Effects of oxidation reduction potential in the bypass micro-aerobic sludge zone on sludge reduction for a modified oxic–settling–anaerobic process

Kexun Li, Yi Wang, Zhongpin Zhang and Dongfang Liu

ABSTRACT

Batch experiments were conducted to determine the effect of oxidation reduction potential (ORP) on sludge reduction in a bypass micro-aerobic sludge reduction system. The system was composed of a modified oxic–settling–anaerobic process with a sludge holding tank in the sludge recycle loop. The ORPs in the micro-aerobic tanks were set at approximately +350, −90, −150, −200 and −250 mV, by varying the length of aeration time for the tanks. The results show that lower ORP result in greater sludge volume reduction, and the sludge production was reduced by 60% at the lowest ORP. In addition, low ORP caused extracellular polymer substances dissociation and slightly reduced sludge activity. Comparing the sludge backflow characteristics of the micro-aerobic tank's ORP controlled at −250 mV with that of +350 mV, the average soluble chemical oxygen (SCOD), TN and TP increased by 7, 0.4 and 2 times, median particle diameter decreased by 8.5 μm and the specific oxygen uptake rate (SOUR) decreased by 0.0043 milligram O_2 per gram suspended solids per minute. For the effluent, SCOD and TN and TP fluctuated around 30, 8.7 and 0.66 mg/L, respectively. Therefore, the effective assignment of ORP in the micro-aerobic tank can remarkably reduce sludge volume and does not affect final effluent quality.

Key words | bypass micro-aerobic tank, effluent quality, ORP, SOUR, sludge particle size, sludge reduction

Kexun Li (corresponding author)
Yi Wang
Zhongpin Zhang
Dongfang Liu
The College of Environmental Science and Engineering,
Tianjin Key Laboratory of Environmental Remediation and Pollution Control,
Ministry of Education Key Laboratory of Pollution and Environmental Criteria,
Nankai University,
Wei Jin Road 94,
Tianjin 300071,
China
E-mail: *Likx@nankai.edu.cn*

INTRODUCTION

The traditional activated sludge treatment process often produces large amounts of excess sludge. The disposal of the excess sludge produced during the wastewater treatment process usually accounts for 25–65% of the total plant operation cost (Pérez-Elvira *et al.* 2006), and how to deal with these large volumes of excess sludge remains a big challenge.

An ideal way to solve excess sludge problems is to reduce sludge volume production by *in situ* technologies. Overall there are three major methods: pre-treatment of the excess sludge through sludge disintegration and/or mineralization using physical treatments (Lee *et al.* 2010); limiting sludge growth within the process by using uncouplers, predators or long sludge age; and modification of a conventional activated sludge (CAS) process (Khursheed & Kazmi 2011). The first two methods face huge investment and operation cost and xenobiotics. By comparison, the oxic–settling–anaerobic (OSA) process, a modification of CAS, may present a cost-effective solution for this issue. The original process only needs to add a single anaerobic tank with limited construction cost to execute this method. Compared with the CAS process, the OSA process has 20–65% greater sludge reduction and it is also effective for phosphorus removal (Chen *et al.* 2003; Saby *et al.* 2003; Ye & Li 2005; Wang *et al.* 2008; Khursheed & Kazmi 2011).

The major difference between OSA and CAS is the added anoxic sludge holding tank. Further, the effect of the anoxic tank is determined by the degree of sludge anoxic exposure, which is mainly related to oxidation reduction potential (ORP) (Chen *et al.* 2001a, b; Saby *et al.* 2003). Saby *et al.* (2003) observed that when the ORP in the anoxic sludge holding tank decreased from +100 to −250 mV the sludge reduction efficiency was increased from 20% to 60%, and concluded that an ORP level lower than +100 mV favored excess sludge reduction. However, other authors have found no significant differences in sludge production in a mixed ORP

doi: 10.2166/wst.2014.135

configuration (Lee & Oleszkiewicz 2003). These conflicting views leave the relationship between sludge production and ORP condition unresolved. Most OSA researches focused on adding an anoxic or anaerobic sludge holding tank to a CAS system and then comparing the sludge yield of OSA with the original CAS; for the sludge holding tank, changing its operation conditions has not received adequate attention. Saby *et al.* (2003) examined the effect of different ORP (+100 to −250 mV) in the sludge holding tank on sludge production. However, this kind of research is limited. There is a lack of research about the influence of ORP in the sludge holding tank on the quality of the effluent and sludge characteristics, and the optimum range of ORP in the tank. The purpose of this study is to fill some of the gaps in information about the OSA process.

This research focuses on: (1) the effect of ORP in the micro-aerobic tank on sludge reduction; (2) the changes in sludge characteristics after passing through the bypass micro-aerobic tank; (3) the effect of the bypass micro-aerobic system on the effluent characteristics. In addition, the possible mechanisms that account for the sludge reduction are discussed.

MATERIALS AND METHODS

Sludge cultivation

In this experiment, five identical sets of experimental bench-scale systems were used. Each system had a conventional sequencing batch reactor (SBR) and a bypass tank, as shown in Figure 1. The bypass tank was aerated and mixed intermittently. In aeration phase, the dissolved oxygen (DO)

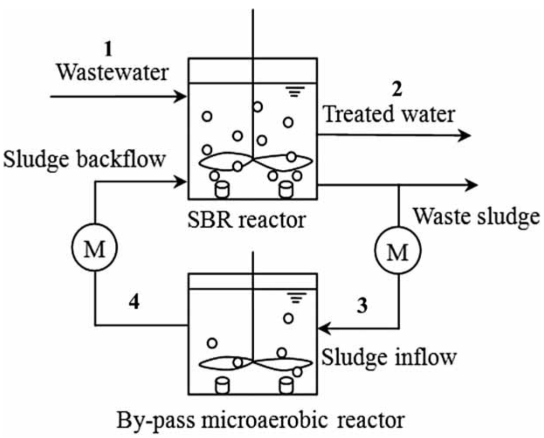

Figure 1 | Schematic diagram of the five experimental systems. (*Note:* M represents peristaltic pump, numbers 1–4 represent sampling points.)

in the tanks was controlled between 1 and 3 mg/L. During mixing alone, the DO was controlled between 0.3 and 0.7 mg/L. Since the operation condition of the bypass tank does not strictly belong to aerobic (DO > 0.5 mg/L), anoxia (0.2 < DO < 0.5) or anaerobic (DO < 0.2 mg/L), 'micro-aerobic' was used to describe the tank to emphasize the limited aeration of its operation characteristic.

The systems were identified by numbering. Numbers were from 0 to 4. System 0 was the reference system and the others were testing systems. The volume of each bypass micro-aerobic tank was 10 L with 8 L working volume. The volume of each SBR reactor was 12 L with 10 L working volume. All the SBR reactors operated on a 6 h cycle which consisted of 0.5 h for wastewater feed in, 4 h aeration and mixing, 1 h settling and 0.5 h for drainage. At the beginning of the experiment, all the reactors were filled with the same activated sludge which was obtained from Jizhuangzi sewage treatment plant in Tianjin, China. Three litres of sludge was added to each SBR and 8 L of sludge was added to each bypass micro-aerobic tank. The initial mixed liquor suspended solids (MLSS) was 10,000 mg/L and initial sludge retention time (SRT) was 25 d. Two litres of distilled water was added into each SBR reactor and then wastewater inflow was started. Experimental water was synthetic wastewater and its characteristics are shown in the Supplementary material, Table S1 (available online at http://www.iwaponline.com/wst/069/135.pdf). Sludge cultivation and subsequent experiments were all conducted at a room temperature of $25 \pm 2\,^{\circ}$C.

System operation and testing

The ORP in each micro-aerobic tank was controlled by different aeration time and standing period. The operating schedules for each micro-aerobic tank are shown in Table 1.

The micro-aerobic tank in the reference system was operated without standing period and aerated during the entire cycle. In order to ensure continuous operation, excess sludge was discharged to the corresponding micro-aerobic

Table 1 | Operating schedule of each micro-aerobic tank (unit: hour)

Experimental systems	Feed	Mixed and aeration	Standing	Mixed	Mixed and backflow
System 0	0.5	5	–	–	0.5
System 1	0.5	0.5	4	0.5	0.5
System 2	0.5	1	3.5	0.5	0.5
System 3	0.5	1.5	3	0.5	0.5
System 4	0.5	2	2.5	0.5	0.5

tank after the sludge in the SBR reactor has settled for 45 minutes. The sludge discharge would last for 30 minutes. Then the discharged excess sludge was mixed and aerated, and left to stand for different time periods. Finally, the same volume of sludge that was fed to the micro-aerobic tanks was discharged back to the corresponding SBR reactor. Every cycle lasted for 6 hours. The volume of the sludge inflow and sludge backflow was set up at 0.25 L per cycle. Hence, the SRT in each bypass micro-aerobic tank was 8 days. During the operation period, DO and MLSS of the SBR reactors were maintained at the same in all systems. DO was controlled between 2.0 and 3.0 mg/L, and MLSS at 3,000 ± 500 mg/L.

Sampling and analytical methods

Sampling was started at the end of the first cycle and lasted for 3 months. Samples were collected at the influent and effluent of SBR, sludge inflow and sludge backflow of the micro-aerobic tank (Figure 1, points 1–4). The parameters examined included soluble chemical oxygen (SCOD), total nitrogen (TN), total phosphorus (TP) for sampling points 1–4 and ORP in the micro-aerobic tanks. The total wasted sludge from each system during the period was also monitored. ORP in the tanks was monitored every 3 days with a Hanna HI8424 PH/ORP meter (Yongcheng Aquarium Equipment Factory). Sampling and testing of the SCOD, TN and TP were measured according to *Standard Methods* (APHA 2012) using a UV-vis spectrophotometer (TU-1900, Beijing Purkinje Instrument Co., Ltd) and performed every day in the first 6 days and once every 6 days thereafter. Before testing, all the samples were centrifuged at 5,000 rpm for 5 minutes and filtered through a 0.45 μm filter. Sludge particle size was tested by a laser particle size analyzer (BT-9300HT, Dandong Baite Instrument Co., Ltd) after all the systems reached stable operation. After the experiment, in order to determine the sludge activity, the specific oxygen uptake rate (SOUR) for the five sludge holding tanks was examined by using a dissolved oxygen analyzer (OXI3210, WTW, Weilheim, Germany) in the manner described by Chen *et al.* (2002).

RESULTS AND DISCUSSION

Effect of ORP in micro-aerobic tank on sludge yield

During the first 6 days of operation a sharp decrease of ORP occurred in the four micro-aerobic tanks of the testing systems (systems 1–4) and then achieved a stable level, while the ORP of the reference system's micro-aerobic tank was maintained at a much higher level than those of the testing systems throughout the operation. Once all the systems reached a stable operation, the ORP in the micro-aerobic tanks of system 0, 1, 2, 3, and 4 were kept at approximately +350, −250, −200, −150 and −90 mV, respectively. In order to simplify operation and analysis, each time the same volume sludge was discharged from one system. Figure 2 summarizes the cumulative excess sludge discharged from the five systems.

Because the initial sludge volume of the SBR reactors was 3 L, and the wasted excess sludge was from the SBR reactors, it can be seen from Figure 2 that the SRTs of the SBR reactors of system 0, 4, 3, 2, 1 were 24, 30, 37, 48 and 56 days, respectively. The cumulative discharged excess sludge volume for each system in the order of largest to smallest was: the system 0> system 4> system 3> system 2> system 1 (Figure 2). Sludge yields calculated from system 0 to system 4 were 0.208, 0.085, 0.104, 0.145 and 0.167 gram total suspended solids per gram COD, respectively. In comparison with the reference system, the sludge production from system 1 to system 4 was reduced by 60, 50, 30 and 20%, respectively. Clearly, low ORP in the micro-aerobic tank promotes sludge reduction. These results are consistent with results of previous research in which sludge production was respectively reduced by 60 and 40% when the ORP was controlled at −250 and −180 mV (Saby *et al.* 2003; Torregrossa *et al.* 2012). During the 3 months of operation, except for the first few days when the system was acclimating, there was no sharp increase or decrease of ORP. However, a sharp increase of ORP during the stabilization phase occurred in a previous similar study (Saby *et al.* 2003), leading to high sludge

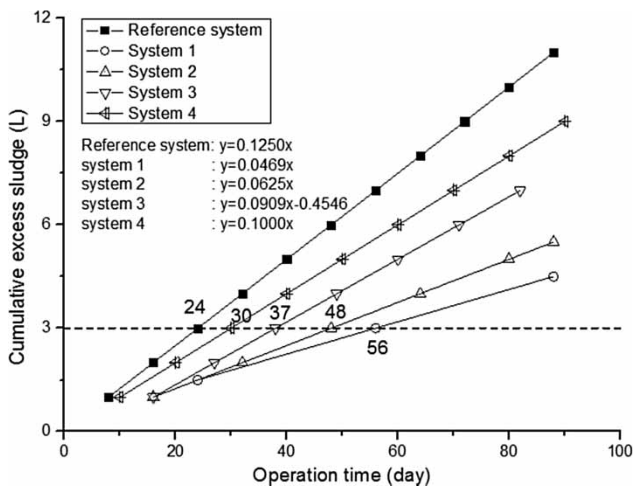

Figure 2 | Cumulative excess sludge discharged from the five systems.

production. In that research a month-long restoration was needed to restore the system. The method presented in the current research may have a certain advantage for maintaining a stable low ORP. Regular mixing and intermittent aeration can achieve desired low ORP and is also simple to implement. Based on this result, the bypass micro-aerobic system can reduce excess sludge effectively, and ORP level in the micro-aerobic tank is one of the keys to achieve the effective sludge reduction. Controlling ORP in the range of −120 to −250 mV is crucial to significant sludge reduction and can achieve 30–60% sludge reduction.

Effect of ORP in micro-aerobic tank on SCOD, TN and TP behavior

SCOD behavior

The total dissolved organic matter change was monitored through SCOD, TN and TP testing. Figure 3 shows the SCOD variation for the five systems.

Figure 3(a) describes the SCOD variation in the sludge inflow of the micro-aerobic tanks. It can be seen that the sludge inflow SCOD in the four testing systems increased during the first 6 days, and then reached a relatively stable value that ranged from 30 to 38 mg/L. The SCOD of the reference system stayed around 27 mg/L throughout the testing period. Figure 3(b) shows the SCOD variation in the sludge backflow of the five micro-aerobic tanks. The figure shows SCOD concentrations in the four testing systems rapidly increased in the first 6 days before they stabilized. The SCOD of the reference system was consistently low at around 15 mg/L during the entire experiment. Using the results shown in Figure 3(b), the SCOD value was calculated in system 1 to system 4 to have increased by 7, 5, 4.5 and 3.5 times, respectively, in comparison with the reference system (after stabilization). A possible explanation for the SCOD increase of testing systems over the reference system is that the sludge decay coefficient under anaerobic condition is accelerated when the ORP is lower than −100 mV (Chon *et al*. 2011). It was surmised that the cause of the COD release may be triggered by a stressful environment with no source of external food (Saby *et al*. 2003). This experiment suggests that the COD release may be due not only to substrate starved sludge, but also to the low ORP. This follow-up conclusion is based on the fact that the sludge in the reference system also struggled in a starved stressful environment: the SCOD concentration in the bypass sludge tank did not increase.

TN and TP behavior

To determine if sludge degradation occurred in the micro-aerobic tank, the TN and TP in the systems were monitored. The dissolved TN and TP variation is shown in Figure 4.

Figure 4(a) shows the soluble TN variation of the sludge inflow in the five micro-aerobic tanks. The soluble TN concentrations in the sludge inflow of the reference system, and system 3 and 4 were low without much variation during the experiment. System 1 and 2 TN values increased after approximate 12 days of operation and were maintained at a relatively higher level. The average concentration of TN from system 0, 4, 3, 2, 1 was 8.4, 8.6, 8.0, 9.5 and 9.5 mg/L, respectively, and that of TP was 0.40, 0.52, 0.66, 0.80 and 0.98 mg/L, respectively

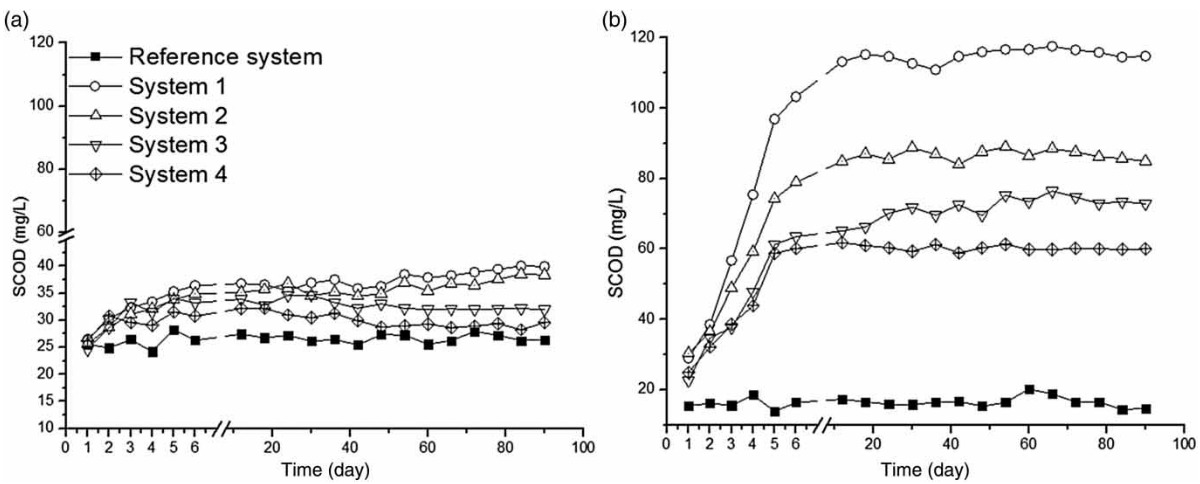

Figure 3 | SCOD variation curves in the sludge inflow and sludge backflow of the five micro-aerobic tanks. (a) SCOD in the sludge inflow, (b) SCOD in the sludge backflow.

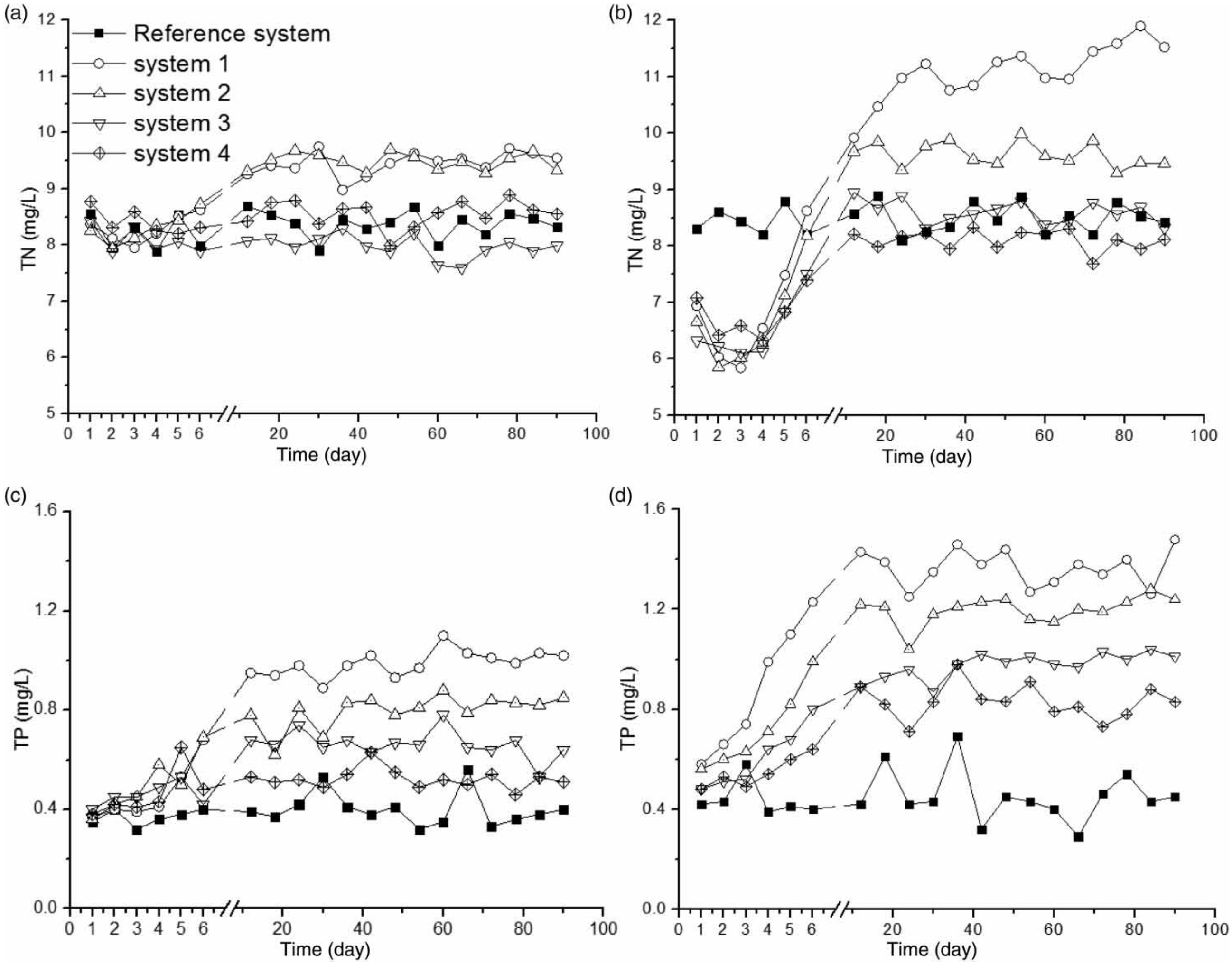

Figure 4 | Soluble TN and TP variation curves in the sludge inflow and sludge backflow of the five micro-aerobic tanks. (a) Soluble TN in the sludge inflow, (b) soluble TN in the sludge backflow, (c) soluble TP in the sludge inflow, (d) soluble TP in the sludge backflow.

(Figure 4(c)). The increased TN and TP concentrations of the sludge inflow could come from soluble proteins produced by sludge degradation and anaerobic phosphorus release that cannot be used by the microorganisms. Figure 4(b) shows that the soluble TN in the four testing micro-aerobic tanks decreased in the first 4 days and then began to increase, while in the reference system the values fluctuated around a single value (8.5 mg/L). After four days of operation a steady-state was established in the micro-aerobic tank. After this point the extracellular polymer substances (EPS) dissociation and sludge degradation speed played dominant roles in the tank at low ORP, leading to dissolved TN increase, which is consistent with the research of Novak *et al.* (2007). In contrast with TN, TP concentrations in the sludge inflow and sludge backflow increased as the ORP decreased (Figure 4(c) and 4(d)). This variation of TP and ORP is similar with the results of previous research (Wang *et al.* 2008). The reason was that under low ORP and low substrate conditions, microorganisms use ATP and polyphosphates as a source of energy. When aeration and substrate are again sufficient, they rebuild their energy reserves (Ye *et al.* 2008).

The figures above show that total dissolved organic matter increased both in sludge inflow and backflow. The increased amount was greater as the ORP in the micro-aerobic tank became lower. The maximum increased values of SCOD, TN and TP were from 15 to 115 mg/L, 8 to 11 mg/L and 0.98 to 1.37 mg/L, respectively, under the lowest ORP (−250 mV). During filtration of the samples of the sludge backflow for test, an increase in color of the samples from system 4 to 1 was observed. The difficulty of filtration was also increasing from system 4 to 1. The cause of the two above phenomena may be the increasing dissolved EPS concentration as ORP decreased in the micro-aerobic tank. Low ORP can cause EPS to dissolve into small molecules which lead to the membrane getting blocked easily during filtration.

Effect of ORP in micro-aerobic tank on sludge characteristics

Variation of the sludge particle size

During the 3 months of operation, no obvious sludge settling problems occurred and there was no significant difference in sludge settling performance between the systems. To further assess the effect of the ORP in the micro-aerobic tank on microbial activity, the variation in sludge size and SOUR were investigated.

The results of the particle size distribution in the five micro-aerobic tanks (Table 2) show that median particle diameter decreases when ORP decreases. The median particle diameter of the sludge in the micro-aerobic tanks from system 0, 4, 3, 2, and 1 were 32.18, 27.13, 27.32, 20.17 and 20.24 μm, respectively. Overall, the specific surface had an inverse relationship with the particle diameter, which is consistent with previous research (Wu *et al.* 2009; Ren *et al.* 2011). The specific surface area increased from 0.11 to 0.16 m^2/g when the particle diameter decreased from 32.18 to 20.17 μm. The results also show that the particle size variation was not obvious under different ORP. By comparison with the reference particle size, the particle size from system 4 to 1 decreased by 0.16, 0.15, 0.37 and 0.37 times, respectively. The settling velocity of granules usually corresponds to their sizes, thereby explaining why the sludge settling performance was not significantly changed from one to another. Thus keeping the ORP ranging from −90 to −250 mV in the micro-aerobic tank will not cause serious negative influence on the sludge, which is similar with the results of Hu *et al.* (2005).

Comparison of SOUR

The SOUR of the sludge in the five micro-aerobic tanks was monitored to evaluate the sludge activity. The SOUR (Supplementary material, Figure S1, available online at http://www.iwaponline.com/wst/069/135.pdf) in the systems was: system 0> system 4 > system 3> system 2 > system 1. The SOUR in system 0, 4, 3, 2, 1 were 0.0193, 0.0176, 0.0165, 0.0152 and 0.0150 milligram O_2 per gram suspended solids per minute, respectively. These results are consistent with the finding of Chen *et al.* (2003) that the lower the ORP, the lower the SOUR of the sludge. Evidence (Torregrossa *et al.* 2012) showed that the OSA process with the ORP in the sludge holding tank kept at −180 mV would cause a slight decrease of the biomass respiratory activity.

From what has been discussed above, a conclusion can be drawn that using alternate aeration and anoxic with proper batch mixing operation is an effective method for maintaining low ORP, which is beneficial for sludge volume reduction. This method is also helpful to achieve a stable sludge settling characteristic and an appropriate sludge activity. The increase of SCOD, TN and TP of the sludge backflow, and the decrease in median sludge particle size and SOUR, with lower ORP confirm that exposure to low ORP conditions in the micro-aerobic tank, coupled with food scarcity, promotes EPS dissolution and sludge decay, which was also proved by Chen *et al.* (2001a, b, 2003). Then the possible mechanism for the sludge reduction in this experiment is that under low ORP, sludge decay is accelerated effectively which facilitates the sludge disintegration, and thus reduces sludge yield.

Effect of ORP in micro-aerobic tanks on removal efficiency of complete system

During the entire experiment, influent COD, TN and TP fluctuated in the range of 370–430, 20–60 and 3–9 mg/L, respectively. The influence of ORP in the micro-aerobic tanks on the quality of the effluent is shown in Figure 5.

Table 2 | Comparison of the sludge particle size in each micro-aerobic tank

Sludge type	ORP (mV)	Particle size interval (μm)	Percentage (%)	Median particle diameter (μm)	Specific surface area (m^2/g)
System 0	+350	21.12–61.61	54.37	32.18	0.11
System 1	−250	12.36–36.07	53.54	20.24	0.17
System 2	−200	12.36–36.07	55.38	20.17	0.16
System 3	−150	17.05–49.74	51.12	27.32	0.12
System 4	−90	17.05–49.74	51.57	27.13	0.12

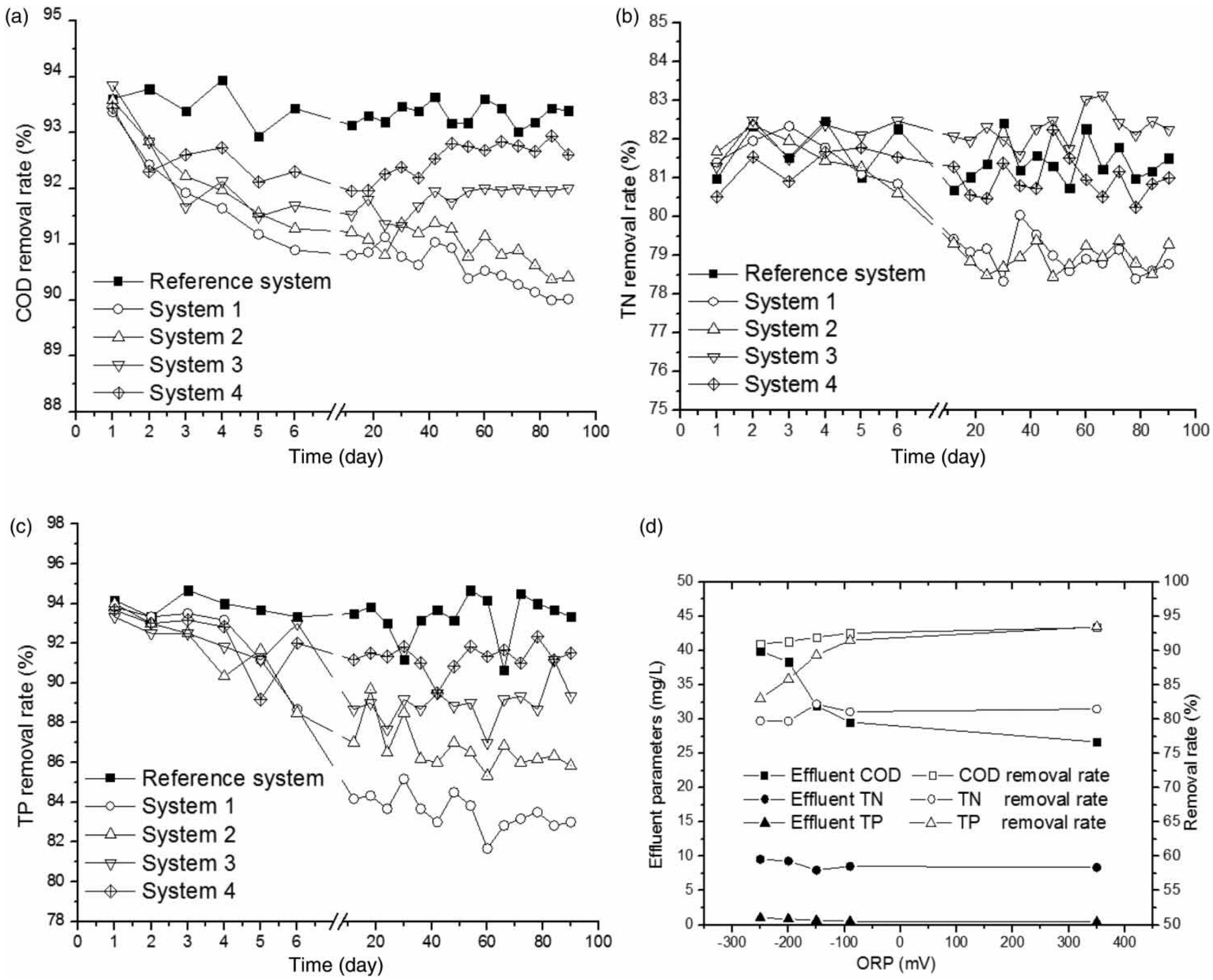

Figure 5 | Effluent COD, TN and TP removal rate variation curves and effluent quality. (a) COD removal rate of the five systems, (b) TN removal rate of the five systems, (c) TP removal rate of the five systems, (d) the effect of ORPs on the removal of effluent COD, TN and TP.

It can be seen that the removal efficiency of effluent COD, TN and TP in the four testing systems decreased during the first 6 days and then kept at a stable level, while the reference system had no sharp change in removal efficiency during the entire experiment. The results show that low ORP in the micro-aerobic tank has a slight negative impact on wastewater treatment efficiency. The removal rate of effluent COD in the reference system was 93% and that of system 4, 3, 2 and system 1 were 92, 92, 91 and 91% respectively. The variation of TN and TP removal rate showed a similar tendency to that of COD. The effluent COD, TN and TP of the testing systems were around 30, 8.7 and 0.66 mg/L, respectively (Figure 5 (d)). In addition, the effluent COD, TN and TP did not exceed 50, 15 and 1 mg/L at any time, i.e. the effluent parameters could meet the requirements of the level 1 class A discharge standard (GB 18918-2002 2002) of all the systems (0–4). Therefore, controlling the ORP in the micro-aerobic tank in the range of −90 to −250 mV will not cause poor performance of the complete system.

CONCLUSION

(1) Low ORP in the bypass micro-aerobic tank plays a vital role in sludge reduction. Proper regulation of ORP ranging from −120 to −250 mV can effectively reduce sludge by 30–60%.
(2) Low ORP caused EPS to dissociate and only slightly reduced sludge activity. After passing through the micro-aerobic tank, the average SCOD, TN, and TP of the four

testing systems increased by 50, 0.49 and 0.35 mg/L, respectively. Compared to the reference system (ORP controlled at +350 mV), the average sludge particle diameter of the testing tanks decreased by 26% and SOUR decreased by 22%.

(3) Low ORP had a slight negative effect on the effluent, but the effluent did not exceed Chinese wastewater discharge limits (GB 18918-2002 2002). The COD, TN, and TP concentrations of the effluent increased from 27 to 38 mg/L, 8.3 to 9.5 mg/L and 0.4 to 0.98 mg/L, respectively, as ORP in the micro-aerobic tank was controlled at −250 mV.

REFERENCES

APHA (American Public Health Association), Rice. Eugene, W. & Bridgewater, L. 2012 *Standard Methods for the Examination of Water and Wastewater*. American Public Health Association/American Water Works Association/Water Environment Federation, Washington, DC, USA.

Chen, G. H., Yip, W. K., Mo, H. K. & Liu, Y. 2001a Effect of sludge fasting/feasting on growth of activated sludge cultures. *Water Research* **35** (4), 1029–1037.

Chen, G. H., Saby, S., Djafer, M. & Mo, H. K. 2001b New approaches to minimize excess sludge in activated sludge systems. *Water Science and Technology* **44** (10), 203–208.

Chen, G. H., Mo, H. K. & Liu, Y. 2002 Utilization of a metabolic uncoupler, 3,3′,4′,5-tetrachlorosalicylanilide (TCS) to reduce sludge growth in activated sludge culture. *Water Research* **36** (8), 2077–2083.

Chen, G. H., An, K. J., Saby, S., Brois, E. & Djafer, M. 2003 Possible cause of excess sludge reduction in an oxic-settling-anaerobic activated sludge process (OSA process). *Water Research* **37** (16), 3855–3866.

Chon, D. H., Rome, M., Rome, Y. M., Park, K. Y. & Park, C. 2011 Investigation of the sludge reduction mechanism in the anaerobic side-stream reactor process using several control biological wastewater treatment processes. *Water Research* **45** (18), 6021–6029.

GB 18918-2002 2002 Discharge standards of pollutants for municipal wastewater treatment plant. Environmental Protection Administration, PR China.

Hu, Z., Ferraina, R. A., Ericson, J. F., MacKay, A. A. & Smets, B. F. 2005 Biomass characteristics in three sequencing batch reactors treating a wastewater containing synthetic organic chemicals. *Water Research* **39** (4), 710–720.

Khursheed, A. & Kazmi, A. A. 2011 Retrospective of ecological approaches to excess sludge reduction. *Water Research* **45** (15), 4287–4310.

Lee, Y. & Oleszkiewicz, J. A. 2003 Effects of predation and ORP conditions on the performance of nitrifiers in activated sludge systems. *Water Research* **37** (17), 4202–4210.

Lee, M. J., Kim, T. H., Yoo, G. H., Min, B. M. & Hwang, S. J. 2010 Reduction of sewage sludge by ball mill pretreatment and Mn catalytic ozonation. *KSCE Journal of Civil Engineering* **14** (5), 693–697.

Novak, J. T., Chon, D. H., Curtis, B. A. & Doyle, M. 2007 Biological solids reduction using the cannibal process. *Water Environment Research* **79** (12), 2380–2386.

Pérez-Elvira, S. I., Nieto Diez, P. & Fdz-Polanco, F. 2006 Sludge minimisation technologies. *Reviews in Environmental Science and Bio/Technology* **5** (4), 375–398.

Ren, R., Li, K. X., Zhang, C., Liu, D. F. & Sun, J. 2011 Biosorption of benzyl dimethyl ammonium chloride on activated sludge: kinetic, thermodynamic and reaction mechanisms. *Bioresource Technology* **102** (4), 3799–3804.

Saby, S., Djafer, M. & Chen, G. H. 2003 Effect of low ORP in anoxic sludge zone on excess sludge production in oxic-settling-anoxic activated sludge process. *Water Research* **37** (1), 11–20.

Torregrossa, M., Gaetano, D. B. & Daniele, D. T. 2012 Comparison between ozonation and the OSA process: analysis of excess sludge reduction and biomass activity in two different pilot plants. *Water Science and Technology* **66** (1), 185–192.

Wang, J. F., Zhao, Q. L., Jin, W. B., You, S. J. & Zhang, J. N. 2008 Performance of biological phosphorus removal and characteristics of microbial community in the oxic-settling-anaerobic process by FISH analysis. *Journal of Zhejiang University-Science* **9** (7), 1004–1010.

Wu, J., Jiang, X. & Wheately, A. 2009 Characterizing activated sludge process effluent by particle size distribution, respirometry and modelling. *Desalination* **249** (3), 969–975.

Ye, F. X. & Li, Y. 2005 Uncoupled metabolism stimulated by chemical uncoupler and oxic-settling-anaerobic combined process to reduce excess sludge production. *Applied Biochemistry and Biotechnology* **127** (3), 187–199.

Ye, F. X., Zhu, R. F. & Li, Y. 2008 Effect of sludge retention time in sludge holding tank on excess sludge production in the oxic-settling-anoxic (OSA) activated sludge process. *Journal of Chemical Technology and Biotechnology* **83** (1), 109–114.

First received 19 September 2013; accepted in revised form 28 February 2014. Available online 14 March 2014