Lead removal from water using organic acrylic amine fiber (AAF) and inorganic-organic P-AAF, fixed bed filtration and surface-induced precipitation

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Granular porous sorbents were normally used for heavy metals removal from water. To search for the new commercial sorbent and treatment strategy, an organic acrylic amine fiber (AAF) and phosphorus loading inorganic-organic AAF (P-AAF) were prepared and used for lead (Pb) removal from water. A new strategy of inorganic-organic coupling technology was proposed for Pb removal, based on the hypothesis of surface-induced precipitation mechanism. The AAF showed a Pb adsorption capacity of 417 mg/g from the Langmuir fitting, while the column filtration technology was further applied to measure the adsorption edge and applications. Effects of different initial Pb concentrations, hydraulic retention time, and co-existing P were considered in the filtration experiments. The presence of 0.8 mg/L Pb in water significantly improved the Pb breakthrough point from 15,000 to 41,000 bed volumes of water spiked with 85 μg/L Pb, while the P-AAF fixed bed showed better removal of Pb than AAF SEM/EDX and XRD spectra were employed for determining the surface functional groups and the formation of surface-induced precipitation of pyromorphite (Pb₃(PO₄)₂OH) on AAF. This study verified the application of AAF sorbent for Pb removal and the enhanced effect of coating P on AAF, thus improved our fundamental understanding and application of the surface chemistry process of Pb with P.

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Introduction

Heavy metals, like lead (Pb), mercury (Hg) et al., in the environment is harmful to humans and ecosystems, due to its high toxicity and insusceptibility to degradation.
exposure to Pb beyond guideline levels may cause health problems like abortion and intellectual disability (Lanphere et al., 2005; Edwards et al., 2009; Lanphere et al., 2018; Li et al., 2019). Researchers found that high levels of Pb in drinking water was partly caused by corrosion of old Pb-containing plumbing systems, i.e. Pb pipeline (Xie and Giammer, 2011; Zhao et al., 2018; Clark et al., 2015). For Pb contamination control, the drinking water standards for Pb is 10 μg/L regarding the World Health Organization (WHO, 2011).

Recent years, adsorption technology has received increasing attention due to its simplicity, high efficiency and low cost for removal of heavy metal ions from aquatic solutions (Liu et al., 2006; Hua et al., 2019; Zhang et al., 2019). In a typical adsorption process, sorbents with activated adsorption sites play a main role to the capacity (Li et al., 2020; Wu et al., 2019). Activated carbon is typically used for Pb removal in point-of-use devices, and the main functional groups on activated carbon can be carboxyl, phenol and carboxylic groups (Figueiredo et al., 1999). The functional groups of organic materials also have different affinities for metal ions (following the order: —O− (enolate) > —NH2 (amine) > —N=O (azo compounds) > —N/ (ring N) > COO− (carboxylate) > —O− (ether) > —C=O (carboxyl) (Charberek and Martell, 1959). Adsorbents containing nitrogen groups such as iminodiacetic group, thiol group and amine group have been highlighted for their potential applications and performance (Vilenisky et al., 2002; Yan ad Bai, 2002; Holt et al., 1991). Moreover, sorbents with amine groups have been focused, because the groups could form chelation coordination complexes with heavy metal ions (Yoshitake et al., 2002; Atia et al., 2003; Jeon and Holl, 2003; Liu et al., 2008). In natural water systems, anions such as phosphate, sulfate, and carbonate are common anions in the water, which may co-exist with Pb and greatly affect the soluble Pb concentration. What’s more, the direct use of Pb-loaded adsorbents also attracted attention (Bacelo et al., 2020).

Pb precipitations usually exist in nature system, like the Pb-salt small particles or adsorbed Pb in natural water as result of low solubility of Pb complexes. In comparison, the solubility products have been measured at10−8.44, 10−7.9, 10−2.2, and 10−12.5 for the Pb compounds chloropyromorphite (Pb5(PO4)3Cl), hydroxylpyromorphite (Pb5(PO4)3OH), anglesite (PbSO4), and cerussite (PbCO3), respectively (Wagman et al., 1982; Lindsay, 1979, Ridge, 1968). This indicates that Pb pyromorphite is less soluble than the other Pb complexes commonly formed in surface water systems (Ruby et al., 1994). Meanwhile, Pb-phosphates (Pb-Ps) are considered as the most stable species and they could form rapidly in case that certain amount of Pb exists (Ridge, 1968). As a result, the presence of phosphate may affect the Pb adsorption process. Meanwhile, phosphate is also used in water and soil treatment for immobilization of Pb (Zhao et al., 2018; Ng et al., 2012).

In this study, the organic sorbent acrylic amine fiber (AAF) and inorganic-organic phosphorus (P) treated AAF (P-AAF) sorbent were prepared and used for the removal of Pb from water. Both batch and column filtration tests were conducted using spiked deionized (DI) water and tap water. Column filtration experiments were conducted at different conditions, especially in the presence or absence of Pb. X-ray diffraction (XRD), and scanning electron microscopy with energy-dispersive X-ray spectrometry (SEM/EDX) were employed for mechanism analysis of Pb effect on Pb removal.

1. Experimental methods

1.1 Materials

The AAF was prepared through the similar procedure described in the previous report (Wei et al., 2020). Briefly, the raw acrylic fiber (AF) was functionalized with tetraethylene pentamine (TEPA) in one step. During the acid-base titration analysis (Kunin and Myers, 1949), the ion exchange capacity of the prepared AAF was measured at 8.20 mmol/g. The as-described AAF adsorbent was used in the following experiments, except for those of using phosphorus treated AAF (P-AAF) for Pb filtration removal. The P-AAF was prepared through the following procedure, AAF was immersed with phosphate solution at pH 3.0–4.0, and rinsed with DI water, dried at 60 °C. Stock solutions of Pb and P were obtained through dissolving Pb(NO3)2 and Na2HPO4 in DI water (18.2 MΩ, Milli-Q), respectively.

1.2 Batch experiments

The kinetics, isotherms, and pH edge tests were conducted at the room temperature of 20 °C. In the adsorption experiment, certain dosages of AAF were added in 50 mL Pb solution at the desired final pH values. The analysis samples were mixed on a circular plate at 50 r/min for a certain time. After separation of the fiber from the solution with settling, Pb concentration was measured by graphite furnace atomic absorption spectrometry (GFAAS, Agilent Z240, USA), and the Pb concentration was determined through inductively coupled plasma optical emission spectrometry (ICP, 5100 ICP-OES, Agilent, USA) analysis.

1.3 Column filtration experiments

In the filtration experiment, the columns (diameter 1 cm) were packed with AAF or P-AAF (BV= 1 mL, 0.4g dry AAF/mL) for treating Pb solutions. The wet bed containing 0.4 g-dry AAF/mL, and the actual density of AAF was measured using the balance cylinder method, which give a density of 4.45 g/mL. The porosity of the working wet AAF bed was measured as Eq. (1).

\[
P = \frac{V_0 - V}{V_0} \times 100\%
\]

Where, P is the porosity constant, \(V_0\) is the total volume of the wet AAF bed (BV=1.0 mL), V is the actual volume of AAF. As a result, 0.4 g dry AAF has the actual V of 0.09 mL, and the porosity of AAF bed is calculated at 91%.

The working solutions are DI water containing 85 μg/L Pb and 0.01 mol/L NaCl, or the actual tap water spiked with 85 μg/L of Pb. The columns worked in a downward flow mode with a certain flow rate as described in the figure notes. The tap water using in the tests containing 87-103 mg/L chloride, 43-53 mg/L sodium, 16-25 mg/L calcium, and 0-5 mg-S/L sulfate. Samples of effluent were collected and acidified using
HNO₃ and kept in 4 °C for analysis. To explore the dynamic Pb adsorption on AAF fixed bed under different modes, filtration tests at different Pb content (1, 10, and 50 mg/L), various flow rates (0.33, 0.80, and 2.0 mL/min) and hydraulic retention time, with the presence or absence of P (0.8 mg/L solution or pretreated with P) were employed. More details of the experiment (i.e, empty bed contact time (EBCT), feed pH, et al.) were marked in the figure caption or the diagram. The adsorption capacities of Pb on AAF or P-AAF (q, mg/g) were calculated based on Eq. (2):

\[ q = \frac{C_0 Q t}{m} \left(1 - \frac{C_t}{C_0}\right) \]  

where Q (L/min) is the influent flow rate, C₀ (mg/L) is the influent Pb concentration, C₁ (mg/L) is the effluent Pb concentration at time t (min), and m (g) is the mass of AAF or P-AAF. Another estimation method is considering the effluent Pb concentration as 0 mg/L before the breakthrough point, thus q equals to C₀Qt/m.

1.4. SEM/EDX and XRD analysis

The spent AAF or P-AAF were rinsed with DI water and dried for further SEM/EDX (LEO 982, LEO Electron Microscopy Inc., Thornwood, NY) and XRD (Rigaku, Ultima IV X-Ray Diffractometer, EquipNet, Inc., USA) analysis.

2. Results and discussion

2.1. Preparation of AAF and adsorption properties on Pb

The morphology of AAF used in the experiment is shown in Fig. 1a, and the diameter of AAF is around 25 μm. To estimate the nitrogen (N) on AAF, SEM-EDX N mapping (Fig. 1b) technology was applied and it reflects the uniform distribution of N. Meanwhile, the detected N was related to the introduced amino group on AAF. The inorganic-organic P-AAF can be considered as the P adsorbed on AAF, through the procedure of P solution treated AAF. The procedure can be expressed as Reaction (3).

\[ \text{R-NH}_2 + \text{H}_2\text{PO}_4^- \rightarrow \text{R-NH}_2\cdot\text{PO}_4^- \]  

Fig. 2a shows the kinetics of Pb removal from three work solutions containing Pb concentrations of 10, 50 and 207 mg/L, respectively. The initial pH was adjusted to 5.3 ± 0.2, and no pH adjustment was made during the experiments. Lead removal reached equilibrium after 60 min of reaction for Pb initial concentrations of 10 and 50 mg/L. It took more than 120 min for the adsorption to achieve an equilibrium when the Pb concentration was 207 mg/L. The solution pH increased slightly to 5.6 during the adsorption, because the weak alkaline amine groups on adsorbent (Deng and Ting, 2005). The results of adsorption kinetics experiments showed the mixing time of 2 and 4 hr were adequate in batch tests to achieve equilibrium for the low and high Pb concentrations, respectively.

Different kinetic models were employed for fitting the Pb adsorption data. As shown in Fig. 2a and Appendix A Table S1 the pseudo-second-order model (Eq. (4)) well describes the adsorption process, with a fitting coefficient of R² > 0.998. The fitting results obtained with the pseudo-second-order kinetic model also shows the rate-controlling step is more related to chemical interaction (Ho and Mckay, 1998).

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

where qₑ (mg/g) and qₜ (mg/g) are the amount of Pb adsorbed at time t (min) and equilibrium stage, respectively; k (g/(mg min)) is the respective rate constant.

Langmuir isotherm model was applied to fit the isotherm data and assessing the adsorption performance (Irving, 1918; Hall et al., 1996). Fig. 2b clearly shows that the amount of adsorption increased with the increase of equilibrium Pb concentration. The Langmuir model fitted the experimental data well for Pb from R² coefficient, it also gave a predicted adsorption capacity close to the experimental result, which confirmed that the adsorption of Pb on AAF occurred by monolayer formation. The maximum capacity of AAF was 417 mg/g from the Langmuir fitting calculation. Therefore, adsorption process of Pb on AAF was represented as Reaction (5).

\[ \text{R-NH}_2 + \text{Pb}^{2+} \rightarrow \text{R-NH}_2\cdot\text{Pb} \]  

Table 1 lists different Pb sorbents and the corresponding capacities on Pb at the labeled conditions. The adsorption capacity of AAF is among the highest, it is also advantageous for its one-step green synthetic method. Besides, raw material AF is a commercial product on the market.

The effect of solution pH was analyzed and the results are shown in Fig. 2c. The experimental solution pH was controlled at 2.5-5.6 to avoid the formation of precipitates at higher pH values. The increasing tendency in Fig. 2c indicated that the adsorption of Pb was strongly affected by the pH value, with higher adsorption at higher solution pH value. At solution pH below 3.2, almost no adsorption was observed, because the protonated amino groups could prevent the association of Pb with the functional groups via strong electrostatic repulsion. Protonation of amine groups and Pb chelation could also compete for the activate sites at low pH under acidic conditions. Meanwhile, we proposed the binding affinity of AAF with Pb is much higher than the repulsion force, so the amine groups could coordinate with the Pb at weak acid solution, and corresponding with adsorption results. Moreover, the adsorption behavior is shown in Fig. 2c also illustrated the better Pb removal at the dosage of 1.0 g/L than 0.2 g/L, and this was related to more active sorption sites at a higher dosage of AAF.
The low adsorption at low pH also indicates that the regeneration of AAF would conduct in acidic solution.

In the natural water system, presence of $\text{PO}_4^{3-}$, $\text{SO}_4^{2-}$ and $\text{CO}_3^{2-}$ in water has significant effect on the chemical species of Pb. The Pb-related species distribution at different solution pH without and with other-co-existing ions was modeled using MINTEQ. (Appendix A Fig. S1, Fig. S2) Special emphasis was placed on Pb-phosphate precipitation products because of their lower solubility in natural water systems than those of sulfate and carbonate. As to the pyromorphite mineral family ($\text{Pb}_5(\text{PO}_4)_3 X$, $X = \text{halide or hydroxide}$), solubility products of $\text{Pb}_5(\text{PO}_4)_3 \text{X}$ consisting of $\text{Cl}$, $\text{Br}$, $\text{OH}$, and $\text{F}$-pyromorphite have been reported at $10^{-84.4}$, $10^{-78.1}$, $10^{-76.8}$, and $10^{-71.6}$, respectively (Nriagu 1974, 1984; Ruby et al., 1994). The stability for $\text{Pb}_5(\text{PO}_4)_3 \text{X}$ was also order as $\text{Pb}_5(\text{PO}_4)_3 \text{Cl} > \text{Pb}_5(\text{PO}_4)_3 \text{Br} > \text{Pb}_5(\text{PO}_4)_3 \text{OH} > \text{Pb}_5(\text{PO}_4)_3 \text{F}$. Therefore, the formation of Pb-Ps might greatly affects the solubility of Pb in solution. Meanwhile, the saturation index was used to evaluate whether a solution system is supersaturated or not.

A simulated $\text{pC} [\text{log} (\text{total Pb})] - \text{pH}$ diagram of Pb precipitation without or with the presence of phosphate was calculated using MINTEQ and shown in Fig. 2d. The $K_{\text{sp}}$ of $\text{Pb(OH)}_2$ (Linus, 1970), $\text{Pb}_5(\text{PO}_4)_3 \text{OH}$ and $\text{Pb}_5(\text{PO}_4)_3 \text{Cl}$ were calculated at $1 \times 10^{-16}$, $10^{-76.8}$ and $10^{-84.4}$, respectively. The simulation yielded a sharp decreasing trend of $\text{Pb(OH)}_2$ and Pb-P precipitation in the pH range of 4-10, and thus translated to a lower Pb solubility. The 99% total Pb precipitation line means that most of the total Pb precipitated in the Pb system without co-existing ions. At pH 7.0, there will be precipitation at Pb concentrations higher than 0.53 mg/L, 6.9 μg/L and 0.028 μg/L, under conditions without co-existing ions, in the presence of 1 mg-P/L, and 1 mg-P/L with 100 mg-Cl/L, respectively (Appendix A Table S2), and corresponding with the results of Ruby (Ruby et al., 1994).

The previous study has reported the adsorption of P on AAF (Wei et al., 2020b), and illustrated that the amine functional groups could adsorb P anions through electrostatic attraction. As a result, the presence of P both in the solution and on the AAF surface (P-AAF) were further studied in the following column filtration tests.

2.2. AAF and P-AAF fixed bed for Pb filtration

Effects of feed concentration and flow rates on Pb filtration removal efficiency using AAF are shown in Fig. 3. At the experimental Pb feed concentrations of 1, 10, and 50 mg/L with a constant flow rate of 0.8 mL/min and a contact time of 1.25 min (Fig. 3a). The breakthrough point of $C_f / C_0$ at 0.1 was recorded for the three columns. Obviously, the column with lower Pb influent concentration gave a higher bed volume at the breakthrough point. When the breakthrough curve is redrawn using the absolute adsorption amount supplied to the bed (Fig. 3b), the comparison of the adsorption amount between the three curves shows that the end-point adsorption amount of 1 mg/L is lower than the columns of 10 and 50 mg/L (Pb concentration). The adsorption capacities of AAF for Pb in the water of 1, 10 and 50 mg/L at the breakthrough point were calculated at 9.3, 14.9 and 26.5 mg/g, respectively. While they are 13.5, 31.8 and 51.6 mg/g, at the saturation point. According to the ratio of adsorption capacity at breakthrough point and saturation point, the utilization efficiency of the three kinds of initial Pb concentration is 68%, 47%, and 51%, and the utilization effi-
Fig. 3 – (a) Column filtration removal of Pb at different initial Pb concentrations, and (b) adsorbed Pb at different initial Pb concentrations; AAF BV 1 mL, pH 5.5. (c) column filtration removal of Pb at different flow rate; and (d) adsorption capacity and bed utilization of AAF beds with different flow rates, AAF BV 1 mL, pH 5.5.

Fig. 4 – (a) Column filtration removal of Pb with or without presence of P using AAF, tap water, BV 1 mL; (b) Comparison of Pb removal using AAF and P-AAF fixed bed, the P-AAF was pretreated with 31 mg/L P solution, 0.1 mol/L NaCl DI, 50 mg/L Pb solution, BV 1 mL.

Fig. 5 – Changes of SEM morphology of AAF surface before and after Pb column filtration. (a AAF before test; b 85 μg/L Pb, 0.8 mg/L P tap water; c 31 mg/L P pretreated AAF, 50 mg/L Pb feed solution)
ciency of low Pb concentration is higher. Besides, the adsorption capacity at high concentrations of 10 mg/L and 50 mg/L is similar to the bed utilization efficiency, indicating that at a certain level, the adsorption capacity of Pb on AAF fixed bed is independent of the initial concentration. Generally, the AAF fixed bed has better Pb removal performance for a dilute solution with relatively low feed concentration.

Fig. 3c shows the breakthrough curve of AAF fixed adsorption bed at different flow rates. At the fixed Pb content of 50 mg/L and three different flow rates of 0.33, 0.8 and 2.0 mL/min, the retention time was 3.33, 1.25 and 0.5 min, respectively. As shown in Fig. 3c, compared with the lower flow rates of 0.33 and 0.8 mL/min, the faster flow rate of 2.0 mL/min obtained a breakthrough point of 30 BV and the saturation point of about 400 BV. As can be seen from Fig. 3d, with the flow rate increasing from 0.33 to 2.0mL/min, the adsorption capacity of AAF bed to Pb at the penetration point and saturation point decreases gradually. Longer residence time is beneficial to sufficient mass transfer and thus lead to a better Pb diffusion in the pore of AAF bed and the surface. Therefore, AAF fixed bed has better Pb removal performance for low flow rate feed.

The influence of solution coexisting phosphate and pre-loading phosphate on the adsorption and Pb removal performance of the AAF column was investigated respectively. Fig. 4a shows that the AAF adsorption column has achieved good Pb removal efficiency for two kinds of influents close to the actual level (10-100 μg/L) of Pb concentration, seepage from pipes (Deshommes et al., 2010; Wang et al., 2012). When the initial Pb concentration is 85 μg/L, the Pb spiked tap water without coexisting ions and containing 0.8 mg-P/L phosphate were used as the influents. The effluent Pb concentration of 10 μg/L (safety standard for drinking water) was chosen as the breakthrough point. Column filtration results showed that the filtered water volume at breakthrough point reached at 15,000 BV in the control group without coexisting P ions, in contrast, that was 41,000 BV under a 0.8mg/L P condition, and the removal capacity of Pb is significantly improved because of P ions introduced. This is corresponding to the promotion of Pb

<table>
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<th>Table 1 – Pb adsorption capacity of various adsorbents.</th>
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<td>Adsorbent</td>
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<tr>
<td>Carbon nanotubes(CNTs)</td>
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<tr>
<td>Activated carbon</td>
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<tr>
<td>Red mud</td>
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<tr>
<td>Chitosan nanoparticles</td>
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<tr>
<td>2D Titanium Carbide</td>
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<tr>
<td>Polymer-supported zirconium P</td>
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<tr>
<td>Hydrazinyl amine</td>
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<tr>
<td>Schiff base chelating resin</td>
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<tr>
<td>Cation exchange resin (sulfonic acid group)</td>
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<td>Cation exchange resin (carboxylic acid group)</td>
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<td>Anion exchange resin (primary/secondary amine group)</td>
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<td>AAF</td>
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in tap water to remove Pb from AAF and the high pH value of tap water.

The results showed a AAF dynamic adsorption capacity of 3.2 mg/g (15,000 BV) in control group without coexisting P ions, 8.7 mg/g (41,000 BV) in the presence of P. Compared with the common active carbon Pb removal materials, which is about 2000 BV and 0.43 mg/g (Shi et al., 2018). AAF has a higher dynamic adsorption capacity, and the presence of phosphate in water can significantly improve the dynamic Pb removal effect of AAF.

Fig. 4b shows the comparison of Pb removal using AAF and P-AAF fixed bed. Under the condition of initial Pb content of 50 mg/L and flow rate of 0.8 mL/min, the original AAF bed can achieve the effect of 600 BV, but after phosphate pretreatment, there is no breakthrough point until 1920 BV, which shows that the adsorption column of P-AAF has better Pb removal performance. The presence of Pb-P surface precipitation on AAF and P-AAF collected form the above two columns were showed in Fig. 5. Compared with the bare AAF (Fig. 5a), small rod crystal of 200 nm were found on the column filtered 85 μg/L Pb with 0.8 mg-P/L tap water (Fig. 5b), while the larger crystals were discovered on P-AAF at a higher Pb concentration feed of 50 mg/L (Fig. 5c).

2.3. SEM-EDX and XRD analysis

SEM/EDX was employed to determining the composition of precipitates on AAF surface, as shown in Fig. 6. Fig. 6a shows the sample with P-Pb at the same condition of Fig. 4a and Fig. 5b. While, Fig. 6b is the image of AAF sample treated with 31 mg/L P solution firstly, and then 207 mg/L Pb bulk solution. The EDX spectrum data clearly show that the precipitate formed on AAF surface is Pb-P precipitate, in which the atomic molar ratio of Pb/P is about 1.6, and there are no other interference ions in the solution system. It is speculated that the precipitate might be Pb phosphate precipitate, and its molecular formula is Pb₅(PO₄)₃OH. The XRD patterns of the AAF and AAF with precipitation also illustrated the formation of pyromorphite on the AAF surface (Fig. 7).

The formation of surface formed pyromorphite on goethite was also demonstrated in the previous report (Zhang et al., 1997; Botto et al., 1997), which corresponds to the formation of pyromorphite on the Pb-adsorbed AAF. When P and Pb were co-existed in water, Pb has been found to promote the removal of Pb. At the same time, a new method trying to distinguish the surface adsorption and surface-induced precipitation accord-
ing to stoichiometry was proposed (Wei et al., 2020a). More explorations, like extended X-ray absorption fine structure (EXAFS) and FTIR analysis of the activated carbon samples prepared in Pb solution and P-Pb binary component solutions showed some differences in the spectra, which was attributed to the formation of a COOPbH$_2$PO$_4$ ternary complex (Shi et al., 2018). Besides, Tiberg also reported that ternary Pb-P complexes were formed on the ferrhydrite surface based on the EXAFS analysis (Tiberg et al., 2013).

A different aspect of our findings is that the positively charged amine group surface could also form surface Pb-phosphate precipitation through a surface-induced effect in the solution containing both Pb and P ions. Meanwhile, both adsorbed Pb and P could transform to surface precipitates when reacted with another ion in the solution (Fig. 8). As a result, another mechanism for Pb enhancement on Pb removal using AAF is surface-induced deposition, i.e. Reactions (6) and (7). Furthermore, research also found the dielectric constant of water has a sharp decrease near the solid surface compared that of free water phase (Fumagalli et al., 2018). Overall, the presence of Pb on surface may worked in the two mechanisms, (a) reduce the surface potential thus reduce the positive surface repulsion force to Pb$^{2+}$; (b) worked as the adsorption site for Pb$^{2+}$.

The AAF pretreated with P showed better Pb removal performance. P on the surface of AAF can form surface precipitation with Pb, then deposited on the surface, thus improving the Pb removal. Due to the extremely low solubility of Pb-P precipitation, the technical path of P pretreatment also provides a reference for the performance optimization of Pb removal materials.

R-NH$_2$ + H$_2$PO$_4^-$ + Pb$^{2+}$ → R-NH$_2$ (PO$_4$-Pb)  \hspace{1cm} (6)

R-NH$_2$PO$_4$ + Pb$^{2+}$ → R-NH$_2$ (PO$_4$-Pb)  \hspace{1cm} (7)

3. Conclusions

To solve the Pb contamination in water system, the organic AAF and inorganic-organic P-AAF with high removal capacity for Pb were prepared. A new concept of using inorganic P coating sorbent for efficient Pb removal was firstly proposed. As to AAF, it can effectively remove Pb from water, and its adsorption kinetics accords with pseudo second-order kinetics model. When the initial Pb concentration is lower than 50 mg/L, the adsorption can reach equilibrium within 60 min. The Langmuir isotherm model fitted well with the Pb adsorption, and its saturated capacity was 417 mg/g. In addition, adsorptive fibers can be used to make woven and nonwoven adsorbents for a broad array of applications. As to column filtration, the dynamic capacity of AAF on Pb was measured at 3.2 mg/g (15,000 BV) in the presence of Pb alone and 8.7 mg/g (41,000 BV) in the presence of P. Most important, the presence of P in water or coating on AAF surface can significantly improve the removal of Pb by AAF. The experimental results using AAF and P-AAF for Pb removal supports a good reference for the treatment of Pb in water, and deepen our understanding of surface chemistry process on Pb removal as well.

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Appendix A Supplementary data

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.jes.2020.08.009.

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