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Enhancement of waste-activated sludge dewaterability using combined Fenton pre-oxidation and flocculation process

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ABSTRACT

A combination of Fenton pre-oxidation and flocculation process was utilized to enhance the dewaterability of waste-activated sludge. The dry solid content of dewatered sludge (DS) cake, supernatant turbidity, sludge specific resistance to filtration (SRF), supernatant of soluble chemical oxygen demand (SCOD), polysaccharides, and proteins concentration were examined to evaluate the sludge dewatering performance and describe the mechanism involved in sludge treatment. The optimal conditions were as follows: 40 mg·g^{-1} DS H_2O_2; 50 mg·g^{-1} DS Fe^{2+}; pH 3; Fenton reaction time, 90 min; and 60 mg·L^{-1} cationic polyacrylamide (CPAM). Under these conditions, the optimal SRF and DS achieved were 1.21×10^9 m·kg^{-1} and 34.20%, respectively. The extracellular polymeric substance structure was destroyed by Fenton oxidation. Consequently, polysaccharide and protein were released into the supernatant, and the SCOD was increased. Morphological property analysis revealed an evident compact sludge floc after the combined Fenton pre-oxidation and flocculation process. The sludge particle treated with the combined processes became larger than that treated with Fenton oxidation alone. The sludge flocs were formed with the addition of CPAM during flocculation due to its excellent charge neutralization and bridging ability. The present results showed that the combination of Fenton pre-oxidation and flocculation is effective in conditioning waste-activated sludge and consequently enhancing sludge dewaterability.

Keywords: Dewaterability; Waste-activated sludge; Fenton; Flocculation; Dry solid content

1. Introduction

With the acceleration of China's urbanization and the improvement of people's living standard, the domestic sewage and industrial wastewater discharge increases constantly, thereby increasing the sludge amount in urban sewage treatment plants [1]. The waste-activated sludge is large in volume and high in water content; therefore, this sludge must be dewatered to facilitate the subsequent disposal [2]. Waste-activated sludge exhibits the characteristics of various kinds of microorganisms, such as high specific surface area and negative electricity [3]. Many previous studies showed that the sludge composition is the main factor affecting the performance of sludge dewatering. The extracellular polymeric substance (EPS) is one of the most important groups in activated sludge, which is an insoluble organic matter secreted by microorganisms in a particular environment [4]. In addition, the polysaccharide and protein content accounts for 50%–90% of the total organic matter in activated sludge,

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which shows a strong water-binding ability [5]. The main component of EPS includes protein and polysaccharide, the concentration of which account for 70%–80% of EPS. These components form a dense gel on the surface of the bacteria, absorb a large amount of water, and reduce the density difference between the sludge flocs and water; consequently, the sludge dewatering performance is reduced and deteriorated [6].

Therefore, the performance of sludge dewatering should be improved to regulate the activated sludge and disintegrate EPS [7]. Currently, the main methods of sludge conditioning include physical, chemical, biological, and combined conditioning [8]. Physical conditioning uses physical technology to break the cells in sludge, thereby changing the surface structure of sludge, reducing the binding ability of sludge and water, and releasing the internal water of sludge [5,9]. Although different physical conditioning methods can reduce the water content of sludge, various physical conditioning methods display their own disadvantages [10,11]. Chemical conditioning also improves sludge dewaterability with the addition of a certain amount of flocculant or other chemical substances to alter the surface properties of activated sludge and damage the structure of sludge colloid so as to release the internally bound water [12,13]. However, considering the strongly bound water of polysaccharide and protein, the traditional flocculants and coagulants cannot completely remove the binding and interstitial water in the sludge flocs; the moisture content of dewatered sludge (DS) can only be reduced to 80%–90% by belt filter press in the actual sewage plant [14]. Accordingly, chemical conditioning can only improve the dehydration rate of sewage; this method cannot significantly increase the degree of sludge dewatering [15]. Hence, many advanced sludge conditioning techniques are used to improve the sludge dewatering performance. Fenton reaction is an advanced oxidation reaction used for organic matter degradation in wastewater treatment [16]. Given the strong oxidation properties, Fenton reaction can reduce the turbidity of decolorization and disintegrate the polysaccharide and protein effectively in activated sludge [17]. Thus, this reaction has been increasingly applied in many fields, including wastewater treatment and sludge pretreatment, to improve the wastewater treatment performance [18]. In such case, the organic polymer addition remains necessary as pretreatment after the Fenton reaction [19]. The combination of Fenton's reagent and cationic polyacrylamide (CPAM) was used in sludge conditioning for the enhancement of sludge dewatering performance, specific resistance to filtration (SRF) and moisture content (MC) at optimum conditions are reduced to minimum values of 1.06×10^{12} m/kg and 58.9% [20]. Red mud is used as an alternative skeleton builder combined with Fenton's reagent for sewage sludge conditioning, and minimum water content of sludge cakes of 59.8% is achieved at the optimal condition [21].

To date, most of the research on sludge conditioning by the combination of Fenton and flocculation processes mainly focuses on either moisture content or SRF. Only few studies investigated soluble chemical oxygen demand (SCOD), polysaccharide, and protein in the supernatant by the combination of Fenton and flocculation processes in conditioning waste sludge. The synergistic mechanisms of these methods are unclear. In this work, Fenton-flocculation process was introduced to enhance the sludge dewatering performance. Fenton reagent was added to degrade polysaccharide and protein and subsequently release the bound water. The effectiveness of Fenton-flocculation process in improving sludge dewaterability was also evaluated by the index of sludge SRF and solid content of sludge cake. The main factors controlling Fenton-flocculation process were also explored. To provide further insights into the dewatering mechanism, the supernatant of SCOD and the concentration of polysaccharide and protein were analyzed. Finally, scanning electron microscopy (SEM) analyses of DS and sludge flocs were used to examine sludge morphology.

2. Experimental reagents and instruments

2.1. Experimental materials

Sludge with 97.5% water content, pH 6.8, and 1.02 kg·L^{-1} mass density was obtained from Jiangxinzhou Wastewater Treatment Plant (Nanjing, Jiangsu), and the detailed characteristics of raw sludge are shown in Table 1. Ferrous sulfate ($FeSO_4 \cdot 7H_2O$), hydrogen peroxide (H_2O_2) (mass fraction of 30%), and other reagents were of analytical purity. All reagents were used without further purification. The flocculant CPAM was synthesized by using acrylamide, methacryloxyethyltrimethyl ammonium chloride, and diallyl dimethyl ammonium chloride and prepared in the laboratory according to our previous study [21].

2.2. Sludge conditioning procedure

The sludge sample of 500 mL was placed in a 500-mL glass beaker, and appropriate amount of $FeSO_4 \cdot 7H_2O$ was added in the beaker at varying pH values. The pH of the raw sludge is adjusted with 0.5 mol/L NaOH and 0.5 mol/L HCl. A predetermined amount of H_2O_2 (30% [w/v]) was added in the sludge sample under vigorous stirring using a magnetic stirrer. Finally, the reaction was stopped after 2 h of stirring, and the pH value of the sludge solution was adjusted to 7. After the Fenton process, the known dosage of flocculant was added into the sludge for flocculation process. In this study, Fenton process alone and the combined Fenton and flocculation processes were conducted separately.

Table 1
The characteristic properties of the sludge

Indexes	Parameters
Moisture content	97.5%
pH	6.8
Supernatant turbidity	60.2 NTU
Transmittance	70.1%
Density	1.02 kg/L
Apparent condition	Dark brown, granular, and pungent
Volatile suspended solids (VSS)	9.1 g/L
Total suspended solids (TSS)	12.9 g/L
SCOD	1.72 mg/g DS
Zeta potential	–23.5 mV
SRF	2.79×10^{12} m/kg

After the reaction, the treated sludge was poured into the Buchner funnel to determine the SRF [22]. The turbidity of the supernatant was determined by using turbidity analyzer (HACH2100Q, HACH, USA). In addition, the zeta potential of the sludge supernatant was measured by zeta potential analyzer (Zetasizer Nano ZS90, Malvern, Britain). The SCOD after sludge dewatering was determined by a chemical oxygen demand measuring instrument (DR1010, HACH, USA). The polysaccharide and protein contents in supernatant were ascertained by the anthrone–sulfuric acid colorimetric method and Coomassie brilliant blue staining assay, respectively [23]. The dry solid content of DS was calculated by gravimetric method [24].

3. Results and discussion

3.1. Effects of Fe^{2+} dosage on sludge dewatering using Fenton process alone

The effects of Fe^{2+} dosage on DS, supernatant turbidity, and SRF are shown in Fig. 1. With the increased Fe^{2+} dosage, the DS first increased and subsequently decreased (Fig. 1(a)); its maximum value was 26.67% at 40 mg·g^{-1} DS H_2O_2 and 50 mg·g^{-1} DS Fe^{2+}. The effects of Fe^{2+}dosage on supernatant turbidity and SRF are illustrated in Figs. 1(b) and (c), respectively. With the increased Fe^{2+} dosage, the turbidity of sludge supernatant and SRF first decreased, subsequently increased, and reached the minimum value of 1.22 NTU and 1.79×10^9 m·kg^{-1} at 40 mg·g^{-1} DS H_2O_2 and 50 mg·g^{-1} DS Fe^{2+}, respectively.

Fe^{2+} plays catalytic and reductive roles in the Fenton reaction. If the Fe^{2+} dosage is considerably small in Fenton reagent, then the amount of hydroxyl radicals generated by Fenton reagent is also small [25]. Fe^{2+}may be adsorbed by sludge flocs, which play no catalytic and reductive roles in Fenton reaction [26]. However, the high Fe^{2+} dosage resulted in high efficiency of sludge dewatering. Evidently, the optimal sludge dewatering performance was obtained when the Fe^{2+} dosage was 40–60 mg·g^{-1} DS. Nevertheless, if the Fe^{2+} dosage is large, the amount of H_2O_2 involved in the Fenton reaction is relatively small, and the sludge dewatering performance decreases [27].

3.2. Effects of Fe^{2+} dosage on sludge disintegration using Fenton process alone

As shown in Fig. 2, the effects of Fe^{2+} dosage on sludge disintegration at different $Fe^{2+}$and H_2O_2 dosages were explored. The concentrations of polysaccharide, protein,

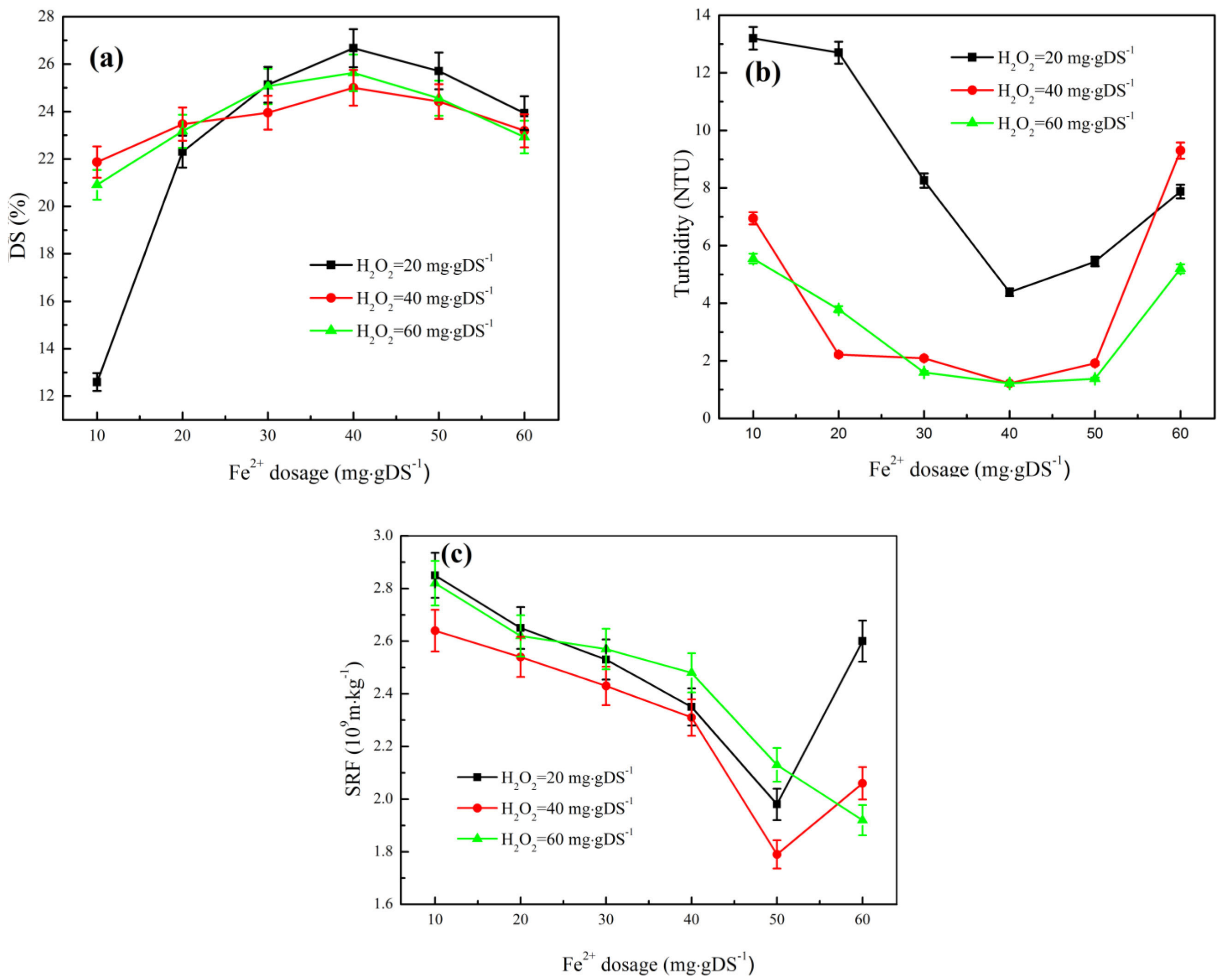

Fig. 1. Effect of Fenton reagent dosage on sludge dewatering performance: (a) DS, (b) turbidity, and (c) SRF.

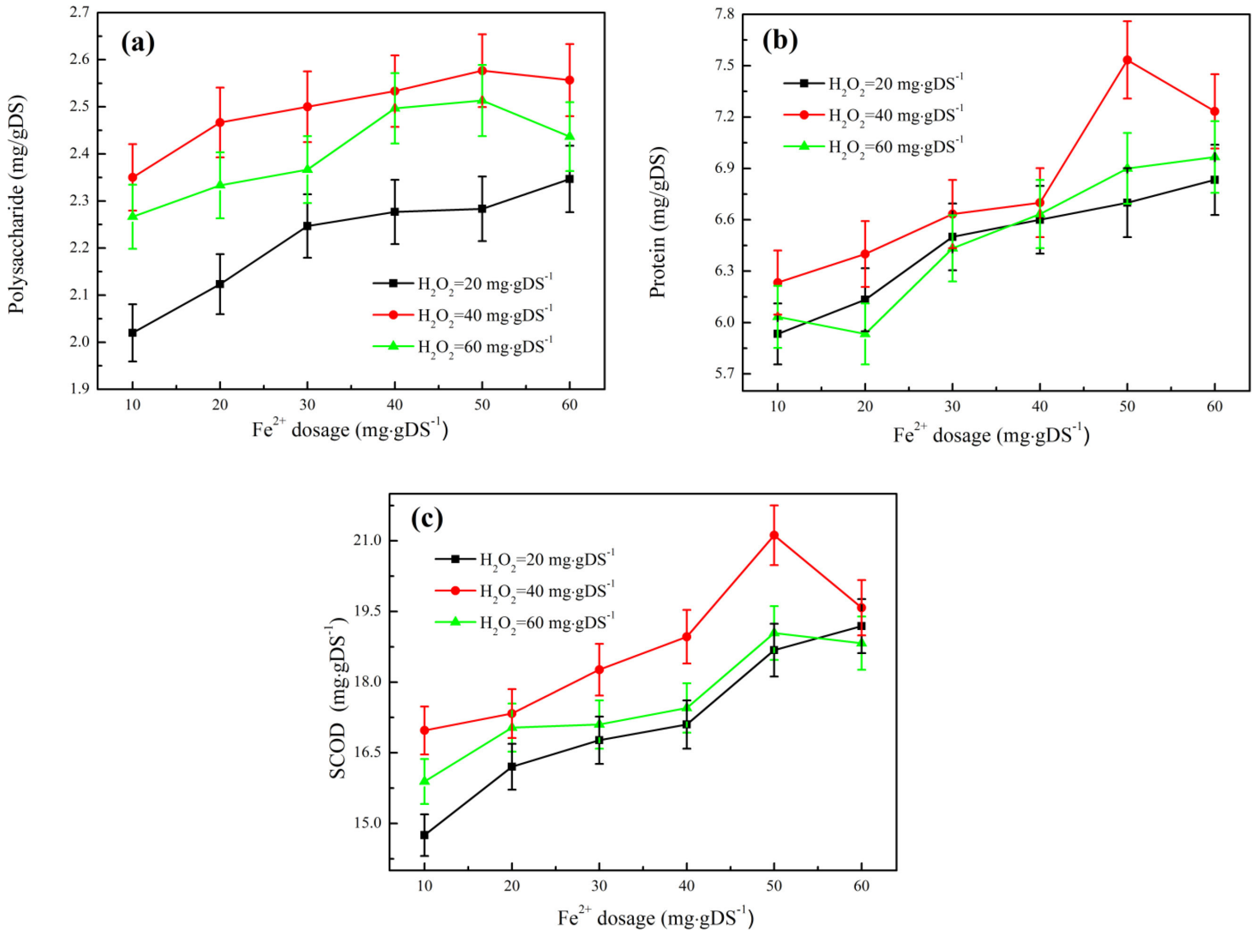

Fig. 2. Effects of Fe^{2+} dosage on extracellular polymeric substance disintegration: (a) polysaccharide, (b) protein, and (c) SCOD.

and SCOD in the filtrate first increased and subsequently decreased with the increased Fe^{2+} dosage under constant H_2O_2 dosage. Significantly, the maximum concentrations of polysaccharide, protein, and SCOD in the filtrate obtained at 50 mg·g^{-1} DS Fe^{2+} and 40 mg·g^{-1} DS H_2O_2 were 2.55, 7.50, and 21.0 mg·g^{-1} DS, respectively. With the continuing addition of Fe^{2+} to the sludge, the contents of filtrate polysaccharide, protein, and SCOD decreased gradually. In the Fenton reaction process, H_2O_2 will generate a large number of OH· in the Fe^{2+} catalysis [28], which can effectively disintegrate the organic compounds in the sludge and release the polysaccharide and protein of the microbial surface [29]. Given that the high molecular weight polysaccharide and protein exerted crucial influence on sludge filtration properties, the SRF increased. When proper amount of Fenton reagent is used to oxidize the sludge, the rigid structure of the polysaccharide and protein is broken, and the sludge flocs are destroyed; consequently, the concentrations of polysaccharide, protein content, and SCOD in the supernatant are increased [30]. When excess amount of Fe^{2+} in the Fenton reagent is added, the H_2O_2 amount is insufficient to oxidize Fe^{2+} to Fe^{3+}, thereby weakening the oxidation ability of Fenton reagent [31]. Therefore, the optimal oxidation disintegration performance can be achieved only when the H_2O_2 and Fe^{2+} dosages reach a certain proportion. Thus, the optimal Fe^{2+} and H_2O_2 dosages are 50 and 40 mg·g^{-1} DS, respectively.

3.3. Effects of Fenton reaction time on sludge dewatering performance using Fenton process alone

The effects of Fenton reaction time on sludge dewatering performance are shown in Fig. 3, and it represented raw sludge at Fenton time 0 min. With the increased reaction time, the DS and SCOD increased gradually, whereas the supernatant turbidity and SRF decreased slowly (Figs. 3(a) and (b)). The DS, supernatant turbidity, SRF, and SCOD gradually stabilized when the reaction time reached 90 min, at which their optimal values were 27.66%, 1.63 NTU, 1.77 ×10^9 m·kg^{-1}, and 21.30 mg·g^{-1} DS, respectively.

Fig. 3(c) illustrates the concentrations of polysaccharide and protein as a function of Fenton reaction time. The polysaccharide and protein contents increased gradually with the extension of reaction time. The maximum contents of polysaccharide (2.62 mg·g^{-1} DS) and protein (7.55 mg·g^{-1} DS) were obtained when the reaction time was 90 min. Nonetheless, the polysaccharide and protein contents became stable when the reaction time was more than 90 min.

The steps in the Fenton reaction in the sludge are the polysaccharide and protein decomposition, cell wall disintegration, and release of bound water and dissolved organic matter, such as polysaccharide and protein. This reaction rapidly increases the concentrations of SCOD, polysaccharide, and protein [5]. However, Fenton reaction requires a certain

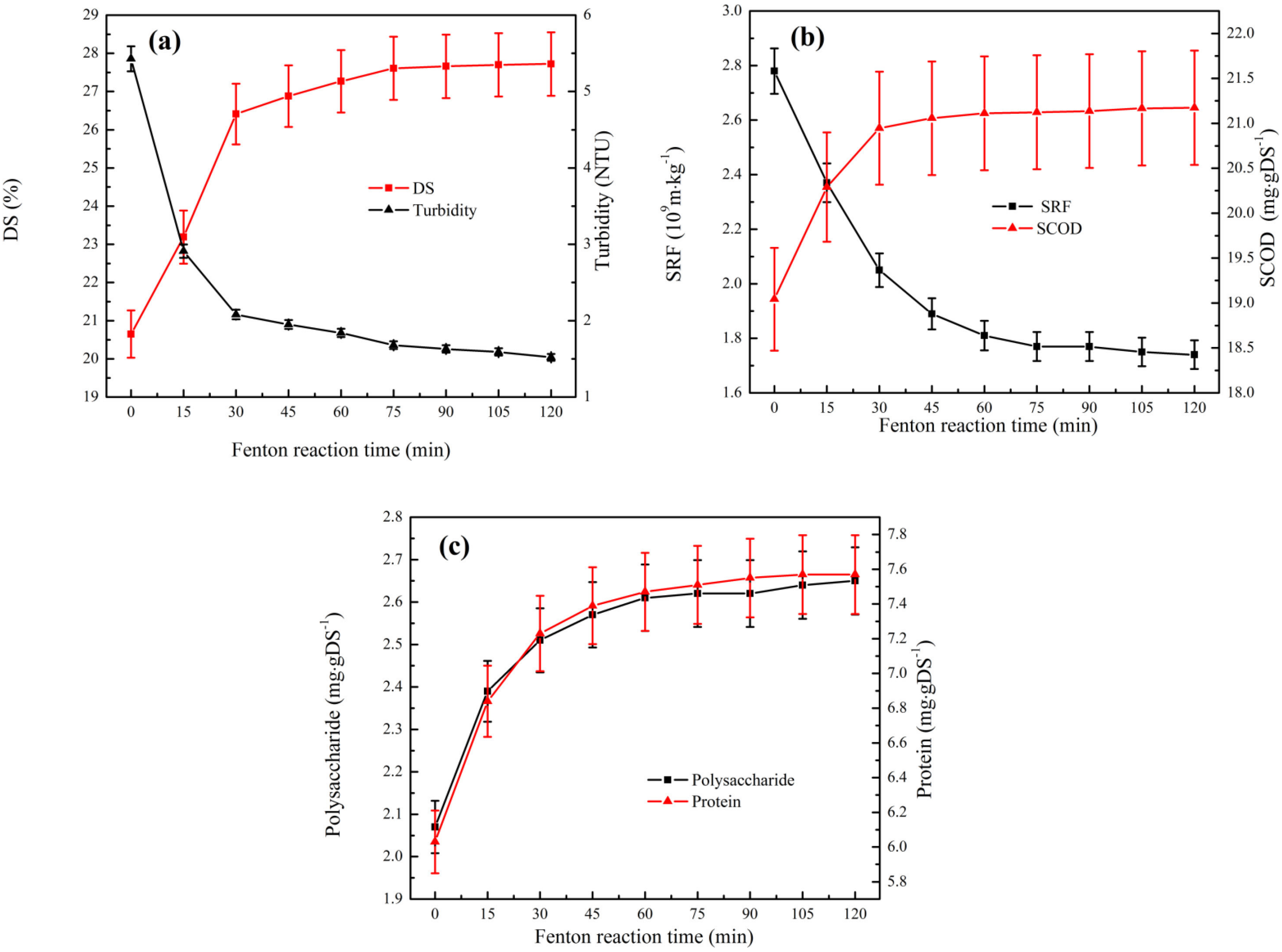

Fig. 3. Effects of Fenton reaction time on sludge dewatering performance: (a) DS and turbidity, (b) SRF and SCOD, and (c) polysaccharide and protein (40 mg·gDS^{-1} H_2O_2, 50 mg·gDS^{-1} Fe^{2+}).

reaction time to produce OH· and consequently oxidize the polysaccharide and protein in the sludge. With the increased reaction time, the OH· concentration and degree of sludge disintegration increase gradually [32]. Therefore, as shown in Fig. 3, with the extension of Fenton reaction time, the DS of sludge increases, the SRF and supernatant turbidity decrease, polysaccharide, protein content, and SCOD in the supernatant increase gradually [30]. When the reaction time reaches a certain extent, the released organic substances (such as protein and polysaccharides) are further oxidized to small molecules, such as volatile fatty acids, H_2O, and CO_2 [33]. Therefore, the optimal reaction time was 90 min.

3.4. Effects of CPAM dosage on sludge dewatering performance in combined Fenton and flocculation process

Fig. 4 shows the effect of CPAM dosage on DS, residual turbidity, SRF, zeta potential, polysaccharide, protein content, and SCOD. With the increased CPAM dosage, the DS of sludge increased first and subsequently decreased, whereas the turbidity of the supernatant first decreased and subsequently increased (Fig. 4(a)). Evidently, the optimal solid content and supernatant turbidity were 33.63% and 1.80 NTU, respectively. With the increased CPAM dosage, the SRF after Fenton and flocculation process decreased first and subsequently increased; the zeta potential gradually changed from a negative to a positive charge (Fig. 4(b)). In addition, the optimal value of SRF was 1.77×10^9 m·kg^{-1} at 60 mg·L^{-1} CPAM. Fig. 4(c) presents that the SCOD and the polysaccharide and protein concentrations increased first and subsequently decreased with gradually increased CPAM dosage. The optimal values of SCOD, polysaccharide, and protein were 21.47, 2.70, and 7.76 mg·g^{-1} DS at 60 mg·L^{-1} CPAM, respectively.

The CPAM can provide the adsorption binding site for sludge particles through Van der Waals force and hydrogen bonds, neutralize the negative charges on the surface of the sludge flocs, and reduce the electrostatic repulsion [34]. With the increased CPAM dosage, the sludge dewatering performance increased gradually. However, when the CPAM dosage was more than 60 mg·L^{-1}, the excess dosage of CPAM hindered the extension of flocculant molecules. Additionally, colloidal particles tend to restabilize due to positive rejection, which is not conducive to the improvement of sludge dewatering performance [35]. The particle size of raw sludge is 100–200 μm, while the sludge particle is more than 400 μm after Fenton-flocculation process. In addition, the sludge

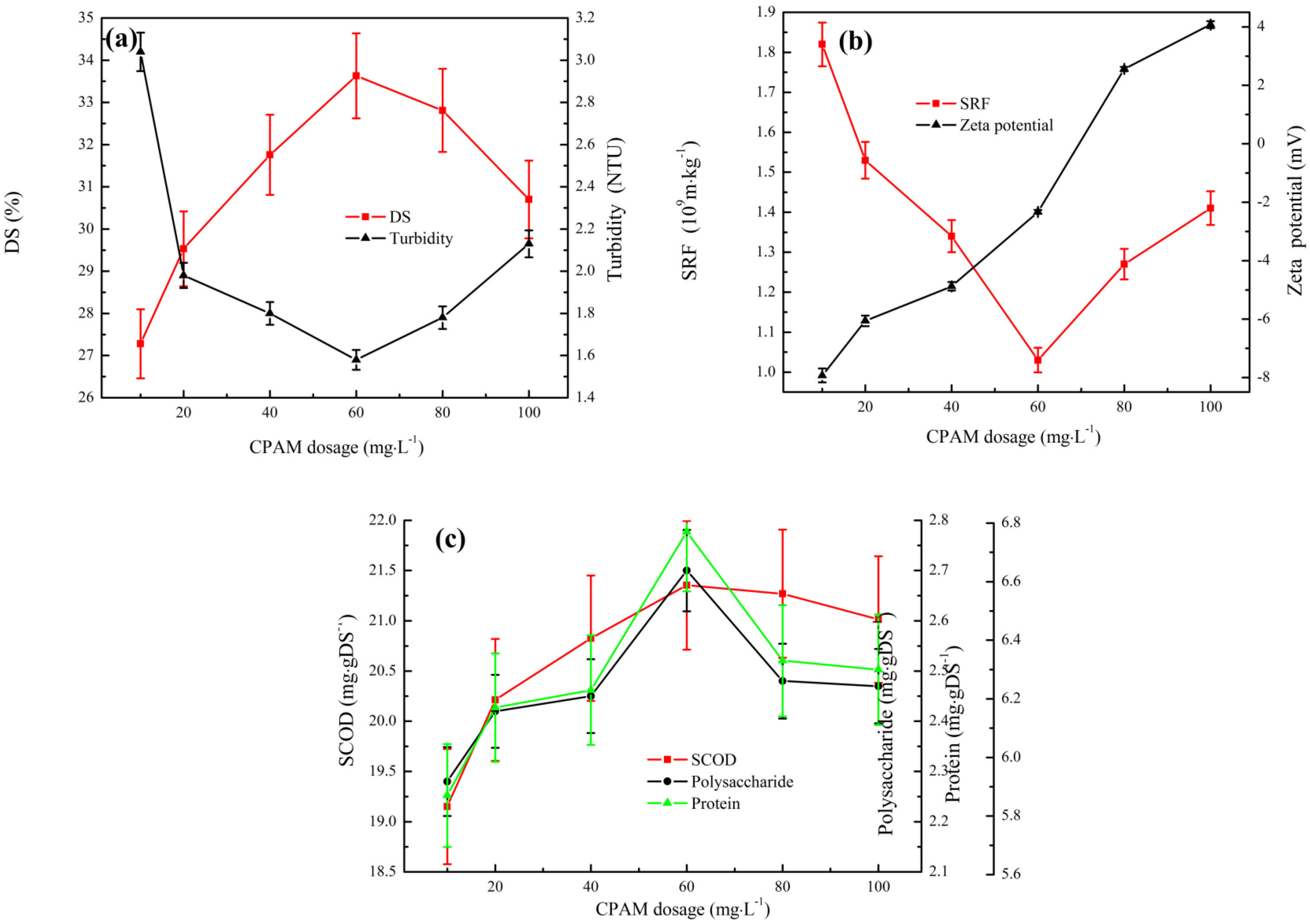

Fig. 4. Effects of CPAM dosage on sludge dewatering performance: (a) DS and turbidity; (b) SRF and zeta potential; (c) SCOD, polysaccharide, and protein (40 mg·gDS^{-1} H_2O_2, 50 mg·gDS^{-1} Fe^{2+}, and Fenton reaction time of 90 min).

particles are aggregated, flocculated into large flocs, and then gathered into pieces with sedimentation. Therefore, when the flocculant dosage was more than 60 mg·L^{-1}, the DS of sludge decreased gradually, and the turbidity of supernatant increased gradually. With its increased dosage, CPAM can adsorb a small amount of polysaccharides and proteins from the surface of sludge particles by flocculation adsorption [36]. When a large amount of flocculant is added to the sludge, the charge in the sludge will be neutralized, and the sludge particle surface will possess the opposite charge, which will stabilize the colloidal particles in the sludge; some of polysaccharide and protein is adsorbed on the surface of the sludge flocs [37]. Consequently, the polysaccharide and protein contents in the supernatant showed a decreasing trend. The experiment found that the optimal flocculation and sludge dewatering performance was obtained at 60 mg·L^{-1} CPAM.

3.5. Effects of pH on the sludge dewatering performance of combined Fenton and flocculation process

The effects of pH 2–11 on sludge dewatering performance were examined, and results are shown in Fig. 5. The optimal pH for sludge dewatering of combined Fenton and flocculation process was 3 (Fig. 5(a)), at which the solid content and turbidity were 34.20% and 1.06 NTU, respectively. Evidently, the difference in DS was less than 5% when the pH of the sludge was acidic. When the sludge solution pH was alkaline, the DS decreased remarkably. This result was attributed to that the sludge pH can change the charge state, ionization degree, and redox potential of the sludge particle surface [38]. The H^+ in the acidic solution can effectively neutralize the negative charges on the surface of the sludge particles, reduce the repulsion between particles, and stabilize the flocs [39]. In addition, Fenton oxidation is strongly dependent on the pH of the solution. pH is an important parameter in the Fenton reagent oxidation reaction because it determines the yield of OH· and the concentration of soluble Fe^{2+} [40]. When the pH of the sludge solution is less than 4, the Fenton reagent can produce a strong oxidation of OH·. Nonetheless, the Fe^{2+} in the sludge solution is hydrolyzed to precipitate and lose its catalytic activity when the sludge solution pH is alkaline [41]. Furthermore, the Fenton oxidation reaction showed the strongest activity when the pH of the sludge solution was 2–4. Hence, the optimal disintegration of polysaccharide and protein was obtained when the sludge solution pH was 2–4.

The effects of sludge solution pH on SRF and zeta potential are shown in Fig. 5(b). The SRF increased slowly with the increase of pH in the range of 2–8. However, the SRF increased abruptly when the sludge solution pH was in the range of 8–11. The minimum SRF value obtained at

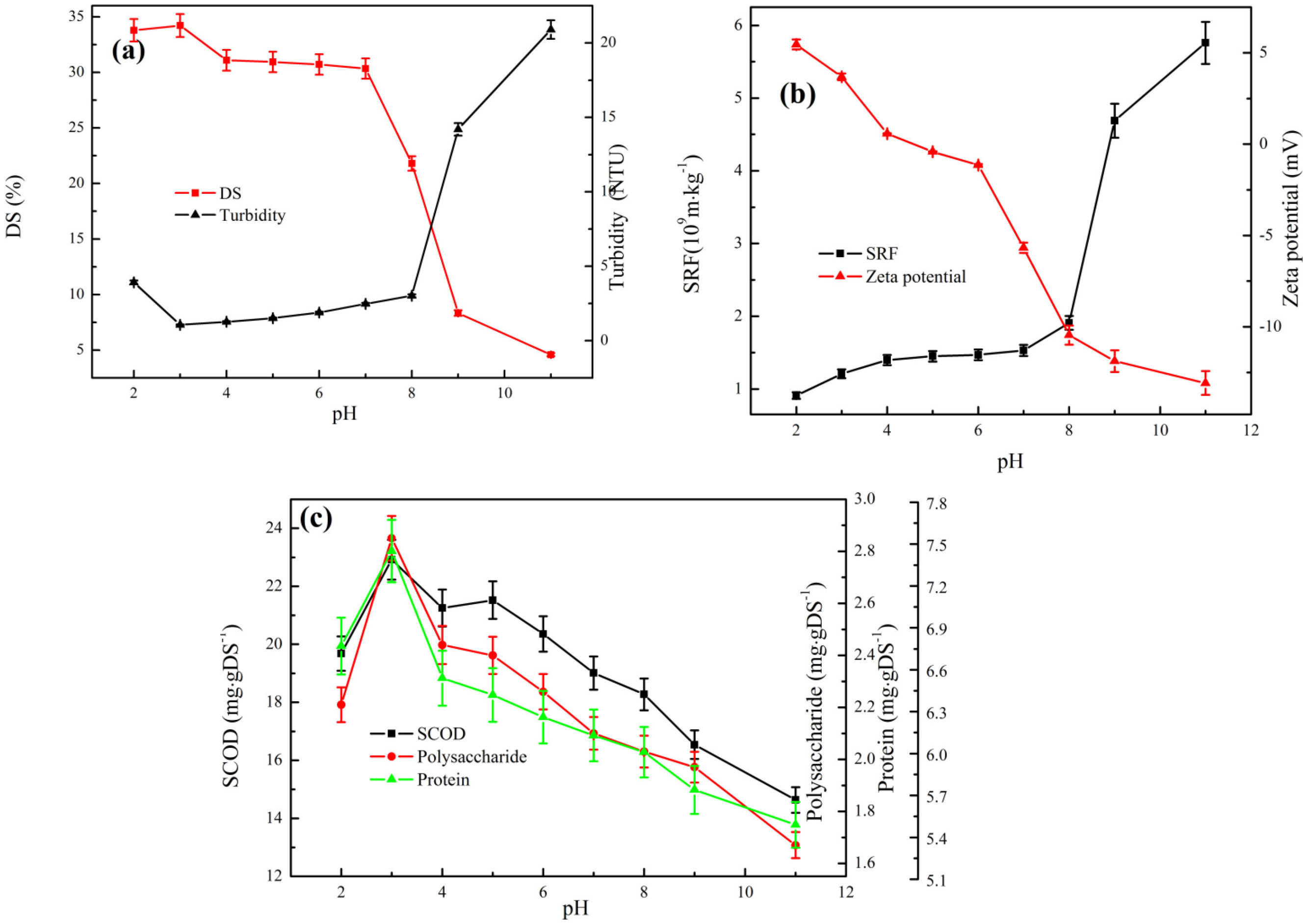

Fig. 5. Effects of pH value on sludge dewatering performance: (a) DS and turbidity; (b) SRF and zeta potential; and (c) SCOD, polysaccharide, and protein (40 mg·gDS^{-1} H_2O_2; 50 mg·gDS^{-1} Fe^{2+}; Fenton reaction time, 90 min; 60 mg·L^{-1} CPAM).

pH 2 and 3 was 0.91×10^9 and 1.21×10^9 m·kg^{-1}, respectively. When the Fenton reagent was applied to sludge conditioning alone, the minimum SRF was 1.79×10^9 m·kg^{-1}. Additionally, the zeta potential decreased gradually with the increased pH value. The zeta potential was close to 0 when the sludge solution pH was in the range of 4–5, at which the optimal sludge dewatering performance was obtained. Given the presence of negative ion, the activated sludge can neutralize H^+ in the acid sludge solution [42]. Therefore, the zeta potential changed slowly when the pH of the sludge solution was acidic. Nevertheless, the zeta potential value remarkably changed when the sludge solution pH was alkaline.

The effects of pH on the polysaccharide, protein, and SCOD are illustrated in Fig. 5(c). With the increased sludge solution pH, the changes in the contents of protein, polysaccharide, and SCOD in the supernatant exhibited a similar trend. When the pH value of sludge solution increased from 2 to 11, the concentrations of polysaccharide, protein, and SCOD increased first and subsequently decreased rapidly. Correspondingly, these concentrations reached the maximum value at pH 3 with 2.85, 7.45, and 25.5 mg·g^{-1} DS, respectively. This result was attributed to that pH is an important parameter in the Fenton reagent oxidation reaction, and it determines the yield of OH· and the concentration of soluble Fe^{2+}. When the pH of the sludge solution is high, the oxidation efficiency of Fenton reaction is low due to the formation of $Fe(OH)_3$ [43]. However, when the sludge solution pH is considerably low, high H^+ concentration retards the formation rate of $FeOOH^{2+}$, which consequently reduces the OH· yield and the oxidation capacity of Fenton reaction [44]. The amount of polysaccharide and protein disintegrated when the sludge was reduced.

3.6. Sludge morphology and mechanism of sludge disintegration and flocculation

Fig. 6 shows the morphology of DS: Figs. 6(a)–(c) show the original sludge, Figs. (d)–(f) the sludge conditioned by Fenton process alone, and Figs. (g)–(i) the sludge conditioned by a combination of Fenton process and flocculation process. SEM analysis results showed that the original sludge was dense, and the sludge structure was loose (Figs. 6(a)–(c)). The original sludge particles were small and uniformly distributed (Figs. 6(b) and (c)). Figs. 6(d)–(f) present the SEM images and photographs of DS conditioned by Fenton process alone. This sludge was characterized with large particle size and smooth surface of granular sludge. Many tiny particles were also present in the sludge flocs (Fig. 6(e)). Most of the organic impurity particles are aggregated under the action of Fe^{3+} [45]. The water content in the residual sludge decreased when the sludge was treated by the Fenton process (Fig. 6(f)), but the dewaterability of the sludge was still poor. Figs. 6(g)–(i) illustrate the characteristics of the sludge conditioned by a combination of Fenton and flocculation processes. The sludge particles in SEM images were larger than that obtained by Fenton oxidation alone. The sludge flocs were evident after the combined treatment of sludge in Fig. 6(h). The dewatering performance was also significantly enhanced in Fig. 6(i).

Remarkably, Fenton reagent and flocculants showed different mechanisms for sludge dewatering.

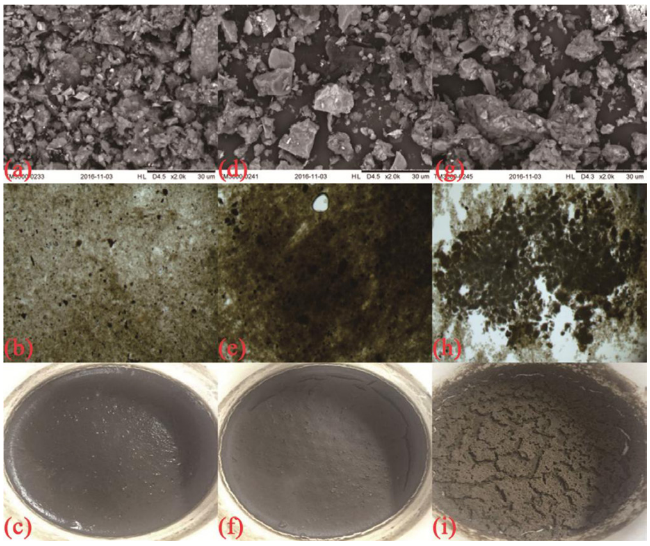

Fig. 6. SEM images and photographs of dewatered sludge: (a–c) original sludge, (d–f) sludge conditioned by Fenton process alone, and (g–i) sludge conditioned by a combination of Fenton process and flocculation process.

On the basis of the present results and analysis, a preliminary description of the possible synergistic mechanisms of combined Fenton and flocculation process for sludge conditioning was obtained and is shown in Fig. 7. In the combined sludge treatment, Fenton reaction disintegrates polysaccharide and protein [46]. However, CPAM causes the flocculation of colloidal particles in the sludge [47]. The Fenton reaction evidently degraded the organic molecules from large molecular sizes into small ones via highly reactive hydroxyl radicals. The flocculant CPAM can flocculate small sludge particles together rapidly by coagulation–flocculation process [21]. First, the Fenton reaction produced a large number of hydroxyl radicals (·OH) with strong oxidation and subsequently caused the disintegration of polysaccharide and protein adsorbed on sludge cell surface. The internal binding water of sludge cells is also released [48]. Afterward, the protein and polysaccharide cracked from polysaccharide and protein diffuse into the sludge solution [44]. Simultaneously, sludge particles are oxidized into small particles [49]. With CPAM addition, the colloidal particles in the sludge solution are rapidly flocculated because of the strong charge adsorption [50]. Finally, the large flocs are gradually formed and settled to the bottom of the beaker with cleared supernatant due to the bridging mechanism of the flocculants [51].

4. Conclusions

This study used a combination of Fenton pre-oxidation and flocculation process to improve the sludge dewatering performance. Parameters affecting sludge dewatering performance, such as H_2O_2 dosage, Fe^{2+} dosage, initial sludge pH, and CPAM dosage, were studied. The improved sludge dewaterability by Fenton oxidation was attributed to the release of both interstitial water trapped between organics and adsorbed and chemically bound water by the degradation of polysaccharide and protein. In addition, the $Fe^{2+}$and

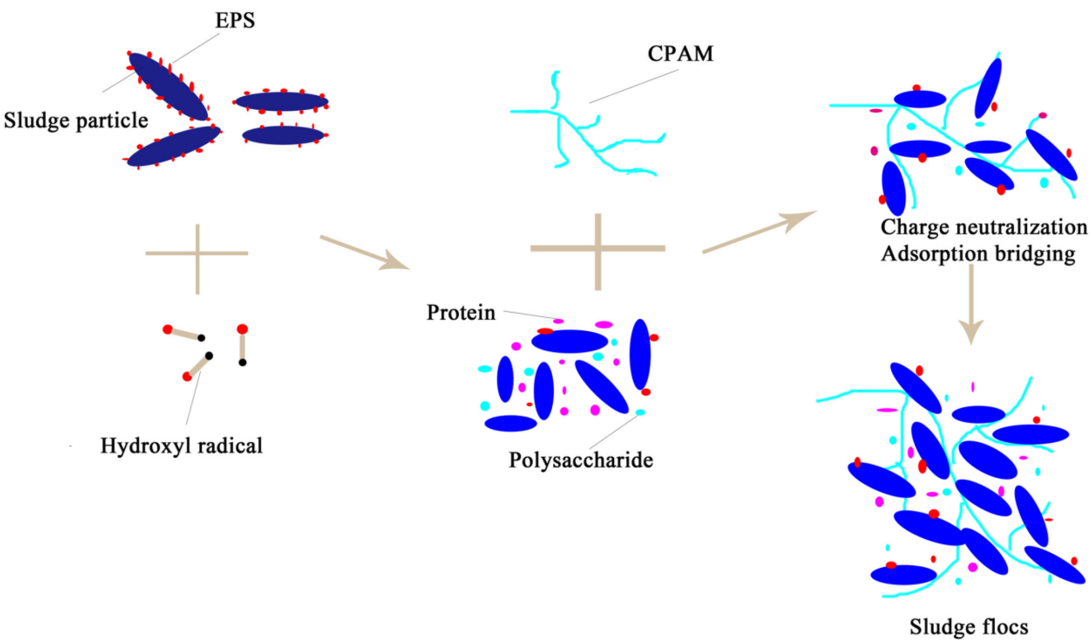

Fig. 7. Possible synergistic mechanisms of combined Fenton and flocculation process for sludge conditioning.

H_2O_2 concentrations and the Fenton reaction time were the main factors affecting the degree of sludge disintegration. Results also showed that the concentrations of SCOD, polysaccharide, and protein in sludge supernatant reached the maximum value at 40 mg·g^{-1} DS Fe^{2+}, 40 mg·g^{-1} DS H_2O_2, and Fenton reaction time of 90 min. pH value exerted a more significant influence on the degradation of polysaccharide and protein than that of Fe^{2+} and H_2O_2 dosages under Fenton oxidation treatment. Fenton oxidation and flocculation also exhibited a significant synergistic effect on improving sludge dewatering. The main mechanism of sludge conditioning by Fenton reaction was the disintegration of polysaccharide and protein and the release of the internal binding water. Microscopic analysis revealed that sludge flocs were formed after flocculation using CPAM.

Acknowledgments

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