High efficient removal of lead(II) and cadmium(II) ions from multi-component aqueous solutions using polyacrylic acid acrylonitrile talc nanocomposite

Mohamed Ragab Abass1 · Wafaa Mohamed El-Kenany1 · Eman Hassan EL-Masry1

Received: 31 December 2021 / Accepted: 18 May 2022 / Published online: 26 May 2022 © The Author(s) 2022

Abstract
This study is interested in the removal of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) onto polyacrylic acid acrylonitrile talc P(AA-AN)-talc nanocomposite. P(AA-AN)-talc was fabricated using γ-irradiation-initiated polymerization at 50 kGy. Different analytical tools were used to investigate the functional groups, morphology, particle size, and structure of this composite. The ability of P(AA-AN)-talc to capture (Pb2+, Cd2+, Co2+, Zn2+, and Sr2+) as multi-component aqueous solutions was performed by a batch method. Saturation capacity and the effect of (agitating time, pH, initial metal concentrations, and temperature) were investigated. The distribution coefficients at different pHs have order: Pb2+ > Cd2+ > Co2+ > Zn2+ > Sr2+. The saturation capacity decreases by increasing heating temperatures. Reaction kinetic obeys the pseudo-second-order model. Sorption isotherms are more relevant to a Langmuir isotherm, and the monolayer sorption capacity is closed to saturation capacity. Thermodynamic parameters (ΔG˚, ΔH˚, and ΔS˚) were endothermic and spontaneous. P(AA-AN)-talc is used for loading and recovery of studied cations in the column system. The study confirms that P(AA-AN)-talc is a promised composite for the sorption of the studied ions from aqueous solutions and should be considered as potential material for decontaminating these ions.

Keywords Gamma-irradiation · Polymerization · Talc · Distribution coefficients · Thermodynamic · Column

Introduction
Radioactive waste is obtained through a wide scale of operations in research facilities, recycling plants, power plants, diagnostic medicine, and isotope manufacturing (Nilchi et al. 2007; Hamed et al. 2016b). The stream of radioactive waste from nuclear facilities includes low, intermediate, and highly radioactive wastes and also includes emitting isotopes. Furthermore, these streams could contain different hazardous elements from decontamination procedures. These wastes must be treated to decrease dangerous element concentrations to levels suitable for discharge into the environment (Nilchi et al. 2007). Recently, the contaminants captured from wastewater have become a major challenge, as their significance is extending with increasing industrial activities (Hamoud et al. 2014). Various technologies can be used to remove pollutants as, for example, chemical precipitation (Ahn et al. 2020), physical treatment like solvent extraction (Brockmeyer et al. 2015; Hurtado-Bermúdez et al. 2018), adsorption (Abass et al. 2021a, b), biosorption (Gupta et al. 2019), membrane (Mehta et al. 2020), and ion exchange (El-Dessouky et al. 2018). Because of the high toxicity of Pb(II) and their widespread presence in the area, lead ions have a particular interest. It is an industrial contaminant, which arrives in the ecosystem through the soil, air, and/or water (Arbabi et al. 2015). Cadmium is common for electroplating,
nuclear reactors, and certain industrial paints (Aglan et al. 2019). Cd(II) has several isotopes along with ¹⁰⁹Cd; its half-life is 1.27 years and is used as rods for controlling and shielding the absorption of neutrons in reactors with supplementary components (Aglan et al. 2019). Co(II) is used in nuclear, medical, enamel, semiconductor industries, and electricity galvanizing. Co(II) can cause adverse effects on health like asthma, heart damage, and destroying the thyroid gland and liver (Kulkarni 2016). Zinc is obtained through effluents discharged from industries, like pigments, electroplating, battery manufacturing, metallurgy, and municipal wastewater management plants. Zn(II) is a toxic ion and can cause damage to human life by bioaccumulating in the food chain (Zhang et al. 2017). Strontium radionuclide is formed by ⁸⁹Sr (t₀.₅ = 51d) and ⁹⁰Sr (t₀.₅ = 29y) and is produced not only from nuclear plants as fission radionuclides but also from nuclear fuel rods in the pre-treatment phase (Ali et al. 2020). The retention of Cd(II), Co(II), Pb(II), Zn(II), and Sr(II) from liquid and radioactive waste solutions has been achieved by many researchers (Wang et al. 2010; Dubey et al. 2014; Kang et al. 2016; Zhang et al. 2017; Aglan et al. 2019; Pyrzynska 2019).

Talc ore is a naturally occurring, low-cost, and small particle size material. Talc ore has a platy structure that consists of three layers, including the Mg-OH layer sandwiched among two (SiO₃)²⁻ layers. Adjacent layers are linked by van der Waals forces. Talc powder surface has a huge number of Si–O-Si, Mg-O, OH, and O-Si–O bonds which will coordinate with transition elements in H₂O or soil to capture them on the talc powder surface (Thi Huong et al. 2020). Talc is commonly used as a filler, coating, and dusting agent in paints, plastics, lubricants, pharmaceuticals, papers, cosmetics, and ceramics manufacture. There are few studies on the adsorption characteristics of heavy metal ions in talc (Andrić et al. 2014; Kalantari et al. 2014). Modified talcum powder using 50% HNO₃ acid was used for sorption of methylene blue from wastewater (Wenlei et al. 2014). Fe₃O₄/Talc nanocomposite was fabricated by the coprecipitation-ultrasonic technique and used for the capture of Cr(IV) from aqueous solutions (Thi Huong et al. 2020). Fe₃O₄/Talc nanocomposite was used for the removal of Cu(II), Ni(II), and Pb(II) ions from aqueous solutions (Kalantari et al. 2014).

Polymeric resins have some advantages like high capacity, non-toxic, high chemical stability, and low production cost, but they have some disadvantages such as low selectivity toward hazardous ions, ability to swell, low radiation, and thermal stability. The sorption efficiency of the talc can be improved by adding polymeric resins. The impregnation of talc ore to organic polymer is more useful due to creating new materials having higher sorption capacity, fast adsorption rate, and selectivity for target elements, as well as reducing some disadvantages present in polymeric resins. The novelty of this study is the impregnation of talc inside polymeric resin layers using the gamma radiation technique as a new nanocomposite with high sorption performance for some metal ions from aqueous solutions, in addition to the need to develop and use new economic materials that have specifications suitable for working in radioactive waste treatment conditions.

In the present work, the fabrication and characterization of P(AA-AN)-talc nanocomposite using gamma radiation at 50 kGy were achieved. The sorption behavior of Cd(II), Co(II), Pb(II), Zn(II), and Sr(II) onto P(AA-AN)-talc nanocomposite is discussed. Chromatographic separations of binary and multi-component systems were applied on P(AA-AN)-talc nanocomposite.

### Experimental

#### Materials

All salts and reagents in this work have an analytical grade and are used without further purification. CoCl₂, 6H₂O, SrCl₂, Pb(NO₃)₂, ZnCl₂, CdCl₂, acrylic acid (AA), and acrylonitrile (AN) from Alpha Chemika, India; HCl and HNO₃ from Merck, Germany; and sodium hydroxide from El-Nasr Co., Egypt. The rate of the dose was ~ 473.35 Gy/h.

#### Gamma cell

Co⁶⁰ γ-cell source of class MC-20 (Russia) was operated as radiation of polymerization process at the Cyclotron factory, Egypt. The rate of the dose was ~ 473.35 Gy/h.

#### Fabrication of P(AA-AN)-talc

At 303 ± 2 K and constant agitation, 10% (AA and AN) solutions were added dropwise to 10% talc solution, and 0.1% N, N-methylene-bisacrylamide (cross-linker) was distributed for the production of P(AA-AN)-talc with volumetric ratios (AA:AN: talc) equal unity. The solution remained agitated for 2 h to overcome complete homogeneity then subject to γ-irradiation at 50 kGy over a time of about 105 h. After radiation, the hydrogel was cut into small fragments, soaked in acetone to eliminate unreacted monomers, then washed with DDW, and dried at 333 ± 2 K. The solid was ground to powder and converted into the H⁺ form by mixing for 1 day with 0.1 M HNO₃. The solid product was decanted and rinsed with DDW to eliminate the excess acid and dried at 333 ± 2 K.
The studied material was analyzed by Fourier transform infrared spectroscopy (FT-IR) (Alpha II Bruker, Germany) at 4000–400 cm\(^{-1}\). X-ray diffraction (XRD) was done (Shimadzu XD-D1, Japan). Differential thermal analysis (DTA) and thermogravimetric analysis (TGA) were done by a Shimadzu DTG-60 H. The surface morphology of the solid was recorded by scanning electron microscope (SEM) model Philips XL 30. The percent of elemental composition was detected by energy-dispersive X-ray (EDX) analysis. Transmission electron microscopy (TEM) and selected area electron diffraction (SAED) pictures were performed using JEM2100, Jeol. s.b, Japan. Atomic absorption spectrophotometer (Buck Scientific, VGP 210) was used to measure the concentrations of Cd(II), Co(II), Pb(II), Zn(II), and Sr(II).

**Chemical stability**

The stability of P(AA-AN)-talc to different solvents was achieved by mixing 0.05 g of solid and 50 mL of H\(_2\)O, HNO\(_3\), HCl, and NaOH in concentrations [1–6 M] in the desired solution with intermittent agitating for about three days at 298 ± 1 K. The filtrates were dried using IR lamps to dry, and then, the residue was tested gravimetrically by weight difference by a sensitive analytical balance (Abou-Mesalam et al. 2018).

**Point zero charge (PZC) determination**

PZC for P(AA-AN)-talc was investigated by using a 10-mL bottle, and 0.1 g of solid was added to 10 mL of desired pH solutions. The pH was adjusted by 0.1 M (HCl and NH\(_3\)) to find the different pHs of (2–12). The pH of the supernatant in each tube was represented as pH\(_i\). The samples were agitating for 24 h using a rotary agitator at 200 rpm. After settling, the pH of the supernatant in each tube was measured and represented as pH\(_f\). The PZC was determined from the plot of ΔpH versus pH\(_i\).

**Adsorption experiments**

**Preliminary study**

The experiment was done by shaking 0.05 g of two samples (talc and P(AA-AN)-talc) with 5 mL of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) (100 mg/L) with V/m = 100 mL/g in a shaker thermostat (Kottermann D-1362, Germany) at 300 ± 1 K. Sorbent and the sorbate solution were agitated in a shaker thermostat at 180 rpm and after sorption (24 h), and the samples were separated from the solution by centrifuge. Ion concentrations in the test solutions were detected before and after sorption by AAS. The uptake percentage is calculated by Eq. (1) (Khataee et al. 2013; Hassani et al. 2014; Hamed et al. 2019):

\[
\% \text{Uptake of } 
\begin{align*}
\text{Cd}^{2+}, \text{Co}^{2+}, \text{Pb}^{2+}, \text{Zn}^{2+}, \text{and } \\
\text{Sr}^{2+}
\end{align*}
\text{ = } \left( \frac{C_i - C_f}{C_i} \right) \times 100
\]

where \(C_i\) and \(C_f\) are the initial and final concentrations of sorbed metal ions in solution, respectively, and the data are tabulated in Table 1. These results reveal that the percent uptake was increased from about 68.2 to 96.9% for Pb(II), from about 61.8 to 91.1% for Cd(II), from about 42.2 to 48.9% for Co(II), from about 21.5 to 29.5% for Zn(II), and from about 8.2 to 11.8% for Sr(II); these data confirm that a great enhancement was noticed in the sorption of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) onto P(AA-AN)-talc, and this modified nanocomposite was used for further experimental work.

Batch sorption studies of the Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) onto P(AA-AN)-talc in H\(^+\) form. The variation in sorption parameters such as pH (1–5), contact time (5–240 min), metal ion concentrations (50–600 mg/L), and temperatures (300, 313, and 333 K) is patterned to get the optimum condition for the sorption process.

The separation factor may be considered as the relative tendency of different ions to be adsorbed in an exchanger from solutions. It is used as a measure of the possibility of chromatographic separation and is also expressed as the ratio of the distribution coefficients of the ions to be separated. The distribution coefficients (\(K_d\)) and separation factors (\(a^a_b\)) as a function of pH are calculated with the Eqs. (2) and (3) (Metwally et al. 2019):

\[
K_d (\text{mL/g}) = \left( \frac{C_i - C_f}{C_f} \right) \frac{V}{m}
\]

\[
a^a_b = \frac{K_d(a)}{K_d(b)}
\]

where \(C_i\) and \(C_f\) are the initial and final concentrations of Cd(II), Co(II), Pb(II), Zn(II), and Sr(II) in solution, respectively. \(V\) is the volume of solution (mL) taken in (5 mL), and \(m\) is the mass of the sorbent (g) taken in (0.05), and \(a\) and \(b\) are two challenging types in a system.

**Table 1** The % uptake of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) sorbed onto talc and P(AA-AN)-talc at 300 ± 1 K

| Samples       | Pb(II) | Cd(II) | Co(II) | Zn(II) | Sr(II) |
|---------------|--------|--------|--------|--------|--------|
| Talc          | 68.2   | 61.8   | 42.2   | 21.5   | 8.2    |
| P(AA-AN)-talc | 96.9   | 91.1   | 48.9   | 29.5   | 11.8   |
Saturation capacity

Repeated equilibration of 200 mg/L studied cations with the P(AA-AN)-talc in \( V/m = 200 \text{ mL/g} \) was carried out for the saturation capacity determination at different heating temperatures (323–673 K). The mixtures were agitated in an agitator at 300 ± 1 K for 2 h. After equilibrium, the solution was separated, and the concentration of the metal ions was measured. This procedure was repeated many times with fresh solutions until the nanocomposite was saturated with metal ions. The saturation capacity \( q \) (mg/g) was calculated from Eq. (4) (Hassan et al. 2021; Ibrahim et al. 2021):

\[
q = \left( \frac{C_i - C_e}{m} \right) X V
\]  

where \( C_e \) is the equilibrium concentration of Cd(II), Co(II), Pb(II), Zn(II), and Sr(II); \( V \) is the volume of solution (L) taken in (0.1 L); and \( m \) is the mass of P(AA-AN)-talc (g) taken in (0.05 g).

Column separation

Chromatographic breakthrough investigations were directed as follows: 0.5 g of solid was packed in a plastic column (0.3 cm radius and 5 cm heights) to give a bed height of 1.1 cm\(^3\) volume. At a rated flow of 4 drops/min, 600 mL of 100 mg/L desired metal ions at pH = 4 was passed through the column beds. A breakthrough capacity (BTC) values were computed by the formula (Abass et al. 2021a):

\[
\text{Breakthrough capacity (mg/g)} = \frac{V_{50\%} C_i}{m}
\]  

where \( V_{50\%} \) is a volume for effluent at 50% breakthrough (L).

Results and discussion

FTIR

FTIR spectrum of P(AA-AN)-talc is shown in Fig. 1(A). This figure shows that bands at 3739 and 3568 cm\(^{-1}\) are due to H\(_2\)O and OH absorbed on P(AA-AN)-talc (stretching vibration) (Abass et al. 2022b). A strong band at 2229 cm\(^{-1}\) is due to the C≡N bond of acrylonitrile (stretching vibration) (Lee et al. 2012a). Two bands at 1574 and 1511 cm\(^{-1}\) are due to N–O (stretching vibration) (Dawood and Li 2014). The band at 1395 cm\(^{-1}\) is due to O–H of carboxylic acid (bending vibration) (Lee et al. 2012b). The band at 1212 cm\(^{-1}\) is due to symmetric C-N (bending vibration) (Dawood and Li 2014). The band at 951 cm\(^{-1}\) is attributed to Si–O-H deformation vibration (Borai et al. 2015) or Si-CH\(_2\) (Mirzayev et al. 2021). The band at 722 cm\(^{-1}\) may be attributed to the Si–O-M (where M = Mg, Ca, and/or Al) (Nabi et al. 2011; Abass et al. 2022a). The band at 478 cm\(^{-1}\) may be attributed to the Si–H (Mirzayev et al. 2021). The three later bands deep-rooted the in-situ precipitation of talc (Mg, Al, and Si) in the net of a polymeric compound as seen later in EDX analysis.

XRD analysis

XRD analysis of P(AA-AN)-talc nanocomposite is shown in Fig. 1(B). This figure indicated that P(AA-AN)-talc has a crystalline structure and exhibits many sharp peaks centered...
at 6.19°, 18.64°, 19.64°, 20.9°, 21.86°, 24.94°, 26.88°, 28.36°, 31.32°, 34.61°, 35.17°, 36.7°, 39.83°, 44.67°, 50.08°, and 60.0° related to Miller index indications (020, 002, 003, 11–1, 111, 11–2, 004, 023, 113, 005, 131, 201, 132, 133, 116, 045, and 060), respectively, with JCPDS number 00–901-0163 confirming their crystalline nature with a monoclinic system, and this result was parallel to the XRD of chlorite (Zanazzi et al. 2007), Fe₃O₄/Talc nanocomposite (Thi Huong et al. 2020), and talcum powder (Wenlei et al. 2014).

**Thermal analysis**

The thermal analysis (TGA and DTA) of P(AA-AN)-talc nanocomposite (Fig. 2), showing that the weight loss process occurred via three steps. From 367 to 583 K due to the lessening of all moisture and free H₂O of P(AA-AN)-talc (Hamoud et al. 2014), the lost weight is 7.5%. From 583 to 733 K due to subtraction of H₂O of crystallization (Abass et al. 2021b) and complete decay of the organic part of P(AA-AN)-talc (El-Aryan et al. 2015), the lost weight is 34.2%. The third stage of weight loss (15.7%), which occurred from 733 to 1073 K caused by heating in N₂ gas and the cycling reaction with C≡N bonds converted to C= N bonds (El-Aryan et al. 2015). DTA shows 2 peaks at 379 K and 664 K (endothermic), due to the subtraction of all surface and matrix-bound H₂O from the polymeric compound and dehydration of carboxylic acid and decarboxylation of acrylic acid, respectively. Two exothermic peaks at 458 K and 592 K correspond to comprehensive decomposition of the organic molecule. From the data of TGA (Fig. 2), the losses weight is continued up to 1073 K. The full losses weight for P(AA-AN)-talc with the heating temperature is 57.7%.

**SEM**

The performance of the particulate composite is detected by particle dispersion inside the matrix; a smooth particle distribution contributes to uniform powder properties, as it is shown in Fig. 3(A). Proper talc dispersion is noticed at diverse points of the sample cross sections with no presence of agglomeration. Also, a favored orientation of talc is appreciated for the composite, resulting from their plate-like structure (Andrić et al. 2014).

**EDX**

Figure 3(B) shows clearly distinct phases of Al, Mg, Si, O, and C, and the resulting relative percentages of different metals at the points are C = 55.3%, O = 20.2%, Si = 7.16%, Al = 6.3%, Mg = 0.29%, and Ca = 1.2. These results confirmed the impregnation of talc in polymeric resin layers as seen in the FTIR analysis mentioned earlier.

**TEM**

TEM images of P(AA-AN)-talc (Fig. 4(A)) were used to estimate small particle size and were extended from 3.1 to 6.34 nm. Also, P(AA-AN)-talc shows a spherical morphology. Also, the TEM image shows a good distribution of talc inside the polymeric resin. The selected area electron diffraction (SAED) pattern (Fig. 4(B)) shows the central, intense, and direct beam such as a pattern, with a sharply focused spot, confirming that P(AA-AN)-talc is at least partly crystalline (Carter et al. 1996). And these results confirm data calculated from XRD as mentioned earlier. The average particle size of the new material is 4.99 nm.
Stability against chemicals

The chemical stability of P(AA-AN)-talc towards different solvents (H₂O, HNO₃, HCl, and NaOH) was performed and the results are tabulated in Table 2 and indicated that the prepared P(AA-AN)-talc is stable in H₂O and HCl; also, the investigated sample is stable in HNO₃ and NaOH up to 3 M; meanwhile, it starts to dissolve at 4 M and greatly dissolves at 6 M (Abou-Mesalam et al. 2018). Also, these results reveal that P(AA-AN)-talc is stable according to sequence order: HCl > HNO₃ > NaOH.

Point zero charge

Point zero charge (PZC) for P(AA-AN)-talc was determined by using plots of ΔpH vs pH (Fig. 5). From this Figure, it is clear that at acidic medium (pH < 7) the ΔpH increases with increasing pH, then the ΔpH decreased at alkaline medium (pH > 7), and PZC was determined at pH value (2.7); this value revealed that the surface charge of P(AA-AN)-talc becomes neutral at this value, and above this value, it becomes negatively charged which leads to improve cationic species, whereas the surface of P(AA-AN)-talc is positively...
charged below this pH value which leads to improve anionic species.

**Adsorption investigations**

**Distribution coefficients (Kd)**

Figure 6(A) shows the $K_d$ difference of $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ (100 mg/L) on P(AA-AN)-talc in H⁺ form, as a function of pH. The $K_d$ improves with the increase in pH. At lower pH lower than three, $K_d$ of the studied cations was inhibited; this is attributed to the existence of excess protons competing with $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ in the solution and preferably occupying the binding sites existing in P(AA-AN)-talc (Metwally et al. 2019). At pH $\geq$ 3, $K_d$ continuously increases with the increase in pH due to the decline in proton competition and optimum uptake achieved at pH 4 as well as uptake was slightly decreased at pH higher than 4, due to the studied cations starting to precipitate, and all experimental work was done at pH 4. Distribution coefficients ($K_d$) and separation factors ($\alpha_{ab}$) for the mentioned cations in different pHs (1–5) are computed and presented in Table 3. The results reveal that $K_d$ values have the sequence: $\text{Pb}^{2+} > \text{Cd}^{2+} > \text{Co}^{2+} > \text{Zn}^{2+} > \text{Sr}^{2+}$ reflecting that the sorption of metal ions was achieved in hydrated ionic radii except $\text{Sr}^{2+}$ applied in unhydrated ionic radii ($\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ have ionic radii 1.2, 0.97, 0.63, 0.74, and 1.13 Å, respectively) according to this sequence (Abou-Mesalam et al. 2020). Separation factor ($\alpha_{ab}$) values were computed and revealed that $\text{Pb}^{2+}$ has very higher $\alpha_{ab}$ by 284.8, 78.1, 33.3, and 3.5 for $\text{Sr}^{2+}$, $\text{Zn}^{2+}$, $\text{Co}^{2+}$, and $\text{Cd}^{2+}$ at optimum uptake (pH 4); these values indicated that Pb(II) ion can very easily be separated from radioactive and industrial waste solutions, which reflected no selectivity for $\text{Sr}^{2+}$.

Non-linear relations between log $K_d$ and pH were observed for studied cations as exposed in Fig. 6(B). The non-ideality of the exchange reaction was clarified from this relation. The difference may be due to the eminence of a mechanism other than ion exchange, such as precipitation and surface sorption (Abou-Mesalam et al. 2018).

**Agitating time impact**

The influence of shaking time changed from 5 to 240 min on the sorption efficiency of $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ (100 mg/L) onto P(AA-AN)-talc was done at pH 4, and the experimental results are given in Fig. 7(A), as a relation between sorption efficiency percent and time. From this figure, it is clear that, as the mixing time increases from 5 to 120 min, the sorption percent of $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ increased from about 72.5 to 96.9% for $\text{Pb}^{2+}$, from about 55.1 to 91.1% for $\text{Cd}^{2+}$, from about 23.8 to 48.95% for $\text{Co}^{2+}$, from about 12.8 to 29.5% for $\text{Zn}^{2+}$, and from about 5.2 to 11.8% for $\text{Sr}^{2+}$. Further, an increase in the mixing time up to 240 min does not affect the removal of $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$. This means that the sorption equilibrium has been achieved at 2 h. Therefore, 2 h represents the preferred time to maximize the $\text{Pb}^{2+}$, $\text{Cd}^{2+}$, $\text{Co}^{2+}$, $\text{Zn}^{2+}$, and $\text{Sr}^{2+}$ sorption using the P(AA-AN)-talc, and this time was used for all experiments.

**Saturation capacity and thermal stability**

The influence of heating temperature on the capacity of studied cations on P(AA-AN)-talc in H⁺ form is detected at 300 ± 1 K, and the data are presented in Table 4. The prominent reduction in the capacity was observed with rising temperature; it is related to the decay of the organic part of the polymeric material with an increase in temperatures as exposed in TGA and DTA data (Gupta et al. 2015). The capacities of studied cations have sequence order: $\text{Pb}^{2+} > \text{Cd}^{2+} > \text{Co}^{2+} > \text{Zn}^{2+} > \text{Sr}^{2+}$; this order reveals that...
Fig. 6  Sorption behavior of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) sorbed onto P(AA-AN)-talc at 300 ± 1 K. (A) Effect of pH on $K_d$ and (B) plots of $\log K_d$ against pH

| pH | $K_d$ (mL/g) and $a^m_p$ | Sr(II) | Zn(II) | Co(II) | Cd(II) | Pb(II) |
|----|--------------------------|--------|--------|--------|--------|--------|
| 1  | $K_d$                     | 1.3    | 12.7   | 15.6   | 6.7    | 17.9   |
|    | $a^m_p$                   | 9.8    | 12.0   | 1.2    | 0.5    | 1.4    |
|    |                           | 0.4    | 1.1    | 0.4    | 0.5    | 0.9    |
| 2  | $K_d$                     | 2.8    | 20.3   | 60.2   | 35.4   | 31.3   |
|    | $a^m_p$                   | 7.3    | 21.5   | 3.0    | 1.7    | 1.5    |
|    |                           | 1.2    | 1.5    | 1.2    | 0.5    | 0.9    |
| 3  | $K_d$                     | 5.8    | 30.0   | 91.5   | 266.3  | 107.1  |
|    | $a^m_p$                   | 5.2    | 15.8   | 3.1    | 8.9    | 3.6    |
|    |                           | 2.9    | 1.2    | 2.9    | 1.2    | 0.4    |
| 4  | $K_d$                     | 13.9   | 50.7   | 119.0  | 1141.9 | 3958.3 |
|    | $a^m_p$                   | 3.6    | 8.6    | 2.3    | 22.5   | 78.1   |
|    |                           | 9.6    | 33.3   | 9.6    | 33.3   | 3.5    |
| 5  | $K_d$                     | 13.4   | 49.7   | 114.8  | 1089.3 | 1875.0 |
|    | $a^m_p$                   | 3.7    | 8.6    | 2.3    | 21.9   | 16.3   |
|    |                           | 9.5    | 37.7   | 9.5    | 37.7   | 1.7    |
the sorption process was achieved according to the lessening in the hydrated ionic radii and hydration energy (Gupta et al. 2003; Rahman et al. 2017). The high capacity of the material investigated for Pb²⁺ and Cd²⁺ may be due to the higher complexing ability of these ions (Abou-Mesalam et al. 2020).

**Kinetic investigation**

The pseudo-first-order and pseudo-second-order equations are specified as (Gürses et al. 2014):

\[
\log (q_e - q_t) = -\left( \frac{K_f}{2.303} \right) t + \log q_e 
\]  

(6)

\[
\frac{t}{q_t} = \frac{1}{K_f q_e^2} + \frac{t}{q_e} 
\]  

(7)

where \( K_f \) (min⁻¹) and \( K_s \) (g mg⁻¹ min⁻¹) are the rates constant of pseudo-first-order and pseudo-second-order, respectively, and \( q_e \) and \( q_t \) (mg/g) are the amounts sorbed per gram at equilibrium and time \( t \). Plotting \( \log (q_e - q_t) \) vs. \( t \).
and \( t \) as shown in Fig. 7(B), the plot shows a linear relationship. From Table 5, it is found that the pseudo-first-order model does not apply to \( \text{Pb}^{2+}, \text{Cd}^{2+}, \text{Co}^{2+}, \text{Zn}^{2+}, \) and \( \text{Sr}^{2+} \) sorption onto P(AA-AN)-talc. When the \( q_e \) calculated from pseudo-first-order plots was compared with the \( q_e \) (experimental) considered, one of the main discrepancies was observed (Sheha and El-Zahhar 2008). However, pseudo-second-order is obtained from plotting \( \frac{t}{q_t} \) against \( t \) for \( \text{Pb}^{2+}, \text{Cd}^{2+}, \text{Co}^{2+}, \text{Zn}^{2+}, \) and \( \text{Sr}^{2+} \) sorption as shown in Fig. 7(C), and it is found that the relationship is linear. The data represented in Table 5 reveal that the correlation coefficients \( (R^2) \) are close to unity which indicates that the sorption procedure follows the pseudo-second-order model; also, the values \( (q_e, K_f, K_s) \) confirm that the pseudo-second-order model applies to \( \text{Pb}^{2+}, \text{Cd}^{2+}, \text{Co}^{2+}, \text{Zn}^{2+}, \) and \( \text{Sr}^{2+} \) sorption onto P(AA-AN)-talc. Similar phenomena have been observed for \( \text{Pb(II)} \) and \( \text{Cd(II)} \) adsorption on biochars (Park et al. 2013), \( \text{Sr(II)}, \) and \( \text{Cs(I)} \) sorbed by irradiated saccharomyces cerevisiae (Tan et al. 2017), cobalt(II) sorbed by orange peel waste (Altunkaynak et al. 2021), and adsorption of \( \text{Zn(II)} \) on natural bentonite (Sen and Gomez 2011).

### Mechanism of adsorption

The adsorption mechanism was examined using the intra-particle diffusion model (Karaca et al. 2013; Dakroury et al. 2021).

\[
q_t = K_{id} t^{0.5} + C \tag{8}
\]

where \( C \) (intercept) and \( K_{id} \) (slope) is the intra-particle diffusion rate constant (mg min\(^{-0.5}\) g\(^{-1}\)). The intra-particle diffusion model for the sorption process of the studied metal ions was established through three steps (Karaca et al. 2013; Dakroury et al. 2021). The first stage includes the diffusion of metal ions from the solution to the surface of the P(AA-AN)-talc (from 5 to 15 min). The second stage (from 30 to 90 min) describes the gradual sorption on the surface of the P(AA-AN)-talc, which may be the rate-limiting step. The third stage (from 120 to 240 min) is the equilibrium saturation. The relation between \( q_t \) and \( t^{0.5} \) is represented in Fig. 7(D), and the three steps of the sorption mechanism are observed in this figure. The intercept which is the thickness of the surface gave information about the contribution of the surface sorption in the rate-determining step. The larger the intercept, the greater its contribution. Table 6 shows the

### Table 5 Kinetic parameters of various metal ions onto P(AA-AN)-talc at 300 ± 1 K

| Metal ions sorbed | Pseudo-1st-order | Pseudo-2nd-order | \( q_e \), exp. (mg/g) |
|-------------------|------------------|------------------|-----------------------|
|                   | \( K_f \times 10^{-3} \) | \( q_e \), (mg/g) | \( R^2 \) | \( K_s \times 10^{-3} \) | \( q_e \), (mg/g) | \( R^2 \) |
| Pb(II)            | 15.7             | 2.32             | 0.94                  | 9.7                 | 10.309            | 0.999   | 9.69 |
| Cd(II)            | 16.3             | 3.78             | 0.968                 | 10.6                | 9.438             | 0.999   | 9.11 |
| Co(II)            | 25.7             | 4.53             | 0.963                 | 18.8                | 5.313             | 0.997   | 4.9  |
| Zn(II)            | 12.7             | 1.82             | 0.976                 | 31.6                | 3.166             | 0.997   | 2.95 |
| Sr(II)            | 9.4              | 0.69             | 0.973                 | 80.2                | 1.246             | 0.991   | 1.18 |

### Table 6 Diffusion models parameters for sorption of various metal ions onto P(AA-AN)-talc at 300 ± 1 K

| Metal ions sorbed | Time range, min | \( K_{id} \), g mg\(^{-1}\) min\(^{-0.5}\) | \( C \) | \( R^2 \) |
|-------------------|-----------------|---------------------------------|--------|--------|
| Pb(II)            | 5–15            | 0.73723                         | 5.52495| 0.87966|
|                   | 30–90           | 0.10071                         | 8.64095| 0.96964|
|                   | 120–240         | -                               | 9.69    | -      |
| Cd(II)            | 5–15            | 0.89314                         | 3.48647| 0.98967|
|                   | 30–90           | 0.17401                         | 7.31939| 0.99632|
|                   | 120–240         | -                               | 9.11    | -      |
| Co(II)            | 5–15            | 0.39982                         | 1.46773| 0.97553|
|                   | 30–90           | 0.2806                          | 2.32585| 0.84142|
|                   | 120–240         | -                               | 4.895   | -      |
| Zn(II)            | 5–15            | 0.30103                         | 0.58301| 0.92805|
|                   | 30–90           | 0.11454                         | 1.7216  | 0.98774|
|                   | 120–240         | -                               | 0.55917 | -      |
| Sr(II)            | 5–15            | 0.11407                         | 0.25413| 0.89901|
|                   | 30–90           | 0.11407                         | 0.25413| 0.89901|
|                   | 120–240         | -                               | 0.55917 | -      |
parameters obtained from the second part of the linear plot. The sorption mechanism of Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\) onto P(AA-AN)-talc was found to be rapid at the initial period of contact time and then to become constant with the increase in contact time. The multi-diffusion step is the main factor in the control of the sorption mechanism, which includes both film and intra-particle diffusion.

**Sorption isotherms**

The Langmuir equation relates the concentration of a medium above a solid phase surface at a constant temperature to cover the sorbed molecules on the solid phase surface (Hamed et al. 2016a). The Langmuir isotherm model suggests an estimation of the maximum adsorption capacity that occurred by complete monolayer adsorption on the adsorbent surface (Hamed 2014). The Langmuir isotherm can be represented in the next equation (Hamed et al. 2016a):

\[
\frac{C_e}{q_e} = \frac{C_e}{Q_{\text{max}}} + \frac{1}{bQ_{\text{max}}} 
\]

(9)

where \(C_e\) is the equilibrium concentration (mg/L), \(Q_{\text{max}}\) is monolayer capacity (mg/g), and \(b\) is the sorption equilibrium constant related to the energy of sorption. The represented relation between \(C_e/q_e\) and \(C_e\) gives straight lines for both Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\) sorbed onto P(AA-AN)-talc nanocomposite as shown in Fig. 8. Table 7 represents the data obtained from the linear form of the Langmuir equation. The correlation coefficients \((R^2)\) are 0.992, 0.998, 0.976, 0.991, and 0.996 for Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\), respectively. The maximum monolayer capacities \((Q_{\text{max}})\) obtained from the Langmuir model were (43.8, 37.91, 28.6, 23.6, and 9.09) for Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\), respectively.

The separation factor \((R_L)\) considers one of the essential characteristics of the Langmuir isotherm model (Hamed et al. 2016a), which can be calculated from the Langmuir constant, \(b\), as the following:

\[
R_L = \frac{1}{1 + bC_e} 
\]

(10)

From data present in Table 7, \(R_L\) values were found to be \(0 < R_L < 1\) reflecting the favorable sorption isotherms of Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\) (Hamed et al. 2016a).

The Freundlich isotherm model is an empirical formula that accurately represents multi-layer sorption on heterogeneous surfaces (Abass et al. 2021a). A Freundlich equation can be represented in linear form as:

\[
\log q_e = \log K_F + \frac{1}{n} \log C_e 
\]

(11)

where \(K_F\) (mg/g) and \(1/n\) are Freundlich constants related to adsorption capacity and adsorption intensity. These constants are calculated from both slope and intercept of the linear plot of \(\log q_e\) versus \(\log C_e\), respectively. The sorption results of Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\) onto

![Figure 8](https://example.com/fig8.png)

**Fig. 8** Sorption behavior of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) sorbed onto P(AA-AN)-talc at 300 ± 1 K. (A) Langmuir isotherm and (B) Freundlich isotherm
P(AA-AN)-talc nanocomposite performed by Freundlich isotherms are shown in Fig. 8. Freundlich isotherm parameters ($K_F$, $1/n$, and $R^2$) are represented in Table 7. The values of $R^2 = 0.959$, $0.915$, $0.97$, $0.974$, and $0.993$ for $Pb^{2+}$, $Cd^{2+}$, $Co^{2+}$, $Zn^{2+}$, and $Sr^{2+}$, respectively, were much lower than the Langmuir isotherm values, reflecting the applicability of Langmuir than Freundlich isotherm.

Comparison of monolayer capacity with different sorbents reported in the literature

The monolayer capacities ($Q_{max}$) of P(AA-AN)-talc for the sorption of $Pb^{2+}$, $Cd^{2+}$, $Co^{2+}$, $Zn^{2+}$, and $Sr^{2+}$ was compared with other sorbents reported in the literature. As represented in Table 8, the monolayer capacities of P(AA-AN)-talc toward the studied cations are higher than the previously reported values which suggested that P(AA-AN)-talc is a promising sorbent to remove $Pb^{2+}$, $Cd^{2+}$, $Co^{2+}$, $Zn^{2+}$, and $Sr^{2+}$ from aqueous solutions (Annadurai et al. 2003; Anitha et al. 2015; Ghasemi et al. 2018; Abdel-Galil et al. 2016). This enhancement in adsorption with increasing the temperature is corresponding to the acceleration of some originally slow adsorption phases and the creation of some fresh exchangeable sites on the adsorbent layers (Abdel-Galil et al. 2016). From both slopes and intercepts of linear relation displayed in Fig. 9, $\Delta H^\circ$ and $\Delta S^\circ$ are computed and tabulated in Table 9. The positive values of both $\Delta H^\circ$ and $\Delta S^\circ$ indicate the endothermic nature and increased

\[
\ln K_d = \frac{\Delta S^\circ}{R} - \frac{\Delta H^\circ}{RT}
\] (12)

where ($\Delta S^\circ$, $\Delta H^\circ$, $R$, and $T$) are the entropy change, enthalpy change, the universal gas constant, and absolute temperature. The $K_d$ of the investigated cations improved with increasing temperature from 300 to 333 K (i.e., the $K_d$ reduced with rising $1000/T$) and similar phenomena for adsorption of Remazol Red RB by modified clay (Karaca et al. 2013). The positive values of both $\Delta H^\circ$ and $\Delta S^\circ$ indicate the endothermic nature and increased

| Metal ions sorbed | Langmuir constants | Freundlich constants |
|-------------------|--------------------|----------------------|
|                   | $Q_{max}$ (mg/g)   | $b$ (L/mg) | $R_L$ | $R^2$ | $1/n$ | $K_F$ (mg/g) | $R^2$ |
| Pb(II)            | 43.80              | 0.075     | 0.04  | 0.992 | 0.40  | 6.14      | 0.957  |
| Cd(II)            | 37.91              | 0.040     | 0.06  | 0.998 | 0.46  | 3.33      | 0.915  |
| Co(II)            | 28.60              | 0.004     | 0.39  | 0.976 | 0.70  | 0.31      | 0.970  |
| Zn(II)            | 23.60              | 0.002     | 0.57  | 0.991 | 0.79  | 0.10      | 0.975  |
| Sr(II)            | 9.09               | 0.002     | 0.62  | 0.997 | 0.77  | 0.04      | 0.993  |

Table 7 Isotherm parameters for sorption of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) sorbed onto P(AA-AN)-talc at 300 ± 1 K

Table 8 Comparison of the monolayer capacity of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) sorbed onto various sorbents
randomness of solid solution interface during the adsorption of these cations on P(AA-AN)-talc, respectively (El-Naggar et al. 2010; Abdel-Galil et al. 2016).

The free energy change of adsorption $\Delta G^\circ$ was computed by the relation:

$$\Delta G^\circ = \Delta H^\circ - T\Delta S^\circ$$

(13)

The negative values of $\Delta G^\circ$ represented in Table 9 reflect that the sorption process is spontaneous and indicates the better sorption of these ions on P(AA-AN)-talc compared with H$^+$ ion (El-Naggar et al. 2010).

### Column investigations

#### Binary system

Figure 10(A) shows breakthrough curves for Pb$^{2+}$ and Sr$^{2+}$ (100 mg/L) onto the P(AA-AN)-talc column. The concentrations of respective ions in the effluent to the feed solution ($C/C_o$) vs. effluent volume, $V$ (L) are plotted on a breakthrough curve. The uptake of Pb$^{2+}$ and Sr$^{2+}$ per gram is computed from Fig. 10(A) using Eq. (5) and found to be 46.3 and 25.0 mg/g for Pb$^{2+}$ and Sr$^{2+}$, respectively. From these consequences, it is found that selectivity followed order: Pb(II) > Sr(II). This order proved that the sorption process was taking place for Pb$^{2+}$ and Sr$^{2+}$ in hydrated ionic radii, similar to the result in the batch method (Abou-Mesalam et al. 2018). And a BTC is lower than the capacity determined from a batch method, due to the fast dynamic motion of investigated cations. From the data above-mentioned, the conclusions can be expected that P(AA-AN)-talc is suitable for the capture of Pb$^{2+}$ from Sr$^{2+}$ at pH=4.

#### Multi-component system

The breakthrough curves for Pb$^{2+}$, Cd$^{2+}$, Co$^{2+}$, Zn$^{2+}$, and Sr$^{2+}$ (100 mg/L for each) onto the P(AA-AN)-talc column are displayed in Fig. 10(B). A breakthrough capacity is computed as mentioned earlier in the binary system from Fig. 10(B) using Eq. (5) and found to be 37.7, 28.9, 13.2, 9.6, and 7.4 mg/g for Pb$^{2+}$, Cd$^{2+}$, Co$^{2+}$, Zn$^{2+}$, and Sr$^{2+}$, respectively. From these data, the selectivity followed order: Pb$^{2+}$ > Cd$^{2+}$ > Co$^{2+}$ > Zn$^{2+}$ > Sr$^{2+}$. This order proved that the sorption was achieved in hydrated ionic radii; hence, the breakthrough capacity was lower than that obtained by the batch technique, due to the fast dynamic motion of studied cations. From the data above-mentioned, the conclusion can be supposed that P(AA-AN)-talc is a more suitable sorbent for the uptake of all studied cations from liquid solutions in the pH=4.

#### Elution

##### Elution of binary system

The elution results for Pb$^{2+}$ and Sr$^{2+}$ are illustrated in Fig. 11(A). The elution of the investigated ions is studied using DDW and dissimilar HNO$_3$ concentrations (0.01, 0.1, and 0.5 M). Sr(II) ions were released using DDW and 0.01 M HNO$_3$ as eluents. However, the separation of Pb(II) ions was released using 0.01 and 0.1 M HNO$_3$ as eluents. By 0.5 M HNO$_3$, the column packed with P(AA-AN)-talc becomes free from any sorbed metal ions and can be reused again for chromatographic separation.

---

**Table 9** Thermodynamic parameters for adsorption of Pb(II), Cd(II), Co(II), Zn(II), and Sr(II) on P(AA-AN)-talc

| Metals | Temp., K | $\Delta H^\circ$ (kJ/mol) | $\Delta S^\circ$ (J/mol K) | $\Delta G^\circ$ (kJ/mol) |
|--------|---------|--------------------------|----------------------------|--------------------------|
| Pb(II) | 300     | 23.6                     | 145.7                      | −20.1                    |
|        | 313     |                          |                            | −21.9                    |
|        | 333     |                          |                            | −24.9                    |
| Cd(II) | 300     | 4.9                      | 75.7                       | −17.8                    |
|        | 313     |                          |                            | −18.8                    |
|        | 333     |                          |                            | −20.3                    |
| Co(II) | 300     | 3.5                      | 51.1                       | −11.9                    |
|        | 313     |                          |                            | −12.5                    |
|        | 333     |                          |                            | −13.6                    |
| Zn(II) | 300     | 10.9                     | 69.3                       | −9.9                     |
|        | 313     |                          |                            | −10.8                    |
|        | 333     |                          |                            | −12.2                    |
| Sr(II) | 300     | 24.8                     | 104.8                      | −6.6                     |
|        | 313     |                          |                            | −8.0                     |
|        | 333     |                          |                            | −10.1                    |
Elution of multi-component system

The elution data for Pb\(^{2+}\), Cd\(^{2+}\), Co\(^{2+}\), Zn\(^{2+}\), and Sr\(^{2+}\) are displayed in Fig. 11(B). The elution of the studied cations is investigated using DDW and different HNO\(_3\) concentrations (0.01, 0.05, 0.1, 0.2, and 0.5 M). Figure 11(B) shows that Sr(II) ions were released using DDW as eluent, Zn(II) ions were released using DDW and 0.01 M HNO\(_3\) as eluents, and Co(II)
ions were released using 0.01 and 0.05 M HNO₃ as eluents. However, Cd(II) and Pb(II) ions were released using 0.05, 0.1, and 0.2 M HNO₃ as eluents by 0.5 M HNO₃, and the column packed with P(AA-AN)-talc becomes free from any sorbed metal ions and can be recycled again for column separation.

**Conclusion**

In this work, P(AA-AN)-talc was fabricated, characterized, and employed for batch sorption of Pb²⁺, Cd²⁺, Co²⁺, Zn²⁺, and Sr²⁺ from multi-component aqueous solutions. P(AA-AN)-talc was synthesized by gamma irradiation-induced preparation at 50 kGy as a new nanocomposite. The distribution coefficients at different pH have selectivity order: Pb²⁺ > Cd²⁺ > Co²⁺ > Zn²⁺ > Sr²⁺. The saturation capacity of the studied cations decreased by increasing the heating temperature. The sorption reaction is fast, and the reaction equilibrium is attained after 2 h. Pseudo-second-order kinetic is applicable with the sorption reaction mechanism. The sorption isotherm belongs to the Langmuir model, and the monolayer capacities of Pb²⁺, Cd²⁺, Co²⁺, Zn²⁺, and Sr²⁺ are 43.8, 37.9, 28.6, 23.6, and 9.09 mg/g, respectively. Thermodynamic parameters displayed that the ion exchange was endothermic and spontaneous. Column investigation reveals that P(AA-AN)-talc is more suitable for binary and multi-system. Finally, P(AA-AN)-talc can be worked as an efficient sorbent used for the separation of Pb²⁺ from Sr²⁺ (binary system) using 0.01 and 0.1 M HNO₃. Also, P(AA-AN)-talc can be used as a promising sorbent for the separation of all studied cations from liquid waste utilizing DDW and different concentrations of HNO₃ as eluents.

**Acknowledgements** This work has been supported by the Egyptian Atomic Energy Authority. Great thanks to all members of the Cyclotron project for helping us with the preparation of the material used.

**Author contribution** M.R. Abass: data writing, writing-original draft review, and editing. W.M. El-Kenany: experimental work and reviewing. E.H. El-Masry: experimental work and editing.

**Funding** Open access funding provided by The Science, Technology & Innovation Funding Authority (STDF) in cooperation with The Egyptian Knowledge Bank (EKB).

**Data availability** Yes.

**Declarations**

**Ethics approval** Yes.

**Consent to participate** Yes.

**Consent to publish** Yes.

**Competing interests** The authors declare no competing interests.

**Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article’s Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article’s Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by/4.0/.

**References**

Abass MR, El-Masry EH, Ibrahim AB (2021a) Preparation, characterization, and applications of polyacrylonitrile/ball clay nanocomposite synthesized by gamma radiation. Environ Geochem Health 43:3169–3188. https://doi.org/10.1007/s10653-021-00813-5

Abass MR, Ibrahim AB, Abou-Mesalam MM (2021b) Retention and selectivity behavior of some lanthanides using bentonite dolomite as a natural material. Chemical Papers 75:3751–3759. https://doi.org/10.1007/s11696-021-01621-y

Abass MR, Diab HM, Abou-Mesalam MM (2022a) New improved thermoluminescence magnesium silicate material for clinical dosimetry. Silicon 14:2555–2563. https://doi.org/10.1007/s12633-021-01049-9

Abass MR, Elmasry E, El-Kenany WM (2022b) Gamma irradiation-induced preparation of polyacrylonitrile acrylamide nano-silica for removal of some hazardous metals. J Inorg Organomet Polym Mater 32:536–546. https://doi.org/10.1007/s10904-021-02156-1

Abass MR, Maree RM, Sami NM (2022c) Adsorptive features of cesium and strontium ions on zirconium tin(IV) phosphate nanocomposite from aqueous solutions. Int J Environ Anal Chem 1-20. https://doi.org/10.1080/03067319.2021.2016728

Abdel-Galil EA, Ibrahim AB, Abou-Mesalam MM (2016) Sorption behavior of some lanthanides on polyacrylamide stannic molybdophosphate as organic-inorganic composite. Int J Ind Chem 7:231–240

Abou-Lilah RA, Rizk HE, Elshorbagy MA et al (2020) Efficiency of bentonite in removing cesium, strontium, cobalt and uranium ions from aqueous solution: encapsulation with alginate for column application. Int J Environ Anal Chem 1-24

Abou-Mesalam MM, Abass MR, Abdel-Wahab MA et al (2018) Polymeric composite materials based on silicate: II. sorption and distribution studies of some hazardous metals on irradiated doped polyacrylamide acrylic acid. Desalin Water Treat 109:176–187. https://doi.org/10.5004/dwt.2018.22084

Abou-Mesalam MM, Abass MR, Ibrahim AB, Zakaria ES (2020) Polymeric composite materials based on silicate. III-Capacity and sorption behavior of some hazardous metals on irradiated doped polyacrylamide acrylonitrile. Desalin Water Treat 193:402–413. https://doi.org/10.5004/dwt.2020.25816

Abou-Mesalam MM, Abass MR, Zakaria ES, Hassan AM (2022) Metal doping silicates as inorganic ion exchange materials for environmental remediation. Silicon 1–9. https://doi.org/10.1007/s12633-021-01568-5

Aglan RF, Hamed MM, Saleh HM (2019) Selective and sensitive determination of Cd(II) ions in various samples using a novel modified carbon paste electrode. J Anal Sci Technol 10:1–11
of cobalt ions from industrial wastewater. J Environ Chem Eng 4:2268–2277

Karaca S, Gürses A, Açılış Ö et al (2013) Modeling of adsorption isotherms and kinetics of Remazol Red RB adsorption from aqueous solution by modified clay. Desalin Water Treat 51:2726–2739

Kayranli B (2021) Mechanism of interaction and removal of zinc with lignocellulosic adsorbents, closing the cycle with a soil condition. J King Saud Univ Sci 33:101607

Khataee A, Alidokht L, Hassani A, Karaca S (2013) Response surface analysis of removal of a textile dye by a Turkish coal powder. Adv Environ Res 2:291–308

Kulkarni SJ (2016) Research and studies on cobalt removal from wastewater. Int J Res Rev 2237:41–44

Lee S, Kim J, Ku B-C et al (2012a) Structural evolution of polyacrylonitrile fibers in stabilization and carbonization. Adv Chem Eng Sci 02:275–282. https://doi.org/10.4236/aces.2012.22032

Lee YK, Jeong JM, Hoigebazar L et al (2012b) Nanoparticles modified by encapsulation of ligands with a long alkyl chain to affect multispecific and multimodal imaging. J Nucl Med 53:1462–1470

Mehta P, Vedachalam S, Sathyaraj G et al (2020) Fast sensing ammonia at room temperature with proline ionic liquid incorporated cellulose acetate membranes. J Mol Liq 305:112820

Metwally SS, Hassan HS, Samy NM (2019) Impact of environmental conditions on the sorption behavior of 60Co and 152+154Eu radio nuclides onto polyaniline/zirconium aluminate composite. J Mol Liq 287:110941

Mirzayev MN, Abdurakhimov BA, Demir E et al (2021) Investigation of the formation of defects under fast neutrons and gamma irradiation in 3C–SiC nano powder. Physica B 611:412842. https://doi.org/10.1016/j.physb.2021.412842

Nabi SA, Shahadat M, Bushra R et al (2011) Synthesis and characterization of nano-composite ion-exchanger; its adsorption behavior. Colloids Surf, B 87:122–128

Nilche A, Atashi H, Javid AH, Saberi R (2007) Preparations of PAN-based adsorbers for separation of cesium and cobalt from radioactive wastes. Appl Radiat Isot 65:482–487. https://doi.org/10.1016/j.apradiso.2006.12.003

Park JH, Choppala G, Lee SJ et al (2013) Comparative sorption of Pb and Cd by biochars and its implication for metal immobilization in soils. Water Air Soil Pollut 224:1–12

Pyrzynska K (2019) Removal of cadmium from wastewaters with low-cost adsorbents. J Environ Chem Eng 7:102795

Rahman N, Haseen U, Rashid M (2017) Synthesis and characterization of polyacrylamide zirconium(IV) iodate ion-exchanger: Its application for selective removal of lead(II) from wastewater. Arab J Chem 10:S1765–S1773. https://doi.org/10.1016/j.arabjc.2013.06.029

Sen TK, Gomez D (2011) Adsorption of zinc (Zn2+) from aqueous solution on natural bentonite. Desalination 267:286–294

Sheha RR, El-Zahhar AA (2008) Synthesis of some ferromagnetic composite resins and their metal removal characteristics in aqueous solutions. J Hazard Mater 150:795–803

Tan Y, Feng J, Qiu L et al (2017) The adsorption of Sr(II) and Cs(I) ions by irradiated Saccharomyces cerevisiae. J Radioanal Nucl Chem 314:2271–2280

Thi Huong N, Son NN, Phuong VH et al (2020) Synthesis Fe3O4/Talc nanocomposite by coprecipitation-ultrasonication method and advances in hexavalent chromium removal from aqueous solution. Adsorpt Sci Technol 38:483–501

Wang FY, Wang H, Ma JW (2010) Adsorption of cadmium(II) ions from aqueous solution by a new low-cost adsorbent Bamboo charcoal. J Hazard Mater 177:300–306

Wenlei L, Shanlin Z, Shuang C et al (2014) Adsorptive characteristics of modified talcum powder in removing methylene blue from wastewater. Chem Speciat Bioavailab 26:167–175

Zanazzi PF, Montagnoli M, Nazzareni S, Comodi P (2007) Structural effects of pressure on monoclinic chloride: A single-crystal study. Am Miner 92:655–661

Zhang X, Hao Y, Wang X, Chen Z (2017) Rapid removal of zinc(II) from aqueous solutions using a mesoporous activated carbon prepared from agricultural waste. Materials 10:1002

Publisher’s note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.