Magnetic Prussian Blue Nanocomposites for Effective Cesium Removal from Aqueous Solution

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ABSTRACT: Magnetic Prussian blue (PB) nanocomposites were synthesized by binding PB to a core of magnetite (Fe₃O₄) nanoparticles for highly efficient and rapid separation of cesium (Cs⁺) from aqueous solution. The average particle size of the magnetic PB nanocomposites was 13.6 nm, and they had a high surface area (322.19 m²/g), leading to efficient Cs⁺ adsorption capability. The nanocomposites showed a maximum sorption capacity of 280.82 mg/g at an initial Cs⁺ concentration of 50 mM, pH 7, and 10 °C, which is much higher than those of previously reported PB-based adsorbents for removing Cs⁺. The adsorption behavior followed pseudo-second-order kinetics and obeyed the Tempkin isotherm. The adsorption capacity of Cs⁺ on magnetic PB nanocomposites remained consistent even at high ionic competition in the simulated seawater. The obtained magnetic PB nanocomposite is a cost-effective adsorbent and can be easily retrieved from an aqueous solution by a magnet after decontamination of cesium. These results showed that the magnetic PB nanocomposite has extensive applicability for the removal of cesium from aqueous solution.

1. INTRODUCTION

In the early period of nuclear energy use, the emphasis was largely on the treatment of operational wastes, but over the past two decades the need to accommodate wastes arising from the decommissioning of nuclear installations has gradually increased. Among various radioactive nuclear wastes, cesium radioisotopes are the most important fission products because of their high fission yield (6.09%), long half-life (30.2 yr), high volatility, high activity, and high solubility. Therefore, effective removal of cesium in aqueous radioactive nuclear waste solutions is crucial for public health and environmental preservation.

In the last few decades, various methods for removing cesium ions from nuclear waste effluents have been proposed, such as solvent extraction, coagulation/sedimentation, sand filtration, electrodialysis, and adsorption. Among those available technologies, adsorption is a simple and economical technique, as it does not require further treatment after cesium ions are removed from the matrix. Many adsorbents, including polymers, biomass, and metal oxides, have been evaluated for the removal of cesium ions from groundwater, seawater, and nuclear waste solutions. On the other hand, there has been continuous investigation aimed at developing better and more easily available materials.

Prussian blue (PB), a class of cyano-bridge coordination polymers based on hexacyanometallates and transition metal ions, has been used as an adsorbent in separation of cesium because of its high selectivity for cesium. This observed result was correlated with a cubic framework of iron centers bound by cyanide bridges such that additional cations can intercalate into the interstitial sites. Small hydrated cations such as K⁺, Rb⁺, Cs⁺, and NH₄⁺ can penetrate the structure, whereas larger hydrated ions such as Na⁺ and Li⁺ cations are blocked. In recent years, there has been growing interest in application of nanoparticles as adsorbents for pollutant removal. The physicochemical properties of nanoparticles differ from those of their large counterparts (milli- or microsized particles); for example, they have a much higher specific surface area, high surface reactivity, increased quantum effects, and strong sorption capability. PB nanoparticles with hydrophilic defect sites exhibited very high cesium adsorption capability, as cesium ions were adsorbed onto the defect sites of the nanoparticle. Nanocomposites based on silica or glass matrices with PB nanoparticles exhibited higher adsorption capacity than the bulk PB analogues and showed a very high distribution coefficient for Cs⁺ ions in seawater. PB nanoparticles with interior hollow cavities and uniform particle sizes were synthesized by a controlled self-etching reaction on solid PB nanoparticles. These PB nanoparticles exhibited higher removal rates of Cs⁺ ions than commercial PB. However, practical application of PB can easily contaminate water owing to its intrinsic property of forming a colloid in water, and it is also hard to separate it from an aqueous solution using centrifugation or filtration after cesium sorption. Moreover, these absorbent materials can accumulate in the discharge, resulting in secondary pollutants. Therefore, developing nanocomposites that make these adsorbents magnetic and more easily separable would significantly improve the process.
In this study, magnetic PB nanocomposites with a core of magnetite (Fe₃O₄) nanoparticles were synthesized to remove cesium from an aqueous solution. The synthesized magnetic PB nanocomposites were characterized by field emission transmission electron microscopy (FE-TEM), a magnetic property measurement system (MPMS), X-ray diffraction (XRD), Fourier transform infrared (FT-IR) spectroscopy, and scanning electron microscopy with energy dispersive X-ray spectroscopy (SEM-EDX). The performance of the nanocomposites in adsorption experiments for cesium separation under different experimental conditions such as pH, temperature, contact time, initial Cs⁺ concentration, and competing ions was investigated, and the experimental results were analyzed using kinetic, thermodynamic, and isotherm models.

2. MATERIALS AND METHODS

2.1. Chemicals and Preparation of Magnetic PB Nanocomposites. Reagent chemicals (ferric chloride hexahydrate, ferric chloride tetrahydrate, potassium ferrocyanide trihydrate, cesium nitrate, etc.) were purchased from Sigma-Aldrich (South Korea) and used as received. Magnetite (Fe₃O₄) nanoparticles were prepared by a modified Massart method. Briefly, 0.7 M ammonia solution in 35 mL was added to an aqueous mixture of 1 M ferric chloride in 40 mL and 2 M ferrous chloride in 10 mL to form magnetite under stirring for 30 min at room temperature. The color of the mixture gradually changed from brown to dark brown. The product was washed with distilled water several times, isolated by centrifugation, and then dried in an oven at 60 °C for 24 h.

Magnetic PB nanocomposites were synthesized by binding PB to a core of magnetite (Fe₃O₄) nanoparticles. In this method, the molar ratio between magnetite and potassium hexacyanoferrate (K₄[Fe(CN)₆]·3H₂O) was set at 1:2.5. The prepared magnetite nanoparticles was redispersed in 300 mL of distilled water for 30 min. 50 mM ferric chloride in 100 mL was injected in the metal precursor solution drop by drop, and then 40 mM potassium hexacyanoferrate in 5 mL was slowly dropped into it under sonication at 30 °C. The color of the mixture rapidly changed from dark brown to dark cyan. The precipitated product was washed several times with methanol and distilled water by centrifugation until the supernatant solution turned neutral. The obtained solids were dried in an oven at 60 °C for 24 h.

2.2. Characterization of Magnetic PB Nanocomposites. The morphology and size of the nanoparticles were analyzed by FE-TEM (Titan G2 ChemiSTEM Cs Probe, FEI Company, Netherlands). The pore size distribution, pore volume, and pore diameter were determined according to the Barrett–Joyner–Halenda sorption model using a specific surface area analyzer (Autosorb-iQ & Quadrasorb SI, USA).
The specific surface area was also measured from the N₂ sorption/desorption isotherm fitted by the Brunauer–Emmett–Teller (BET) model. The presence of binding sites and their involvement in adsorption were investigated by FT-IR spectroscopy (Spectrum 100, PerkinElmer, USA). XRD patterns were obtained using Cu Kα radiation (λ = 1.54 Å) on a multipurpose X-ray diffractometer (X’pert PRO MRD, PANalytical, Netherlands). A MPMS (SQUID-VSM QM02, magnetometer) was used at room temperature with an applied field between −15,000 and 15,000 Oe. SEM (S-4200, HITACHI, Japan) with EDX (Horiba, E-MAX EDX detector) analysis was performed to examine the surface morphology and chemical composition of the adsorbents.

2.3. Adsorption Experiments. Inactive cesium nitrate (CsNO₃) from Sigma-Aldrich (USA) was used for the adsorption experiments. Batch equilibrium experiments were conducted in 50 mL conical tubes of solutions with different initial Cs⁺ concentrations containing 0.1 g of the synthesized magnetic PB nanocomposites. The conical tubes were constantly shaken for 24 h in a shaking incubator at 120 rpm. The samples were taken at predetermined time intervals and filtered. Sampling was followed by filtering through a 0.2 μm syringe filter and analysis of residual Cs⁺ in the solution. The sampling supernatant Cs⁺ solution was determined by inductively coupled plasma optical emission spectrometry (Optima 2100 DV, PerkinElmer Co., USA). Each sample was analyzed in triplicate, and the average values were used to calculate the sorption capacities. The pH values were adjusted to 2−9 using 0.1 M HNO₃ and 0.1 M NaOH. To investigate the competing ion effect, the magnetic PB nanocomposite was added to 50 mL of an enriched seawater solution containing NaCl (1.36 g) and MgCl₂ (0.19 g) at different initial Cs⁺ concentrations, and the test tubes were constantly shaken for 24 h in a shaking incubator at 150 rpm. Kinetic studies were conducted using cesium solutions with various initial concentrations (1, 10, 20, 30, and 50 mM) for different time intervals. The mechanism of cesium sorption kinetics was analyzed using pseudo-first-order and pseudo-second-order
kinetic models. Equilibrium isotherms for the Langmuir, Freundlich, and Temkin models were also obtained by varying the initial Cs+ concentration (1, 10, 20, 30, or 50 mM) at 10, 20, and 30 °C in a shaking incubator with a thermostat.

3. RESULTS AND DISCUSSION

3.1. Characterization of Magnetic PB Nanocomposites. Magnetite nanoparticles were successfully synthesized by the modified Massart method. The formation of magnetic nanoparticles was indicated by the development of a dark brown color. The formed nanoparticles had a mean diameter of approximately 5 ± 0.5 nm (Figure 1(a)).

Then, magnetic PB nanocomposites were synthesized by binding PB to a core of magnetite (Fe3O4) nanoparticles. The synthesized magnetic PB nanocomposites were isotropic (i.e., they have a low aspect ratio) in shape and size, as shown in the FE-TEM images, and their mean diameter was approximately 13.6 ± 0.5 nm (Figure 1(b)). BET analysis showed that the nanocomposites had a high surface area of 322.19 m²/g. The specific pore volume and pore diameter of the nanocomposites were 0.119 cm³/g and 2.579 nm, respectively. The XRD patterns of the magnetic nanoparticles and magnetic PB nanocomposites are shown in Figure 2(a).

Seven diffraction peaks around 2θ = 30.3°, 35.7°, 43.3°, 53.7°, 62.7°, and 74.8° were observed for magnetite nanoparticles, which could be assigned to the (220), (311), (400), (422), (440), (511), and (533) planes of the cubic spinel crystal structure of Fe3O4 (JCPDS card no. 19-0629). After PB was coated on the magnetite surface, four new diffraction peaks were observed at 2θ = 17.4°, 24.8°, 39.8°, and 50.4°, which further confirmed the existence of PB in the composites (JCPDS card no. 52-1907).

The FT-IR spectra of the magnetite nanoparticles and magnetic PB nanocomposites are shown in Figure 2(b). The characteristic stretching vibration band at 2082 cm⁻¹ shows the existence of the −C≡N− group, and the stretching band at 499 cm⁻¹ is attributed to the formation of Fe²⁺−CN−Fe³⁺, which indicates the presence of PB. In addition, the stretching bands near 3415 and 1560 cm⁻¹ refer to the O−H stretching mode and H−O−H bending mode, respectively, indicating the presence of interstitial water in the magnetic PB nanocomposites. However, the Fe−O bond at 570 cm⁻¹ in magnetite nanoparticles shifted to a higher wavenumber of 587 cm⁻¹ after PB was added. As the nanoparticles were reduced to nanoscale dimensions, the absorption bands of the FT-IR spectra shifted to higher wavenumber.

MPMS profiles of the magnetite nanoparticles and magnetic PB nanocomposites showed a typical magnetic hysteresis loop (Figure 3). The saturation magnetization of magnetite nanoparticles was 57.32 emu/g at room temperature. After PB was coated on the surface of the magnetite nanoparticles, the saturation magnetization decreased to 12.07 emu/g. The magnetic PB nanocomposites showed almost zero coercivity and zero remanence in the magnetization curve, indicating supermagnetic properties. The synthesized magnetic PB nanocomposites were quickly attracted toward a magnet, and the reaction solution became clear. Thus, the magnetic PB nanocomposites could be effectively collected from contaminated water by magnetic separation.

Figure 4 shows SEM-EDX peaks before and after Cs+ adsorption on magnetic PB nanocomposites. The mechanism of cesium ion binding on PB remains controversial. Some researchers assume that cesium ions are adsorbed into the crystal cage structure of metal hexacyanoferrate as an ion pair with a cation, whereas others consider that cesium ions are exchanged with potassium ions, especially on the surface layer of the adsorbent. The EDX peaks of magnetic PB nanocomposites before and after Cs+ adsorption confirmed that the total amount of potassium ions remained constant. Therefore, the experimental results confirmed that cesium ions were captured by the cage of the PB lattice structure, as there was no ion exchange between potassium and cesium ions.

3.2. Effect of pH. Metal adsorption from aqueous solutions can be greatly influenced by the solution pH, which affects both the binding sites and the metal chemistry. In this study, the effect of pH in the range from 2 to 9 on the adsorption capacity was investigated (Figure 5). A sharp increase in the adsorption was observed with increasing pH from 2 to 7, eventually reaching the highest value at pH 7. As the pH was increased further from 7 to 9, a significant negative effect of alkaline pH on the adsorption capacity was observed. A negative effect of alkaline pH on PB stability was reported. This low stability might be due to the strong interaction between ferric ions and hydroxyl ions at an alkaline pH, which results in Fe−CN−Fe bond cleavage in PB.

3.3. Effect of the Contact Time and Initial Cs+ Concentration. The effect of the contact time on the adsorption of Cs+ was investigated at different initial Cs+ concentrations (Figure 6). The adsorption of Cs+ on the magnetic PB nanocomposites increased with increasing contact time. The high removal rate at the start of the contact time was attributed to the large number of vacant binding sites available for the adsorption of Cs+. As the exterior surface became exhausted, the cesium ion uptake rate began to decrease, and finally an apparent equilibrium was reached within 4 h depending on the initial Cs+ concentration. This relatively short contact time might also provide an advantage for large-scale application of the adsorption method to real nuclear waste solutions. The adsorption capacity increased from 55.12 to 280.82 mg/g as the initial Cs+ concentration was increased from 1 to 50 mM at 10 °C. Therefore, the initial Cs+ concentration significantly affected its removal from aqueous solution. A high initial Cs+ concentration could result in a high mass gradient pressure between the solution and adsorbent, which provides a driving force to overcome the resistance to
mass transfer between the aqueous and solid phases. The Cs⁺ adsorption capacity of the magnetic PB nanocomposites in this study was compared with those obtained in previous studies of Cs⁺ adsorption using different PB-based adsorbents (Table 1). The adsorption capacity of the nanocomposites was higher than those of most of the adsorbents reported previously.

3.4. Effect of Competing Ions. The effect of other metal ions in seawater on the adsorption of Cs⁺ on magnetic PB nanocomposites was evaluated. The adsorption affinity of Cs⁺ on the nanocomposites was represented in terms of the distribution coefficient $K_d$ (mL/g), which is calculated as

$$K_d = \frac{(C_0 - C_e)V}{m}$$

where $C_0$ and $C_e$ are the initial and equilibrium concentrations of Cs⁺ (mg/L), $V$ is the volume of the solution (mL), and $m$ is the weight of sorbent used (g). The distribution coefficient gives the chemical binding affinity of the target metal ion to an adsorbent at dilute metal concentrations. Higher $K_d$ values indicate stronger binding affinity. A $K_d$ value of 5,000 or above is generally considered good, and a $K_d$ value greater than 50,000 is considered excellent. In this study, the $K_d$ values of Cs⁺ were evaluated in the presence of competing cations (Na⁺ and Mg²⁺) in the simulated seawater. The $K_d$ values were in the range of $(3-6) \times 10^5$ at initial cesium concentrations ranging from 1 to 50 mM at pH 7 and 10 °C. The experimental results suggest that common cations in seawater do not affect the binding of Cs⁺ on the magnetic PB nanocomposites.

3.5. Effect of Temperature. To examine the effect of temperature on the cesium adsorption performance of the magnetic PB nanocomposites, experiments were conducted at 10, 20, and 30 °C at initial cesium concentrations ranging from 1 to 50 mM at an initial pH of 7 (Figure 6). The temperature strongly affected the adsorption within the temperature range.
of 10−30 °C. For example, as shown in Figure 6, the adsorption of cesium ions decreased remarkably from 55.12 to 34.58 mg/g with increasing temperature from 10 to 30 °C at an initial Cs⁺ concentration of 1 mM. These results indicate the exothermic nature of Cs⁺ adsorption on PB nanoparticles.

The Gibbs energy $\Delta G^0$, enthalpy $\Delta H^0$, and entropy $\Delta S^0$ can be calculated from the following equations:

$$\Delta G^0 = -RT \ln K_c$$

(2)

where $K_c = q_e/C_e$, and

$$\ln K_c = \frac{\Delta S^0}{R} - \frac{\Delta H^0}{RT}$$

(3)

where $\Delta G^0$, $\Delta H^0$, and $\Delta S^0$ are the standard free energy change, standard enthalpy change, and standard entropy change, respectively; $K_c$ is the equilibrium constant, $q_e$ (mg/g) is the equilibrium concentration of Cs⁺ on the adsorbent, $C_e$ (mg/L) is the equilibrium concentration of Cs⁺ in solution, $R$ is the ideal gas constant, and $T$ (K) is the adsorption temperature.

The $\Delta H^0$ and $\Delta S^0$ values were obtained from a linear plot of $\ln K_c$ vs $1/T$. The equilibrium constant $K_c$ and $\Delta G^0$ changed from 1.46 to 1.14 and from $-3.39$ to $-1.03$ kJ/mol, respectively, with increasing temperature from 10 to 30 °C, which indicates the exothermic nature of Cs⁺ adsorption on the nanocomposites.

The negative value of $\Delta H^0$ ($-7.89$ kJ/mol) shows that heat was released during adsorption, indicating the exothermic nature of adsorption. The negative $\Delta S^0$ value ($-1.44$ J/mol·K) showed a decrease in randomness at the interface during adsorption of Cs⁺ on the nanocomposites.

3.6. Adsorption Kinetics. Pseudo-first-order and pseudo-second-order kinetic models and an intraparticle diffusion model were used to understand the dynamics of Cs⁺ adsorption. The pseudo-first-order model can be written in linearized form as follows:

$$\ln(q_e - q_t) = \ln q_e - k_1 t$$

(4)

where $q_t$ is the Cs⁺ concentration (mg/g) at any time ($t$), and $k_1$ is the first-order rate constant (min⁻¹). The Lagergren first-order rate constant ($k_1$) and equilibrium Cs⁺ concentration ($q_e$) values were calculated from the model (Table 2). The adsorption data were then analyzed using the pseudo-second-order kinetic model, which can be written in linearized form as follows:

$$t/q_t = 1/k_2 q_e^2 + t/q_e$$

(5)

where $k_2$ is the second-order rate constant (g/mg·min). The values of the regression coefficient were very high ($r^2 > 0.998$), and the experimental $q_{exp}$ values were in good agreement with the theoretical values calculated from the pseudo-second-order equation (Table 2). The pseudo-second-order kinetic model exhibits better correlation with the adsorption of Cs⁺ on the nanocomposites.
magnetic PB nanocomposites at different temperatures than the pseudo-first-order model. Therefore, the rate-limiting step in the adsorption process is the chemical interactions between the functional groups of adsorbent and cesium ions.

On the other hand, the pseudo-second-order model does not identify the diffusion mechanism. Therefore, the kinetic data were used to examine the presence or absence of intraparticle diffusion and determine whether intraparticle diffusion was the rate-limiting step. The Weber–Morris intraparticle diffusion model is commonly expressed using the following equation:

$$q_t = k_f t^{1/2} + C$$

where $k_f$ is the intraparticle diffusion rate constant (mg/g min), and $C$ (mg/g) is the intercept. The plot of the Cs$^+$ adsorption density (mg/g) as a function of $t^{1/2}$ was not linear over the entire time range, suggesting that more than one mode of adsorption occurred in the uptake of Cs$^+$ by the PB nanoparticles (Figure 7). This could be separated into two linear regions. The initial linear portion might represent external surface adsorption, in which cesium ions diffused through the solution to the external surface of the adsorbent, and the uptake rate was quite high. The later linear portion refers to gradual adsorption due to intraparticle diffusion in the micropores as the process approached the equilibrium stage. The intraparticle diffusion rate constants were calculated from the slope of the multilinear plots at different temperatures (Table 3). The external mass transfer rate ($k_i$) in the first linear portion was much higher than the intraparticle diffusion rate ($k_2$) in the second linear portion. The values of the intercept C

in the second linear portion (intraparticle diffusion) provide information on the thickness of the boundary layer. The boundary layer thickness decreased with increasing temperature (Table 3). This suggests that surface diffusion is more important at high temperatures because of the greater random motion associated with the increased thermal energy. The plot of $q_t$ vs $t^{1/2}$ using initial kinetic data for up to 2 h produced high correlation ($r^2$) values (Table 3), which indicates that intraparticle diffusion might play a significant role in the initial stage of Cs$^+$ adsorption onto the magnetic PB nanocomposites.

### 3.7. Adsorption Isotherms.

The equilibrium distribution of Cs$^+$ between the adsorbent and liquid phase is important for determining the maximum adsorption capacity of the magnetic PB nanocomposites for Cs$^+$ and understanding the adsorption mechanism. Nonlinearized forms of three different adsorption models, the Langmuir, Freundlich, and Tempkin isotherms, were investigated to examine the experimental data. The Langmuir isotherm assumes that adsorption occurs at specific homogeneous sites on the surface of the adsorbent and can be given as:

$$q_e = \frac{q_{max} K_L C}{1 + K_L C}$$

where $q_{max}$ (mg/g) is the maximum sorption capacity of the adsorbent, and $K_L$ (L/mg) is the Langmuir constant related to the free energy of sorption. The Freundlich equation based on sorption on a heterogeneous surface is given as:

$$q_e = K_F C_e^{1/n}$$

where $K_F$ and $n$ are Freundlich constants.
where $K_F$ (L/g) is the Freundlich constant related to the sorption capacity of the adsorbent, and $n$ (dimensionless) is the Freundlich exponent, which indicates the adsorption intensity. The Tempkin isotherm assumes that the sorption energy during the sorption process decreases linearly rather than exponentially with increasing saturation of sorption sites. The Tempkin isotherm is given as:

$$q_e = \frac{RT}{b \ln(AC_0)}$$

(9)

To evaluate the fitness of the isotherm equations, the chi-squared ($\chi^2$) values and average percentage errors (APEs) were calculated using the following equations

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

(10)

$$\text{APE} = 100 \sum_{i=1}^{n} \left( \frac{q_{e,\text{cal}} - q_{e,\text{exp}}}{q_{e,\text{exp}}} \right)$$

(11)

where $n$ is the number of observations, and $q_{e,\text{exp}}$ and $q_{e,\text{cal}}$ are the experimental and calculated values, respectively. The nonlinearized isotherm coefficients were estimated using a nonlinear regression method (MATLAB 7.8). Figure 8 shows the three equilibrium adsorption isotherm models applied to the experimental data. The adsorption data were fitted better by the Tempkin isotherm than by the Langmuir and Freundlich isotherms; this was confirmed by the higher correlation coefficients $r^2$ and lower APE and $\chi^2$ values (Table 4). The maximum adsorption capacity ($q_{\text{max}}$) of the magnetic PB nanocomposites for Cs$^+$ was 280.82 mg/g at an initial Cs$^+$ concentration of 50 mM, which is in line with the value of 294.39 mg/g theoretically calculated using the nonlinear Langmuir model. To confirm the adsorption favorability, the separation factor $R_L$ was obtained using the following equation:

$$R_L = \frac{1}{1 + K_C C_0}$$

(12)

where $R_L$ is dimensionless and indicates the shape of the isotherm, $C_0$ is the initial Cs$^+$ concentration, and $K_C$ is the obtained Langmuir constant. The value of $R_L$ indicates that an isotherm is irreversible ($R_L = 0$), linear ($R_L = 1$), unfavorable ($R_L > 1$), or favorable (0 < $R_L < 1$). As the calculated $R_L$ values were between 0 and 1, Cs$^+$ adsorption on the magnetic PB nanocomposites was favorable.

4. CONCLUSIONS

In this study, an environmentally friendly magnetic PB nanocomposite was developed for removing cesium ions from aqueous solution. XRD, FE-TEM, and FT-IR observations all indicated successful coating of magnetite cores with PB. The magnetic PB nanocomposites exhibited a much higher Cs$^+$ adsorption capacity than other PB-based adsorbents reported previously owing to their smaller particle size (13.6 nm) and higher surface area (322.19 m$^2$/g). The adsorption behavior of the PB nanoparticles was fitted well with the Tempkin isotherm and the pseudo-second-order kinetic model. The obtained magnetic PB nanocomposite is a cost-effective adsorbent and can be easily retrieved from an aqueous solution by a magnet after decontamination of cesium. Therefore, magnetic PB nanocomposites are considered to be an excellent adsorbent for the removal of Cs$^+$ in aqueous solution.

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Notes

The authors declare no competing financial interest.

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REFERENCES


Table 4. Comparison of Nonlinearized Isotherm Models for the Adsorption of Cs$^+$ onto Magnetic PB Nanocomposites

<table>
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<tr>
<th>isotherm model</th>
<th>isotherm parameters</th>
<th>$\chi^2$</th>
<th>APE</th>
<th>$r^2$</th>
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<td>Langmuir</td>
<td>$q_{\text{max}} = 294.39$, $K_L = 0.18$</td>
<td>4.19</td>
<td>5.71</td>
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<td>Freundlich</td>
<td>$K_F = 77.39$, $n = 2.98$</td>
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<td>Tempkin</td>
<td>$A = 55.82$, $b = 2.70$</td>
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<td>1.99</td>
<td>0.992</td>
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</table>

$^{10}$C, initial pH = 7.0, 120 rpm.
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