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Effects of sludge retention time on oxic-settling-anoxic process performance: Biosolids reduction and dewatering properties

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HIGHLIGHTS

- Biosolids reduction in OSA process was evaluated using real wastewater.
- Optimum SRT improved volatile solids destruction in the external anoxic reactors.
- OSA achieved >35% sludge reduction in main aerobic tank at optimum SRT (20 d).
- Further increasing SRT over 20 d did not reduce sludge yield.
- OSA improved sludge dewaterability (lower CST and higher cake solids content).

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ABSTRACT

In this study, the effect of sludge retention time (SRT) on oxic-settling-anoxic (OSA) process was determined using a sequencing batch reactor (SBR) attached to external aerobic/anoxic reactors. The SRT of the external reactors was varied from 10 to 40 d. Increasing SRT from 10 to 20 d enhanced volatile solids destruction in the external anoxic reactor as evidenced by the release of nutrients, however, increasing the SRT to 40 d did not enhance volatile solids destruction further. Relatively short SRT (10–20 d) favoured the conversion of destroyed solids into inert products. The application of an intermediate SRT (20 d) of the external reactor showed the highest sludge reduction performance (>35%). Moreover, at the optimum SRT, OSA improved sludge dewaterability as demonstrated by lower capillary suction time and higher dewatered cake solids content.

1. Introduction

Conventional activated sludge (CAS) is the most widely used process for wastewater treatment. However, CAS produces a large amount of sludge that is inherently difficult to stabilise and dewater (Mowla et al., 2013). Sludge management is an expensive exercise and excessive sludge production can result in high operating cost in wastewater treatment plants (WWTPs). Therefore, it is desirable to reduce sludge production in order to minimise the costs associated with downstream processing of sludge (*e.g.*, dewatering and digestion). Various methods have been employed to control sludge production. They include the manipulation of operation conditions such as dissolved oxygen (DO) and sludge retention time (SRT) of the aeration tank, the addition of chemicals to minimise biomass growth, and the use of advance oxidation processes to destroy biomass. Some of these methods require significant capital investment and operating cost and/or only result in a marginal biosolids reduction (Foladori et al., 2010). A promising alternative is the oxic-settling-anoxic (OSA) process, which modifies CAS by placing external anoxic reactor/s in the return activated sludge (RAS) loop. OSA allows RAS to be partially biodegraded in the external reactor, which has low DO and substrate concentration, before it is returned to the aeration tank. The interchange of sludge between conditions that are rich (the aeration tank) and deficient (the external anoxic reactor/s) in oxygen and substrate results in net excess sludge reduction. The appeal of OSA is in its simple configuration, which can be readily set up in existing or new plants with minimal capital and operating cost (Semblante et al., 2014).

Despite its potential, the wide-scale use of OSA is hindered by inconsistent performance which is evident in the literature. Laboratory-scale OSA operated using synthetic wastewater reportedly achieves over 40% sludge reduction (Chon et al., 2011b; Saby

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et al., 2003). However, these high sludge reduction values are rarely observed in pilot- or full-scale systems or when real sewage is used as the feed (Coma et al., 2013). This is probably because of poor operational control stemming from knowledge gaps about the mechanisms governing sludge reduction (Khursheed et al., 2015). It is only recently that laboratory-scale studies using domestic sewage demonstrated the key steps occurring in OSA. For example, Semblante et al. (2015) showed that OSA causes destruction of volatile solids in the external anoxic reactor/s as well as a decline in the sludge yield (*i.e.*, mass of biomass produced per mass of substrate consumed) of the main bioreactor.

Previous research suggests that sludge reduction by OSA may be mainly due to its long SRT. The addition of external reactor/s that temporarily hold RAS results in an increase of the total SRT of activated sludge (Semblante et al., 2014). SRT is inversely proportional to sludge yield due to the diversion of energy towards cell maintenance rather than synthesis (Liu and Tay, 2001). However, contradicting reports have been reported regarding the relationship of SRT and OSA performance. For example, Saby et al. (2003) observed that biosolids reduction (23–58%) was directly proportional to the SRT of the external anoxic reactor of OSA (11–17 d). On the contrary, Ye et al. (2008) found that biosolids reduction (14–33%) had an inverse relationship with the SRT of the external anoxic reactor, although the range of SRTs investigated was much shorter (5.5–11.5 h) than that of Saby et al. (2003). These studies were conducted with synthetic wastewater, and furthermore the SRTs reported were scattered, ranging from very short (*e.g.*, less than 1 day) (Ye et al., 2008) to significantly long (*e.g.*, 70–80 d) (Novak et al., 2007). It is difficult to establish a correlation between SRT and OSA performance based on the available literature, especially since the reports are based on varying wastewater, operation conditions, and methods of quantifying sludge reduction.

In addition to reducing biosolids, there is evidence that OSA may affect sludge properties. For example, some studies that used either synthetic (Saby et al., 2003; Ye et al., 2008) or real wastewater report that OSA decreased sludge volume index (SVI) and improved sludge settleability (Coma et al., 2013). The impact of OSA on sludge dewaterability has not been reported in literature. Sludge dewatering, which is one of the most challenging downstream processes associated with biosolids treatment, is influenced by several factors including the concentration and composition of extracellular polymer substances (EPS) that serve as the framework of sludge flocs (Mowla et al., 2013). OSA causes disintegration of EPS in the external reactor/s (Chon et al., 2011a; Semblante et al., 2015) and therefore may have implications on sludge dewatering characteristics, but this is yet to be studied systematically.

This study aims to determine the impact of SRT of the external anoxic reactors on biosolids reduction in an OSA system fed with real wastewater. Volatile solids reduction and associated other biological reactions, namely, release and fate of nutrients in the external reactors were closely monitored. Additionally, this study compares the dewaterability of waste activated sludge (WAS) with and without OSA. A systematic investigation concentrating on these topics has not been reported in literature. The results of this study will shed light on the underlying mechanisms in OSA, and will provide critical information on how OSA performance can be improved.

2. Materials and methods

2.1. Wastewater characteristics

Domestic unsettled sewage was collected from the beginning of the primary sedimentation channel of Wollongong WWTP fortnightly and stored at 4 °C prior to use. The properties of domestic sewage are provided in Supplementary Table S1. The use of domestic sewage ensures the cultivation of biomass possessing realistic properties.

2.2. Reactor configuration and operation

The OSA system consisted of a sequencing batch reactor, SBR_{OSA} (5 L), attached to an external aerobic/anoxic (2 L) and an additional anoxic reactor (2 L) (Fig. 1a). The control system consisted of $SBR_{control}$ (5 L) attached to a single-pass aerobic digester (2 L) (Fig. 1b).

$SBR_{control}$ and SBR_{OSA} were fed with domestic sewage (Section 2.1). They were operated at 4 cycles/day and a HRT of 12 h. Each cycle comprised of 15 min of filling, 5 h and 30 min of aeration, 1 h of settling, and 15 min of decanting. The SRT of both SBRs was maintained at 10 d by regular sludge wastage (W) (Fig. 1). The average pH, DO concentration, and ORP of $SBR_{control}$ were 6.8 ± 0.6 ($n = 62$), 5.9 ± 2.4 mg/L ($n = 62$), and 117.7 ± 20.5 mV ($n = 34$), respectively, while those of SBR_{OSA} were 6.8 ± 0.8 ($n = 62$), 5.4 ± 1.7 mg/L ($n = 62$) and 129.7 ± 28.2 mV, respectively. These measurements were taken during the aeration period.

The aerobic/anoxic reactor of the OSA system (Fig. 1a) was intermittently aerated (*i.e.*, 8/16 h aeration on/off) using an air diffuser placed at the bottom of the reactor. The anoxic reactor was kept airtight using a silicone-lined cap with inlet and outlet ports. The pH of the aerobic/anoxic reactor was 6.7 ± 0.5 ($n = 62$), whereas its DO concentration when aeration was turned on and off was 4.6 ± 1.0 mg/L ($n = 62$) and 0.4 ± 0.2 mg/L ($n = 62$), respectively. The aerobic/anoxic reactor was fed with sludge from SBR_{OSA} thickened to 5–10 g/L (q_1) by centrifugation (Beckman Coulter, USA) at 3728*g* and 25 °C for 10 min. Thirty-three percent (33%) of sludge from the aerobic/anoxic reactor was transferred to the anoxic reactor (q_2), and 67% was discharged (q_3). A sufficient amount was discharged from the external aerobic/anoxic reactor to vary the total SRT of the external reactors according to the following sequence: 20, 40, 20, and 10 d. The sludge discharged from the aerobic/anoxic reactor was thickened to 16–24 g/L by centrifugation (Beckman Coulter, USA) at 3728*g* and 25 °C for 10 min. The supernatant was returned to SBR_{OSA}, and the pellet was discarded. Sludge from the anoxic reactor was returned to the aerobic/anoxic reactor (q_4) and SBR_{OSA} (q_5).

The aerobic digester of the control system (Fig. 1b) was continuously aerated using an air diffuser. The pH and DO concentration were 6.0 ± 1.7 ($n = 62$) and 6.2 ± 0.19 mg/L ($n = 62$), respectively. The SRT of this digester was varied by regular sludge wastage (Q_{out}) according to the following sequence: 20, 40, 20, and 10 d. The aerobic digester was fed with sludge from $SBR_{control}$ (Q_{in}) thickened to 5–10 g/L by centrifugation (Beckman Coulter, USA) at 3728*g* and 25 °C for 10 min. The supernatant produced by the thickening step was discarded.

2.3. Calculation of sludge yield

Sludge reduction was determined by comparing the sludge yield of the SBRs under parallel operation conditions. The experimental sludge yield (Y) of the SBRs was defined as

$$Y = \frac{P}{C} = \frac{g\ MLVSS}{g\ tCOD} \tag{1}$$

wherein P is the sludge produced in terms of mixed liquor volatile suspended solids (MLVSS) and C is the substrate consumed in terms of tCOD. Sludge yield was derived from the slope of the linear regression of the cumulative sludge produced versus the cumulative substrate consumed. Cumulative values were obtained by incrementing the variations in sludge production and substrate