Insight into the comparison of thermally and Fe(II) activated persulfate on sludge dewaterability and disintegration

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ABSTRACT

The effects of thermally and Fe(II) activated potassium persulfate (PPS) on sludge dewatering performance were compared systematically. Sludge dewaterability was monitored by measuring capillary suction time (CST) and sludge specific resistance to filtration (SRF), and the degradation effect was characterized by chemical oxygen demand (COD), total organic carbon (TOC), ammonia nitrogen (NH₄⁺-N) and extracellular polymeric substances (EPS). The change of extracellular polymer substance (EPS) including soluble, loosely bound and tightly bound EPS (S-EPS, LB-EPS and TB-EPS) with time and PPS dosage was monitored to discuss the oxidation efficiency of thermally and Fe(II) activated PPS. Sludge supernatant was analyzed by three-dimensional fluorescence excitation–emission spectra (3D-EEM) to confirm the protein transformation. The result showed that sludge dewaterability in terms of CST and SRF were enhanced with increasing PPS dosage and condition time of two activated methods. While Fe(II) activated PPS could reduce sludge CST and SRF to preferred values at low PPS dosage and short condition time. Meanwhile, sludge degradation effect was also more obvious. Mechanically, sludge TB-EPS in proteins and polysaccharides converted to SB-EPS was faster with Fe(II) activated PPS. In addition, thermally activated PPS tended to oxidize the protein in the supernatant first.

Key words: activated methods, extracellular polymeric substances (EPS), potassium persulfate (PPS), sludge condition, sludge dewatering performance

HIGHLIGHTS

- Thermal and Fe(II) activated potassium persulfate were used to condition sludge.
- Fe(II) activated potassium persulfate showed higher efficiency with lower dosage.
- Thermally activated persulfate obtained consistent results over a long duration.
1. INTRODUCTION

The treatment and disposal of surplus sludge from sewage treatment plants is a serious challenge in current environmental management. The moisture content of sludge is as high as to 95% (Xu et al. 2020); even after mechanical dewatering processes, such as centrifugal dewatering and belt type pressure filter with simply conditioning, it still up to around 80%. In order to improve sludge dewatering efficiency, the surplus sludge usually will be conditioned before dewatering.

As a very promising pretreatment for sludge disposal, advanced oxidation processes have been used to improve sludge dewaterability and degradability, and have received the extensive concern of many researchers, because the oxidant could destroy the sludge structure and degrade organic matter in sludge (Feng et al. 2014). Among them, ozone oxidation (Zhang et al. 2016), peroxide oxidation (Kim et al. 2009), chloride oxidation (Wang 2014), persulfate oxidation (Xiao et al. 2017), and Fenton reagent oxidation (Xiao et al. 2018; Cai et al. 2019) could degrade macromolecular organic matter and reduce the sludge volume, some of which could also improve sludge dewatering performance. The oxidants listed above are based on the generation of highly reactive free radicals, such as superoxide free radicals, hydroxyl free radicals, and sulfate radicals. Among them, advanced oxidation based on persulfate system has attracted increasing attention in sludge pretreatment because of its significant advantages in modifying sludge dewatering effect and application conditions, such as its high oxidation–reduction potential, long duration of oxidation, and wide range of applicable pH (Zhen et al. 2012a, 2012b). For instance, when Waclawek et al. used persulfate system to treat digested sludge, the oxidant not only degraded the sludge, but also enhanced sludge dewaterability (Wacławek et al. 2016). With the development of research and increasingly stringent social requirements, researchers have explored many synergistic effects of activated persulfate coupled with other methods to condition sludge, in order to enhance dewatering performance further (Liu 2019). For example, the microwave-persulfate(PS)/Fe(II) system was used to improve sludge dewatering performance and reduce sludge capillary sucking time (CST) by 94.6% (Zhen et al. 2019); When PS/Fe(II) was combined with polymeric material for sludge pretreatment, sludge specific resistance to filtration (SRF) was significantly reduced, with a decreased percentage of nearly 94.9% (Wang et al. 2017).

However, it should be pointed out that persulfate can only produce a large number of reactive groups-sulfate radicals (SO₄²⁻) in a certain activation mode. The reported persulfate activating methods included transition metal activation, thermally activation, alkali activation, and photoactivation (Shi et al. 2015; Min et al. 2016). Due to the complexity of sludge composition and poor light transmittance, at present, only Fe(II) and thermally activation systems have been widely investigated to improve sludge dewatering performance (Xiao et al. 2018), and the application of other activating methods is relatively rare, so more efforts are needed.

As mentioned above, persulfate needs to be activated to develop the oxidation effect, in addition the enhancement of sludge dewaterability also depends on the activation method (Zhen et al. 2013), such as using iron (Shi et al. 2015; Li et al. 2018) and
heat (Min et al. 2016) to activate persulfate for condition sludge, can significantly improve sludge degradation and dewatering properties at the same time, and reduce dewatered sludge moisture content and solid content (Wang et al. 2009). It is worth noting that in the research process of improving the dewatering performance of sludge, the migration and transformation of organic matter in sludge and the physicochemical effect are two inevitable issues to be discussed. The composition of sludge is complex, and the mechanisms of persulfate activation using heat and ferrous iron are different. In addition, the effect of heat and iron ions on sludge is also different, so the results of sludge treatment may be different. According to current studies, there have been much research on thermally or ferrous ion activated persulfate to condition sludge (Kim et al. 2016), but relatively few studies have compared thermally and Fe(II) activated methods. Therefore, there is a need to carry out research on sludge dewatering performance and degradation by different activation methods.

In summary, an in-depth comparative study of thermally activated and ferrous activated persulfate in sludge conditioning should be carry out. Therefore, this project aimed to build thermally/Fe(II) activated persulfate oxidation processes to treat sludge, and explore the effect and mechanism of improving sludge dewatering performance at first; then, systematically investigate the migration and transformation of EPS in sludge before and after oxidation treatment and its correlation with sludge dewaterability.

2. MATERIALS AND METHODS

2.1. Experimental materials
The sludge was taken from the secondary sedimentation tank of a waste water treatment plant in Wuhan, China. The collected sludge was pretreated by sieving through a 30 mesh sifter to get rid of large particles. Then the sludge sample was immediately transferred to the laboratory and stored in a plastic container at 4 °C prior to use. The ranges of sludge pH, moisture content, suspended solid (SS) and volatile suspended solid (VSS) were 7.2–7.4, 96.8–97.4%, 25–29 g/L and 14–16 g/L respectively.

2.2. Experimental procedures
The experiment parameter setting is shown in Table 1. The waste activated sludge treatment experiments were conducted in a batch reactor using a 500 mL Erlenmeyer flask with 300 mL raw sludge. The reaction was initiated by adding an aliquot of the persulfate stock solution into the reactor containing waste activated sludge. The concentrations of persulfates for the experiments were 0.2, 0.5, and 1.0 mmol/g DS of sludge. Upon the addition of persulfate, the reactor was immediately heated with stirring at 120 rpm by a ceramic hot plate stirrer. The reactor was continuously heated until the temperature reached 80 °C (for 5 min), and the temperature was controlled constant (± 1.0 °C) by intermittent heating during the entire reaction (60 min). The sludge samples were withdrawn at predetermined time intervals and immediately immersed into an ice bath to quench further reaction. The samples were cooled for 3 min in the ice bath prior to analysis. The dewaterability of sludge was monitored using CST and SRF.

2.3. EPS extraction
EPS in the sludge was stratified into soluble EPS (S-EPS), loosely bound EPS (LB-EPS), and tightly bound EPS (TB-EPS) by the EPS extraction protocol slightly modified from a method described in the literature (Min et al. 2016). The sludge samples of 20 mL were centrifuged at a speed of 4,000 r/min for 10 min. The soluble organic matter in the supernatants after filtering

Table 1 | Experiment parameters setting

| Groups | Potassium persulfate (mmol/gDS) | Ferrous sulfate (mmol/gDS) | Temperature (°C) |
|--------|-------------------------------|--------------------------|-----------------|
| 1      | 0                             | 0                        | 80 (± 1)        |
| 2      | 0.2                           | 0                        | 80 (± 1)        |
| 3      | 0.5                           | 0                        | 80 (± 1)        |
| 4      | 1.0                           | 0                        | 80 (± 1)        |
| 5      | 0.2                           | 0.3                      | 25 (± 1)        |
| 6      | 0.5                           | 0.3                      | 25 (± 1)        |
| 7      | 1.0                           | 0.3                      | 25 (± 1)        |
through a 0.45 μm membrane were collected as soluble EPS (S-EPS). The bottom pellets were resuspended to the original volume using phosphate buffered saline (PBS, pH 7). Then the samples were centrifuged for 10 min at 10,000 r/min. The organic substances in supernatant were collected as loosely bound EPS (LB-EPS). The bottom pellets were resuspended using PBS and ultrasonicated at 20 kHz and 480 W for 10 min while maintaining a temperature of 4 °C in an ice bath. The suspensions were centrifuged at 10,000 r/min for 15 min, and the supernatants were collected as TB-EPS. Protein (PN) and polysaccharide (PS) were quantified in each fraction of the extracted EPS. For the analyses of PN and PS, the EPS samples were filtered through 0.45 μm polytetrafluoroethylene (PTFE) membrane syringe filters, and the filtrates were used to detect PN and PS concentration. PN and PS, which were assumed to be dominant components of EPS, were quantified in S-EPS, LB-EPS and TB-EPS. Protein concentrations were detected using the rapid Lowry method (Shanghai, China) with bovine serum albumin as the standard. Polysaccharide contents were measured using the anthracene ketone–sulphuric acid method, with glucose as standard substance (Dubois et al. 1956). All the physicochemical analyzes were performed in triplicate.

2.4. Other indicators

CST was monitored using a CST instrument (Triton, Model 304M, Essex, UK). SRF was measured using the Brucellae funnel method. As for other characterizations, sludge particle size could reflect the sludge accumulation or diffuse state which was analyzed by a laser particle analyzer (Mastersizer 2000, Pennsylvania-based firm, UK). Mixed liquor suspended solids (SS), VSS, and moisture contents were assessed using the gravimetric method (Association 1967). Zeta potential was carried out on a Zeta sizer nano S90 (Malvern, UK). Three dimensional fluorescence excitation–emission matrix (3D-EEM) spectra of SMP were scanned using a fluorescence spectrophotometer (F7000, Hitachi, Japan). TOC was analyzed using an organic carbon analyzer (multi N/C 2100, Jena, Germany). COD and ammonia nitrogen (NH₄⁺-N) were measured using international standard methods. Free radicals were detected by electron spin resonance (ESR) spectroscopy coupled with 5,5-dimethyl-1-pyrroline-N-oxide (DMPO) as a spin-trapping agent. Fe(II) in the filtrate was determined using an o-phenanthroline method. Total Fe in the filtrate was determined by reducing Fe(III) to Fe(II) first and then measuring the total Fe(II). Fe(III) content was the difference between total Fe and Fe(II). Fe in dewatered cake was extracted by nitric acid wet digestion firstly, then the iron species were analyzed in the leaching solution.

3. RESULTS AND DISCUSSION

3.1. Sludge dewaterability under different conditions

The result of sludge dewaterability in terms of CST is shown in Figure 1(a). Sludge CST increased significantly with increasing thermal treatment time individually. At the present dosing of PPS at 80 °C, CST increased slightly during the 60 min heating process when PPS dosage was fixed at 0.2 mmol/gDS. When PPS dosage was increased to 0.5 mmol/gDS or above, CST increased at first, then decreased after conditioning for 30 min. CST increased to 114.3 s in the first 15 min, then decreased

![Figure 1](http://iwaponline.com/wst/article-pdf/84/6/1464/942186/wst084061464.pdf)
sludge dewaterability could be modified with thermally activated PPS dosage to surpass a certain amount (such as 0.5 mmol/gDS PPS) and a relatively long condition time (30 min), but the improving efficiency did not increased profoundly continuously. For dosing PPS/Fe(II) to condition sludge, CST decreased when PPS and Fe(II) dosages were 0.2 mmol/gDS and 0.3 mmol/gDS, respectively. CST decreased from 96 s to 71.1 s in the first 15 min, and finally reached 57.4 s. When dosing at 0.5 and 1.0 mmol/gDS PPS, CST plummeted in 15 min, then changed relatively little, and finally went down to just 27.1 and 25.1 s, respectively. In comparative analysis of the two activated methods, both thermally and Fe(II) activated PPS could reduce CST efficaciously. When dosing 1.0 mmol/gDS and conditioning for more than 30 min, CST decreased to about 30 s for both Fe(II) and thermally activated PPS. However, sludge dewaterability could be modified with fewer PPS dosage and shorter condition times using Fe(II) activated PPS.

As shown in Figure 1(b), sludge SRF increased sharply from 1.94 × 10⁹ S²/g to 6.2 × 10⁹ S²/g with increasing time under solely thermal treatment conditions, indicating that sole heat treatment would deteriorate sludge dewaterability much seriously, which was consistent with the observations in earlier reports (Audrey et al. 2011; Min et al. 2016). While for thermally activated PPS conditioned sludge, SRF firstly increase with lower PPS dosage and then decrease visibly with higher PPS dosage and longer conditioning time. SRF dropped down to 1.92 × 10⁸ S²/g with a 90% reduction when dosing 1.0 mmol/gDS PPS under 80 °C for 60 min. For Fe(II) activated PPS pretreated sludge, SRF declined with increasing PPS dosage and conditioning time. SRF was reduced to 1.75 × 10⁸ S²/g with only dosing 0.5 mmol/gDS PPS for 30 min. The result confirmed the above inference that Fe(II) activated PPS could enhance dewaterability in terms of SRF more efficient at a lower PPS dosage and shorter condition time.

3.2. Sludge degradation under different conditions

The change of TOC concentrations in the supernatant of sludge conditioned thermally and by Fe(II) activated PPS is shown in Figure 2(a). TOC concentration increased with the increasing treating time within 60 min under 80 °C individually. For thermally activated PPS treated sludge, TOC concentration decreased at the first 15 min when dosing 0.2 and 0.5 mmol/gDS PPS. Then the concentration increased after 30 min. When the PPS dosage reached 1.0 mmol/gDS, TOC concentration increased during the experiment duration. The results indicated that part TOC in the supernatant of raw sludge was oxidized by activated PPS under low dosage and short thermal treating time. Then thermal interaction and oxidation made EPS release, and the amount was beyond that of the degraded TOC. While for Fe(II) activated PPS, the TOC in the supernatant increased continuously.

NH₄⁺-N contents in the supernatant of different sludge samples are shown in Figure 2(b). NH₄⁺-N concentration in the same group presented tended to increase with the increase of conditioning time. As to thermally activated PPS, NH₄⁺-N concentration also increased with increasing PPS dosage. For Fe(II) activated PPS, the evident change appeared at a lower PPS dosage of less than 1.0 mmol/gDS. The results indicated that more PPS and longer treatment time were beneficial for sludge degradation, which could produce more sulfate radicals for oxidizing sludge and supply an adequate reaction time. The analysis suggested that organic substances in the sludge, especially protein-like substances, began to hydrolyze under the action of activated PPS, producing ammonia nitrogen and releasing it into the liquid phase.

The time-dependent variation of COD concentration (as shown in Figure 2(c)) was monitored in sludge supernatant with different PPS dosages. The COD content in raw sludge increased from 649.0 mg/L to 1,477.6 mg/L with increasing condition time at 80 °C. In the presence of heat or Fe(II) activated PPS, the COD concentration decreased firstly and then increased with the same PPS dosage. While for the same condition time, the higher the PPS dosage was, the larger the COD concentration would be. Both heat and Fe(II) activated PPS could reduce the COD in sludge supernatant, and also make sludge release COD into the liquid phase. The difference was that Fe(II) activated PPS could make COD increase even more significantly, which might be because Fe(II) activation could produce more free radicals to oxidize COD in supernatant and degrade organic material in sludge flocs more quickly.

The change of TOC, NH₄⁺-N and COD concentrations in the supernatant illustrated that sludge disintegration was connected with thermal treatment and oxidation of activated PPS. The thermally activated PPS reacted with the organic matter in the liquid phase and then oxidized the organic matter in the sludge. However, Fe(II) activated PPS tends to oxidize organic matter in sludge flocs more obvious. The thermal elution of organics without PPS made COD and TOC present the maximum values. While the oxidation of PPS played a key role in releasing ammonia nitrogen by oxidizing nitrogenous
organic matter, such as proteins-like substances. Further determination of the reason for heat and Fe(II) activated PPS improved sludge dewaterability and disintegration is needed through the migration and transformation of organic matter in sludge flocs.

3.3. The migration of organic substances
According to the 3D-EEM spectra of different sludge sample supernatants (as shown in Figure 3) and the fluorescence substances of different Ex/Em regions (as shown in Table 2), the main fluorescence characteristic peaks are T₁ (λex/λem: 220–225/307), T₂ (λex/λem: 225/343–348), T₃ (λex/λem: 275/306), T₄ (λex/λem: 275–280/340–346), representing aromatic proteins (Zhu et al. 2015), soluble microbial by-products (Bourven et al. 2012), tryptophan-protein substances (Pang et al. 2014), humic acid substances (Wang et al. 2009), respectively.

In the 3D-EEM diagram of raw sludge supernatant, the main peaks of soluble microbial by-products, aromatic proteins, tryptophan-protein substances and humic acid substances were not obvious, when the sludge was treated at 80 °C the peaks became more evident with extending treatment time. The results indicated that organic matter in sludge such as EPS was degraded into soluble matter, especially protein-like substances. When dosing 0.2 mmol/gDS PPS under 80 °C, the four peaks also became more and more evident, while dosing 0.5 or more mmol/gDS PPS, the four peaks were very obvious at first for 15 min, then became fuzzy after 30 min. For the Fe(II) activated PPS group, the peaks were relative distinct.

Figure 2 | TOC (a), NH₄⁺-N (b) and COD (c) in sludge supernatant under different conditions.
Figure 3 | 3D-EEMs fluorescence spectra of different sludge supernatant samples with heat and Fe(II) activated PPS conditioning.
when dosing 0.2 mmol/gDS PPS, but which were more ambiguous than the thermally activated PPS group with the same condition time. When dosing 0.5 mmol/gDS PPS, the peaks changed slightly. For 1.0 mmol/gDS PPS, the peaks were very inconspicuous, and were on a trend to become significant. The results showed that protein, humic acid and microbial by-products in sludge supernatant changed with condition time, PPS dosage and activated methods. Thermal treatment could increase protein and humic acid substances in the supernatant, at low PPS dosage (0.2 mmol/gDS), the thermal effect still dominated the changing trend of the above substances. When PPS dosage reached 0.5 mmol/gDS, the oxidation of activated PPS played the role in controlling the migration and transformation of organic matter of the supernatant. As shown in the EEM diagram, thermally activated PPS needed more time to reduce the protein concentration in the supernatant, while Fe(II) activated PPS needed a larger dosage.

The protein and polysaccharide content changes in the three fractions of EPS are shown in Figure 4. The protein (Figure 4(a)) and polysaccharide (Figure 4(d)) concentrations in S-EPS of both Fe(II) and thermally activated groups increased with extending condition time and increasing PPS dosages. The result indicated that the activated persulfate-dominated advanced oxidation process induced a higher removal rate for proteins in the LB-EPS and TB-EPS than conventional coagulants, thus enhancing dewaterability (Masihi & Badalians Gholikandi 2018). But for dosing Fe(II) groups, the growth degree were more significant with short conditioning time at an equivalent PPS dosage. In contrast, the PN concentration of LB-EPS increased slightly less in Fe(II) activated PPS group than in the thermally group at a relatively small PPS dosage. As for the PN in TB-EPS, the concentrations decreased with the extending condition time and increasing PPS dosage, but for Fe(II) activated groups, PN concentration decreased faster than thermally treated group. A similar changing trend was observed for polysaccharides concentration.

Sludge treatments using advanced oxidation processes produce numerous oxidizing radicals, which result in the breakdown of EPS and changes in the chemical properties of EPS (Tahir et al. 2019). The final changed extents of protein or polysaccharide concentration were almost the same for Fe(II) and thermally activated PPS conditioned sludge EPS, however the progress was not uniform. In other words, Fe(II) activated PPS had higher efficiency in oxidizing TB-EPS to LB-EPS, then changing to S-EPS, which was beneficial for sludge dewatering performance and disintegration.

The zeta potential change of sludge conditioned by different methods is shown in Figure 5. For the thermally activated PPS treated sludge, zeta potential increased with increasing pretreatment time, but changed slightly with different PPS dosage in the same condition times. Conversely, zeta potential increased with both increasing PPS dosage and condition time when using Fe(II) activated PPS to condition sludge. This result indicated that the oxidation of activated PPS induced charge neutralization. This might be because the oxidation process disrupted flocs and reduced the amount of negatively charged organic matter, such as protein in EPSs, in addition the positive ions would reduce the electro-static repulsion of flocs, leading to charge neutralization (Wang et al. 2018a; Liang et al. 2020). In particular, when dosing 1.0 mmol/gDS PPS and 0.5 mmol/gDS Fe(II), zeta potential changed from negative to positive, and reached 0.53 mv. A similar result was observed in an earlier report (Li et al. 2020). There are two main reasons to explain the phenomenon, firstly, in the PPS activating process, much produced Fe(III) would neutralize some negative charge; secondly, an extracellular polymeric substance was oxidized and degraded by activated PPS, especially protein-like substances, which were supposed to be negatively charged.

The iron species in the filtrate and dewatered cake are shown in Figure 6. When the PPS dosage was 0.2 or 0.5 mmol/gDS, Fe(II) and Fe(III) in the filtrate and Fe(III) in dewatered cake did not change with condition time clearly as shown in Figure 6(a), 6(b) and 6(d). Fe(II) content in dewatered cake was 1.9 mg/gDS and was nearly the same at different PPS.
Figure 4 | Changes of protein and polysaccharide concentrations in EPSs of sludge conditioned by heat and Fe(II) activated PPS.
dosage and condition time, but which were much higher than in raw sludge. When dosing 1.0 mmol/gDS, Fe(II) in filtrate decreased from about 45.1 mg/L to 5.0 mg/L, and Fe(III) increased most in the first 15 min, then decreased to about 24 mg/L after 30 min. Corresponding, Fe(III) in dewatered cake increased significantly and was higher than in raw sludge,

Figure 5 | Zeta potential of sludge conditioned by heat and Fe(II) activated PPS.

Figure 6 | Fe$^{2+}$ (a) and Fe$^{3+}$ (b) in filtrate and Fe$^{2+}$ (c) and Fe$^{3+}$ (d) in dewatered cake.
especially conditioned for 30 and 60 min. The results indicated that at low PPS dosage (less than 0.5 mmol/gDS), 0.3 mmol Fe(II)/gDS was sufficient to activate PPS, and Fe(II) and Fe(III) in dewater cake increased after conditioning by Fe(II)/PPS.

3.4. Mechanism analysis of the difference between thermally and Fe(II) activated PPS in conditioning sludge

ESR spectra of Fe(II) and thermally activated PPS is shown in Figure 7. The results showed that the intensity of the DMPO-SO₄ adduct signal was very weak, while the relatively strong DMPO-OH adduct signal appeared in the activated PPS system, but the free radicals were mainly sulfate radicals (Wang et al. 2018b). The mechanism based on the persulfate oxidation system to improve sludge dewatering and disintegration performance was discussed by detecting the sludge extracellular polymeric substances (EPS) and zeta potential, preliminarily determining the oxidation of target organic substances by activated persulfate (Guo et al. 2019). The results presented in this study at the same time revealed and confirmed the mechanism, which agreed with current researchers’ opinion that sludge dewatering and degradation performance improved by persulfate is due to the sulfate radicals produced in the activation process (Niu et al. 2013), which moved to the oxidative mineralization of eluted organic substances. The mechanism supposed that activated persulfate destroyed the sludge floc structure, made the extracellular polymeric substance or intracellular matters degrade from insoluble macromolecular substance into small molecules soluble matters, and released bound water inside the sludge particles at the same time (Li et al. 2019). A higher EPS content, especially proteins (Yu et al. 2008), can lead to a higher affinity for bound water and a smaller number of large pores.

Figure 7 | ESR spectra of Fe(II) and thermally activated PPS.

Figure 8 | Mechanism of Fe(II) and thermally activated PPS conditioning sludge for improving dewaterability and disintegration effect.
which can hamper the separation of water from the sludge cake during mechanical dewatering (Wang et al. 2020). This also illustrated that the sludge dewatering performance is closely related to the properties of organic matter (such as molecular weight) in the sludge (Niu et al. 2013).

A mechanism graph of Fe(II) and thermally activated PPS conditioning sludge is shown in Figure 8. Fe(II) and thermally activated PPS exhibited similar functions on modified sludge dewaterability during relatively long condition times and high PPS dosage, however, this effect was much different for the short condition time duration. The results may be because Fe(II) activated PPS would produce strong enough oxidation compared with heat in a shorter time duration, meaning that the reaction rate of Fe(II) activated PS was larger than that of thermally activated PPS in conditioning sludge. In addition, Fe(II) was oxidized into Fe(III) which could react with sludge fleetingly. Furthermore, the thermal treatment deteriorated the sludge dewaterability more significantly than thermally activated PPS improved dewaterability in the short condition time.

4. CONCLUSIONS

Thermally and Fe(II) activated PPS has a similar effect on improving sludge dewaterability with adequate and the same condition time and PPS dosage. Activated persulfate compared with the thermal effect or produced Fe(III) played the main role in improving sludge dewatering performance. Activated PPS could destroy sludge flocs and TB-EPS, change protein types and sludge zeta potential, which was beneficial for dewaterability. However, sludge CST and SRF decreased faster when using Fe(II) activated PPS than thermally activated PPS to condition sludge. The difference between the two conditioning methods was reflected in the early effects on sludge properties, for example thermally activated persulfate first oxidizes protein-like substances in liquid phase and then oxidizes sludge TB-EPS, meanwhile COD and TOC in sludge supernatant decreased first and then increased. However for Fe(II) activated PPS, it was more inclined to oxidize sludge TB-EPS, so that COD and TOC in the supernatant basically increased at all times.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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