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Simultaneous denitrification and denitrifying phosphorus removal in a full-scale anoxic–oxic process without internal recycle treating low strength wastewater

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ABSTRACT

Performance of a full-scale anoxic–oxic activated sludge treatment plant (4.0×10^5 m^3/day for the first-stage project) was followed during a year. The plant performed well for the removal of carbon, nitrogen and phosphorus in the process of treating domestic wastewater within a temperature range of 10.8°C to 30.5°C. Mass balance calculations indicated that COD utilization mainly occurred in the anoxic phase, accounting for 88.2% of total COD removal. Ammonia nitrogen removal occurred 13.71% in the anoxic zones and 78.77% in the aerobic zones. The contribution of anoxic zones to total nitrogen (TN) removal was 57.41%. Results indicated that nitrogen elimination in the oxic tanks was mainly contributed by simultaneous nitrification and denitrification (SND). The reduction of phosphorus mainly took place in the oxic zones, 61.46% of the total removal. Denitrifying phosphorus removal was achieved biologically by 11.29%. Practical experience proved that adaptability to gradually changing temperature of the microbial populations was important to maintain the plant overall stability. Sudden changes in temperature did not cause paralysis of the system just lower removal efficiency, which could be explained by functional redundancy of microorganisms that may compensate the adverse effects of temperature changes to a certain degree. Anoxic–oxic process without internal recycling has great potential to treat low strength wastewater (*i.e.*, TN < 35 mg/L) as well as reducing operation costs.

Introduction

Domestic wastewater contains compounds including nutrients and organic matter that if not properly treated, can increase water eutrophication (Gulati and van Donk 2002), alter the ecological balance of water systems, threaten aquatic organisms, and risk public health. Therefore, adequate removal of nutrient and organic matter removal are major concerns for wastewater treatment plants (WWTPs) that need to meet increasingly stringent discharge requirements. These increased requirements are often combined with needs to reduce the energy consumption and minimize operational costs of wastewater treatment (Guerrero et al. 2012; Plosz 2007). Many existing full-scale WWTPs must be adapted to improve the efficacy of existing processes or new treatment facilities must be designed and constructed; a better understanding of

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the process is essential to make systems more effective and efficient.

Conventional biological nutrient removal (BNR) processes require chemical oxygen demand (COD), which is often the limiting substrate in the incoming wastewater (Zeng et al. 2003). Several methods have been examined to enhance nitrogen removal, including internal carbon source addition (Biradar et al. 2010; Kampas et al. 2007; Park et al. 2011a) and external carbon source addition (Gong et al. 2013; Modin et al. 2007; Quan et al. 2005). However, carbon source addition could increase operation cost and carbon footprint. Simultaneous nitrification and denitrification (SND) is an attractive pathway for nitrogen reduction in comparison with traditional method because of saving carbon source and reducing aeration consumption. Zhao et al. (1999) found that the nitrogen loss due to SND in the aeration tank, contributed 10% to 50% of the total Kjeldahl nitrogen to the overall nitrogen removal.

As a method of removing phosphorus biologically by polyphosphate-accumulating organisms (PAOs) rather than chemically, enhanced biological phosphorus removal (EBPR) has stimulated much interest in the study of the removal mechanisms and the microbiology of the systems (Henze et al. 2008; Oehmen et al. 2007). Recent research has shown that phosphorus uptake in the presence of nitrate (*i.e.*, under anoxic conditions) does occur, and simultaneous denitrification and phosphorus removal can be achieved. Several denitrifying phosphorus removal systems have been developed including the sequencing batch reactor (SBR) system (Merzouki et al. 2001), the University of Cape Town (UCT) system (Kuba et al. 1997), and the Anaerobic/Anoxic/Oxic (A/A/O) multiple reactor system (Kishida et al. 2006; Xu et al. 2011). The importance and overall benefits of denitrifying PAO (DPAO) in activated sludge systems has also been widely recognized (Kuba et al. 1994). However, studies on DPAO's anoxic activities in full-scale anoxic/oxic (A/O) WWTPs are rare.

Plenty of intensive studies were carried out on the performance of full-scale BNR WWTP to evaluate the effect of operating parameters and environmental conditions on nutrient removal (Fernandes et al. 2013; Lopez-Vazquez et al. 2008). Nevertheless, the factors that may influence full-scale BNR WWTPs are far from being fully understood leading to unacceptable discharge levels of pollutants. Therefore, it is important to deepen the understanding of biochemical transformation and degradation processes in full-scale WWTPs and increase the detailed knowledge on the process and control.

Understanding the impact of temperature change on bioreactor performance could allow for the evaluation of the microorganisms' interaction with the environmental conditions. The wastewater treatment in A/O processes can contain a quite diverse microbial community whose activity depends on temperature for nutrient removal. The knowledge of the temperature involved in wastewater purification becomes relevant for the efficiency of treatment. However, the question of how temperature variation influence on the plant stability is an uncertain issue.

The objective of the present article is to evaluate the overall performance and wastewater characteristics may influence the carbon and nutrient removal at a full-scale BNR WWTP. These findings then are applied in a discussion of pollutant removal mechanisms and factors affecting plant performance. This work was performed to gain a better fundamental understanding of nutrient removal and to provide insights into new strategies for future optimization of this exciting technology.

1. Materials and methods

1.1. *Wastewater treatment plant*

The Jiangxinzhou full-scale WWTP in this study is in operation and treating domestic wastewater in Nanjing, central east China. A performance assessment of this WWTP was conducted recently, and design parameters are presented in Table 1.

The plant began operation in 1996, and was upgraded in 2003 to include BNR systems. The first-stage project has an average flow rate at 4.0×10^5 m^3/day. Wastewater from primary clarifiers (PCs) is fed to the anoxic tanks (ATs) with mechanical mixers, where denitrification takes place. In the oxic tanks (OTs), nitrification is performed, where adequate air is uniformly distributed across the length of the tanks with air diffusers. The settled sludge from the secondary clarifiers (SCs) is recycled to the head of the anoxic zones to keep the microbial biomass active. The excess sludge is disposed in the sludge treatment room and then transported outside. The sludge recycling flow rate was 0.8 times the influent flow rate at 3.2×10^5 m^3/day. The mixed liquor suspended solid (MLSS) varied from 3800 to 6000 mg/L in the biochemical pools. The total hydraulic retention time (HRT) of the wastewater in the system (including anoxic and aerobic zones) was 6.7 hr. The sludge retention time (SRT) was maintained at 8–12 days by controlling excess sludge. Dissolved oxygen (DO) concentrations in the anoxic and oxic zones were less than 0.25 mg/L and more than 0.50 mg/L, respectively. Fig. 1 shows a flow diagram of the biological treatment units of the WWTP.

Table 1 – Design parameters of the full-scale WWTP (the first-stage project).

Process unit	Volume (m^3) for each	Number	Depth (m)	HRT (H)	Main equipment
Aerated grit chambers	350	4	4	0.083	Jet aerator
Primary clarifiers	8333	4	4.7	2	Mud scraper
Anoxic tanks	7000	4	6	1.7	Mixers
Oxic tanks	21,000	4	6	5.0	Micropore aerators
Secondary clarifiers	6250	8	4.8	3	Mud scraper
Sludge thickening tanks	5000	6	10	720	None

WWTP: wastewater treatment plant; HRT: hydraulic retention time.

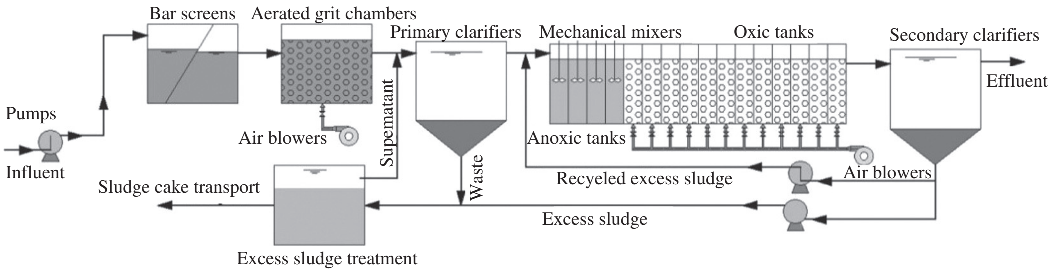

Fig. 1 – Diagram of a continuous full-scale activated sludge treatment plant.

1.2. *Wastewater characteristics and temperature changes*

We collected data weekly from the Jiangxinzhou full-scale WWTP for a year (from March 2013 to February 2014). The average influent raw wastewater characteristics are described in Table 2. Internal loading was caused by a thickening supernatant (600–700 m^3/day) and reject water from sludge dewatering (1000 m^3/day), which was pumped into the PCs. Annual water temperature variation for primary effluent is shown in Fig. 2. As shown in Fig. 2, water temperature begins to increase in spring, until it reaches the maximum on August 20th, 2013, and the average change rate of rising temperature is 0.104°C/day. Then the water temperature decreases to the minimum on February 11th, 2014, and the average change rate of dropping temperature is 0.113°C/day. When the change of temperature is greater than the average values (namely, base rate) mentioned above, it can be considered as a sudden change. The rest conditions refer to gradual changes.

1.3. *Analytical methods*

Chemical oxygen demand (COD_(Cr)), ammonium (NH_4^+–N), nitrate (NO_3^-–N), nitrite (NO_2^-–N), total nitrogen (TN), and total phosphorus (TP) were all measured according to standard methods (APHA 2005). The dissolved oxygen (DO) and temperature were measured using YSI 6600 V2 Multi-Parameter Water Quality Sonde (made in USA).

1.4. *Pollutant mass balance*

In this study, the pollutant mass balance in anoxic zones, aerobic zones and SCs were calculated according to Eqs. (1), (2) and (3).

Table 2 – Characteristics of raw wastewater.

Contents	Range	Average	Times *
COD (mg/L)	78.0–217.0	129.5	43
NH_4^+-N (mg/L)	10.9–28.4	20.3	43
NO_2^-N (mg/L)	0–0.14	0.06	43
NO_3^-N (mg/L)	0.02–2.82	0.40	43
TN (mg/L)	15.0–35.6	25.0	43
TP (mg/L)	1.20–3.03	2.10	43

COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus.
* The number of measurements.

For anoxic zones

$$Q_0 \cdot C_{\text{inf.C,N,P}} + Q_r \cdot C_{\text{r.C.N.P}} = (Q_0 + Q_r) \cdot C_{\text{eff.C,N,P}} + \Delta A_{\text{an.C,N,P}} \quad (1)$$

For oxic zones

$$(Q_0 + Q_r) \cdot C_{\text{inf.C,N,P}} = (Q_0 + Q_r) \cdot C_{\text{eff.C,N,P}} + \Delta A_{\text{ae.C,N,P}} \quad (2)$$

For SCs

$$(Q_0 + Q_r) \cdot C_{\text{inf.C,N,P}} = Q_r \cdot C_{\text{r.C.N.P}} + Q_0 \cdot C_{\text{eff.C,N,P}} + \Delta A_{\text{sc.C,N,P}} \quad (3)$$

where, $\triangle A_{an,C,N,P}$, $\triangle A_{ae,C,N,P}$, $\triangle A_{sc,C,N,P}$ denote the capacity of pollutants changed in anoxic zones, aerobic zones and SCs; Q_0 and Q_r represent influent flow and sludge returned recirculation flow, respectively; C_r represents pollutant concentrations of returned sludge from SCs; C_{inf} and C_{eff} represent pollutant concentrations of influent and effluent; C, N, P denote the concentration of COD, TN and TP, respectively.

2. Results and discussion

2.1. *Overall performance*

Fig. 3 shows the influent and effluent concentrations and the removal efficiency of COD, NH_4^+–N, TN and TP during the study period. Although the influent COD fluctuated between 78.0 and 217.0 mg/L, the removal performance was relatively stable with an average effluent concentration of 15.6 mg/L. The influent fluctuation had minimal effect on COD removal during a year (Fig. 3a). NH_4^+–N and TN removal performance are shown in Fig. 3b, c, respectively. Ammonia oxidation was incompletely accomplished with a mean effluent NH_4^+–N of 1.78 mg/L. The TP concentration in the influent fluctuated from 1.20 to 3.03 mg/L, and the effluent average concentration of TP was 0.98 mg/L (Fig. 3d). Overall, the influent fluctuation had almost no distinct effect on effluent quality which met with class 1B of Chinese discharge regulations (MEP, 2002) in terms of COD, NH_4^+–N, TN and TP (COD < 60 mg/L, NH_4^+–N < 8 mg/L, TN < 20 mg/L, TP < 1.5 mg/L).

Water temperature changes did not affect the effluent COD concentration. However, sudden temperature variations (Fig. 2) account for transient impaired biological activity, leading to lower removal efficiency of nitrogen and phosphorus (Fig. 3c, d). On the one hand, sudden dropping of temperature had obvious adverse effects on nutrient removal. In April, when the water temperature suddenly went down from 19.4°C (April 17th) to 17.2°C (April 23rd) with change rate of 0.367°C/day, the

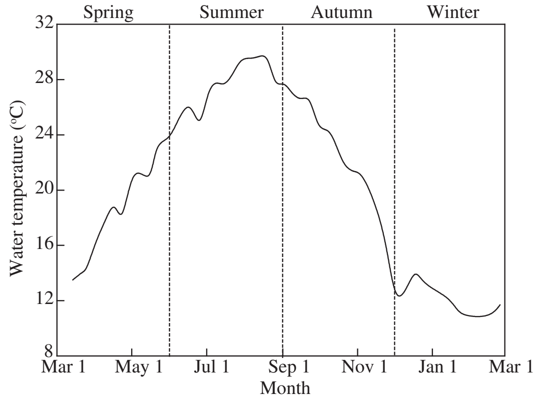

Fig. 2 – Annual water temperature variation for primary effluent (March 2013 to February 2014. Vertical dash lines separate different seasons).

removal efficiency of NH_4^+–N and TN declined from 96.7% to 82.4% and from 55.0% to 30.3%, respectively. In May, while the water temperature dropped from 22.2°C (May 2nd) to 20.2°C (May 16th) with change rate of 0.143°C/day, removal efficiency of NH_4^+–N and TP came down from 98.4% to 92.5% and from 77.2% to 43.6%, respectively. In summer, the water temperature suddenly fell from 27.1°C (June 3rd) to 24.0°C (June 17th) with change rate of 0.221°C/day resulted in lower removal efficiency of TN and TP, which decreased from 62.6% to 27.7% and from 68.1% to 35.8%, respectively. On 19th, November, the water temperature of 18.2°C quickly fell to 11.9°C on 2nd, December. Change rate reached 0.485°C/day, leading to poor situation that TN and TP removal efficiency plunged dramatically from 57.2% to 40.1% and from 70.1% to 31.7%, respectively.

On the other hand, sudden rise in temperature also could cause adverse behaviors of the plant on nutrient removal. In April, when the water temperature increased from 17.7°C (April 9th) to 19.4°C (April 17th) with change rate of 0.213°C/day, TP removal rate decreased from 76.3% to 64.2%. In May, the water temperature rose from 20.2°C (May 16th) to 23.2°C (May 20th) with change rate of 0.750°C/day, the removal rate of NH_4^+–N and TN decreased from 92% to 89% and from 63.5% to 51.7%, respectively. Similar results were observed in the period from July 22nd to July 29th of the observation as well as in some other experiments, showing that sudden increase of temperature brought about lower nutrient removal efficiency.

The effluent TN and TP concentrations were satisfactory within a temperature range of 10.8°C to 30.5°C during a year, indicating that the microbial populations could adapt to conditions of gradual temperature change. However, sudden temperature variations led to lower removal efficiency. Interestingly, sudden changes in temperature did not cause paralysis of the system, which could be explained by functional redundancy of microorganisms that may compensate the adverse effects of temperature changes to a certain degree, which was in an agreement with a previous study (Siripong and Rittmann, 2007).

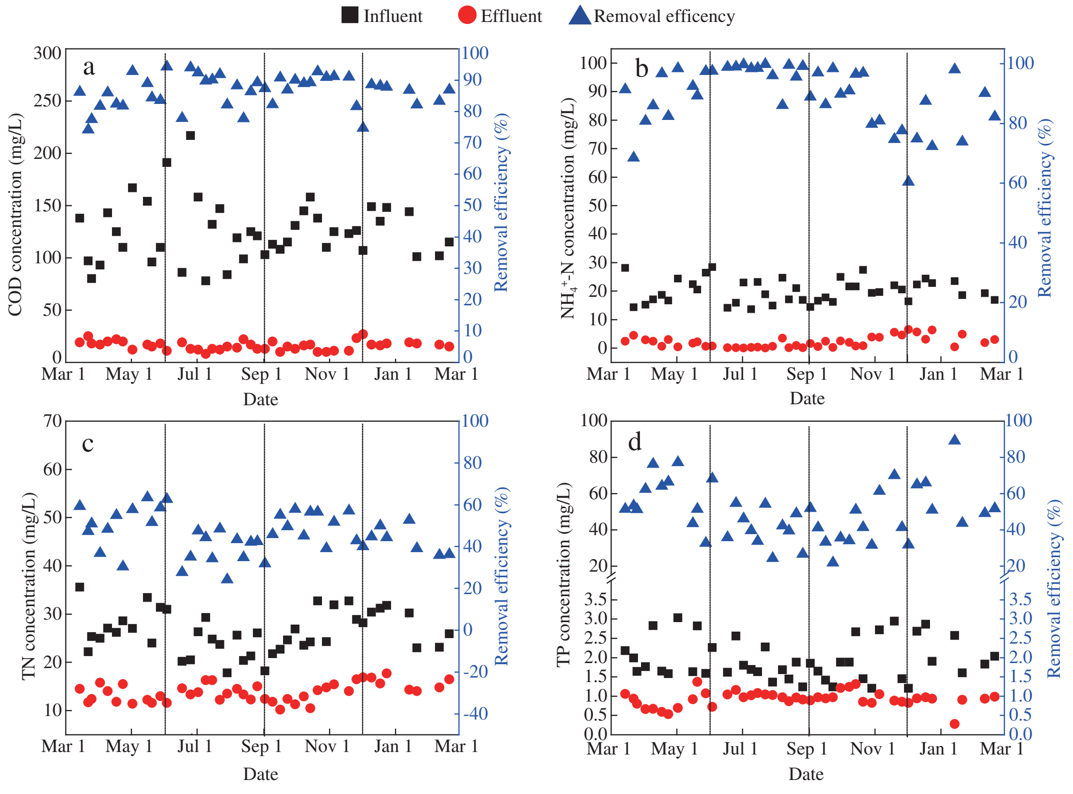

Fig. 3 – Removal performance of (a) COD, (b) NH_4^+–N, (c) TN, and (d) TP during a year (vertical dash lines separate different seasons). COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus.

2.2. *Pollutant changes along the length of the plant*

The changes in COD, nitrogen and phosphorus along the length of the full-scale WWTP are shown in Fig. 4. The pollutant concentrations of PC effluent were higher than that of the plant influent due to the internal loading pumped into the PCs. The concentrations of pollutants leaving anoxic zones were lower than that of PC effluent, at least partly due to dilution caused by the returned sludge flow. The COD concentration and utilization performance throughout the plant were investigated (Fig. 4a). The results indicated that COD utilization mainly occurred in anoxic zones and only a small fraction of total COD was consumed in the oxic zones. Fig. 4b shows that NH_4^+–N concentration in the effluent of oxic zones was considerably lower than that of AT effluent confirming that autotrophic bacteria play an important role in the oxidation of ammonia nitrogen in an aeration environment. TN and TP concentration decreased along the length of the plant starting from the effluent of PCs (Fig. 4c, d).

In the traditional sense, the role of SCs is to separate the water and sludge, but important biochemical reactions occur within the SCs in this study, because average TN concentration of SC effluent with 13.31 mg/L was lower than that of OT effluent with 14.60 mg/L (Fig. 4c), indicating that denitrification might occur. However, denitrification in the SCs could cause sludge floatation (Amanatidou et al. 2015). Therefore, it is important to strengthen the operation and management of the WWTP to prevent the deterioration of the effluent water quality.

2.3. *Mass balance*

Fig. 5 illustrates the overall mass balances of COD, NH_4^+–N, TN, and TP in the full-scale treatment system. The COD removal amount for the plant was 5.470×10^4 kg/day on average, as shown in Fig. 3a. The anoxic zones accounted for 88.20% of the total COD removal with 4.824×10^4 kg/day. The remaining COD removal occurred in the oxic zones, 10.27%, and in the SCs, 1.53% (Fig. 5a). This likely occurs because influent COD was extensively reduced before entering the oxic zones, which could explain the reason why COD in effluent did not change significantly. Similarly, Cao et al. (2013) showed that 74% of COD was efficiently utilized for denitrification and phosphorus release in a modified four step-feed wastewater treatment system.

In the plant, nitrification occurred mainly in the oxic area accounting for the majority, 78.77% of the total ammonia removal, as shown in Fig. 5b. According to nitrogen mass balance, 57.41% of nitrogen removal took place in the anoxic phase shown in Fig. 5c, which shows that anoxic denitrification capabilities play prominent roles in TN removal. The denitrifying bacteria in anoxic zones utilize nitrate or nitrite as electron acceptors, and raw sewage provides carbon source as a source of electron donors to allow the reduction of nitrate or nitrite to gaseous nitrogen. Although the anoxic zones were the main places of nitrogen removal pathway, nitrogen loss in the aerobic tanks (including microbial assimilation and SND) accounted for 31.73% of TN total removal. Actually, the contribution of nitrogen in biosolids accounted for only 3.3% (Lauver and Baker, 2000). Therefore, SND was the main nitrogen removal pathway in aerobic phase after balancing the data. The same SND phenomenon occurred in the modified UCT step-feed process, studied by Peng and Ge (2011), who reported 35% of TN removed through SND in oxic tanks due to unevenly distributed DO. Puznava et al. (2000) proposed that SND occurs within microbial flocs as a result of DO concentration gradients arising from diffusional limitations. Because of the limitation of oxygen diffusion, DO

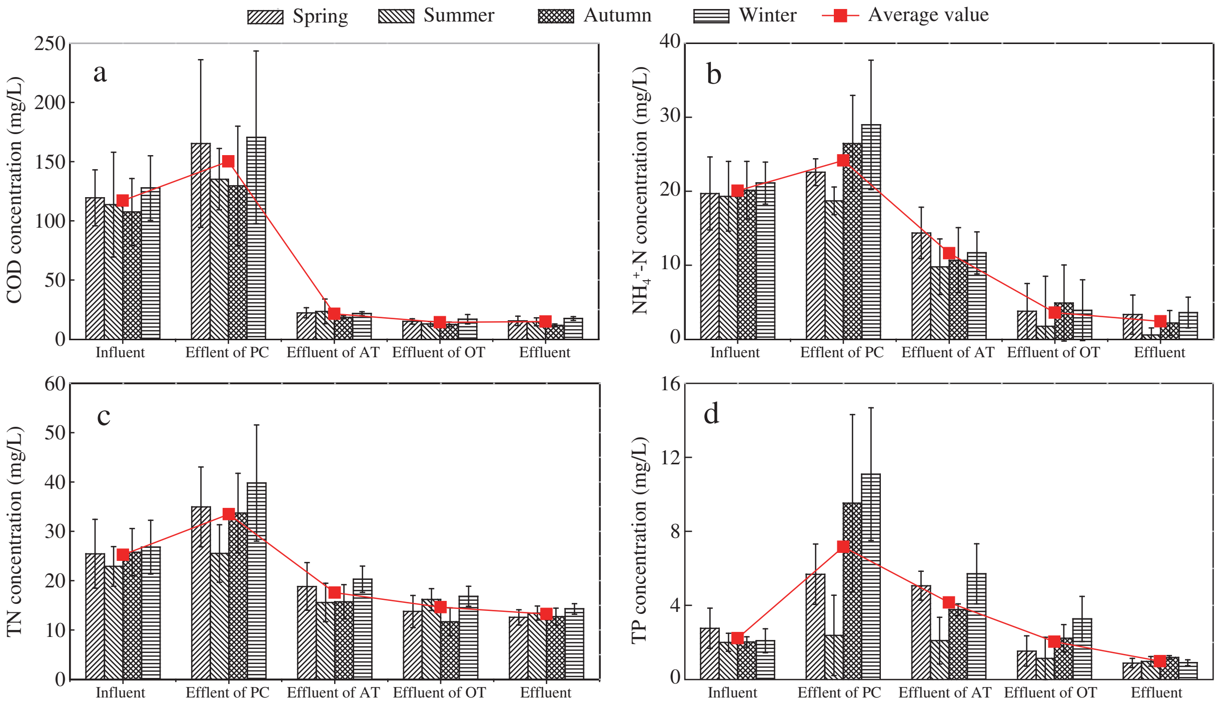

Fig. 4 – Evolution and removal of (a) COD, (b) NH_4^+–N, (c) TN, and (d) TP in the full-scale process. COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus.

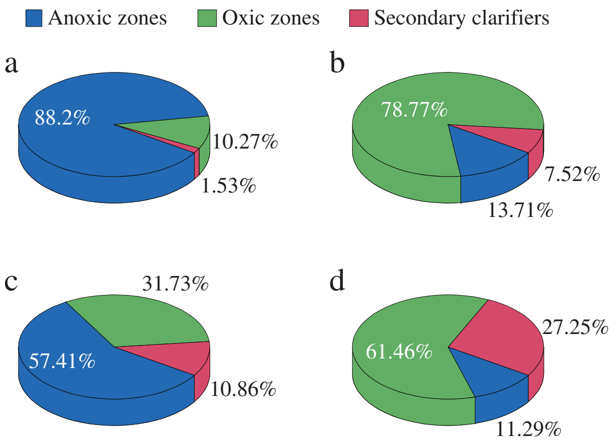

Fig. 5 – Contribution ratio of different structures for (a) COD removal, (b) NH_4^+–N removal, (c) TN removal, and (d) TP removal. Removing amount is the difference between the effluent pollutant concentration of the PCs and that of the SCs during a year period. COD: chemical oxygen demand; TN: total nitrogen; TP: total phosphorus; PCs: primary clarifiers; SCs: secondary clarifiers.

concentration is distributed on a gradient in the sludge floc, resulting in a hypoxic microenvironment in the center of the sludge floc, and an altered distribution of the heterotrophic bacteria for denitrification. SND is significant at very low DO concentrations (0.2 mg/L), while at high concentrations (more than 1.0 m/L), aerobic denitrification is strongly inhibited (Pochana et al. 1999). Another possible explanation is that aerobic denitrifying bacterium may be capable of converting nitrates to nitrogen gas consistent with the finding that the denitrification rate of the isolate bacteria (DCB-T6) was not affected by the presence of oxygen under oxic conditions (DO, 2–4 mg/L) (Pai et al. 1999). Therefore, aerobic denitrification could exhibit good and stable performance in the oxic tanks on different DO levels.

Phosphorus removal from wastewater is achieved through the removal of waste activated sludge containing high polyphosphate. In this study, more than 60% of net phosphorus was removed during oxic phase, as shown in Fig. 5d. In addition, there was significant phosphorus removal in anoxic zones, which might be associated with denitrifying phosphorus removal. PAOs can consume volatile fatty acids from the influent substrate to synthesize polyhydroxyalkanoate (PHA) and release phosphate. However, the denitrifying bacteria were able to utilize the limited carbon source in the influent to a greater extent than were the PAOs (Wang et al. 2012). When carbon inputs decreased, some PAOs (*i.e.*, DPAOs) could use nitrate or nitrite as electron acceptors and, therefore, perform phosphorus uptake and denitrification simultaneously. In the plant, the organic matter in the SCs is very low, generally less than 10 mg/L, while the nitrate concentration was always more than 10 mg/L, therefore, DPAOs could use nitrate to achieve nitrogen removal and phosphorus reduction in the SCs, as shown in Fig. 4c, d.

2.4. Factors affecting performance

2.4.1. Temperature variation and the counteraction

Temperature not only affects the metabolic activities of the microbial population but also can influence the diversity of the bacterial community (Flowers et al. 2013). The temperature changed significantly with the season in the Jiangxinzhou full-scale WWTP. The average temperatures of sewage in the Nanjing region are around 20.2°C, with a minimum of 10.8°C and a maximum of 30.5°C. Carbon, nitrogen, and phosphorus removal activities of the full-scale WWTP at different seasonal temperatures on average are shown in Table 3. There was no obvious difference in COD removal efficiency between different seasons. The plant exhibited good performance of nitrogen removal activities in spring and autumn seasons. However, nitrogen removal amount was the lowest in summer during a year, likely due to low influent nitrogen loading. When the temperature was as low as 12.2°C on average in winter, the TN removal was still comparable with other seasons in which water temperature was higher, indicating that microorganism could adapt to conditions of gradual temperature change and, therefore, counteract adverse influence caused by low temperature (Fig. 2).

The plant achieved a high level of phosphorus removal (5.44×10^2 kg/day) in winter (Table 3). Phosphorus removal activities were poorer in summer and autumn compared to the other two seasons likely due to lower influent loading and higher water temperature. Sudden change of temperature was also one of the reasons for the reduction of phosphorus removal activities. As the temperature rises, the portion of energy required for activity maintenance of PAOs increases substantially which reduces the energy availability for cell reproduction (Panswad et al. 2003). Erdal et al. (2003) proposed that better performance of the system was due to a reduced competition for substrate in the non-oxic zones, which resulted in an increased PAO population as temperature decreased that coincided with better performance of EBPR at lower temperatures. A high catalytic activity of PAOs at low temperatures is the main physiological adaptation to cold at the enzymatic level (Feller, 2003), which could also explain why the WWTP exhibited the best phosphorus elimination from wastewater at the lower temperatures in winter.

2.4.2. Nitrate recirculation

Nitrogen removal is not possible without nitrification according to the conventional two-step nitrogen removal theory. In the

Table 3 – Average carbon, nitrogen and phosphorus removal activities of the full-scale WWTP (the first-stage project) during different seasons.

Season	Activity (kg/day)		
	Carbon (COD)	Nitrogen (N)	Phosphorus (P)
Spring (Mar, Apr, May)	4.06×10^4	4.96×10^3	4.92×10^2
Summer (Jun, Jul, Aug)	3.75×10^4	3.80×10^3	3.92×10^2
Autumn (Sep, Oct, Nov)	4.22×10^4	4.88×10^3	2.48×10^2
Winter (Dec, Jan, Feb)	4.07×10^4	3.90×10^3	5.44×10^2

WWTP: wastewater treatment plant.

nitrification step, ammonia is transferred into nitrate and/or nitrite in the oxic phase. Then in the second step of denitrification, nitrate and/or nitrite is internally circulated from oxic zones to anoxic zones, and is reduced to gaseous nitrogen using influent substrate as a carbon source. In this study, there was no internal recycling and the main source of NO_3^-–N came from the recycled sludge for denitrification, because the anoxic reactor was fed with influent with very low nitrate and nitrite concentrations (0.40 and 0.06 mg/L respectively (Table 2)). Moreover, the anoxic–oxic process without nitrate recirculation provided significant advantages over the conventional process of pumping mixed liquid from aerobic to anoxic chambers, because the high DO concentration in the nitrate liquid from the aeration pools made it difficult to maintain the ideal environment for the denitrifying bacteria in the anoxic zones. Such, a lack of nitrate recirculation could reduce the impact of DO on the anoxic environment. Furthermore, denitrifying phosphorus removal occurred in the full-scale plant because of the satisfactory anoxic environmental.

2.4.3. *Internal loading*

Comparisons of carbon to nutrient ratios over a range of internal carbon sources are illustrated in Table 4. In this study, there was no significant difference between raw wastewater and a mixture of raw wastewater and internal loading in terms of the C/N ratio. The C/P ratio decreased from 59.94 in the raw wastewater to 20.95 in the mixture. Throughout the observation period, the processes of denitrification and SND were exhibited successfully for nitrogen removal, and both denitrifying phosphorus removal and oxic phosphorus uptake were noticeable. Wang et al. (2013) achieved complete phosphorus removal in the anaerobic–anoxic/nitrification (A2N) two-sludge SBR, where the C/P ratio was 20.6. Park et al. (2011b) considered that ozonolysate of excess sludge as a carbon source could be enhanced for phosphorus removal with a C/P ratio of 42–57 using a modified intermittently decanted extended aeration process. However, the addition of disintegrated sludge resulted in low C/P ratio of less than 13.6 that would slow the process (Kampas et al. 2007).

2.5. *Proposals for plant operation*

For a specific application, the available alternatives for nutrient elimination need to be evaluated for a multitude of parameters, including costs, chemical and energy requirements, bio-energy recovery, process reliability, and environmental impact. In the term of a full-scale WWTP, the cost effectiveness of operation is particularly important. Modification of process parameters in response to changing conditions should be performed to increase performance and lower operational costs, rather than continual operation under fixed conditions. When the influent TN concentration is low (*i.e.*, less than 35 mg/L), an anoxic–oxic activated sludge system without internal nitrate recirculation would be competitive alternative for nutrient removal. Recycled sludge could be used to recycle nitrified mixed liquor for denitrification in the anoxic zones. Furthermore, a lack of internal recycling allows operation cost savings.

Among the factors affecting the performance of the full-scale WWTP, DO and SRT are likely the key contributors. Maintenance of appropriate DO concentration could promote the growth of denitrifying bacteria in the anoxic phase, transforming nitrate

Table 4 – Comparisons of carbon to nutrient ratios for different conditions.

Contents	C/N ratio	C/P ratio	Results	Reference
Influent[a]	4.65	59.94	This study	This study
Effluent of PCs[b]	4.48	20.95	This study	This study
Disintegrated surplus activated sludge	9.0[c]	8.0[d]	The carbon source addition improved phosphate release and denitrification. Overall, this type of sludge allowed BNR and had lower carbon to nutrient ratios, but had the potential for negative effects on nutrient recycling in BNR.	Soares et al. (2010)
Ozonolysate of excess sludge	11.2–15.2	42-57[d]	The removals of nitrogen and phosphorus were simultaneously enhanced by addition of the ozonolysate.	Park et al. (2011b)
Mechanical sludge disintegration	~70[e]	7.0–13.6[d]	The increase in the concentration of phosphorus reached 480 mg/L for 15 min of retention time and led to a relatively low ratio of SCOD:PO_4^{3-}–P (~13.6), which indicated that the addition of disintegrated sludge would deteriorate the phosphorus removal.	Kampas et al. (2007)

C: carbon; N: nitrogen; P; phosphorus; PCs: primary classifiers; BNR: biological nutrient removal.

[a] Raw wastewater.

[b] Mixture of raw wastewater and internal loading.

[c] Soluble chemical oxygen demand (SCOD):NO_3^-–N.

[d] SCOD:PO_4^{3-}–P.

[e] SCOD:NH_4^+–N.

and/or nitrite to gaseous nitrogen. DO should be below 0.25 mg/L in the anoxic tanks and aeration should be reduced when the DO concentration exceeds 2 mg/L in the aerobic pool. In principle, using the method of gradually reducing aeration along the lengths of aerobic pool to enhance complete nitrification is reasonable. On the other hand, SRT is a prerequisite for the existence of denitrifying bacteria, PAOs, and DPAOs.

A larger ratio of anoxic to total volume might be efficient to enhance denitrifying phosphorus removal and achieve an acceptable effluent quality even at limiting carbon (Sun et al. 2014). Besides, if the exiting configuration was modified to add anaerobic zones for the full release of phosphorus, further phosphorus removal efficiency might be achieved. Those proposals need to be evaluated in future studies.

3. Conclusions

This study provides an evaluation of performance and nutrient removal mechanisms in the treatment of low strength wastewater. Conventional denitrification was the main pathway for TN removal in the anoxic tanks by 57.41%. Nitrogen elimination in the oxic tanks was mainly contributed by SND. There was significant phosphorus removal in anoxic zones, which might be associated with DPAOs achieving phosphorus uptake and denitrification simultaneously. Practical experience demonstrated that the microbial populations could adapt to conditions of gradual temperature change, which maintained the plant overall stability. Sudden changes in temperature did not cause paralysis of the system just lower removal efficiency, which could be explained by functional redundancy of microorganisms that may compensate the adverse effects of temperature changes to a certain degree. Anoxic–oxic process without internal recycling has great potential to treat low strength wastewater (*i.e.*, TN < 35 mg/L) and reduce operation costs.

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