes, since it has many remarkable properties such as high specific surface area, good stability, chemical inertness, as well as porous and tunable structure [15, 16]. However, the use of mesoporous silica as an adsorbent requires a tedious separation process such as filtration and centrifugation to separate the used adsorbent from the treated solution.

The incorporation of magnetic iron oxide nanoparticles on the mesoporous silica material has been attracting much attention for adsorption process, due to its high specific surface area, pore volume, tunable structure, functionalizable surface and easy separation [17–19]. Magnetic iron nanoparticles immobilized on mesoporous silica is expected to create a synergistic effect in removing pollutants from aqueous solution by combining the best properties of both materials [20]. In addition, the presence of magnetic nanoparticles on mesoporous silica will eliminate the filtration or centrifugation process needed after the adsorption process. The used adsorbent can be easily separated from the aqueous phase using an external magnetic force, providing a cleaner and fast method [18, 21].

Magnetic mesoporous silica (MMS) is frequently synthesized using two steps: magnetic iron nanoparticle synthesis, followed by surface modification of the nanoparticles. In this study, MMS was synthesized in a one-pot system using various alkanolamine compounds as a basic catalyst. The fabrication of MMS composites was carried out using CTAC surfactant as a template and alkanolamine as a basic catalyst in the one-pot system. The synthesis procedure is illustrated in Fig. S2. MMS was prepared using the following procedure: 100 ml of H2O was purged with nitrogen gas for 30 min and 410 mg of FeCl3.6H2O and 232.6 mg of FeCl2.4H2O were dissolved into it. After all of the powders had dissolved, 1 mL of 28% ammonia was added into the solution dropwise and the mixture was stirred for 2 h. After that, 320 mg of CTAC was added and the mixture was stirred for another 10 min. 2.24 mmol of alkanolamine compound (TEA, DEA or TRIS) was added and the mixture was sonicated for 30 min. The molecular structure of each alkanolamine is given in Fig. S3. The mixture was heated to 80 °C and stirred during heating. During this step, 0.5 mL of TEOS as silica precursor was added into the solution mixture dropwise. The solution was stirred for 3 h afterward. After the reaction was complete, the products were washed with ethanol and water subsequently. The produced MMS materials were calcined at 500 °C for 5 h to remove the templates. The composites synthesized using TEA, DEA and TRIS were called MMSTEA, MMSDDEA and MMSTRIS, respectively.

Characterization of MMS

A scanning electron microscope (SEM, Hitachi, S-4300) combined with an EDX (Horiba, EMAX) analyzer was used to capture the surface morphology and roughly estimate the composition of the materials. The average size and morphology of the materials were observed with a transmission electron microscope (TEM, JEOL, JEM-2100). The specific surface area, total pore volume, and pore size of the materials were determined by N2 adsorption–desorption isotherm at 77 K using a micrometer (Tristar II 3020, Micrometrics). A Fourier-transform infrared spectroscopy (FTIR, Perkin Elmer Spectrum 400) instrument was used to characterize the functional groups present on the materials. The phase structure of the materials was characterized using an X-ray powder diffraction (XRD, RigakuUltima II/PC) instrument with CuKα radiation. The surface chemistry of the materials was analyzed using an X-ray photoelectron spectroscopy (XPS, ULVAC, Quantera XSM) instrument.

Experimental

Reagents and chemicals

All the chemicals used in the experiments were of analytical grade and used as received. Cetyltrimethylammonium chloride (CTAC), triethanolamine (TEA), diethanolamine (DEA), ferric chloride hexahydrate (FeCl3·6H2O) and ferrous chloride tetrahydrate (FeCl2·4H2O) were purchased from FUJIFILM Wako Pure Chemicals Corporation (Osaka, Japan). Tris(hydroxymethyl)aminomethane (TRIS), 28% ammonia and brilliant green (BG) were obtained from Nacalai Tesque, INC (Kyoto, Japan). Tetraethyl orthosilicate (TEOS) was obtained from Tokyo Chemical Industry Co., Ltd. (Tokyo, Japan). Deionized water (DI) was used to prepare all the solutions.
Adsorption experiments

Adsorption experiments were carried out in a batch system. Adsorption isotherm of BG onto MMS composites was studied at 25 °C with the initial concentration of BG ranging from 10 to 90 mg L⁻¹. A fixed amount of adsorbent was added into the BG solution and stirred for 60 min. Adsorption kinetics of BG onto MMS composites was investigated by mixing 30 mg of adsorbent with 60 mL of 200 mg L⁻¹ BG solution. The residual concentration of BG in the mixture was measured at certain time intervals (0–120 min). All adsorption experiments were conducted in the pH of the original BG solution unless stated otherwise. The adsorption experiment was repeated three times and the result was obtained from the average value of the triplicate. The residual concentration of BG in the aqueous solution was measured by using UV–Vis spectrophotometer (Hitachi, U-4100 Spectrophotometer) at a wavelength of 624 nm. The adsorption efficiency and adsorbed amount of dye \( q_e, \text{ mg g}^{-1} \) were calculated using Eqs. 1 and 2:

\[
\text{Adsorption efficiency (\%) = } \left( \frac{C_0 - C_t}{C_0} \right) \times 100%,
\]

\[
q_e = \frac{V(C_0 - C_e)}{m},
\]

where \( C_0 \) (mg L⁻¹) is the initial concentration of BG and \( C_t \) (mg L⁻¹) is the concentration of BG in aqueous solution at time \( t \); \( C_e \) (mg L⁻¹) is the concentration of BG in aqueous solution at equilibrium, \( V \) is the volume of solution (L), and \( m \) is the adsorbent mass (g).

The recyclability of MMS composites as adsorbents for BG dye was evaluated by employing acidified ethanol to desorb BG from MMS composites. After sequential washing using distilled water and oven-drying, the adsorbent was reused for adsorption of BG in solution. This adsorption–desorption process was carried out for four cycles.

Results and discussion

Characterization of MMS

SEM and TEM images of all composites in Fig. 1 showed that all of the MMS composites have a spherical shape and agglomeration was observed in all composites: MMS\textsubscript{TRIS} showed the least agglomeration. From TEM images, the average particle size of the composites was around 10 nm. The rough estimation of the percentage (w/w) of O, Fe and Si atoms in the composites from EDX measurement showed that MMS\textsubscript{TEA} had the highest Fe content among all composites, while MMS\textsubscript{TRIS} showed the lowest Fe content (Table S1).

The X-ray diffractogram of all MMS composites is presented in Fig. 2a. The presences of characteristics peaks at 30°, 35°, 43°, 54°, 57° and 62° were ascribed to (220), (311), (400), (422), (511), and (440) planes of Fe\textsubscript{3}O\textsubscript{4}, respectively (JCPDS:75-0033), indicating the presence of magnetic iron oxide in all composites. This result showed that the magnetic iron oxide immobilized in silica material maintains its crystalline phase. All composites showed a relatively similar pattern of X-ray diffraction without any obvious peaks of silica material.

The FTIR instrument further confirmed the presence of functional groups in the synthesized composites. The FTIR spectra of MMS composites are presented in Fig. 2b. Each composite showed similar absorbance peak with different intensity, indicating the presence of similar functional groups on the composites. The absorption band at 1090 cm⁻¹ was due to the asymmetric stretching vibration of Si–O–Si bonding. The peak at 800 cm⁻¹ was attributed to symmetric stretching of Si–O–Si. The appearance of Si–O–Si vibration peaks on the materials indicated that the condensation of silicon alkoxide has occurred [22]. The absorbance peak at 570 cm⁻¹ which corresponded to the vibration of the Fe–O bonding also suggested iron oxide's presence in the materials. The FTIR analysis results demonstrated the presence of SiO\textsubscript{2} and Fe\textsubscript{3}O\textsubscript{4} in all of the synthesized MMS.

The specific surface area and porosity of the composites were determined by liquid N\textsubscript{2} adsorption (at 77 K). The N\textsubscript{2} adsorption–desorption isotherms obtained for the composites are shown in Fig. 2c. The N\textsubscript{2} adsorption–desorption isotherm curves of all composites exhibited a type IV isotherm and an H1-type hysteresis loop which corresponded to mesoporous materials with a narrow range of mesopores [23].

Table 1 summarizes the textural properties of the synthesized MMS. The surface area of the composites not only depends on the particle size of the material, but also the amount of iron nanoparticles immobilized in the materials. The more the iron contents in the composite, the lower is the surface area due to the blockage of mesopores by the iron nanoparticles. This data agreed with the EDX data in Table S1. MMS\textsubscript{TRIS} with the largest specific surface area showed the lowest iron content. All of the synthesized composites showed an average pore size in the range of mesopore size. The MMS\textsubscript{TEA} and MMS\textsubscript{DEA} showed relatively similar BET surface area, pore volume, and BJH average pore size, while the MMS\textsubscript{TRIS} showed the highest values among all. These results suggested that the use of alkanolamines with different basicities and structure could influence the properties of the synthesized MMS composites. Alkanolamines could influence the hydrolysis and polycondensation reaction of silica parent material [22]. They could act both as the...
basic reactant and hydrolysis-retarding agent (complexing agent), which keeps the seeds separated during the condensation process [24].

XPS spectra of MMS composites (Fig. 3a) showed the peaks of Si, C, O and Fe elements. The binding energies of Fe(III)2p3/2 and Fe(III)2p1/2 can be seen at 710.5 eV and 724 eV, respectively. This indicated that the iron oxide present in all composites is in the form of magnetite [25]. The presence of silica was shown by Si 2p peak at 102 eV [26]. The XPS spectra of MMS after the adsorption of BG showed a slight increase of the binding energies of Fe(III) in the iron oxide, as shown in Fig. 3b. The binding energies of Fe(III)2p3/2 and Fe(III)2p1/2 were observed at 715 eV and 730 eV, respectively. This increase was due to the decrease of electron density of iron after binding with N+ of BG molecule [27].

**Adsorption studies**

The adsorption performance of the MMS composites was studied using BG as a model pollutant. The effect of solution pH on the adsorption efficiency was studied under the initial solution pH of 3 to 6. At pH lower than 3, the BG solution started to decolorize while at pH higher than 6, and
the turbidity and precipitation started to occur. Initial pH of the solution plays an important role in the adsorption efficiency of BG onto MMS composites, since the stability and structure of BG molecule and the dissociation of functional groups on the composites are predominantly controlled by the solution pH. It was found that the optimum pH of BG adsorption onto all composites was 3; however, since the initial concentration of BG on pH 3 was significantly reduced (Fig. S4), the adsorption studies were conducted under the original pH of BG solution which was around 4.

The adsorption kinetics study provides information about the efficiency of adsorption process and is an important aspect of an adsorption process related to the industrial batch application [28]. In this work, the kinetic data were evaluated using pseudo-first-order and pseudo-second-order kinetic models, given in Eqs. 3 and 4, respectively.

\[
\ln (q_e - q_t) = \ln q_e - k_1 t, \tag{3}
\]
\[
\frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{t}{q_e}. \tag{4}
\]

Parameter \( t \) is time (min), \( q_e \) and \( q_t \) (mg g\(^{-1}\)) are adsorption capacities at equilibrium and time \( t \), respectively, and \( k_1 \) (min\(^{-1}\)) and \( k_2 \) (g mg\(^{-1}\) min\(^{-1}\)) are rate constants.

Experiment data were fitted to both kinetic models (Fig. 4) and all obtained kinetic parameters are given in Table 2. It is seen that the adsorption of BG onto all composites followed the pseudo-second-order kinetic model with \( R^2 \) values greater than 0.99. Moreover, the adsorption

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**Fig. 2** a XRD diffractogram; b FTIR spectra; c nitrogen adsorption–desorption isotherm of MMS composites

| Table 1 Textural properties of MMS composites |
|---------------------------------------------|
| MMS sample  | BET surface area (m\(^2\) g\(^{-1}\)) | Pore volume (cm\(^3\) g\(^{-1}\)) | BJH average pore size (nm) |
| MMS\(_{TEA}\) | 228.5 | 0.416 | 2.73 |
| MMS\(_{DEA}\) | 232.0 | 0.401 | 2.68 |
| MMS\(_{TRIS}\) | 280.2 | 0.546 | 3.14 |
capacities obtained from the experiment \((q_{\text{exp}})\) were similar to the values calculated \((q_{\text{cal}})\) from the linear graph equation of the pseudo-second-order model. This implies that BG’s adsorption onto the surface of MMS composites was mainly chemisorption or surface complexation [29]. The pseudo-second-order rate constant of BG adsorption on MMS\textsubscript{TRIS} is the highest among other composites, followed by the rate constant of BG adsorption on MMS\textsubscript{TEA} and MMS\textsubscript{DEA}. It is notable that the larger the average pore size of the composite, the higher is the adsorption rate constant of BG on the MMS composite.

The Langmuir and Freundlich isotherm models were used to describe the adsorption process of BG onto the MMS composites. The Langmuir isotherm model assumes that the adsorption process is monolayer, while the Freundlich isotherm model applies to the multilayer adsorption
process. The non-linear equations of Langmuir and Freundlich model are given in Eqs. 5 and 6, respectively.

\[
q_e = q_{\text{max}} K_L \frac{C_e}{1 + K_L C_e},
\]  

(5) \[q_e = K_F C_e^{1/n}.
\]  

(6)

Parameter \(C_e\) (mg L\(^{-1}\)) is the concentration of the dye at equilibrium, \(q_e\) (mg g\(^{-1}\)) is the adsorption capacity at equilibrium, \(q_{\text{max}}\) (mg g\(^{-1}\)) is the maximum adsorption capacity, \(K_L\) (L mg\(^{-1}\)) is the Langmuir constant, and \(K_F\) and \(n\) are the Freundlich adsorption constants.

The adsorption experiment data of BG onto all MMS composites were fitted using both isotherm models. The fitting curves are shown in Fig. 5, and the adsorption isotherm parameters are given in Table 3. The adsorption of BG onto MMS composites fitted well both Langmuir and Freundlich isotherm models \((R^2 > 0.95)\) in the studied adsorptive concentration range. The \(1/n\) value obtained from Freundlich isotherm was greater than 0 for all composites, indicating that the adsorption of BG on all prepared MMS composites was favorable [30]. The maximum adsorption capacity of BG onto all MMS composites in this work was comparable to some previously reported works (Table S2).

Adsorption kinetic study suggested that the adsorption of BG onto MMS composites was mainly through the chemisorption process. Adsorption of BG onto MMS composites could involve electrostatic interaction between the protonated ammonium of BG molecule and deprotonated hydroxyl group available on the surface of the composites. In addition, hydrogen bonding between hydroxyl groups of the adsorbent and amine group of BG might also proceed.

| Isotherm model | Parameters | Adsorbents |
|---------------|------------|------------|
| Langmuir      |            | MMS\(_{\text{TEA}}\) | MMS\(_{\text{DEA}}\) | MMS\(_{\text{TRIS}}\) |
| \(q_{\text{max}}\) (mg g\(^{-1}\)) | 208.9 | 339.7 | 71.7 |
| \(K_L\) (L mg\(^{-1}\)) | 0.019 | 0.006 | 0.12 |
| \(R^2\)        | 0.979 | 0.988 | 0.998 |
| Freundlich    |            |            |            |
| \(K_F\)        | 21.36 | 3.69 | 1.72 |
| \(n\)          | 1.50 | 4.79 | 1.36 |
| \(R^2\)        | 0.980 | 0.993 | 0.992 |

[31] as illustrated in Fig. S5. Further study is necessary to elucidate a detailed adsorption mechanism of BG onto the prepared MMS composites.

The regeneration ability of the MMS composites was studied, since regeneration and recyclability properties of an adsorbent are crucial for the implementation in the real wastewater. The result of recyclability studies of all MMS composites is given in Fig. 6. The adsorption capacity of all MMS composites decreased quite significantly from one cycle to another. After the fourth cycle, the adsorption capacity of MMS decreased by 73%, 59%, and 82% for MMS\(_{\text{TEA}},\) MMS\(_{\text{DEA}},\) and MMS\(_{\text{TRIS}},\) respectively. However, after the third cycle, all composites still showed a good adsorption capacity of around 90–100 mg g\(^{-1}\). A complete recovery of BG from MMS composite was hard to achieve, most likely due to the strong affinity of MMS for BG molecule. In addition, after sequential adsorption–desorption and washing processes, the active sites of MMS might be blocked or changed, causing the decrease of the adsorption capacity.
Conclusions

Magnetic mesoporous silica (MMS) composites have been synthesized using various alkanaolamines as a basic catalyst in a one-pot system. The use of different alkanoamines resulted in MMS with different surface area, pore volume, and average pore size. The characterization analysis showed the presence of magnetic nanoparticles in all composites. The application of the prepared composites for the removal of brilliant green (BG) from solution showed satisfactory results with excellent maximum adsorption capacities and adsorption rate. The main plausible mechanisms of adsorption of BG onto MMS were the chemical interaction such as hydrogen bonding and electrostatic interaction between BG and the hydroxyl groups of the composites. All of the prepared composites still showed high adsorption capacity toward BG after the third adsorption–desorption cycle. The results showed the potential of MMS as an adsorbent for the removal of BG dye from wastewater.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s44211-022-00183-7.

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References

1. E. Zablocka-Godlewska, W. Prystaś, Water Air Soil Pollut. 231, 75 (2020). https://doi.org/10.1007/s11270-020-4441-1
2. O. Tuńcu, H. Tanaci, Z. Aksu, J. Hazard. Mater. 163, 1 (2009). https://doi.org/10.1016/j.jhazmat.2008.06.078
3. M.M. Hassan, C.M. Carr, Chemosphere (2018). https://doi.org/10.1016/j.chemosphere.2018.06.043
4. J. Sharma, S. Sharma, V. Soni, Reg. Stud. Mar. Sci. 45, 101802 (2021). https://doi.org/10.1016/j.rsma.2021.101802
5. A.K. Moorthy, B.G. Rathi, S.P. Shukla, K. Kumar, V.S. Bharti, Environ. Toxicol. Pharmacol. 82, 103552 (2021). https://doi.org/10.1016/j.etap.2020.103552
6. M. Hernández-Zamora, H.V. Perales-Vela, C.M. Flores-Ortiz, R.O. Cañizares-Villanueva, Ecotoxicol. Environ. Saf. (2014). https://doi.org/10.1016/j.ecoenv.2014.05.030
7. S.H. Chen, A.S.Y. Ting, J. Environ. Manag. (2015). https://doi.org/10.1016/j.jenvman.2014.09.014
8. H. Li, S. Liu, J. Zhao, N. Feng, Colloids Surf. A: Physicochem. Eng. Asp. (2016). https://doi.org/10.1016/j.colsurfa.2016.01.048
9. Y. Kang, H. Yoon, C. Lee, E. Kim, Y. Chang, Water Res. (2019). https://doi.org/10.1016/j.watres.2018.12.038
10. Y. Zhen, G. Yao, Q. Cheng, S. Yu, M. Liu, C. Gao, Desalination (2013). https://doi.org/10.1016/j.desal.2013.08.009
11. B.K. Nandi, A. Goswami, M.K. Purkait, J. Hazard. Mater. 161, 1 (2009). https://doi.org/10.1016/j.jhazmat.2008.03.110
12. I. Konstantinou, T.A. Albanis, Appl. Catal. B. 49, 1 (2004). https://doi.org/10.1016/j.apcatb.2003.11.010
13. M.T. Yagub, T.K. Sen, S. Afroze, H.M. Adv. Colloid Interface Sci. (2014). https://doi.org/10.1016/j.cis.2014.04.002
14. V.K. Gupta, P.J.M. Carroll, M.M.L. Ribeiro Carroll, Crit. Rev. Environ. Sci. Technol. (2009). https://doi.org/10.1080/10643
15. V.B. Cashin, D.S. Eldridge, A. Yu, D. Zhao, Environ. Sci. Water Res. Technol. (2018). https://doi.org/10.1039/C7EW00322F
16. C.T. Kresge, M.E. Leonowicz, W.J. Roth, C. Vartuli, J.S. Beck, Nature (1992). https://doi.org/10.1038/359710a0
17. X. Chen, K.F. Lam, Q. Zhang, B. Pan, M. Arruebo, K.L. Yeung, J. Phys. Chem. C 113, 22 (2009). https://doi.org/10.1021/jp9018052
18. E. Egodawatte, A. Datt, E.A. Burns, S.C. Larsen, Langmuir 31, 27 (2015). https://doi.org/10.1021/acs.langmuir.5b01483
19. G. Li, Z. Zhao, J. Liu, G. Jiang, J. Hazard. Mater. 192, 1 (2011). https://doi.org/10.1016/j.jhazmat.2011.05.015
20. J. Ye, D. Nyobe, B. Tang, L. Bin, P. Li, S. Huang, F. Fu, Y. Cai, G. Guan, X. Hao, J. Mol. Liq. 303, 112656 (2020). https://doi.org/10.1016/j.molliq.2020.112656
21. P. Chaikhan, Y. Udman, R.J. Ampiah-Bonney, W. Chuachuad Chaiyasinth, Anal. Sci. 37, 7 (2021). https://doi.org/10.2116/anas ci.20P383
22. S. Nandy, D. Kundu, M.K. Naskar, J. Sol-Gel Sci. Technol. 72, 1 (2014). https://doi.org/10.1016/j.sglst.2014.3420-7
23. M. Thommes, K. Kaneko, A.V. Neimark, J.P. Olivier, F. Rodriguez-Reinoso, J. Rouquerol, K.S.W. Sing, Pure Appl. Chem. 87, 9 (2015). https://doi.org/10.1515/pac-2014-1117
24. J. Kobler, K. Möller, T. Bein, ACS Nano 2, 4 (2008). https://doi.org/10.1021/nn700008s
25. T. Radu, C. Iacovita, D. Benea, R. Turcu, Appl. Surf. Sci. (2017). https://doi.org/10.1016/j.apsusc.2017.02.002
26. M. Kokate, K. Garadkar, A. Gole, J. Mater. Chem. A 1, 16 (2013). https://doi.org/10.1039/C2TA00951J
27. M. Ganguly, P.A. Ariya, ACS Omega 4, 7 (2019). https://doi.org/10.1021/acsomega.9b00757
28. W. Yu, H. Li, L. Zhang, J. Liu, F. Kong, W. Wang, Anal. Sci. 36, 10 (2020). https://doi.org/10.2116/analsci.20P383
29. A. Kamari, W.S. Wan Ngah, M.Y. Chong, M.L. Cheah, Desalination 249, 3 (2009). https://doi.org/10.1016/j.desal.2009.04.010
30. M.A. Al-Ghouti, D.A. Daana, J. Hazard. Mater. 393, 122383 (2020). https://doi.org/10.1016/j.jhazmat.2020.122383
31. R. Nicola, S. Muntean, M. Nistor, A. Putz, L. Almásy, L. Săcărescu, Chemosphere 261, 127737 (2020). https://doi.org/10.1016/j.chemosphere.2020.127737

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