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Comparison of adsorption behaviors of lignite and its fly ash for the removal of bisphenol A from aqueous media

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ABSTRACT

This paper presents a study on the batch adsorption of bisphenol A (BPA) from aqueous solution on raw lignite (BC) and its fly ash (FA) to explore their potential use as low-cost adsorbents for water purification. The main parameters influencing BPA adsorption, including initial adsorbent dose, pH, and ionic strength were investigated. The pseudo-first-order, pseudo-second-order, and intraparticle diffusion models were used to describe the kinetic data, and rate constants were evaluated. The results obtained showed that the rate of adsorption followed a pseudo-second-order kinetic model with adsorption rate constants (k_2) of 0.0034 and 0.0058 g/μmol min for BC and FA, respectively. The equilibrium data were analyzed using the Freundlich, Langmuir, Temkin, and Dubinin–Radushkevich isotherms. The adsorption data fit well with the Langmuir isotherm model, with the maximum monolayer adsorption capacity of 13.34 mg/g for BC and 10.88 mg/g for FA, respectively. The results in this study indicated that lignite and fly ash are attractive materials for the removal of BPA from water.

Keywords: Adsorption; Bisphenol A; Low-cost adsorbent; Lignite; Fly ash

1. Introduction

Bisphenol A (BPA) is widely used in the production of polycarbonate plastics, epoxy resins, and synthetic polymer used for the production of everyday products, including food-contact applications [1,2]. Bisphenol A has attracted high concern because of its toxic and endocrine-disrupting effects, and its widespread occurrence [3]. Therefore, its removal from the aquatic environment is very important and is an object of interest for environmental scientists. Bisphenol A is removed from water by various biological [4–6], chemical [7–9], and physical [8,10] wastewater treatment processes. However, among the methods available for its removal from water, adsorption is most preferred. The adsorption technique is efficient and effective at different concentration levels and is inexpensive. Activated carbons are well-known and the most popular materials used for the removal of organic pollutants from water due to their high specific surface area and well-developed porous structure. Unfortunately, commercially available activated carbons are expensive and require a complex regeneration system. As a result, there is growing interest in alternative adsorbents that are simultaneously cheap and efficient in removing pollutants from water. As proposed by Crini et al. [11], these adsorbents can be classified into five groups: (1) natural materials, (2) modified natural materials, (3) manufactured materials, (4) agricultural solid wastes, and industrial by-products, and (5) biosorbents.

Lignite (brown coal) is a natural material that has the potential to be used as an adsorbent in the treatment

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of wastewaters because it is cheap and readily available. Lignite is a good source (precursor) for the preparation of activated carbons [12,13] or cokes [14,15] and also in an unprocessed (raw) form, it can be quite an effective adsorbent. In recent years, lignite was used for the removal of water pollutants including ammonium [16], dyes [17,18], and phenols [19,20]. First of all, lignite is used as fuel, and it is burned to produce ash. Fly ash as industrial waste can also be used as a potential adsorbent for the removal of air and water pollutants [21–25]. The leaching behavior of lignite fly ash indicates that the potentially toxic and/or leachable elements content is lower than of the same elements in standardized European fly ashes [26]. Toxic metal ions are released in extremely low quantities due to the alkaline nature of the fly ash [27]. The lignite fly ashes can be accepted as inert wastes due to the ettringite formation. Since ettringite stabilizes several toxic elements such as As, Cr, Se, and Sb by incorporating them into its structure [28]. The maximum concentration of the majority of the trace metals in the leachates was within the prescribed limits. The leaching of trace metals from lignite fly ash was governed by their concentrations, association with the ash particles, leaching duration, and pH of the leachate (most influencing parameter) [29]. It was shown that lignite mixed with ground limestone was efficient in immobilizing Pb and Cu. Not only adsorption but also complexation and precipitation were involved in the immobilization process [30].

The above-mentioned areas of use of lignite and ashes as sorbents in relation to some pollutants do not exhaust all the possibilities of their application. It is important to broaden the knowledge in the field of assessing the suitability of the present materials, for example as sorbents for specific micropollutants. This is the area where the present work, in which the effectiveness of the removal of bisphenol A from aqueous solutions using raw brown coal and its fly ash, was assessed. Contrary to other studies, the presented research uses lignite from one of the largest deposits in Poland, therefore both this material and the ash generated as a result of its combustion are available in large quantities and relatively cheap. The results of the presented research can be used to indicate new directions of their development, for example as fillings of sorption barriers.

In the present study, the efficiency of removing bisphenol A from aqueous solutions using raw lignite (BC) and its fly ash (FA) was evaluated taking into account the effects of various parameters, including the adsorbents' dose, solution pH, and ionic strength. The novelty of this work is the parallel study of adsorption properties of both materials used.

2. Experimental

2.1. Materials and methods

Bisphenol A (≥98%) was purchased from Sigma-Aldrich (St. Louis, USA). All the other reagents used in the experiments were of analytical grade and were received from Chempur (Piekary Śląskie, Poland).

The samples of lignite (BC) and fly ash (FA) were obtained from the Bełchatów (Poland) deposit and the local electrical power plant, respectively. The raw lignite was crushed in a mortar and sieved to a particle size of less than 250 μm. The raw fly ash was sieved to the same particle size (<250 μm). Both materials were then dried in an oven at 120°C for 24 h and kept in a desiccator for further study.

A scanning electron microscopy (SEM) analysis of the tested materials was performed with a scanning electron microscope QUANTA 250 FEG (FEI). For chemical (elemental) microanalysis of adsorbent surfaces, energy-dispersive X-ray spectroscopy (EDS) was used. Water and benzene vapor adsorption isotherms at 25°C were determined gravimetrically using a dynamic vapor sorption apparatus, DVS Advantage (Surface Measurement Systems Ltd., London, UK). The pH drift method was used to determine the point of zero charges (pH_{pzc}) of the adsorbents [31].

2.2. Batch adsorption studies

The batch adsorption experiments were performed within glass Erlenmeyer flasks containing 0.02 L BPA solution and an appropriate amount of the adsorbent. The flasks were mechanically stirred at 25°C at 200 rpm and the samples were taken after 6 h or at different preset contact time intervals (kinetic studies) and filtered through 0.2 μm filter paper. The BPA concentration was measured by high-performance liquid chromatography with UV detection (Shimadzu LC-20, Kyoto, Japan) according to a procedure described elsewhere [32]. The experiments (except for adsorption isotherms) were carried out for the initial BPA concentration of 250 μmol/L. The pH effect experiments were conducted at pH from 2.0 to 11.0; the solution pH was adjusted with 0.01 mol/L NaOH or HCl solutions. The influence of ionic strength was examined using varying NaCl concentrations from 0 to 0.1 mol/L. All batch experiments were conducted in duplicate, and the average of the two test results was used for further calculations.

The removal efficiency of BPA on the lignite and fly ash was evaluated using the following equation:

$$\text{Removal\%} = \frac{\left(C_0 - C_e\right)}{C_0} \times 100 \tag{1}$$

The amounts of BPA adsorbed at equilibrium (q_e) and at time t (q_t) were calculated by the following formulas:

$$q_e = \frac{\left(C_0 - C_e\right)V}{m} \tag{2}$$

$$q_t = \frac{\left(C_0 - C_t\right)V}{m} \tag{3}$$

where: C_0, C_e and C_t (μmol/L) are concentrations of BPA in the solution at the starting time, at equilibrium, and at any time t, respectively; V (L) is the volume of the solution, and m (g) is the mass of adsorbent used.

3. Results and discussion

3.1. Characterization of adsorbents

The SEM images of the lignite and the fly ash were performed at a magnification of 320× at 15 kV, and are shown in Fig. 1. SEM images show that the lignite grains are of an

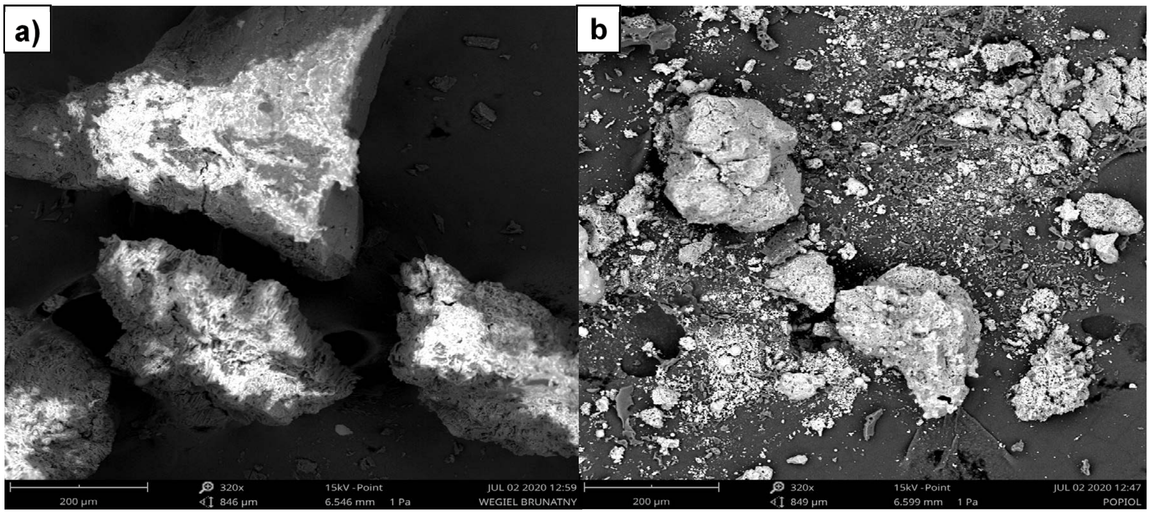

Fig. 1. SEM micrographs of the lignite (a) and fly ash (b) were performed at a magnification of 320× at 15 kV.

angular shape, while in the case of ash they are rounded. These grain shape differences result from the combustion of brown coal at high temperatures and the sintering of the minerals in the fly ash. The results of EDS analysis for both of the materials are presented in Table 1. The proportions of the contents of the main components (Ca, Al, Si, and Fe) apart from O and C in the lignite and the fly ash are similar, that is, approx. 8:1. Also, the proportions between the contents of the main components (Ca, Al, Si, and Fe), of course except for O and C, in the lignite and fly ash are preserved.

Table 1
EDS analysis for the lignite and fly ash

Chemical composition (wt.%)	Lignite	Fly ash
O	28.32	53.79
Ca	1.90	14.97
Si	1.37	10.95
Al	1.15	8.95
C	64.41	4.80
Fe	0.33	2.49
Mg	0.11	0.37
S	1.10	0.93
Ti	0.11	0.10
Na	0.44	0.32

The adsorption isotherms of water and benzene vapors on BC and FA are presented in Fig. 2. The specific surface areas (S_{BET}), as well as the micropore (V_{mi}) and mesopore (V_{me}) volumes of the materials, were calculated from the benzene adsorption isotherms, and results are listed in Table 2.

In the case of fly ash, the oxygen content is mostly bound in the form of oxides: mainly calcium, silicon, aluminum, and iron. On the other hand, in the case of lignite, a significant part of the oxygen is bound to the carbon matrix in the form of surface oxygen functional groups. They are numerous centers of water vapor adsorption. This is the reason for the much higher course of the water vapor adsorption isotherm on lignite compared to that determined for fly ash (Fig. 2a).

In the case of benzene vapor adsorption isotherms (Fig. 2b), the difference in their course results from the properties of both adsorbents. After burning, the fly ash contained in the lignite is very finely divided. This results in a great difference in surface development and porosity (Table 2).

3.2. Effect of initial adsorbent dosage

To investigate the effect of adsorbent dose, aqueous solutions of BPA with three initial concentrations of 25, 50, and 250 μmol/L (V = 0.02 L), and an adsorbent dosage equal to 1.25, 2.5, 5.0, 7.5 and 10 g/L were placed in Erlenmeyer flasks and agitated. The percentage removal of BPA at different adsorbent doses is given in Fig. 3, which shows that the adsorption of the adsorbate onto both adsorbents increases with increasing adsorbent dose. The increase in the percentage adsorption of the BPA with increasing the adsorbent amount may contribute to the availability of the larger adsorbent surface to the fixed amount of BPA. For all other experiments, the BC and FA amounts were fixed at 0.1 g/0.02 L (5 g/L).

3.3. Effects of solution pH and ionic strength

The pH and ionic strength of the solution are among the most important parameters greatly influencing the adsorption process. Solution pH determines the surface charge of the adsorbent and the degree of ionization and specification of adsorbate molecules. Similarly to the pH dependence, the presence of an inorganic salt in the solution might be a factor controlling the adsorbent-adsorbate electrostatic interactions, thereby increasing or decreasing the adsorption. The effects of solution pH on the adsorption of BPA on both of the adsorbents are shown in Fig. 4a. The adsorption of BPA on the brown coal remained stable at a pH range of 2–8 and decreased with the further increase in the pH from 8 to11. The adsorption capacity of the fly ash was insignificantly

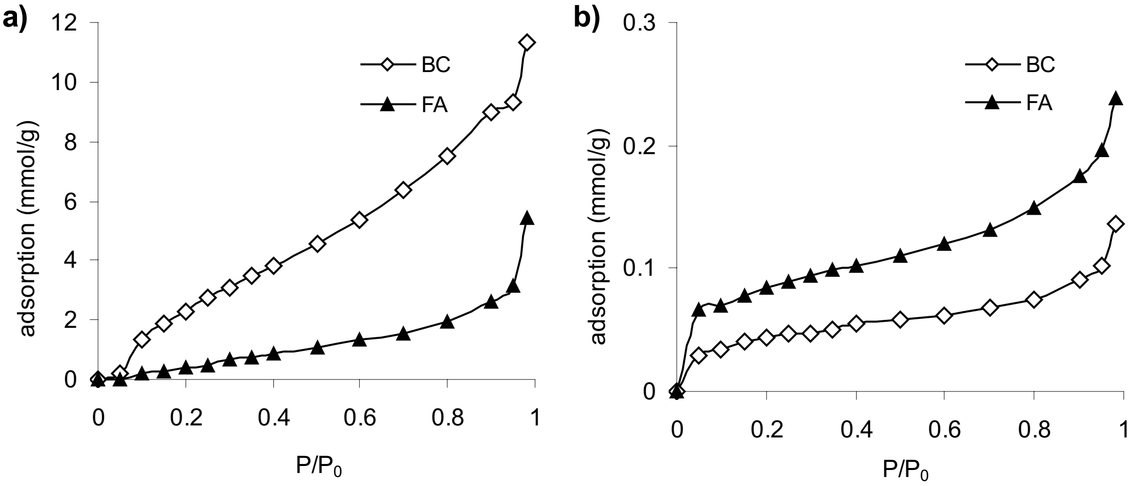

Fig. 2. Isotherms of water (a) and benzene (b) adsorption on the lignite (BC) and fly ash (FA).

Table 2
Textural properties of the lignite and fly ash

Parameter	BC	FA
S_{BET} (m^2/g)	8.3	15.4
V_{mi} (cm^3/g)	0.0036	0.0069
V_{me} (cm^3/g)	0.0091	0.0176

affected by the solution pH from 2 to 11. This difference is probably due to the higher pH_{pzc} value of the fly ash (11.0) compared to the lignite (7.1). The pH_{pzc} is the pH at which the net charge on the adsorbent's surface is equal to zero. At a pH lower than pH_{pzc}, the adsorbent's surface is positively charged, while at a pH greater than pH_{pzc}, the surface has a net negative charge. In this case, at pH values below 7.1, the lignite surface became positively charged, attracting negative species. On the other hand, at pH values above 7.1, it became negatively charged, attracting positive species. The pKa of BPA is 9.6; this means that at a pH greater than the pKa value (>9.6), the adsorbate molecules exist predominantly in anionic forms as negatively charged ions. So, the large reduction in the BPA adsorption on the lignite at higher pH occurs because of repulsive interactions between the negatively charged adsorbent surface and the negatively charged BPA molecules. The adsorption process on the fly ash was less sensitive to changes in solution pH since in the tested pH range (2 to 11), the FA surface is always positively charged.

The influence of ionic strength on BPA adsorption is shown in Fig. 4b. As can be seen, the increase in ionic strength from 0 to 0.1 mol/L NaCl resulted in a slight change in the amount of BPA adsorbed on both of the adsorbents. This suggests that electrostatic attractions play a negligible role in the mechanisms of BPA adsorption on the lignite and fly ash.

3.4. Adsorption kinetics

Fig. 5a shows the variation of the BPA amount (q_t) adsorbed as a function of time. The rate of adsorption on both of the materials was high at initial times of adsorption;

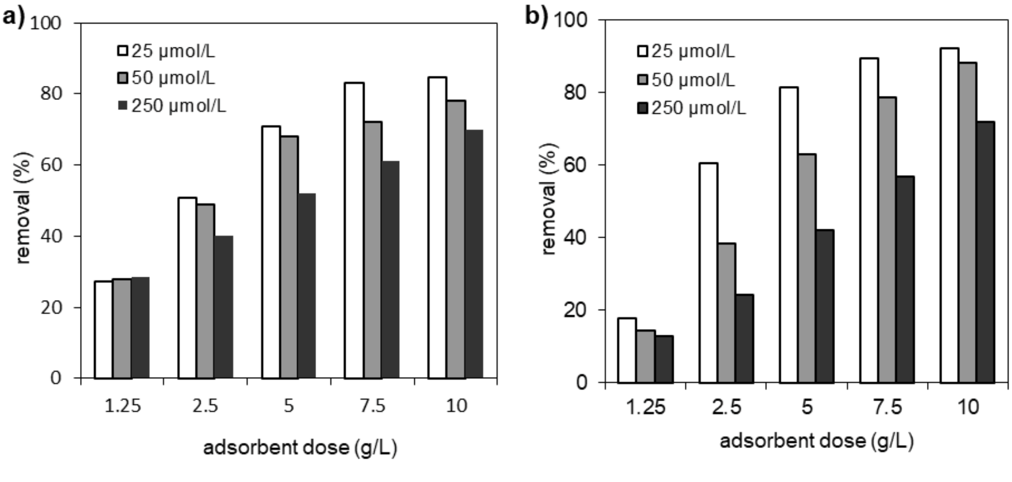

Fig. 3. Effect of adsorbent dose on the adsorption of BPA on the lignite (a) and fly ash (b).

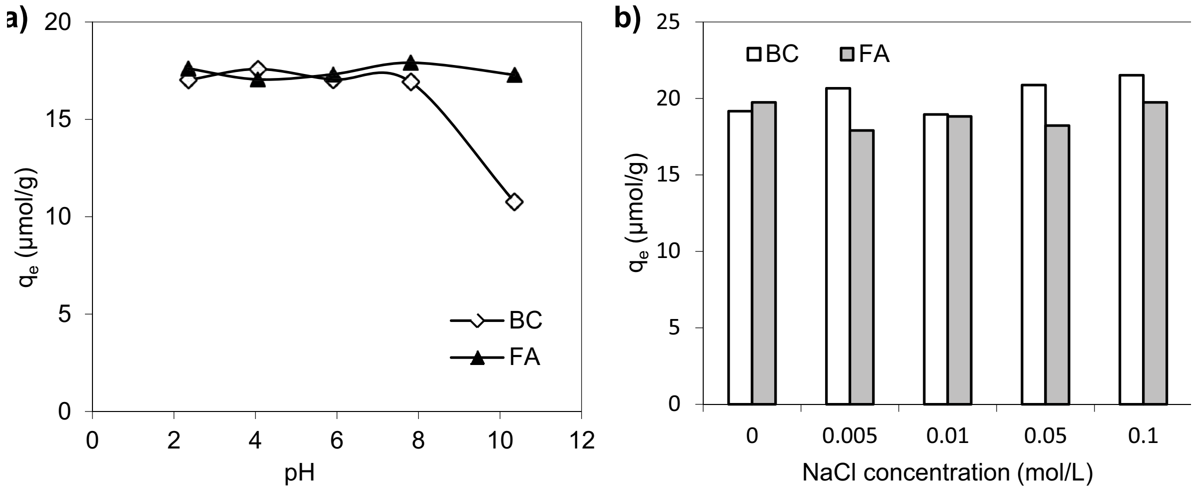

Fig. 4. Effect of solution pH (a) and ionic strength (b) on the adsorption of BPA on the lignite (BC) and fly ash (FA).

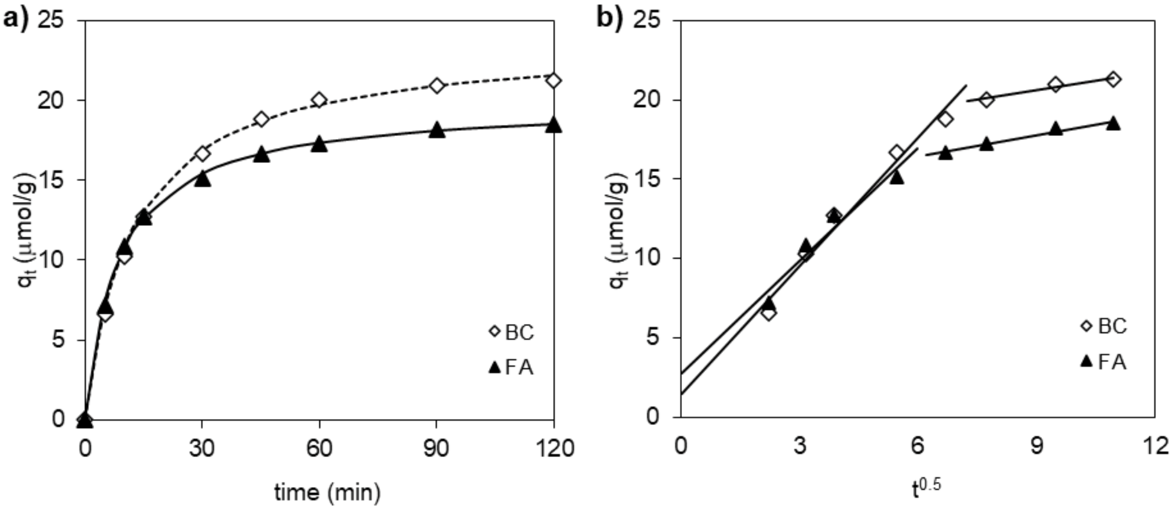

Fig. 5. Adsorption kinetics of BPA on the lignite and fly ash (a), and intraparticle diffusion model plots for the adsorption of BPA from aqueous solutions (b).

most of the adsorption took place within 15 min, and the adsorption equilibrium was reached after about 60 min. To further analyze the adsorption kinetics, the experimental data was fitted by pseudo-first-order Eq. (4), pseudo-second-order Eq. (5), and intraparticle diffusion Eq. (6) models:

$$\log\left(q_e - q_t\right) = \log q_e - \frac{k_1}{2.303}t \tag{4}$$

$$\frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{1}{q_e}t \tag{5}$$

$$q_t = k_i t^{0.5} + C_i \tag{6}$$

where k_1 and k_2 are the rate constants of pseudo-first-order (min^{-1}) and pseudo-second-order adsorption models (g/µmol min), respectively; k_i is the intraparticle diffusion rate constant (µmol/g min$^{-0.5}$), and C_i is the thickness of the boundary layer.

The chi-square test (χ^2) Eq. (7), along with the correlation coefficient (R^2) Eq. (8), were applied to determine the model that best fitted the experimental data.

$$\chi^2 = \sum_{i=1}^{n} \frac{\left(q_{e(\mathrm{exp})} - q_{e(\mathrm{cal})}\right)^2}{q_{e(\mathrm{cal})}} \tag{7}$$

$$R^2 = \frac{\sum_{i=1}^{n}\left(q_{e(\mathrm{cal})} - q_{e(\mathrm{mean})}\right)^2}{\sum_{i=1}^{n}\left(q_{e(\mathrm{cal})} - q_{e(\mathrm{mean})}\right)^2 + \sum_{i=1}^{n}\left(q_{e(\mathrm{cal})} - q_{e(\mathrm{exp})}\right)^2} \tag{8}$$

where: $q_{e(\mathrm{cal})}$ is model/predicted value, $q_{e(\mathrm{exp})}$ is experimental value, and $q_{e(\mathrm{mean})}$ is mean value.

Table 3 depicts the kinetic parameters for BPA adsorption on both of the adsorbents. According to the results, the pseudo-second-order kinetic model better described

Table 3
Kinetic parameters of pseudo-first-order, pseudo-second-order, and intraparticle diffusion models for the adsorption of BPA on the lignite and fly ash

Kinetic model	Adsorbent	
	Lignite	Fly ash
$q_{e(\mathrm{exp})}$ (μmol/g)	21.27	18.52
Pseudo-first-order model		
k_1 (min^{-1})	0.0892	0.0935
$q_{e(\mathrm{cal1})}$ (μmol/g)	30.85	29.24
R^2	0.789	0.768
χ^2	11.21	10.13
Pseudo-second-order model		
k_2 (g/μmol min)	0.0034	0.0058
$q_{e(\mathrm{cal2})}$ (μmol/g)	23.75	19.88
R^2	0.999	0.999
χ^2	0.039	1.717
Intraparticle diffusion		
k_{i1} (μmol/g min$^{-0.5}$)	2.694	2.370
R^2	0.976	0.939
k_{i2} (μmol/g min$^{-0.5}$)	0.385	0.440
R^2	0.947	0.965

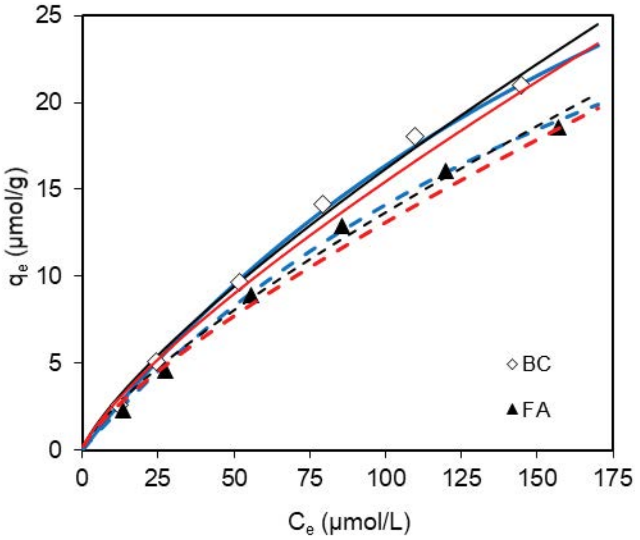

Fig. 6. Adsorption isotherms of BPA on the lignite (BC) and fly ash (FA) from aqueous solutions. Experimental results are shown together with fittings to the Freundlich (black line), to the Langmuir (blue line), and the Temkin isotherm models (red line).

the adsorption of BPA on the BC and FA than the pseudo-first-order model, due to the higher R^2 and lower χ^2 values that it exhibits. Furthermore, the experimental values of $q_{e(\mathrm{exp})}$ were found to be similar to the respective calculated ones using the pseudo-second-order equation. The pseudo-second-order adsorption rate constants (k_2) were 0.0034 and 0.0058 g/μmol min for BC and FA, respectively. It can be concluded that the adsorption process occurs faster on the fly ash. This is probably due to the higher mesopore volume of the FA (0.0176 cm^3/g) compared to the BC (0.0091 cm^3/g).

Fig. 5b presents the plots of q_t vs. $t^{0.5}$ of the intraparticle diffusion model. Straight lines with relatively high R^2 values can be found (Table 3). These straight-line plots do not pass through the origin, and they are non-linear over the whole time range. This suggests that the intraparticle diffusion involves the adsorption process and that the BPA adsorption is controlled by two or more mechanisms occurring simultaneously.

3.5. Adsorption isotherms

The equilibrium data were modeled with four isotherms: Freundlich, Langmuir, Temkin, and Dubinin–Radushkevich models [33].

The Freundlich isotherm is commonly given by the following linear equation:

$$\ln q_e = \ln K_F + \frac{1}{n}\ln C_e \tag{9}$$

where K_F ((μmol/g)(L/μmol)$^{1/n}$) and n are the Freundlich isotherm constants.

The Freundlich adsorption isotherm parameters were calculated from the slope and intercept of the linear plots of lnq_e vs. lnC_e.

The Langmuir equation is given as follows:

$$\frac{C_e}{q_e} = \frac{1}{q_m}C_e + \frac{1}{q_m b} \tag{10}$$

where q_m (μmol/g) and b (L/μmol) are the Langmuir parameters.

Both constants were obtained by plotting C_e/q_e vs. C_e.

The Temkin isotherm equation is expressed as:

$$q_e = \frac{RT}{b_T}\ln A_T + \left(\frac{RT}{b_T}\right)\ln C_e \tag{11}$$

where b_T (J/mol) is the Temkin isotherm parameter related to the heat of adsorption, A_T (L/g) is the Temkin isotherm constant, R is the gas constant (8.314 J/mol K), and T is the temperature (K).

The b_T and A_T parameters were calculated from the slope and intercept of the linear plots of q_e vs. lnC_e.

The Dubinin–Radushkevich isotherm is given by equation:

$$\ln q_e = \ln Q_m - K\varepsilon^2 \tag{12}$$

where K (mol^2/kJ^2) is a constant related to the adsorption energy, Q_m (μmol/g) is the theoretical saturation capacity, ε is the Polanyi potential, calculated from Eq. (13):

$$\varepsilon = RT\ln\left(1 + \frac{1}{C_e}\right) \tag{13}$$

A plot of lnq_e vs. ε^2 enables to determine the constant K and Q_m.

The parameters obtained from fitting of these models on adsorption data are compiled in Table 4. The applicability of the isotherm models were adjudged by the closeness to the unity of the R^2 values and lowest χ^2 values.

The results show that the Langmuir, Freundlich as well as Temkin isotherm models fitted reasonably well the adsorption data. However, the Langmuir model exhibited slightly better correlation coefficients and lower chi-square values than other models. The Dubinin–Radushkevich model turned out to be the least useful. For this model, the regression coefficient R^2 values were 0.815 for BC, and 0.851 for FA, respectively. The higher R^2 and lower χ^2 of the Langmuir model indicate the homogeneous adsorption of the BPA on the BC and FA surfaces leading to the formation of a monolayer coverage of the adsorbate molecules on the adsorbent surface. The monolayer adsorption capacities of the BC and FA for BPA were 58.43 and 47.68 μmol/g (13.34 and 10.88 mg/g), respectively. The best results were obtained for the BC despite its lower surface area compared with the FA. This suggests that the adsorption depended on the chemical properties of the adsorbent surface rather than its porous structure.

A dimensionless separation factor (R_L) can be used to predict a favorable or unfavorable nature of the adsorption process. The R_L can be defined by the following relationship:

$$R_L = \frac{1}{1 + bC_0} \tag{14}$$

where R_L is the constant separation factor of a solid–liquid adsorption system, b is the Langmuir equilibrium constant, and C_0 is the initial BPA concentration.

The value of R_L was used to predict whether the adsorption system was linear ($R_L = 1$), irreversible ($R_L = 0$), unfavourable ($R_L > 1$) or favourable ($0 < R_L < 1$). In this study, R_L ranging from 0.506 to 0.911 for BC and from 0.488 to 0.905 for FA indicates favorable adsorption for the adsorbate–adsorbent system. This is also confirmed by the n constant of the Freundlich equation (Table 4), which characterizes the strength of adsorption (the adsorption isotherm is linear when $n = 1$, favorable when $n < 1$, and unfavorable when $n > 1$).

Table 5 summarizes some low-cost adsorbents used for the removal of BPA from aqueous solutions. When comparing our results for raw lignite and its fly ash with the results of others, it can be concluded that these materials have adsorbed BPA as effectively as the other adsorbents listed.

4. Conclusion

The batch removal of bisphenol A from aqueous solution using low-cost adsorbents such as lignite and its fly ash under different experimental conditions was investigated in this study. The effects of initial adsorbent dose, solution pH, and ionic strength were investigated. The adsorption kinetics was investigated according to the pseudo-first-order, pseudo-second-order, and intraparticle diffusion models. The results obtained show that the adsorption kinetics on both adsorbents can be explained by a second-order equation better than the first-order model. The adsorption process occurs in two kinetic steps, and the intraparticle diffusion is not the rate-determining step. The equilibrium data

Table 4
Freundlich, Langmuir, Temkin and Dubinin–Radushkevich isotherm parameters for BPA adsorption on the lignite and fly ash

Isotherm model	Adsorbent	
	Lignite	Fly ash
Freundlich		
K_F ((μmol/g)(L/μmol)$^{1/n}$)	0.442	0.401
n	1.279	1.306
R^2	0.984	0.982
χ^2	0.385	0.504
Langmuir		
q_m (μmol/g)	58.43	47.68
b (L/μmol)	0.0039	0.0042
R^2	0.998	0.997
χ^2	0.085	0.109
R_L	0.506–0.911	0.488–0.905
Temkin		
b_T (J/mol)	338.5	364.3
A_T (L/g)	0.092	0.083
R^2	0.955	0.965
χ^2	1.859	1.189
Dubinin–Radushkevich		
Q_m (μmol/g)	14.34	12.93
K (×10^{-5} mol^2/kJ)	5.000	6.000
R^2	0.851	0.815
χ^2	10.25	13.25

Table 5
Comparison of BPA adsorption on various low-cost materials

Adsorbent	Adsorption capacity (mg/g)	Ref.
Lignite	13.34	This study
Fly ash	10.88	This study
Raw bentonite	3.41	[34]
H_2SO_4-coconut shell	4.16	[35]
H_2SO_4-durian peel	4.18	[35]
H_2SO_4-coir pith	4.31	[35]
H_2SO_4-coconut bunches	4.66	[35]
Modified Spanish broom cellulose fibers	4.86	[36]
Fe_3O_4	5.08	[37]
Rice husk ash	8.72	[38]
Surfactant modified bentonite	10.45	[34]
Barley husk	15.51	[39]
Raw fibric peat	15.97	[40]
Tyre pyrolysis char	17.10	[41]
Rice husk biomass	25.98	[42]
Chitosan	27.02	[43]
Coconut peels activated carbon	49.61	[44]
CO_2 activated tyre pyrolysis char	108.0	[32]

were analyzed using the Freundlich, Langmuir, Temkin, and Dubinin–Radushkevich isotherm models. The equilibrium process was described well by the Langmuir isotherm model and the maximum adsorption capacities were found to be 13.34 mg/g for BC and 10.88 mg/g for FA, respectively. The results indicated that lignite and fly ash are interesting materials for the removal of BPA from water. The lignite fly ashes can be accepted as inert wastes because the majority of the trace metals in their leachates were within the prescribed limits. Both materials used in this study exhibit adsorption efficiency better or comparable as many other low-cost adsorbents. Their great advantage is their availability in large quantities from lignite opencast mines or nearby power plants. Both materials can be used to remove organic pollutants from the soil-water environment.

The research results are important when considering the possibility of using the tested materials as fillings for sorption barriers or as additives to natural sorption barriers. These materials can be useful in the rehabilitation of degraded areas. In this case, the possibility of using brown coal both to restore the humic substance in the topsoil and to bind impurities is combined. It is also possible to use the tested sorbents as sleeve fillers to isolate pollution foci.

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