Sludge reduction by direct addition of chlorine dioxide into a sequencing batch reactor under operational mode of repeatedly alternating aeration/non-aeration

Hong Peng, Weiyi Liu, Yuanmei Li and Hong Xiao

ABSTRACT

The effect of direct addition of chlorine dioxide (ClO2) into a repeatedly alternating aeration/non-aeration sequencing batch reactor (SBR) on its sludge reduction and process performance was investigated. The experimental results showed that the sludge reduction efficiency was 32.9% and the observed growth yield (Yobs) of SBR was 0.11 kg VSS/kg COD (chemical oxygen demand) for 80 days’ operation at the optimum ClO2 dosage of 2.0 mg/g TSS (total suspended solids). It was speculated that cell lysis and cryptic growth, uncoupled metabolism and endogenous metabolism were jointly responsible for the sludge reduction in this study. COD, NH3-N, total nitrogen (TN) and total phosphorus (TP) in the effluent increased on average 29.47, 4.44, 1.97 and 0.05 mg/L, respectively. However, the effluent quality still satisfied the first-class B discharge standards for municipal wastewater treatment plants in China. In that case, the sludge maintained fine viability with the specific oxygen uptake rate (SOUR) being 14.47 mg O2/(g VSS·h) and demonstrated good settleability with the sludge volume index (SVI) being 116 mL/g. The extra cost of sludge reduction at the optimum ClO2 dosage was estimated to be 2.24 CNY (or 0.36 dollar)/kg dry sludge.

Key words | alternating aeration/non-aeration, chlorine dioxide, SBR, sludge reduction

INTRODUCTION

According to statistics from the Ministry of Housing and Urban-Rural Development of the People’s Republic of China, 3,340 municipal wastewater treatment plants (WWTPs) had been put into operation nationwide by 2012, the total capacity of which reached 142 million cubic metres per day. Given excess sludge (ES) production accounts for 0.4‰ of the mass of treated municipal wastewater, the corresponding daily ES production is 56.8 thousand tons per day. Herein, the ES produced from municipal WWTPs using the sequencing batch reactor (SBR) technology occupies a large proportion. Large amounts of non-stably disposed ES pose environmental risks and hence require safe disposal (Lin et al. 2012).

The treatment and ultimate disposal of ES is expensive and usually represents 50–60% of the operating costs of WWTPs (Campos et al. 2009). The approaches to treat or dispose of ES can be divided into two categories: (1) post-treatment, including anaerobic digestion, sludge composting, land application, landfill, incineration, etc.; and (2) in-situ reduction, i.e. applying other strategies in the water line to reduce sludge production (Yang et al. 2011). Of the two categories of approaches, the latter has attracted increasing attention.

The mechanisms for sludge in-situ reduction techniques are commonly classified into five groups: cell lysis and cryptic growth, uncoupled metabolism, endogenous metabolism, microbial predation and hydrothermal oxidation (Wei et al. 2003). In recent years, the first two mechanisms have been intensively studied. For cell lysis and cryptic growth, cell lysis plays a critical role. Cell lysis could be realized by various methods such as enzymatic hydrolysis (Foladori et al. 2010), ultrasound (Zhang et al. 2007; Zhang et al. 2009), ozonation (Chu et al. 2009), Fenton oxidation (He & Wei 2010), chlorination (Fazelipour et al. 2014) and chlorine dioxide (ClO2) oxidation (Wang et al. 2011; Zuriaga-Agustí et al. 2012). From the engineering application point of view, ClO2 oxidation exhibits more outstanding performance in comparison with other methods due to its good economical feasibility, simple operation or minimal formation of harmful chlorinated organic compounds. As regards uncoupled metabolism, addition of chemical uncouplers was a
common method (Low et al. 2000; Ye et al. 2003; Ye & Li 2010). Although good sludge reduction performance was achieved, the biggest drawback of uncouplers lies in their toxicity or non-biodegradability, which may cause secondary environmental pollution. Fortunately, uncoupled metabolism may also be carried out when microorganisms are subjected to a physiological shock created by lack of oxygen and substrate (Chudoba et al. 1992).

In the present work, ClO2 was directly dosed into SBRs to achieve cell lysis. An extraordinary aeration strategy of repeated coupling of aeration and non-aeration was adopted so as to foster uncoupled metabolism. Cell lysis and uncoupled metabolism were hereby coupled for sludge reduction. To our best knowledge, this new sludge reduction method has never been reported. In our experiment, besides the sludge reduction efficiency, the effluent quality (considering effluent chemical oxygen demand (COD), NH3-N, total nitrogen (TN) and total phosphorus (TP)) and the sludge settling property were comprehensively investigated.

**MATERIALS AND METHODS**

**Laboratory equipment**

Five identical laboratory SBRs were employed. The total ($V_T$) and the reaction ($V_R$) volumes of each SBR were 2 L and 1.6 L, respectively. The aeration system comprised an air compressor (ACO-008, Resun Group, Guangdong, China), a gas flow meter (LZB-6, Zhenxing Flow Instrument Factory, Jiansu, China) and a rubber-film disc diffuser (Benyuan Environmental Protection Equipment Co., Ltd, Jiansu, China). Aeration was controlled by a programmable-logic-controller (PLC) with the aeration strength being adjusted by the gas flow meter. Dissolved oxygen (DO) was maintained at approximate 3.0 mg/L during aeration phases. Mixing was provided by a mechanical stirrer (JJ-6, Kemai Experimental Equipment Co., Ltd, Jiansu, China) run at 120 rpm. The influent was directly poured into SBR. The effluent drainage and ES withdrawal were carried out by a self-made siphoning installation.

**Simulated wastewater and chlorine dioxide**

The feed solution consisted of 350 mg/L of anhydrous dextrose, 165 mg/L of ammonium sulfate and 45 mg/L of trisodium phosphate 12-hydrate. Chemicals were dissolved in tap water. COD, NH3-N, TN and TP of the simulated wastewater were 349.33 ± 7.02, 35.23 ± 0.47, 35.50 ± 0.53 and 3.49 ± 0.04 mg/L, respectively. 3,000 mg/L of ClO2 solution was prepared through mixing sodium chloride and sodium bisulphate. The chemical reaction took place as in Equation (1):

$$\text{2NaClO}_2 + \text{Na}_2\text{S}_2\text{O}_8 \rightarrow 2\text{ClO}_2 + 2\text{Na}_2\text{SO}_4$$

Different volumes from these solutions were added to establish the specified ClO2 dosages in SBRs. These dosages were referred to the total suspended solids (TSS) concentration. Dosages of 1.0, 1.5, 2.0 and 2.5 mg ClO2/g TSS were adopted in this study.

**Experimental procedure**

Five SBRs were operated in parallel. Four SBRs were used as test SBRs (with ClO2 addition) and one SBR was used as the control SBR (with no ClO2 addition). All experiments were conducted at room temperature (27 ± 2°C). Three operational periods were run per day for each SBR. The detailed operational strategy is illustrated in Figure 1.

In all cases, the oxidant volume to be dosed was previously calculated according to the measured TSS. All SBRs were seeded with activated sludge taken from a municipal WWTP in Chengdu, China. The acclimation period to the simulated wastewater lasted 15 days. Subsequent experiments started with the same TSS concentration of 3.0 g/L and lasted 80 days for all SBRs. The volume exchange ratio (VER) is defined.
as the quotient between the fill volume and the reaction volume. The hydraulic retention time (HRT) is defined for SBR as in continuous flow activated sludge systems (VR/Q). In this work, VER and HRT of all SBRs were 1/2 and 16 h, respectively.

ES was withdrawn once every 4 days during 80 days of operation with TSS maintained at a constant concentration of 3.0 g/L.

Calculations

\[ Y_{obs} \text{of an SBR system can be determined as in Equation (2) (Ma et al. 2012):} \]

\[ Y_{obs} = \frac{1000 \times W}{Q \times (COD_{in} - COD_{eff})} \]

where \( W \) is the excess activated sludge discharged from the system, kg VSS/d; \( Q \) is the flow-rate of wastewater treated by the SBR system, m³/d; \( COD_{in} \) and \( COD_{eff} \) are the COD concentrations of influent and effluent of the SBR system, kg/m³.

SRT (d) of a SBR can be defined as the total biomass in the system divided by the mass rate of biomass withdrawn from the system, as in Equation (3):

\[ SRT = \frac{X \times V_R}{\Delta X/\Delta t} \]

where \( X \) is VSS concentration in SBR, mg/L; \( V_R \) is the working volume of SBR, L; \( \Delta X/\Delta t \) is the mass of VSS withdrawn from SBR per day, mg/d.

The sludge reduction efficiency (\( R_{sludge} \)) can be calculated as in Equation (4):

\[ R_{sludge} = \frac{\text{sludge production in control SBR} - \text{sludge production in test SBR}}{\text{sludge production in control SBR} \times 100\%} \]

The biomass reduction efficiency (\( R_{biomass} \)) can be calculated as in Equation (5):

\[ R_{biomass} = \frac{Y_{obs} \text{ of control SBR} - Y_{obs} \text{ of test SBR}}{Y_{obs} \text{ of control SBR} \times 100\%} \]

The removal efficiency (\( R \)) of pollutants can be calculated as in Equation (6):

\[ R = \frac{C_i - C_e}{C_i \times 100\%} \]

where \( C_i \) and \( C_e \) (mg/L) are the pollutants’ concentrations in influent and effluent, respectively.

Analysis

COD, VSS, TSS, TN, NH₃-N and TP were measured according to Standard Methods (APHA, AWWA and WEF 2005). DO was continuously monitored by WTW, pH/oxi340i meter with DO probes (WTW Company, Germany). Protein content was determined using Lowry’s method and polysaccharide content was analyzed by the Anthrone method (Raunkjær et al. 1994). Trihalomethanes were determined according to the method described by Zuriaga-Agustí et al. (2012).

Viability assessment

Viability of the sludge samples with and without ClO₂ addition was assessed by oxygen uptake rate (OUR) measurements. The OUR was determined during respirometric experiments as described by Spanjers et al. (1996). To make comparison between the results from different respirometric tests, the specific oxygen uptake rate (SOUR) was calculated as in Equation (7):

\[ SOUR \text{ (mgO}_2/\text{(g VSS \cdot h)}) = \frac{OUR}{X} \]

where \( X \) is VSS concentration of the mixed liquid in the respirometric test (g/L).

RESULTS AND DISCUSSION

Excess sludge reduction performance

Four test SBRs were denoted as SBR(1.0), SBR(1.5), SBR(2.0) and SBR(2.5), respectively. The control SBR was denoted as SBR(0.0). The number in the parentheses next to SBR indicates the dosage (with unit of mg ClO₂/g TSS) of ClO₂ added into a SBR. Figure 2 illustrates the mass of sludge withdrawn from five SBRs.

As is shown in Figure 2, the mass of sludge withdrawn from SBR(1.0) was nearly the same as that from SBR(0.0). Increasing ClO₂ dosage from 1.0 to 1.5 mg ClO₂/g TSS, the accumulative mass of withdrawn sludge from SBR(1.5) was 19.7% less than that from SBR(0.0) for the duration of 80 days. As the ClO₂ dosage further increased, the accumulative mass of withdrawn sludge continued to reduce. Compared with SBR(0.0), the sludge reduction efficiencies of SBR(2.0)
and SBR(2.5) reached 32.9% and 40.6%, respectively. Taking the effluent quality into account, the optimal ClO2 dosage for sludge reduction in this study was 2.0 mg ClO2/g TSS, which was much lower than that reported by Wang et al. (2011) based on similar sludge reduction efficiency. In their study, a ClO2 dosage of 10 mg ClO2/g dry sludge was used to reduce 36% of the waste activated sludge with an initial concentration of 15 g TSS/L in a ClO2 oxidation reactor. However, it is not difficult to explain this difference of ClO2 dosage in both studies. Basically, the sludge disintegration rate depends largely on the contact between the ClO2 molecules and sludge cells. The stirring by a mechanical stirrer was operated in our study but not in theirs. Mechanical stirring accompanying the dosing of ClO2 in our study enhanced the contact effect, which resulted in a lower ClO2 dosage. The fact that a relative low ClO2 dosage can disintegrate sludge effectively was also proved by Zuriaga-Agustí et al. (2012), who achieved a 20.2% reduction in the sludge production at a dosage of 2.0 mg ClO2/g TSS.

In fact, biomass reduction is the core of sludge reduction. The diminution degree in $Y_{obs}$ can better reflect the reduction efficiency in biomass production. The $Y_{obs}$ calculation results are shown in Figure 3.

Figure 3 shows that the $Y_{obs}$ decreased as the ClO2 dosage increased. In this study, the $Y_{obs}$ values of SBR(0.0) and SBR(2.0) were 0.17 and 0.11, respectively. According to Equation (6), the biomass reduction efficiency was 35.3%.

In order to compare the sludge reduction efficiency in our study with that in other literature, the observed sludge yields obtained are presented in Table 1.

In Table 1, the $Y_{obs}$ values ranged from 0.11 to 0.329, which were lower or far lower than the $Y_{obs}$ (0.4–0.7 kg VSS/kg COD) of a conventional activated sludge (CAS) process (Tchobanoglous et al. 2005). The $Y_{obs}$ of 0.11 kg VSS/kg COD for SBR (2.0) in this study is the lowest one. This result can be explained from two aspects. Firstly, a long SRT (43.9 d) of SBR(2.0) enhanced the endogenous metabolism of microorganisms, and therefore led to less sludge production. Secondly, cell lysis-cryptic growth also contributed to decrease in the biomass yield. As shown in Figure 4, the proteins and polysaccharide in the effluent of SBR(2.0) were $2.12 \pm 0.07$ and $12.2 \pm 0.58$ mg/L, respectively, whereas those in the effluent of SBR(0.0) were only $0.47 \pm 0.03$ and $4.65 \pm 0.08$ mg/L. Such a result indicated that cell lysis happened in SBR(2.0).

It is worth noting that a quite low $Y_{obs}$ (0.17 kg VSS/kg COD) was obtained in SBR(0.0). Besides the endogenous metabolism caused by a relatively long SRT (29.4 d), uncoupled metabolism was supposed to work in this case. As pointed out by Chudoba et al. (1992), uncoupled metabolism may be triggered by subjecting microorganisms to a physiological shock created by lack of oxygen and substrate. In their work, an oxic–settling–anaerobic (OSA) process, which is a modification of a CAS process that inserts an anaerobic side-stream reactor within the returned activated sludge circuit after starvation conditions (setting tank), may reduce ES production by 40–50%. Otherwise, repeatedly coupling of the aerobic and anaerobic (rCAA) process was demonstrated to have good performance in sludge reduction (Yu et al. 2006). Inspired by the ideas of OSA and rCAA processes, it is speculated that SBR characterized by sequencing...
batch feeding and repeated coupling of the aeration and non-aeration may also fulfill uncoupled metabolism effectively.

In summary, cell lysis and cryptic growth, uncoupled metabolism and endogenous metabolism were jointly responsible for sludge reduction by direct dosing of ClO₂ into a SBR under the operational mode of repeatedly alternating aeration/non-aeration.

**Effluent quality**

One-sided pursuit of high sludge reduction efficiency is unwise. Attention should also be paid to the effluent quality when evaluating a sludge reduction method. To investigate the effect of ClO₂ dosing on the effluent quality of SBR, COD, NH₃-N, TN and TP in effluent were monitored every other day for 80 days’ operation. The data are presented in Figure 5.

Figure 5(a) shows the effect of ClO₂ dosage on the effluent COD concentration and COD removal efficiency. When the influent COD was 349.33 ± 7.02 mg/L, the effluent COD

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**Table 1** Comparison of $Y_{obs}$ from other literature and this study

<table>
<thead>
<tr>
<th>Process</th>
<th>SRT (d)</th>
<th>$Y_{obs}$ (g VSS/g COD)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Modified-SBR + sludge holding tank</td>
<td>100</td>
<td>0.14</td>
<td>Datta et al. (2009)</td>
</tr>
<tr>
<td>CAS system + (lysis – cryptic growth)</td>
<td>–</td>
<td>0.27</td>
<td>Ma et al. (2012)</td>
</tr>
<tr>
<td>SBR + high-pressure-homogenization</td>
<td>–</td>
<td>0.308</td>
<td>Lan et al. (2013)</td>
</tr>
<tr>
<td>BIMINEX</td>
<td>17.4</td>
<td>0.329</td>
<td>Coma et al. (2013)</td>
</tr>
<tr>
<td>SBR(2.0)</td>
<td>43.9</td>
<td>0.11</td>
<td>This study</td>
</tr>
<tr>
<td>SBR(0.0)</td>
<td>29.4</td>
<td>0.17</td>
<td>This study</td>
</tr>
</tbody>
</table>

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**Figure 4** Polysaccharide and protein concentrations in the effluent at different dosages of ClO₂. Data were the mean values obtained from SBRs with different dosages of ClO₂. Error bars represented standard deviations of statistical analysis. $n$ meant measurement times.
of SBR(0.0) and SBR(1.0) were 24.30 ± 5.23 and 25.90 ± 5.55 mg/L, respectively. Meanwhile, the COD removal efficiencies of SBR(0.0) and SBR(1.0) were 93.1 ± 1.4% and 92.6 ± 1.8%, respectively. The difference between them was minimal. It is supposed that when the ClO2 dosage was 1.0 mg/g TSS, ClO2 only destroyed some zooglea structures by oxidizing the EPSs (extracellular polymeric substances) and bridging material. The cell cytoderm was hardly destroyed and the intracellular material was hardly released. The small difference between SBR(0.0) and SBR(1.0) in terms of polysaccharide and proteins concentrations in the effluent (as shown in Figure 4) seemed to support this supposition. Another supposition was that ClO2 dosage as low as 1.0 mg/g TSS probably had a minor impact on the heterotrophic bacteria responsible for COD degradation. As the ClO2 dosage increased from 1.5 to 2.5 mg ClO2 /g TSS, the effluent COD concentration increased evidently. For SBR(2.0), the effluent COD increased to 53.77 ± 2.61 mg/L, however, it still met the first-class B discharge standards for WWTPs in China (<60 mg/L). As regards SBR(2.5), the effluent COD increased to 69.17 ± 3.21 mg/L, which exceeded the COD limit of 60 mg/L. Therefore, in order to ensure the effluent quality, the dosage of ClO2 should not exceed 2.0 mg ClO2/g TSS.

As is shown in Figure 5(b) and 5(c), the effect of ClO2 dosage on NH3-N and TN removal performance exhibited a similar tendency. The influent concentrations of NH3-N and TN were 35.23 ± 0.47 and 35.50 ± 0.53 mg/L, respectively. When the ClO2 dosage was 1.0 mg/g TSS, the effluent NH3-N and TN in SBR(1.0) were 4.47 ± 0.25 and 13.30 ± 0.44 mg/L, respectively; while those in SBR(0.0) were 2.17 ± 0.31 and 11.93 ± 0.31 mg/L, respectively. Obviously, the removal efficiency of NH3-N and TN was influenced more while that of COD was influenced less under this ClO2 dosage. This result indicated that nitrifiers
and denitrifying bacteria may be more sensitive than heterotrophic bacteria. As the ClO2 dosage increased, the effluent concentrations of NH3-N and TN in test SBRs became higher. Taking the ClO2 dosage of 2.0 mg/g TSS for example, the effluent NH3-N and TN in SBR(2.0) were 6.61 ± 0.26 and 13.90 ± 0.35 mg/L, respectively; however, they both still complied with the first-class B discharge standards for WWTPs in China (NH3-N <15 and TN <20 mg/L). As the ClO2 dosage was further increased to 2.5 mg/g TSS, the corresponding NH3-N and TN removal efficiencies descended to 54.1 ± 2.9% and 48.0 ± 0.9%, respectively. Although the effluent TN (18.43 ± 0.49 mg/L) still met its corresponding limit of 20 mg/L, the effluent NH3-N (16.17 ± 1.15 mg/L) exceeded its limit of 15 mg/L. It is obviously shown in Figure 4(b) and 4(c), when the ClO2 dosage was increased from 2.0 to 2.5 mg/g TSS, the effluent NH3-N and TN sharply increased; meanwhile, the removal efficiencies of NH3-N and TN steeply decreased. The reasons may lie in: firstly, the ClO2 dosage of 2.5 mg/g TSS destroyed the cell cytoderm effectively and released more intracellular nitrogenous material. Secondly, such a ClO2 dosage caused significant harm to bacteria responsible for nitrification and denitrification. Therefore, from the perspective of nitrogen removal, the dosage of ClO2 should not exceed 2.0 mg/g TSS, which is in agreement with the results considering COD removal performance.

Figure 5(d) shows the effect of ClO2 dosage on the effluent TP concentration and TP removal efficiency. When the influent TP was 3.49 ± 0.04 mg/L, the effluent TP in the control SBR was 0.27 ± 0.02 mg/L. For SBR(1.0), SBR(1.5), SBR(2.0) and SBR(2.5), the effluent TP concentrations were 0.28 ± 0.09, 0.29 ± 0.03, 0.32 ± 0.05, 0.36 ± 0.04 mg/L, respectively. Comparing SBR(2.0) with SBR(0.0), the average TP concentration increased by 0.05 mg/L (18.5%) and the average TP removal efficiency declined by 1.2% at an ES reduction rate of 32.9%. An increase of phosphorus concentration was unavoidable because phosphorus was mainly removed by bioabsorption and bioaccumulation in excess sludge. Wang et al. (2011) reported a 42.3% increase of effluent phosphorus in the lysis–cryptic growth system at an ES reduction rate of 55%, while Lin et al. (2012) observed a 43% increase of effluent phosphorus when the ES was reduced by 58%. In this study, despite the adverse effect of ClO2 on TP removal, the effluent TP concentration satisfied the first-class B discharge standards for WWTPs in China (<1.0 mg/L).

It has to be highlighted that trihalomethane (THM) concentration in the effluents of test SBRs was always below the detection limit (1 × 10^-4 mg/L), which confirmed the lack of THM formation reported in the literature (Li et al. 1996; Zuriaga-Agustí et al. 2012). Therefore, the ClO2 used in this study was a safe and environment-friendly chemical reagent.

**Viability of sludge**

The SOUR values of the sludge sampled from SBR(0.0), SBR(1.0), SBR(1.5), SBR(2.0) and SBR(2.5) were determined as 16.23, 16.15, 15.80, 14.47, 9.46 mg O2/(g VSS·h), respectively; it could be concluded that ClO2 dosages lower than 1.5 mg/g TSS did little harm to the viability of sludge. In fact, the SOUR value of sludge sampled from SBR(2.0) was still higher than 10 mg O2/(g VSS·h), indicating fairly good biological viability (Henze 2002). These results were in good agreement with the COD and NH3-N removal performance of SBR(1.0), SBR(1.5) and SBR(2.0). As ClO2 dosage increased to 2.5 mg O2/(g VSS·h), the SOUR of the sludge decreased sharply, suggesting that the viability of the sludge suffered serious damage. In that case, the sludge cells were lysed to release intracellular material; however, cryptic growth was hindered. As a result, the COD and NH3-N removal performance slipped dramatically.

**Sludge settling properties**

Despite the effluent quality, the sludge’s settling properties should also be concerned. Whether certain sludge reduction technologies would do harm to sludge settleability aroused much controversy. Negative and positive conclusions were made by different researchers (Deleris et al. 2002; Caravelli et al. 2006; Novak et al. 2007; Yan et al. 2013). The sludge volume index (SVI) is an important parameter to evaluate the settleability of the activated sludge. In the present study, the seed sludge came from a secondary settling tank in a municipal WWTP, the SVI of which was 106 mL/g. After 15 days’ acclimation of the simulated wastewater, the sludge’s SVI changed to 125 mL/g. The initial SVI of sludge in the test SBRs and the control SBR was the same. After 80 days of operation period, the sludge SVI became 134, 130, 121, 116 and 75 mL/g for SBR(0.0), SBR(1.0), SBR(1.5), SBR(2.0) and SBR(2.5), respectively. The decrease of the sludge SVI upon ClO2 addition may be explained as follows: firstly, ClO2 destroyed some zoogela structures and produced smaller floc; which became more compact; secondly, ClO2 oxidation promoted the mineralization degree of sludge flocs. Although a lower SVI implies a better settleability, it doesn’t necessarily mean a better viability. In contrast, a very low SVI caused by a high level of inorganic matter content usually exhibits poor sludge...
viability. Commonly, SVI between 100 to 150 mL/g is reasonable. Therefore, from the perspectives of sludge settleability and viability, ClO₂ dosage should not exceed 2.0 mg/g TSS.

Cost estimation

Taking ES reduction efficiency and effluent quality into comprehensive consideration, the optimal dosage of ClO₂ was determined as 2.0 mg/g TSS. In relation to SBR(0.0), SBR(2.0) required extra cost, which primarily came from chemical agent consumption. During 80 days’ operation, 6,305.0 and 11,747.5 mg sludge were drained from SBR (2.0) and SBR(0.0), respectively. Meanwhile, 768.0 mg ClO₂ was consumed in SBR(2.0). Therefore, to achieve 1 kg sludge reduction, 0.14 kg of ClO₂ was needed. According to Equation (1), 1.67 kg sodium chlorite (purity of 80%) and 1.80 kg sodium bisulfate (purity of 98%) are needed to prepare 1 kg ClO₂. The average domestic market prices of sodium chlorite (purity of 80%) and sodium bisulfate (purity of 98%) are 3,900 and 5,400 CNY/ton, respectively. Thus, the ClO₂ generation cost is 16.0 CNY/kg (1 CNY = 0.16 dollar). Based on the above calculations, the extra cost for sludge reduction in SBR(2.0) can be determined as 2.24 CNY (or 0.36 dollar)/kg dry sludge. The cost was slightly higher than the 2.18 CNY/kg dry sludge reported by Lin et al. (2012), who use the method of ClO₂-ultrasonication disruption to achieve sludge reduction. However, the method of direct dosing ClO₂ into a SBR is more preferable due to its lower install investment and simpler operation. In Guangdong, China, the conventional treatment of excess sludge costs up to 2.60 CNY/kg dry sludge, including dewatering, transportation and ultimate disposal, but not counting environmental risk (Lin et al. 2012). Therefore, the method in this study is economically feasible in sludge reduction.

CONCLUSIONS

Sludge reduction by direct dosing ClO₂ into a SBR under operational mode of repeatedly alternating aeration/non-aeration was proved to be technically and economically feasible. At an optimal dosage of 2.0 mg ClO₂/g TSS, the Y₁obs of SBR(2.0) was 0.11 kg VSS/kg COD and the sludge reduction efficiency was 32.9% comparing SBR(2.0) with SBR(0.0), meanwhile, the effluent quality could meet the first-class B discharge standards for municipal wastewater treatment plants in China during 80 days’ operation.

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