Pyrite activated peroxymonosulfate combined with a physical–chemical conditioner modified biochar to improve sludge dewaterability: analysis of sludge floc structure and dewatering mechanism

Wenjian Gao1 · Lei Song1· Zehao Wang1 · Lili Xuan1

Abstract

In this study, we proposed an advanced oxidation process of pyrite (FeS2) and peroxymonosulfate (PMS) and prepared a modified polyaluminum chloride biochar (P-BC). The motivation is to use the combination of FeS2 + PMS + P-BC to improve waste activated sludge (WAS) dewaterability. The method to improve the sludge dewatering effect with the combination of FeS2 + PMS + P-BC is as follows: in the first step, pour 0.75 g/g TSS FeS2 and 0.6 g/g TSS PMS into the sludge, and stir for 15 min. Then, add P-BC and stir for 5 min; complete the entire WAS processing process. The vacuum filtration test was used to evaluate the dehydration effect. The water content (Wc) and specific resistance to filtration (SRF) of the raw sludge can be reduced from the original values of 92% and 2.36 × 1013 m/kg to 67% and 9.89 × 1011 m/kg, respectively. The results showed that the combination of FeS2 + PMS + P-BC can effectively improve the sludge dewatering effect through oxidation.

A laser particle size analyzer is used to observe changes in sludge particle size. The median diameter of sludge particles increased from 55.37 to 64.56 μm. A zeta analyzer to is used observe changes in sludge particle size. The median diameter of sludge particles increased from 55.37 to 64.56 μm. A zeta analyzer to is used observe changes in sludge zeta potential. The zeta potential of sludge particles increased from −15.8 to 0.4 mV. In the analysis of extracellular polymeric substances (EPS) of sludge, it was found that protein (PN) and polysaccharide (PS) in EPS decreased significantly. To further analyze the phenomenon of PN and PS drop, excitation-emission-matrix spectra (3D-EEM) was used. To observe the changes of sludge functional group, X-ray photoelectron spectroscopy was used. It was found that FeS2 + PMS + P-BC can destroy the functional groups of sludge, such as O−H, C−C, and O═C—NH− related to proteins and polysaccharides.

Keywords Sludge dewatering · FeS2/PMS oxidation · Re-flocculation · Modified biochar · Hydrophobic

Introduction

Extensive research has shown that waste activated sludge (WAS) treatment and disposal is still the biggest challenge faced by wastewater treatment plants (WWTP) (Raheem et al. 2018). WAS is a multiphase colloidal system composed of organic pollutants, colloids, cellular material, and water. WAS is characterized by a complex composition and carries a variety of substances that are harmful to the environment, for example, a large number of microorganisms, bacteria, and even the most harmful new coronavirus, as well as various refractory organic pollutants, dioxins, heavy metals, etc. (Maryam et al. 2021). These characteristics have brought great economic burden and environmental challenges to the treatment and utilization of sludge. Therefore, to find a method to reduce the difficulty of WAS dehydration and avoid environmental
burden, it has the advantages of industrial urgency and scientific significance.

Methods of dealing with WAS can be divided into physical methods and chemical methods. Physical methods include the addition of skeleton filter aids, thermal methods, freeze–thaw methods, the use of microwaves, and ultrasonic conditioning. Rao et al. (Rao et al. 2018) studied the relationship between the moisture content of sludge and freeze–thaw pretreatment. The Wc of untreated sludge and freeze-thawed sludge were 42% and 38%. Mobaraki et al. (Mobaraki et al. 2018) conducted a high-power ultrasonic vibration–enhanced sludge dewatering experiment, and the study showed that high-power ultrasonic can improve the dewatering performance of sludge.

Reports on chemical methods, such as bioleaching (Liu et al. 2016a), acid and alkaline pretreatment (Li et al. 2009; MacDonald et al. 2018), and advanced oxidation treatment (Li et al. 2016), have been mentioned in the literature to improve the dehydrolysis capacity. There are a number of important differences between hydroxyl radicals (•OH) through the activation on hydrogen peroxide (H₂O₂) and generate sulfate radicals (SO₄²⁻•) through the activation on peroxymonosulfate (PMS). Generate sulfate radicals higher oxidation potential (2.5–3.1 eV), higher stability, longer life time (30–40 µs), and wider pH range (3.0–10.0) compared with hydroxyl radicals (•OH)–based advanced oxidation process (AOPs) (Min et al. 2016). have been widely used to process WAS (Alexopoulou et al. 2019). At present, the main method for activating PMS is by using transition metals represented by Fe²⁺. However, the method of PMS activation by Fe²⁺, there are certain limitations to our study. The reaction product of Fe²⁺ is ferric hydroxide precipitate, precipitated Fe hydroxides, collected after sedimentation and filtration and are hard to dewater (Marouek et al. 2022). This will further reduce resource utilization and increase actual cost (Wang et al. 2019). Pyrite (FeS₂) comes from the natural iron mineral on the earth’s surface, it is characterized by abundant reserves, cheap and wide sources. Since FeS₂ can release Fe²⁺ to activate PMS, it is considered to be a potential activator. Formulas 1–3 are the chemical reaction process of FeS₂ releasing Fe²⁺. Liang proved that FeS₂ + PMS treatment is an effective method to improve WAS dehydrolysis capacity and triclosan removal efficiency (Liang et al. 2021). In the study of WAS, FeS₂ was used to activate PMS given that a new activated combination requires scientific attempts to enrich the theoretical studies on sludge treatment and disposal.

\[2\text{FeS}_2 + 7\text{O}_2 + 2\text{H}_2\text{O} \rightarrow 2\text{Fe}^{2+} + 4\text{H}^+ + 4\text{SO}_4^{2-} \quad (1)\]

\[4\text{Fe}^{2+} + \text{O}_2 + 4\text{H}^+ \rightarrow 4\text{Fe}^{3+} + 2\text{H}_2\text{O} \quad (2)\]

\[\text{FeS}_2 + 14\text{Fe}^{3+} + 8\text{H}_2\text{O} \rightarrow 15\text{Fe}^{2+} + 2\text{SO}_4^{2-} + 16\text{H}^+ \quad (3)\]

Although chemical conditioning methods play a great role in sludge dewatering, the final effect of AOP on WAS treatment is limited and has several negative effects. When AOP releases the bound water (BW) in the extracellular polymeric substances (EPS), the floc structure will be destroyed, and organic substances, such as protein (PN) and polysaccharide (PS), will escape. The destruction of the floc structure will increase the clogging of the microfiltration medium, resulting in a decreased filterability of the sludge (Wu et al. 2015). In addition, the escaped PN and PSs are the main hydrophilic substances in the sludge, and they are important factors influencing sludge polarity. Wu et al. believed that after the AOP treatment, the spatial distribution of extracellular PN is the key to determining the removal of interstitial water from sludge (Wu et al. 2017). Therefore, to further improve the effect of dehydration, some studies began exploring AOP combined with other processes. To improve the filterability of sludge, scholars have developed various framework particles to adjust the sludge to improve its dewatering performance (Cao et al. 2019; Wang et al. 2017). In these studies, the framework additives formed a permeable and hard lattice structure in the sludge, and the sludge cake remained permeable during the compression and filtration process, thereby improving the dewaterability of the sludge. However, the addition of framework additives is a single-mechanism method and has limitations, used with chemical regulators (thus complicating the search for the best combined dosage) or large amounts of physical regulators (which will greatly increase the solid content of the sludge).

The annual output of crop straw in my country is as high as 0.5–0.8Gt, accounting for about 30% of the world’s total straw. The main method of disposal of straw is incineration. However, this method will cause environmental pollution, and it also violates my country’s national ecological policy of carbon neutrality. The biochar made of straw has the advantages of hard structure and good water permeability and is a good skeleton additive. Using straw to make biochar (BC) can avoid environmental pollution and reduce carbon dioxide emissions. This reflects the principles of waste utilization and environmental protection. In this study, targeting with BC as modifier, the prepared modified polyaluminum chloride (PAC) biochar (P-BC) can be used as a framework assistant to increase the rigidity of the sludge and prevent clogging of the filter pores while exerting chemical flocculation; the limitations can be overcome. The sludge can be re-flocculated through reactions such as adsorption and charge neutralization. These processes will promote the accumulation of PN and other organic substances in the supernatant, improve the sludge hydrophobicity, and further enhance the dehydration effect.

Therefore, this study aimed to evaluate FeS₂/PMS combined with P-BC as an innovative technology to improve the dehydration capacity of WAS. The water content (Wc) and
specific resistance to filtration (SRF) were used as the main indicators to measure the dehydration performance. To conduct a comprehensive study on the principle of dehydration, we analyzed the changes in the WAS floc structure before and after pretreatment by scanning electron microscopy (SEM) along with zeta potential and particle size. Different EPS components were extracted, and their respective functions and distributions in sludge dewatering were revealed by UV spectrophotometry, three-dimensional fluorescence spectroscopy (3D-EEM), and X-ray photoelectron spectroscopy (XPS). The underlying mechanisms of EPS and sludge-floc microstructural changes were revealed. After dehydration, the calorific value of the sludge cake and the recovery of FeS₂ were analyzed. An economic analysis of energy consumption during the research process was carried out with reference to “Networked, Smart, and Responsive Devices in Industry 4.0 Manufacturing Systems” and “Deep Learning-enabled Smart Process Planning in Cyber-Physical System-based Manufacturing” to evaluate the practical application of this research potential.

Materials and methods

Materials

Waste activated sludge

WAS samples were taken from the aeration tank of a sewage treatment plant in Hohhot, Inner Mongolia, China. The sewage treatment plant adopts the CASS reactor treatment process, and the daily water treatment capacity is about 50,000 m³. In order to get good experimental results, we used a 4-mm sieve to remove large impurities in the sludge before the experiment, and stored the sludge in the refrigerator at 4 °C. The main characteristics of the original sludge are shown in Table 1.

Experimental drug

The drugs used in this experiment are shown in Table 2. All the drugs in the table are from China, and the purity is of analytical grade.

P-BC preparation

All procedures for material preparation were done at the School of Civil Engineering, Inner Mongolia University of Technology.

Preparation of BC

The original biochar (BC) was prepared with waste straw as the raw material and at Muffle (Japan, MOV-212F-PC, Panasonic Health and Medical Equipment Co., Ltd.) furnace 500 °C for 40 min. The BC was washed with deionized water for ash and then dried in an oven at 105 °C. Then, BC was ground and sieved to obtain a particle size of 200 µm.

Preparation of P-BC

a. The BC was soaked in 1 mol/L HCl solution at a ratio of 1:10 (g:mL) for 12 h and then centrifuged at 6000 r/min for 10 min in a centrifuge (China, TG16-WS, Changsha Xiangyi Co., Ltd.).

b. The precipitate was collected and dried. Then, the precipitate was immersed in 0.5 mol/L PAC at a ratio of 1:10 (g:mL), stirred with a magnetic stirrer (China, DZKW, Shanghai Luda Experimental Instrument Co., Ltd.) at a reaction temperature of 70 °C for 4 h and then cooled naturally.

c. After soaking for 12 h, the sample was centrifuged for 10 min at 3000 r/min; the sediment was collected, dried naturally, ground, and sieved to obtain P-BC.

Table 1 Characteristics of raw sludge

| Wc (%) | SRF (10¹³ m/kg) | TSS (g/L) | VSS (g/L) | pH |
|--------|----------------|-----------|-----------|----|
| 99.39  | 1.83           | 5.545     | 8.776     | 6.32 |

Table 2 Chemical reagents

| Drug name          | Molecular formula | Manufacturer                                |
|--------------------|-------------------|---------------------------------------------|
| Pyrite             | FeS₂              | Jereh Minerals Co., Ltd                     |
| Polyaluminum Chloride | [Al₉(OH)₉Cl₆-n]m | Mingchuang Environmental Protection Materials Co., Ltd |
| Bovine serum albumin | —                 | Hefei Qiansheng Biotechnology Co., Ltd         |
| Folin Phenol Reagent | —                 | Xinyu Biotechnology Co., Ltd                  |
| Methanol           | CH₃O              | Sinopharm Group Chemical Reagent Co., Ltd     |
| Tert-butanol       | C₄H₁₀O            | Sinopharm Group Chemical Reagent Co., Ltd     |
| 5,5-Dimethyl-1-oxypyrrole | 5,5-Dimethyl-1-oxypyrrole | Sinopharm Group Chemical Reagent Co., Ltd     |
Sludge conditioning

If no special requirements were needed, a series of experiments was carried out with 100 mL WAS samples in a 250-mL beaker. First, FeS_2 + PMS according to the ratio of 1:0.8 reagents of different preset concentrations were added to the WAS sample. The mixture of sludge and FeS_2 + PMS was stirred at 300 r/min for 10 min and allowed to stand for 1 min. Then, the pretreated sludge was poured onto another beaker. In a beaker, the remaining undissolved FeS_2 particles were recovered by precipitation. Then, P-BC with different preset concentrations was added to the sludge, and the mixtures were stirred for 5 min at 150 r/min. For comparison, BC and P-BC were added for comparison under the same conditions.

Dewatering procedure and performance evaluation of the sludge

A vacuum filtration device (China, HH-1, Shanghai Luda Experimental Instrument Co., Ltd.) with a 9-cm Buchner funnel and qualitative filter paper was used for the dehydration treatment of WAS. Take a 100 mL sample of the treated sludge and pour it into a Buchner funnel for dehydration. A constant pressure of 0.04 MPa is used with a vacuum pump, and the sample is filtered through 0.45-μm filter paper. After continuous filtration for 5 min, the effluent of the filtrate was continuously collected, the flux data were collected every 5 s, and the filtration time and volume of the filtrate were recorded until no filtrate exuded. SRF and Wc were calculated using the method provided by Liu et al. (Liu et al. 2016b).

EPS extraction and analysis

The three-layer EPS is extracted according to the method provided by Li et al. (Li et al. 2019). A three-dimensional fluorescence spectrometer (Japan, F-7100, Hitachi) was used to detect the 3D-EEM spectrum of the EPS extract.

The Folin–phenol method was used to determine the PN content, the anthrone colorimetric method was used to determine the PS content, and the mechanism was analyzed through the changes in the substances of EPS.

Other methods

The total solids, volatile solids, Wc, and pH were determined in accordance with standard methods. A combination of differential scanning calorimeter (DSC) (DSC-214 Polyma GER) and thermogravimetric analyzer (TGA) (TGA-Q600 USA) was used to measure BW and free water (FW) and total water (TW). SEM (Japan, S-4800, Hitachi) was used to observe the microstructure of P-BC and flocs. A zeta potential (UK, Nano Zeta SE, Malvern) meter and laser particle size analyzer (China, NKT5200-H) were used to analyze the sludge structure before and after pretreatment. XPS (China, Ultim Extreme, Oxford Instruments) was used to identify the chemical states of C, O, N, and Si in the WAS samples.

Economic analysis

The economic cost has a very important influence on the application potential of a new process. The cost of this study primarily included the cost of FeS_2 and PMS and the cost of preparation of P-BC. Inner Mongolia is rich in pyrite and produces a large amount of waste ore every year. The cost of FeS_2 can be saved through the recovery and recycling of waste ore and straw recycling through farmland. These are not included in the cost. Table 3 is the cost and energy consumption breakdown of the consumables used.

Results and discussion

Influence of single and combined process treatment on sludge dewatering performance

Analysis of sludge Wc and SRF in single and combined processes

Batch experiments were carried out to study the effects of FeS_2 + PMS, P-BC, and BC on the dehydration performance of WAS. As shown in Figure S1, first, single, and combined dehydration performance tests were performed on the three substances, and the best Wc and SRF results for each were summarized (Fig. 1). Figure S1 shows that the addition of FeS_2 + PMS can significantly reduce Wc and SRF. With the increase in dosage, when FeS_2 + PMS was added to 0.6/0.48 and 0.75/0.6 g/g TSS, the SRF and Wc reached the optimal values of 5.37 × 10^{12} m/kg and 77.9%, respectively. This result is due to the use of FeS_2 as an activator to effectively activate PMS to produce a large amount of SO_4^2−.

Table 3 Chemical reagents

| Materials and equipment | Unit price (RMB) | Energy consumption and equipment depreciation (RMB) |
|-------------------------|------------------|---------------------------------------------------|
| FeS_2                   | —                | —                                                 |
| PMS                     | 600/t            | —                                                 |
| Straw                   | —                | —                                                 |
| Muffle furnace          | —                | 50,000/year                                       |
| PAC                     | 800/t            | —                                                 |
activate PMS according to the discussion: First, according to the expression in the introduction and the analysis of the chemical equation, since FeS$_2$ can release Fe$^{2+}$ to activated PMS. This reaction process creates a prerequisite for FeS$_2$ to activate PMS. The released Fe$^{2+}$ then undergoes a redox reaction with PMS to generate SO$_4^{2-}$•. According to chemical Eqs. (4)–(7).

$$\text{Fe}^{2+} + \text{HSO}_3^- \rightarrow \text{Fe}^{3+} + \text{SO}_4^{2-} + \text{OH}^- \quad (4)$$

$$\text{SO}_4^{2-} + 2\text{H}_2\text{O} \rightarrow \text{SO}_4^{2-} + \text{H}^+ \quad (5)$$

$$\text{Fe}^{2+} + \text{SO}_4^{-} \rightarrow \text{Fe}^{3+} + \text{SO}_4^{2-} \quad (6)$$

$$\text{SO}_4^{-} + \text{HSO}_3^- \rightarrow \text{SO}_5^- + \text{SO}_4^{2-} + \text{H}^+ \quad (7)$$

The oxidation capability of SO$_4^{2-}$• can effectively crack EPS and release the internal BW, thereby enhancing the dehydration capability of WAS. When Wc reached the lowest value, SRF rebounded slightly, possibly due to the large amount of SO$_4^{2-}$•, which led to the destruction of the floc structure and clogging of filter holes during the suction filtration process.

Figure 1 also reveals that the use of BC and P-BC alone can improve the dehydration performance of WAS. However, when FeS$_2$/PMS was combined with AOP and BC, a more significant dehydration effect was observed compared with a single AOP or BC. When AOP was combined with BC (Fig S1), FeS$_2$+PMS was under the premise of an optimal dosing ratio of 0.75/0.6 g/g TSS. When the dosing amount of BC reached 3.5 g/L, the Wc and SRF ranged from 92.3% and 2.36×10$^{13}$ m/kg to 75.1% and 2.88×10$^{12}$ m/kg, respectively. Given the synergistic effect of AOP and carbon-based framework additives, the void structure and water permeability of BC can form a channel for releasing water, which prevents the clogging of filter pores during suction filtration and promotes water penetration and release (Guo et al. 2019). This result is also consistent with the findings of previous research (Shi et al. 2016; Wang et al. 2021). However, the strong oxidation effect of AOP is not selective. As a result, while oxidizing EPS to release BW, SO$_4^{2-}$• also oxidizes microbial cells to release more intracellular substances, such as PN and DNA. These substances are often hydrophilic, which will further deteriorate the dehydration performance. BC as a conventional physical aid cannot solve these negative effects produced by AOP.

As shown in Fig. 1, when FeS$_2$/PMS was combined with P-BC, the Wc and SRF decreased from 92.3% and 2.36×10$^{13}$ m/kg to 67.4% and 9.89×10$^{11}$ m/kg, respectively. Thus, the combination of AOP with P-BC has a very positive effect on improving the dehydration efficiency. We believe that P-BC can improve the structure of flocs decomposed after the AOP treatment and re-flocculate the flocs of WAS into larger and looser versions. P-BC itself, as a skeleton aid, will form a strong skeleton structure in the mud cake to improve the release of sludge BW, thereby improving the dewatering effect of WAS.

**Changes in water distribution of WAS flocs**

The filtrate cake water was analyzed by DSC and TGA to verify that the combined process bound water liberation. DSC peak area graph and TGA change graph are in the attachment. According to DSC analysis, the temperature from 20 to 20 °C free water undergoes an endothermic and exothermic process, and the peak area of the DSC curve represents the free water content. Compared with the peak area of the original sludge, the peak area of the sludge treated by FeS$_2$+PMS, FeS$_2$+PMS+BC, and FeS$_2$+PMS+P-BC all became smaller, among which FeS$_2$+PMS+P-BC became the most obvious. According to TGA analysis, when the temperature increased from 20 to 150 °C, the weight of the original sludge was 86%, and the weights after conditioning by FeS$_2$+PMS, FeS$_2$+PMS+BC, and FeS$_2$+PMS+P-BC were reduced to 81%, 73%, and 58%, respectively. Sludge becomes easier to dewater after conditioning. FW and BW contents were calculated, and the results are shown in Fig. 2. The figure shows that after the AOP treatment alone, TW, FW, and BW dropped from the original value of 12.63, 8.46, and 4.17 g/g TSS to 6.03, 3.97, and 2.06 g/g TSS, respectively. These findings provide additional evidence for SO$_4^{2-}$• from the FeS$_2$/PMS oxidation broke the WAS flocs and bound water liberation. As a result, bound water were released into the liquid phase.

When AOP was combined with P-BC, TW, FW, and BW can be reduced to 2.08 g/g TSS, 1.21 g/g TSS, and 0.87 g/g TSS, respectively. Compared with the original mud, the
contents of the three water types were reduced by 83.5%, 85.6%, and 79.1%. Compared with the use of AOP alone or in combination with BC, AOP combined with P-BC can improve the dehydration effect better. One of the reasons for the better sludge dewatering effect, we think it may be that AOP decreased the size of sludge particles when it released BW through oxidation. Therefore, in the following content, we go back to explore the changes in sludge particles. Although the addition of BC can play a certain framework support effect, it cannot change the negative effect of the deteriorating hydrophobicity of the sludge and the destruction of floc structure. P-BC is a modified BC, and the surface-loaded PAC has a very high positive charge density and good coagulation performance (Duan and Gregory 2003, Wu et al. 2008). Aside from playing the role of framework additives, it can promote the release of BW and increase the strength of the flocculated structure of the sludge (Cao et al. 2021). Therefore, when AOP is combined with P-BC, it can effectively offset the negative effects of AOP itself, take advantage of P-BC, improve the filterability of the sludge, and further improve the dehydration efficiency. By sharp contrast, the use of several inorganic or organic flocculants (such as iron or aluminum salts) alone (Saveyn et al. 2005; Zhang et al. 2017) will hardly destroy the EPS structure and convert the combined water into FW; in certain cases, it can reduce the dehydration efficiency. Such a phenomenon occurs because as a flocculant, it can interact with EPS through charge neutralization and bridging reactions; however, BW remains embedded in the sludge floc and is difficult to release (Yan et al. 2013).

**Effect of initial pH value on sludge performance**

Figure 3a shows the pH value changes of sludge during the FeS$_2$ + PMS conditioning sludge process at different initial pH values. It can be seen from Fig. 3a that with the reaction time, the pH value of the sludge shows a downward trend, down to 6.61, 6.06, 5.79, 4.27, and 3.68. The main reason for this result is that the activation of PMS by FeS$_2$ is a redox reaction, and FeS$_2$ will generate a large amount of H$^+$ in the process of being oxidized; at the same time, SO$_4$$^{2-}$ will consume OH$^-$ in water and release H$^+$ (Fang et al. 2013). This phenomenon is consistent with Yuan et al.’s study using ferrous sulfide particles to activate persulfate to degrade chloroaniline. Figure 3b shows the effect of different initial pH
values on the sludge regulation effect of FeS$_2$+PMS + P-BC. It can be seen from Fig. 3b that lower pH value, especially pH value lower than 6.0, obviously improves the dewaterability of sludge. According to Fig. 3a, when the initial pH value of the sludge is less than 6, after 15 min of reaction, the pH value is basically maintained at an acidic condition of about 4. Acidic conditions will promote the continuous release of Fe$^{2+}$ from FeS$_2$, while when the pH value of the solution is greater than 4, iron hydroxide precipitates will be formed and adsorbed on the surface of FeS$_2$, inhibiting the release of Fe$^{2+}$. In general, when the initial pH value of the sludge is maintained below 7, the dewatering performance of the sludge is not significantly affected, and the initial pH value of the raw sludge used in the experiment satisfies this condition. Therefore, in the advanced oxidation pretreatment sludge during the process, there is no need to adjust the initial pH value of the sludge.

**Free radical analysis of FeS$_2$-activated PMS**

Electron paramagnetic resonance (EPR) and radial scavenging experiments were used to analyze the source of the oxidation capacity of FeS$_2$ + PMS. As shown in Fig. 4a, the process of sludge treatment, methanol (MeOH) and tert-butanol (TBA) were added to determine which free radical played a leading role. It can be seen from the figure that when TBA was added to the sludge, the Wc of the sludge increased from 77.1 to 78.3% after 15 min, which indicated that only 1.2% of the sludge dewatering effect was related to OH. When MeOH was added to the sludge, the dewatering effect of the sludge deteriorated significantly. Finally, Wc rises to 91.1%, which means that 12.8% of sludge dewatering effect is related to SO$_4$•−. To be more sure of the presence of free radicals, EPR analysis of the sludge was also performed, and the results are shown in Fig. 4b. There were obvious SO$_4$•− and ·OH signals in the sludge. In summary, FeS$_2$ + PMS can generate SO$_4$•− and ·OH, in which SO$_4$•− plays a dominant role. Through these active free radicals, the EPS of the sludge is deoxidized, and the dewatering effect of the sludge is improved.

**Changes in the microstructure of WAS flocs by observation**

**Zeta potential analysis of sludge**

The charged characteristics of sludge flocs are characterized by measuring the zeta potential. The zeta potential of sludge flocs is usually between −30 and −10 mV (Horan and Eccles 1986). The zeta potential will affect the aggregation and sedimentation level of sludge particles. Generally speaking, the stronger
Fig. 6 Changes in sludge particle size: a cumulative percentage, b change in median diameter D50

Fig. 7 Mud cake structure changes under different sizes: a raw; b FeS₂ + PMS; c FeS₂ + PMS + BC; d FeS₂ + PMS + P-BC; e BC; f P-BC
the negative charge of the sludge, the more stable the flocs, the more difficult it is to dehydrate (Elakneswaran et al. 2009). Figure 5 indicates the changes in zeta potential over time. In order to control the influencing factors of the experiment, pH at which the zeta potential was kept at 6 throughout. The zeta potential of the original sludge was $-15.8 \text{ mV}$. After AOP treatment, the zeta potential rose to $-12.4 \text{ mV}$ within 5 min. However, over time, the best zeta potential of WAS after the AOP treatment remained at $-10.4 \text{ mV}$, which differs from previous results (Ai et al. 2021; Wang et al. 2021; Zhu et al. 2018). This situation might have resulted from the continuous release of negatively charged organic matter from internal sludge cells and flocs to the outer space of the sludge and the release of biopolymers with the same charge, which repel each other in the AOP. This condition further hindered the flow of water, aggravated the filterability of the sludge, and deteriorated the dehydration performance (Liu et al. 2001, Zhang et al. 2009). Moreover, using FeS$_2$ as the activator, the unsaturated S atoms on the surface can further improve the activation efficiency and accelerate the Fe$^{2+}$/Fe$^{3+}$ conversion (Chhowalla et al. 2013; Xing et al. 2018). This phenomenon can not only further improve the activation effect but also reduce the precipitation of ferric hydroxide sludge. However, given that the conversion of Fe$^{2+}$/Fe$^{3+}$ was accelerated, the Fe$^{3+}$ content in the filtrate decreased, which neutralized the negative charge on the floc surface, resulting in a certain effect. When AOP was combined with BC, the negative charge of zeta potential was further increased, possibly due to the oxidization of organic substances in BC and release of PN by sulfate radicals. Over time, analogs entered the filtrate, thereby reducing the zeta potential of the sludge (Wang et al. 2021).

When AOP was combined with P-BC, the negative charge decreased sharply after 5 min, and the zeta potential rose to $-4.3 \text{ mV}$ and finally stabilized at $0.4 \text{ mV}$ with the extension of the reaction time. Given that P-BC can sufficiently neutralize negatively charged sludge particles, the solubility was reduced, and the EPS aggregation increased, leading to the flocculation of colloidal sludge particles. These results also prove that AOP and P-BC have a very good synergistic effect in improving the sludge dewatering process.

### Particle size analysis of sludge

The particle size of the sludge is closely related to the dewatering capacity of the sludge (Shao et al. 2009). Figure 6 shows the changes in WAS particle size distribution before and after pretreatment. Figure 6a shows the cumulative percentage of sludge particles with a particle size in the range of 0–200 µm was investigated. It can be seen from the figure that compared with the original sludge, the cumulative percentage curve of the sludge particle size after FeS$_2$+PMS treatment obviously moves to the left, which proves that the sludge particle size after FeS$_2$+PMS conditioning is obviously smaller. To intuitively reflect the change process of particle size, this study selected the particle size change with a representative median diameter D(50). Figure 6b shows the change in D(50). The particle size of the sludge was significantly reduced after the AOP treatment, and the median diameter (D50) of the sludge was reduced from 55.37 µm of the original sludge to 42.46 µm. This finding indicates that the SO$_4$$^{2-}$ produced by AOP can effectively oxidize and decompose large-particle sludge flocs into small particles. However, the destruction of the floc structure during the oxidation process increased the surface area of the sludge and exposed more hydrophilic groups, thereby enhancing the affinity of the sludge colloid for water and further aggravating the dewatering of the sludge (Shi et al. 2016; Wang et al. 2021). Therefore, in...
the combined treatment of AOP and BC, the adsorption of BC can increase the particle size, and the median diameter (D50) increased from 42.46 to 47.13 µm. This finding is consistent with previous research results (Guo et al. 2019; Liu et al. 2017; Xiong et al. 2017). However, difficulty arises in the effective re-flocculation of the damaged floc structure by relying on the adsorption and framework support of conventional carbon-based framework additives. When AOP was combined with P-BC, the median diameter (D50) reached 64.56 µm. The reason is that P-BC cooperates with the adsorption capacity of BC itself, which can better play the role of charge neutralization and bridging induction. Thus, the combined treatment of AOP and P-BC can re-flocculate the floc structure after cracking the EPS and releasing the BW and further improve the dehydration efficiency.

**WAS microstructure analysis**

The samples were analyzed by SEM to understand the surface morphology of WAS before and after pretreatment. Figure 7a shows the SEM image of WAS before preprocessing. Before treatment, the WAS presented a dense and smooth surface structure, showing a compacted shape without remarkable voids or channels. Figure 7b shows the WAS after the AOP treatment. The internal structure changed from a complete plate-like structure to a broken block-like one, and the surface became rough, with evident cracks and cavities. This finding is similar to the above-mentioned change in particle size and caused by the EPS cracking and destruction of the floc structure resulting from oxidation. Figure 7c shows the microstructure after the combined treatment of AOP and BC. BC was distributed on the surface of WAS as a framework assistant, which can prevent fine sludge particles from clogging the filter pores during the suction filtration process and improve the dehydration efficiency. Figure 7d displays the WAS after the joint processing of AOP and P-BC. The figure also shows that the surface of the flocs appeared sparser and more porous after the cooperative treatment of WAS. Because of the negative charge nature of EPS itself, it is easy to agglomerate with P-BC through electrostatic interaction and bridging. The porous sludge morphology is the performance that P-BC makes the floc structure destroyed after AOP treatment re-aggregate (Cai et al. 2018). This result is consistent with the statement in the previous section. Figure 7d shows that compared with the AOP treatment alone, relatively more hydrophobic channels were observed on the sludge surface after the AOP and P-BC co-treatment. Before P-BC flocculation, the AOP pretreatment decomposed EPS and caused the release of BW into the liquid phase, resulting in a high degree of dispersion of sludge particles. Then, the cracked sludge flocs were easily bridged by P-BC and re-flocculated to form a more porous structure (Ai et al. 2021).

**Effect of AOP and P-BC on WAS organic matter under co-processing**

**Protein and polysaccharide contents of different EPS fractions**

EPS is mainly composed of PN and PS, which is considered to be one of the most important factors affecting dewaterability of sludge (Nouha et al. 2018; Sheng et al. 2010). According to the different binding strength of EPS and microbial cells, EPS is usually divided into three layers: S-EPS, LB-EPS, and TB-EPS. Figure 8 shows the concentration of PN and PSs in different EPS components. As shown in Fig. 8a, in S-EPS, LB-EPS, and TB-EPS, the PN concentrations of raw mud totaled 64.86, 49.72, and 407.13 mg/L, respectively. After the AOP treatment, the PN content of TB-EPS was significantly reduced to 126.47 mg/L, whereas that of S-EPS increased to 209.41 mg/L. The change in the LB-EPS content was relatively low, increasing to 64.74 mg/L. After the AOP treatment, the total amount of PN in the raw mud decreased from 521.71 mg/L to 400.62 mg/L. Correspondingly, in Fig. 8b, the trend of PS concentration changes was consistent with that of PN. This result indicates that the SO$_{4}^{2-}$ released by FeS$_{2}$-activated PMS has strong oxidizing properties, which can change the distribution of different EPS components and at the same time oxidatively degrade part of the soluble organic matter; thus, TB-EPS was degraded into dissolved PN and PSs (Yu et al. 2016) and released. In the supernatant, the contents of PN and PS in S-EPS and LB-EPS increased, whereas the total amount of PN and PS decreased. This finding is consistent with the previous work of Li et al. (Li et al. 2019; Zhen et al. 2012, 2013). When AOP was used in combination with BC, the change trend of PN and PS contents was consistent with the results of AOP alone. The overall contents of PN and PS increased slightly because BC itself contained PN and PS, which were released into the three EPS filtrates. The situation is similar in other studies of AOP combined with matrix additives (Guo et al. 2019; Wang et al. 2021).

Notably, after the AOP pretreatment, with the further addition of P-BC, the PN and PS contents of the three-layer EPS component decreased sharply, the total PN decreased...
to 200.01 mg/L, and the total PS decreased to 141.51 mg/L. This finding may be due to the PAC loaded on the surface of P-BC itself; this PAC acted as an inorganic flocculant, possessing strong charge neutralization and hydrophobic association, which can effectively compress the negatively charged EPS layer and interact closely with the PN components in the EPS (Yan et al. 2019). On the other hand, as a monomer form of aluminum hydroxy, PAC can coordinate with the PN and PS in EPS and change its secondary structure, thereby promoting the aggregation of PN and PS molecules (Cao et al. 2021). Therefore, P-BC can be combined with hydrophilic organic substances, such as PN and PS, in the supernatant, and large amounts of organic substances, such as PN and PS, are adsorbed on its surface, resulting in a sharp decrease in the PN and PS contents in the supernatant.

During the AOP, several cells are oxidized to break and release intracellular substances, such as PN, PSSs, and DNA. These substances enter the liquid phase to increase the viscosity of the filtrate and are usually negatively charged and hydrophilic, which are not conducive to flocc aggregation in the sludge and the separation of water molecules (E. Neyens et al. 2002). Therefore, the combination of AOP and P-BC can not only change the composition distribution of EPS but also drastically reduce organic matter, such as PN, in the liquid phase, which can effectively reduce the negative charge of the sludge and is conducive to the formation of sludge flocs. Such combination can also improve the hydrophobicity of the sludge to easily achieve the separation of water molecules. This condition is important for improving the filterability of the sludge while releasing the BW inside it.

**3D-EEM fluorescence analysis**

We applied the 3D-EEM method to further investigate the possible mechanisms during sludge conditioning, which is a semi-quantitative characterization of organic components in EPS (Sheng et al. 2010). Figure 9 reveals the typical EEM fluorescence spectra of EPS from raw sludge samples and sludge treated with FeS$_2$/PMS synergistic with P-BC. According to previous research (Li et al. 2016; Lv et al. 2019), the 3D-EEM response is usually divided into four major regions, namely, aromatic PN ($\lambda_{ex/em} = 200–250/280–380$ nm), tryptophan-like PN ($\lambda_{ex/em} = 250–400/280–380$ nm), HA-like fluorophores ($\lambda_{ex/em} = 250–400/>380$ nm), and fulvic acid (FA)-like fluorophores ($\lambda_{ex/em} = 200–250/>380$ nm). It can be seen from Fig. 9c that the high fluorescence intensity peaks detected in the raw mud–untreated TB-EPS were aromatic PN, consistent with the deterioration of raw sludge dewatering. According to previous studies, the two peaks with high fluorescence intensity are associated with PN-like substances (Chen et al. 2016; Jacquin et al. 2017). The main fluorescence peaks observed in EPS also had Ex/Em = 200–250/>380 nm FA-like fluorophores and 250–300/>380 nm HA-like fluorophores. It can be seen from Fig. 9f that the combined conditioning by AOP and P-BC, the peak intensity of TB-EPS was significantly weakened. Through the comparison between Fig. 9a, Fig. 9b, Fig. 9d, and Fig. 9e, it is found that the peak intensities of S-EPS and LB-EPS are obviously weakened after FeS$_2$ + PMS + P-BC treatment. The fluorescence peak intensities of LB-EPS and S-EPS increased because the pre-treated sludge TB-EPS was destroyed, and PN and PS organic materials flowed to S-EPS and LB-EPS. This also proves that SO$_4^{2-}$ can destroy EPS, allowing the innermost TB-EPS to release bound water and organic substances such as PN and PS.

**Decisive composition and structure linked to WAS dewaterability**

**XPS analysis of sludge**

We applied the XPS measurements to further investigate the chemical changes in C, N, and O on the surface during sludge conditioning. In Fig. 10a, C1s high-resolution XPS spectrum showed the raw sludge appearance of the C-containing species. In Fig. 10d, C1s high-resolution XPS spectrum showed the treated sludge appearance of the C-containing species. Corresponding to the typical peaks of aliphatic/ aromatic (C−C, C−H, and C=C) at 284.5 and 286.1 eV (Dai et al. 2017) and bound by C−O and C−N. Representation according to Fig. 10b, Fig. 10c and Fig. 10e, Fig. 10f. This peak appeared at the 532.3 eV peak in the O1s high-resolution spectrum and 399.6 and 400.8 eV peaks of the N1s high-resolution spectrum. The peak at 288.5 eV was linked to the O−C=O bond of carboxyl acids. The main components in the WAS matrix were PN and PSSs, with a small amount of DNA, lipids, etc. The peaks indicated by C−C, C−H, C−N, C=O, and O=C−NH− provided the direct evidence for the existence of PN (Zhu et al. 2012). Peaks related to O−H and C−O, the two most common functional groups in carbohydrates, reflected the presence of carbohydrates. The results in Fig. 8 show that the spectra of the three elements considerably differed after pretreatment. Compared with the original sludge, it is found that the peak intensities of C, N, and O of the sludge treated by FeS$_2$ + PMS + P-BC are obviously decreased. This shows that the content of organic substances such as protein and polysaccharide in the sludge treated by FeS$_2$ + PMS + P-BC is reduced. This is because FeS$_2$ + PMS can produce SO$_4^{2-}$− deoxidize PN and PS and other organic substances, and mineralize protein substances into organic substances with lower molecular weight. Liangliang et al. (Liangliang et al. 2019) also studied the hydrophilicity and hydrophobicity of sludge by Fe$^{2+}$-activated PMS. It has been proved that advanced oxidation can reduce the properties of proteinaceous organic matter and improve sludge dewatering performance.

© Springer
One more reason, according to the research by Zhang et al. (2021), PAC had high densities of positive charge and strong capacities of coordination and complexation, which intensified their abilities to bind with amino groups in proteins. On the surface of P-BC, a large amount of PAC is loaded on the surface of P-BC, which can be combined with

![Figure 10](image)

Fig. 10 XPS survey scans of the sludge samples: raw WAS (a) (b) (c); the WAS treated (d) (e) (f)
organic compounds such as PN and PS. Accordingly, the interaction of biopolymers (i.e., proteins and carbohydrates) with hydrogen bonds (such as –OH and –NH bonds) greatly affects the water binding properties. The treatment of sludge by FeS2 + PMS + P-BC can lead to the breaking of biopolymers and related hydrogen bonds, thereby improving the hydrophobicity of sludge.

**Economic analysis and environmental analysis of sludge**

**Analysis of calorific value of sludge cake**

A higher LHV means that the fuel has a better energy production effect. It can be seen from the table that the LHV and HHV of the sludge added with P-BC are significantly increased, which proves that FeS2 + PMS + P-BC can increase the calorific value of the sludge and add environmental value to the sludge (Table 4).

**Economic analysis**

A preliminary cost analysis was performed for this study. Assuming a sewage treatment plant with a treatment population of 100 k and a daily treatment capacity of 50 km³/day, the material costs involved are shown in Table 5.

The traditional Fenton technology needs to spend 0.0295/m³ for sludge treatment. According to the simulation data, the annual sludge treatment cost of the traditional Fenton technology is about 538,375 RMB, which is much higher than FeS2 + PMS + P-BC cost used by BC. Therefore, the combined conditioning process of FeS2 + PMS + P-BC has certain advantages in terms of cost. But the sludge treatment procedure of FeS2 + PMS + P-BC also has shortcomings and limitations. During the reaction between FeS2 and PMS, the impact of sulfide intermediates on the environment is still lacking in tracking analysis. The preparation process of P-BC is limited to the laboratory, and the consumption of BC is too high during the preparation process, which may also become a barrier to commercialization. Therefore, for the sludge treatment method in this study, overcoming the above-mentioned commercial obstacles, further reducing resource consumption, and dynamically tracking the

---

### Table 4 Change of calorific value before and after sludge conditioning

| Samples | Element mass percentage (%) | Calorific value (kcal/kg) |
|---------|-----------------------------|--------------------------|
|         | C  | H  | N  | S  | O  | HHV | LHV |
| RAW     | 20.11 | 3.48 | 3.39 | 0.37 | 23.03 | 1806.79 | 1045.04 |
| FeS2 + PMS + P-BC | 36.74 | 7.31 | 6.81 | 0.85 | 38.5 | 3790.08 | 3013.88 |

### Table 5 Economic analysis of the FeS2 + PMS + P-BC conditioning for sludge dewaterability

| Process release | Material | Price (RMB) | Total cost (RMB) |
|-----------------|----------|-------------|------------------|
| FeS2 + PMS + P-BC | FeS2 | —— | 262,000 |
|                  | PMS | 90,000 | |
|                  | Combustion cost and equipment consumption | 10,000 | |
|                  | PAC | 162,000 | |

PS: in FeS2 + PMS + P-BC, FeS2 = 1.2 g/g TSS, PMS = 0.2 g/g TSS, P-BC = 0.75 g/g TSS, PAC annual consumption 202.5 t
environmental effects of process treatment are the future research directions.

Conclusions

This study showed that FeS$_2$ + PMS combined with P-BC as a novel combination has a significant effect on improving the dehydration capacity of WAS. The following conclusions were drawn:

FeS$_2$ + PMS combined with P-BC can significantly improve the dewatering performance of sludge, especially its filterability. Under optimal conditions, Wc and SRF decreased from 92.3% and 2.36 × 10$^{13}$ m/kg to 67.4% and 9.89 × 10$^{11}$ m/kg, respectively. This is also shown in TW, FW, and BW which dropped from the original value of 12.63 g/g TSS, 8.46 g/g TSS, and 4.17 g/g TSS to 6.03 g/g TSS, 3.97 g/g TSS, and 2.06 g/g TSS, respectively. FeS$_2$ + PMS can produce SO$_4$$^•-$ and ·OH two free radicals. SO$_4$$^•-$ plays a major role: as an oxidizing species, it destroys extracellular polymeric substance, releasing the moisture inside.

As a modified BC, P-BC can function as a physical and chemical regulator simultaneously. When P-BC acts as a framework aid, it can play a framework support role in mud cake, improve the permeability of mud cake, and increase the dewatering capacity of the sludge. The results of zeta potential, sludge particle size, and floc structure showed that P-BC can also act as a chemical regulator, helping to re-flocculate the negatively charged components in the sludge floc and enhancing the sludge settling and filterability. FeS$_2$ + PMS + P-BC can destroy the structure of EPS and release bound water. The free radicals generated by FeS$_2$ + PMS + P-BC and the PAC supported by P-BC can change the structure of organic substances such as PN and PS. Prove this point with XPS. After FeS$_2$ + PMS + P-BC treatment, the peak intensities of functional groups related to PN and PS were significantly weakened.

FeS$_2$ + PMS + P-BC can increase the calorific value of the sludge and add environmental benefits, also able to consume waste straw and iron ore. Composite country’s carbon neutrality policies and waste utilization principles.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11356-022-21074-4.

Acknowledgements The authors wish to thank the Gongzhufu waste-water treatment plants for supply of samples for testing.

Author contributions Data analysis and the lead in writing the manuscript were performed by WG. LS contributed to the study conception and design. Data collection was performed by ZW. LX commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Funding This study was financially supported by the National Natural Science Foundation of China (No.21107041) and the National Science Foundation of Inner Mongolia (No. 2020MS05028).

Data availability All data generated or analyzed during this study are included in this published article.

Declarations

Ethics approval and consent to participate Not applicable.

Consent for publication Not applicable.

Competing interests The authors declare no competing interests.

References

Ai J, Wang Z, Dionysiou DD, Liu M, Deng Y, Tang M, Liao G, Hu A, Zhang W (2021) Understanding synergistic mechanisms of ferrous iron activated sulfite oxidation and organic polymer flocculation for enhancing wastewater sludge dewaterability. Water Res 189:116652

Alexopoulou C, Petala A, Frontistis Z, Drivas C, Kennaou S, Kondarides DI, Mantzavinos D (2019) Copper phosphide and persulfate salt: A novel catalytic system for the degradation of aqueous phase micro-contaminants. Appl Catal B 244:178–187

Cai MQ, Hu JQ, Wells G, Seo Y, Spinney R, Ho SH, Dionysiou DD, Su J, Xiao R, Wei Z (2018) Understanding Mechanisms of Synergy between Acidification and Ultrasound Treatments for Activated Sludge Dewatering: From Bench to Pilot-Scale Investigation. Environ Sci Technol 52(7):4313–4323

Cao B, Wang R, Zhang W, Wu H, Wang D (2019) Carbon-based materials reinforced waste activated sludge electro-dewatering for synchronous fuel treatment. Water Res 149:533–542

Cao B, Zhang T, Zhang W, Wang D (2021) Enhanced technology based for sewage sludge deep dewatering: A critical review. Water Res 189:116650

Chen Z, Zhang W, Wang D, Ma T, Bai R, Yu D (2016) Enhancement of waste activated sludge dewaterability using calcium peroxide pre-oxidation and chemical re-flocculation. Water Res 103:170–181

Chhowalla M, Shin HS, Eda G, Li LJ, Loh KP, Zhang H (2013) The chemistry of two-dimensional layered transition metal dichalcogenide nanosheets. Nat Chem 5(4):263–275

Dai X, Xu Y, Dong B (2017) Effect of the micron-sized silica particles (MSSP) on biogas conversion of sewage sludge. Water Res 115(may15):220

Duan J, Gregory J (2003) Coagulation by hydrolysing metal salts. Adv Colloid Interface Sci 100:475–502

Elakneswaran Y, Nawa T, Kurumisawa K (2009) Zeta potential study of paste blends with slag. Cement Concr Compos 31(1):72–76

Fang GD, Zhou DM, Dionysiou DD (2013) Superoxide mediated production of hydroxyl radicals by magemite nanoparticles: demonstration in the degradation of 2-chlorobiphenyl. J Hazard Mater 250:68–75

Guo S, Liang H, Bai L, Qu F, Ding A, Ji B, Wang X, Li G (2019) Synergistic effects of wheat straw powder and persulfate/Fe(II) on enhancing sludge dewaterability. Chemosphere 215:333–341
Horan NJ, Eccles CR (1986) Purification and characterization of extracellular polysaccharide from activated sludges. Water Res 20(11):1427–1432

Jacquin C, Lesage G, Traber J, Pronk W, Heran M (2017) Three-dimensional excitation and emission matrix fluorescence (3DEEM) for quick and pseudo-quantitative determination of protein- and humic-like substances in full-scale membrane bioreactor (MBR). Water Res 118:82–92

Jin B, Wilén B-M, Lant P (2004) Impacts of morphological, physical and chemical properties of sludge flocs on dewaterability of activated sludge. Chem Eng J 98(1–2):115–126

Li H, Jin Y, Nie Y (2009) Application of alkaline treatment for sludge decrement and humic acid recovery. Bioresour Technol 100(24):6278–6283

Li Y, Yuan X, Wu Z, Wang H, Xiao Z, Wu Y, Chen X, Zeng G (2016) Enhancing the sludge dewaterability by electrolysis/electrocoagulation combined with zero-valent iron activated persulfate process. Chem Eng J 303:636–645

Li H, Song L, Han B, Song H, Bai R, Li H, Wang Q, Lin Z, Hu W (2019) Highly efficient enhancement of municipal sludge dewaterability using persulfate activation with nZVI/HA. Water Sci Technol 79(7):1309–1315

Liang J, Zhang L, Zhou Y (2021) Pyrite assisted peroxymonosulfate sludge conditioning: Uncover triclosan transformation during treatment. J Hazard Mater 413:125368

Liangliang W, Ding J, Xue M et al. (2019) Adsorption mechanism of ZnO and CuO nanoparticles on two typical sludge EPS: Effect of nanoparticle diameter and fractional EPS polarity on binding. Chemosphere 214:210–219

Liu JC, Lee CH, Lai JY (2001) Extracellular polymers of ozonized waste activated sludge. Water Sci Technol 44(10):137–142

Liu H, Yang S, Shi J, Xu X, Liu H, Fu B (2016a) Towards understanding the dewatering mechanism of sewage sludge improved by bioleaching processing. Sep Purif Technol 79(7):1309–1315

Liu J, Yang Q, Wang D, Li X, Zhong Y, Li X, Deng Y, Wang L, Li K, Zeng G (2016b) Enhanced dewaterability of waste activated sludge by Fe(II)-activated peroxymonosulfate oxidation. Bioresour Technol 206:134–140

Liu H, Xiao H, Fu B, Liu H (2017) Feasibility of sludge deep-dewatering with sawdust conditioning for incineration disposal without energy input. Chem Eng J 313:655–662

Lv H, Liu D, Xing S, Wu D, Wang F, Yang J, Wu X, Zhang W, Dai X (2019) The effects of aging for improving wastewater sludge electro-dewatering performances. J Ind Eng Chem 80:647–655

MacDonald BA, Oakes KD, Adams M (2018) Molecular disruption through acid injection into waste activated sludge – A feasibility study to improve the economics of sludge dewatering. J Clean Prod 176:966–975

Maroukova J, Marouková A, Zoubek T, Barto P (2022) Economic impacts of soil fertility degradation by traces of iron from drinking water treatment. Environ Dev Sustain Multidiscip Approach Theory Practice Sustain Dev 24:1–10

Maryam A, Zeshan, Badshah M, Sabeeh M, Khan SJ (2021) Enhancing methane production from dewatered waste activated sludge through alkaline and photocatalytic pretreatment. Bioresour Technol 79(7):1309–1315

Min SK, Lee KM, Kim HE et al. (2016) Disintegration of waste activated sludge by thermally-activated persulfates for enhanced dewaterability. Environ Sci Technol 50(13):7106

Moharaki M, Semken R, Scott, Mikkola et al (2018) Enhanced sludge dewatering based on the application of high-power ultrasonic vibration. ULTRASONICS–LONDON THEN AMSTERDAM–Netherlands

Neyen E, Baeyens J, Weemaes M, Heyder BD (2002) Advanced Biosolids Treatment Using H2O2-Oxidation. Environ Eng Sci 19(1):27–35

Nouka, K. Kumar RS, Balasubramanian S, Tyagi RD (2018) Critical review of EPS production, synthesis and composition for sludge flocculation. J Environ Sci (china) 66:225–245

Raheman A, Sikarwar VS, He J, Dastyar W, Dionysiou DD, Wang W, Zhao M (2018) Opportunities and challenges in sustainable treatment and resource reuse of sewage sludge: A review. Chem Eng J 337:616–641

Rao B, Zhu Y, Yu M, Lu X, Wan Y, Huang G et al (2018) High-dry dewatering of sludge based on different pretreatment conditions. Process Saf Environ Prot 122:288–297

Saveyn H, Meersseman S, Thas O, Van der Meeren P (2005) Influence of polyelectrolyte characteristics on pressure-driven activated sludge dewatering. Colloids Surf, A 262(1–3):40–51

Shao L, Peipei HE, Guanghui YU, Pinjing HE (2009) Effect of proteins, polysaccharides, and particle sizes on sludge dewaterability. J Environ Sci English(1):6

Sheng GP, Yu HQ, Li XY (2010) Extracellular polymeric substances (EPS) of microbial aggregates in biological wastewater treatment systems: a review. Biotechnol Adv 28(6):882–894

Shi Y, Yang J, Liang S, Yu W, Xiao J, Song J, Xu X, Li Y, Yang C, Wu X, Hu J, Liu B, Hou H (2016) Principal component analysis on sewage sludge characteristics and its implication to dewatering performance with Fe2+/persulfate-skeleton builder conditioning. Int J Environ Sci Technol 13(9):2283–2292

Wang S, Yang YK, Chen XG, Lv JZ, Li J (2017) Effects of bamboo powder and rice husk powder conditioners on sludge dewatering and filtrate quality. Int Biodeterior Biodegradation 124:288–296

Wang J, Yang M, Liu R et al. (2019) Anaerobically-digested sludge conditioning by activated peroxymonosulfate: Significance of EDTA chelated-Fe2+. Water Res 160(SEP.1):454–465

Wang Q, Song L, Hui K, Song H (2021) Iron powder activated peroxymonosulfate combined with waste straw to improve sludge dewaterability. Environ Technol 42(8):1302–1311

Xiu W, Wang D, Ge X, Tang H (2008) Coagulation of silica microspheres with hydrolyzed Al(III)—Significance of Al13 and Al13 aggregates. Colloids Surf, A 330(1):72–79

Wu C, Jin L, Zhang P, Zhang G (2015) Effects of potassium ferrate oxidation on sludge disintegration, dewaterability and anaerobic biodegradation. Int Biodeterior Biodegradation 102:137–142

Wu B, Ni BJ, Horvat K, Song L, Chai X, Dai X, Mahajan D (2017) Occurrence State and Molecular Structure Analysis of Extracellular Proteins with Implications on the Dewaterability of Waste-Activated Sludge. Environ Sci Technol 51(16):9235–9243

Xiong M, Xu W, Dong C, Bai Y, Zeng J, Zhou Y, Zhang J, Yin Y (2018) Metal Sulfides as Excellent Co-catalysts for H2O2 Decomposition in Advanced Oxidation Processes. Chem 4(6):1359–1372

Xiong Q, Zhou M, Yang H, Liu M, Wang T, Dong Y, Hou H (2017) Improving the Dewaterability of Sewage Sludge Using Rice Husk and Fe2+ Sodium Persulfate Oxidation. ACS Sustain Chem Eng 6(1):872–881

Yan WL, Wang YL, Chen YJ (2013) Effect of conditioning by PAM polymers with different charges on the structural and characteristic evolutions of water treatment residuals. Water Res 47(17):6445–6456

Yan ZR, Meng HS, Yang XY, Zhu YY, Li XY, Xu J, Sheng GP (2019) Insights into the interactions between triclosan (TCS) and extracellular polymeric substance (EPS) of activated sludge. J Environ Manage 232:219–225

Yu W, Yang J, Shi Y, Song J, Shi Y, Xiao J, Li C, Xu X, He S, Liang S, Wu X, Hu J (2016) Roles of iron species and pH optimization on sewage sludge conditioning with Fenton’s reagent and lime. Water Res 95:124–133

Zhang G, Yang J, Liu H, Zhang J (2009) Sludge ozonation: disintegration, supernatant changes and mechanisms. Bioresour Technol 100(3):1505–1509
Zhang W, Chen Z, Cao B, Du Y, Wang C, Wang D, Ma T, Xia H (2017) Improvement of wastewater sludge dewatering performance using titanium salt coagulants (TSCs) in combination with magnetic nano-particles: Significance of titanium speciation. Water Res 110:102–111

Zhang WM, Tang D et al. (2021) Effects of alkalinity on interaction between EPS and hydroxy-aluminum with different speciation in wastewater sludge conditioning with aluminum based inorganic polymer flocculant. J Environ Sci 100(02):259–270

Zhen G, Lu X, Zhao Y, Chai X, Niu D (2012) Enhanced dewaterability of sewage sludge in the presence of Fe(II)-activated persulfate oxidation. Bioresour Technol 116:259–265

Zhen GY, Lu XQ, Li YY, Zhao YC (2013) Innovative combination of electrolysis and Fe(II)-activated persulfate oxidation for improving the dewaterability of waste activated sludge. Bioresour Technol 136:654–663

Zhou Y, Wang X, Zhu C, Dionysiou DD, Zhao G, Fang G et al (2018) New insight into the mechanism of peroxymonosulfate activation by sulfur-containing minerals: role of sulfur conversion in sulfate radical generation. Water Res 142(oct.1):208–216

Zhu L, Qi HY, Lv ML, Kong Y, Yu YW, Xu XY (2012) Component analysis of extracellular polymeric substances (EPS) during aerobic sludge granulation using FTIR and 3D-EEM technologies. Bioresour Technol 124:455–459

Zhu X, Yang Q, Li X, Zhong Y, Wu Y, Hou L, Wei J, Zhang W, Liu Y, Chen C, Wang D (2018) Enhanced dewaterability of waste activated sludge with Fe(II)-activated hypochlorite treatment. Environ Sci Pollut Res Int 25(27):27628–27638

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