Simultaneous phosphorus uptake and denitrification by EBPR-r biofilm under aerobic conditions: effect of dissolved oxygen
Pan Yu Wong, Maneesha P. Ginige, Anna H. Kaksonen, Ralf Cord-Ruwisch, David C. Sutton and Ka Yu Cheng

ABSTRACT
A biofilm process, termed enhanced biological phosphorus removal and recovery (EBPR-r), was recently developed as a post-denitrification approach to facilitate phosphorus (P) recovery from wastewater. Although simultaneous P uptake and denitrification was achieved despite substantial intrusion of dissolved oxygen (DO > 6 mg/L), to what extent DO affects the process was unclear. Hence, in this study a series of batch experiments was conducted to assess the activity of the biofilm under various DO concentrations. The biofilm was first allowed to store acetate (as internal storage) under anaerobic conditions, and was then subjected to various conditions for P uptake (DO: 0 – 8 mg/L; nitrate: 10 mg-N/L; phosphate: 8 mg-P/L). The results suggest that even at a saturating DO concentration (8 mg/L), the biofilm could take up P and denitrify efficiently (0.70 mmol e^- /g total solids*hr). However, such aerobic denitrification activity was reduced when the biofilm structure was physically disturbed, suggesting that this phenomenon was a consequence of the presence of oxygen gradient across the biofilm. We conclude that when a biofilm system is used, EBPR-r can be effectively operated as a post-denitrification process, even when oxygen intrusion occurs.

Key words | electron acceptors, phosphate accumulating organisms, post-denitrification

INTRODUCTION
Low effluent concentrations for total phosphorus (TP) and total nitrogen (TN) are increasingly being imposed on wastewater treatment plants worldwide. Europe and North America in particular have enforced discharge limits of 0.1 mg/L for TP and 3 mg/L for TN (Boltz et al. 2012). While the strict TP limits are largely achieved through chemical precipitation, biological post-denitrification is applied to meet the TN discharge limit. With much of the readily biodegradable organic carbon in the influent being oxidised during upstream aerobic/anoxic oxidative processes, post-denitrification is heavily reliant on the addition of carbon sources (e.g., methanol) (Wei et al. 2014). The cost of adding external carbon is a significant burden to the wastewater industry, and one way to offset this cost is through resources recovery from wastewater.

Among many resources in wastewater, phosphorus (P) is of interest because it is a non-renewable resource, and its scarcity for agricultural purposes could potentially threaten global food security (Cordell et al. 2011). Wong et al. (2013) proposed and validated a post-denitrification process, termed enhanced biological phosphorus removal and recovery (EBPR-r), that facilitates nitrogen (N) removal and also enables P recovery from wastewater. The EBPR-r process involves two steps in which a biofilm consisting of phosphate accumulating organisms (PAOs) is alternately exposed to a wastewater stream and a separate recovery stream. As the PAOs can use nitrate (NO3^-) as a final electron acceptor for P uptake, the first step of the process facilitates both denitrification and P removal from the wastewater. In the absence of soluble carbon in wastewater, the internal carbon storage polymers (i.e., such as poly-β-hydroxy-alkanoates; PHA) act as electron donors and an energy source to facilitate this step. In the subsequent anaerobic step, the biologically captured P is released into a smaller recovery stream. External carbon (i.e., acetate) is added to facilitate P release, and the biofilm simultaneously replenishes its carbon reserves by storage of carbon supplied to the recovery stream. The
volume difference maintained between the wastewater and the recovery stream (e.g., a ratio of 4:1) enables recovery of P as a concentrated solution.

It has been suggested that dissolved oxygen (DO) concentrations >0.2 mg/L inhibit denitrification (Gerardi 2003). In conventional post-denitrification processes, anoxic conditions prevail largely because of the rapid consumption of oxygen (O₂) by heterotrophic bacteria during carbon oxidation. Facilitating an anoxic environment in a similar manner is not feasible in the EBPR-r process, as external carbon is supplied only to the recovery stream. Except for the intracellularly stored carbon, no soluble carbon is available in the wastewater stream. Under these conditions, Wong et al. (2013) observed elevated DO concentrations (DO >6 mg/L) during N and P removal, but surprisingly the high DO concentrations did not appear to inhibit denitrification and P removal.

Storage-driven denitrification is commonly observed in simultaneous nitrification, denitrification and phosphorus removal processes (SNDPR) (Zeng et al. 2003; Lemaire et al. 2008) in which nitrifiers make use of the O₂ to facilitate partial nitrification under low DO concentration (<1 mg/L). Hence, they are largely responsible for creating the anoxic conditions for denitrification. Findings based on the SNDPR process are not directly useful for understanding denitrification in the EBPR-r process, which has been observed to take place at much higher levels of DO (>6 mg/L) with little contribution of nitrifiers (Wong et al. 2013). If large-scale EBPR-r is to be implemented, a clear understanding of the impact of DO on post-denitrification and P removal is critical.

Hence, the aim of this study was to explore the impact of DO on simultaneous storage-driven denitrification and P removal by an EBPR-r biofilm. The specific objectives included assessment of the importance of the biofilm structure, and the levels of DO that can be tolerated by the bacteria without impeding denitrification. First, batch experiments were conducted to quantify P uptake, NO₃⁻ removal and O₂ consumption kinetics by an intact biofilm exposed to various concentrations of DO (0–8 mg/L) and NO₃⁻ (0–50 mg-N/L). Second, the EBPR-r biofilm was physically disturbed to investigate the effect of biofilm structure on the P and N removal efficiencies.

**MATERIALS AND METHODS**

**Reactor configuration and synthetic wastewater**

A laboratory-scale sequencing batch biofilm reactor (master reactor) was operated continuously in EBPR-r configuration for a 2-year period, as described previously (Wong et al. 2013). A schematic diagram of the reactor process is shown in Figure S1 in the supporting information (available online at http://www.iwaponline.com/wst/072/325.pdf). In brief, 1,000 biofilm carriers (Kaldnes® K1 polyethene) were equally distributed among eight adjoining stainless steel mesh compartments. Over a 6-h cycle the biofilm carriers were alternately exposed for 4 h to a wastewater stream (7.2 L) for P uptake, and for 2 h to a separate recovery stream (1.8 L, 25% of the volume of the wastewater stream) for anaerobic P release.

Both the wastewater and recovery streams contained a standard growth medium consisting of (per L of deionised (DI) water): 39 mg MgSO₄, 20 mg CaCl₂·2H₂O, 11 mg NH₄Cl (5 mg/L NH₄⁺-N), 200 mg NaHCO₃ and 0.5 mL of a trace element solution (Wong et al. 2013). The wastewater stream also contained 8 mg-P/L phosphate (supplemented as 1 M phosphate buffer), 10 mg-N/L nitrate (as sodium nitrate) and 11.6 mg/L N-Allylthiourea, the latter added to prevent nitrification during the P-uptake phase (Ginestet et al. 1998). To restore intracellular PHA reserves during the anaerobic P release, 520 mg/L sodium acetate was added to the recovery stream, which corresponded to 400 mg/L chemical oxygen demand. Concentrated stock solutions (15×) of the media comprising each of the streams were prepared, and the pH was adjusted to 7.0 ± 0.2 using 2 M HCl. Defined volumes of the stock solution and DI water were simultaneously pumped into the reactor at the beginning of each phase to achieve the desired concentrations.

**Kinetic experiments using intact biofilm**

To elucidate the use of O₂ and NO₃⁻ by the EBPR-r biofilm when both electron acceptors were present, two sets of experiments were performed in duplicate (Figure 1). (1) The activity of the biofilm was investigated using an initial NO₃⁻ concentration of 10 mg-N/L, but the bulk DO...
concentration was varied (0, 2, 4, 6 and 8 mg/L). (2) Constant influent bulk DO concentration of 8 mg/L was maintained, but the initial NO\textsubscript{3}– concentration was varied (0, 5, 10, 20, 30 and 50 mg-N/L).

In each experiment, biofilm carriers (~330) were removed from two compartments of the master reactor at the end of the anaerobic P-release phase, at which time the biomass had stored PHAs (Bond et al. 1999). The carriers were immediately transferred into a column reactor (440 mL working volume; diameter 45 cm, height 300 cm; Figure 1), where biological P uptake was triggered by recirculating (7.85 L/h; Masterflex\textsuperscript{®}, Masterflex, Vernon Hills, IL, USA) a P-containing wastewater stream (2.4 L, 8 mg-P/L) for 4 h. Two luminescent DO probes (PDO\textsubscript{2}; Barben Analyser Technology, Barben Analyser Technology, Carson City, NV, USA) were installed in the recirculation line, one before (DO\textsubscript{in}) and one after (DO\textsubscript{out}) in the column reactor. The influent DO\textsubscript{in} was controlled at 0–8 mg/L by sparging air or nitrogen into the aeration vessel (2.0 L), while NO\textsubscript{3}– was added (as 4 M NaNO\textsubscript{3}) into the wastewater stream to give an initial concentration of 0–50 mg-N/L. The monitoring and control of DO were performed using a programmable logical controller and software (LabVIEW, National Instruments, LabVIEW, National Instruments, Austin, TX, USA).

Liquid samples were collected from the reactor every 15–45 min and immediately filtered using a 0.22 μm pore size syringe filters (Acrodisc\textsuperscript{®} PF; Pall Corporation, Pall Corporation, Port Washington, NY, USA). The concentrations of soluble NO\textsubscript{2}–N, NO\textsubscript{3}–N, and PO\textsubscript{4}–P in the filtrates were determined using ion chromatography (ICS-3000, DIONEX, Sunnyvale, CA, USA). Changes in the concentrations of PO\textsubscript{4}–P, NO\textsubscript{2}–N, NO\textsubscript{3}–N + NO\textsubscript{2}–N, and NO\textsubscript{3}–N were plotted against time, and the specific P uptake rate (PUR) and the NO\textsubscript{x} removal rate (expressed as mg/L×h) were recorded as the slope of the steepest part of the curves. These rates were normalised using the respective total solids (TS) concentrations, and expressed in mmol/gTS×h. The TS was obtained by subtracting the weight of 50 biofilm-free carriers from the dry weight (dried at 60 °C) of 50 EBPR-r carriers supporting biofilm (Wong et al. 2015). To compare the reduction (electron-accepting) kinetics of NO\textsubscript{3}– and O\textsubscript{2} using the storage reducing power (i.e., PHAs), both the oxygen uptake rate (OUR) and the NO\textsubscript{x} removal rate were transformed into a common unit, termed the electron-accepting rate (mmol e–/gTS×h). The percentage of electrons used for O\textsubscript{2} and NO\textsubscript{3} reduction was calculated by assuming that all the electrons from the storage were captured by either O\textsubscript{2} or NO\textsubscript{3}. The details of the calculations are given in the supporting information (available online at http://www.iwaponline.com/wst/072/325.pdf).

**Kinetic experiments using dislodged biomass**

To confirm whether the observed denitrification in the presence of O\textsubscript{2} was due to the presence of an oxygen gradient across the biofilm, kinetic experiments were conducted using biomass dislodged from the carriers. To obtain the biomass, biofilm carriers (~330) were removed from the master reactor at the end of a P-uptake phase (low in PHA storage) and placed into growth medium in a 500 mL flask. Attached biofilm was physically removed by shaking the carriers in standard medium for 2 min. To break down the size of the flocs, the suspended biofilm was repeatedly drawn up and expelled through a needle (gauge 19–1/2) using a 50 mL syringe. An acetate-containing recovery stream (200 mL) was added to the dislodged biomass for PHA replenishment for 2 h. Thereafter, the biomass was concentrated by centrifugation and washed twice with standard medium (without N, P and C) under anaerobic conditions to remove any excess acetate. The PHA-rich biomass was then resuspended in 220 mL standard medium under anaerobic conditions for use in batch experiments.

Four batch reactors (250 mL Schott bottles, Mainz, Germany) were operated in parallel to compare the denitrification ability of the biomass in the presence of 10 mg-N/L NO\textsubscript{3}– for 4 h: (1) oxygenated, with 8 mg-P/L phosphate; (2) oxygenated, without phosphate; (3) anoxic, with 8 mg-P/L phosphate; and (4) anoxic, without phosphate. To initiate the experiment, 50 mL of suspended biomass was added to 200 mL of synthetic wastewater. The oxygenated and anoxic condition was achieved by continuously sparging air and nitrogen into the liquid, respectively. Mixing was achieved using a multi-position magnetic stirrer (400 rpm; RT10, IKA, Rawang, Selangor, Malaysia). To confirm the observation of denitrification under oxygenated conditions, on a different day the batch tests for the aforementioned conditions (1) and (2) were repeated.

Liquid samples were withdrawn and filtered for analysis of soluble NO\textsubscript{2}–N, NO\textsubscript{3}–N, and PO\textsubscript{4}–P. The mixed liquor suspended solids (MLSS) value for the suspended biomass was measured according to the standard method (American Public Health Association, American Water Works Association & Water Environment Federation (USA) 1995). The PUR, NO\textsubscript{x} removal rate and the NO\textsubscript{2} accumulation rate were normalised with the solids concentration, and expressed as mmol/gMLSS×h. The size distribution of the suspended biomass flocs, measured using a laser particle sizer (Malvern Mastersizer 2000 Analyzer, Malvern, Worcestershire, UK), was determined by an external laboratory (CSIRO, Division of Mineral Particle Analysis Service, Waterford, Australia).
RESULTS AND DISCUSSION

Storage-driven denitrification and P uptake at very high DO concentrations

In a previous study we observed that an EBPR-r biofilm could uptake phosphate and remove NO3 in the presence of saturating DO (Wong et al. 2015). However, as both NO3 and O2 were provided to the biofilm, we were unable to distinguish the independent effects of these electron acceptors. Hence, in the present study batch experiments were conducted to assess the influence of each of O2 and NO3 as sole electron acceptors on P removal. The EBPR-r biofilm could readily use either O2 or NO3 for P uptake, or both (Figure S2 in the supporting information, available online at http://www.iwaponline.com/wst/072/325.pdf). The highest PUR was observed when O2 was provided, either alone (0.038 mmol-P/gTS*h) or in combination with NO3 (0.043 mmol-P/gTS*h). When NO3 was used as the sole electron acceptor, the PUR was markedly reduced by 30%, from 0.043 to 0.030 mmol-P/gTS*h (Table 1). According to Kuba et al. (1996), the energy (adenosine triphosphate) production during oxidative phosphorylation with NO3 as the electron acceptor is approximately 40% less than occurs with O2 as the acceptor. While the reduced energy from oxidative phosphorylation based on NO3 could have contributed to the 30% reduction in PUR, the possible role of a low-abundance denitrifying PAO population in the biofilm should not be overlooked.

As expected, the highest NOx removal rate (0.076 mmol-N/gTS*h) was observed when NO3 was supplied as sole electron acceptor. When supplemented with DO (8 mg/L), 70% of the denitrification efficiency of the biofilm was retained (Table 1), indicating that the enriched EBPR-r biofilm could denitrify under a very high DO concentration.

The oxygen gradient across the biofilm enabled denitrification in the presence of DO

To elucidate the effect of DO and NO3 loading on the P uptake and denitrification behaviour of the EBPR-r biofilm, two sets of experiments were conducted in which the concentration of one electron acceptor was maintained constant while the concentration of the other was varied. The concentration profiles for each experiment set are illustrated in the supplementary data (Figure S3, available online at http://www.iwaponline.com/wst/072/325.pdf).

At an NO3 concentration of 10 mg-N/L, increasing the bulk DO concentration from 0 to 8 mg/L increased the PUR by 43% (from 0.030 to 0.043 mmol/gTS*h) (Figure 2(a)). A linear relationship ($R^2 = 0.999$) was obtained between the OUR and the applied bulk DO concentration (Figure 2(b)).

### Table 1 | The result for the intact EBPR-r biofilm under three electron acceptor scenarios

<table>
<thead>
<tr>
<th>Electron acceptors</th>
<th>O2</th>
<th>O2 + NO3</th>
<th>NO3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phosphate uptake rate (mmol-P/gTS*h)</td>
<td>0.038 ± 0.002</td>
<td>0.043 ± 0.002</td>
<td>0.030 ± 0.004</td>
</tr>
<tr>
<td>NO3 removal rate (mmol-N/gTS*h)</td>
<td>72 ± 6</td>
<td>77 ± 1</td>
<td>53 ± 1</td>
</tr>
<tr>
<td>N removal efficiency (%)</td>
<td>n.a.</td>
<td>38 ± 5</td>
<td>54 ± 1</td>
</tr>
<tr>
<td>P/N ratio (g-P/g-N as NO3)</td>
<td>1.36 ± 0.05</td>
<td>0.54 ± 0.03</td>
<td></td>
</tr>
</tbody>
</table>

Figure 2 | Effect of bulk DO (0–8 mg/L) and initial NO3 (0–50 mg-N/L) concentration on: (a) and (e) phosphate uptake rate; (b) and (f) OUR and NO3-N removal rate; (c) and (g) electron accepting rate for O2 and NO3; and (d) and (h) the percentage of electrons used for PHA oxidation using electrons generated by O2 and NO3 reduction.
Such a first order kinetic behavior suggested that the biofilm was limited by O$_2$ (Meyer et al. 2005). Increasing the bulk water DO concentration from 0 to 8 mg/L could result in deeper penetration of O$_2$ into the biofilm, triggering the higher OURs (from 0 to 0.091 mmol/gTS*h) and the lower denitrification rates (1.5-fold reduction from 0.076 to 0.052 mmol-N/gTS*h) (Figure 2(b)). This result confirms the presence of an oxygen gradient in the biofilm, with the inner anoxic environment facilitating the observed denitrification despite the bulk water being saturated with oxygen.

In contrast, when the bulk DO concentration was fixed at 8 mg/L, increasing the NO$_3$ concentration from 0 to 50 mg-N/L resulted in an increase in the denitrification rate (from 0 to 0.096 mmol-N/gTS*h) (Figure 2(f)). This result was expected because increased NO$_3$ availability in the bulk water could facilitate the penetration of NO$_3$ into the deeper anoxic layers of the biofilm, as observed in the conventional EBPR process under anoxic conditions (Ahn et al. 2001; Yuan & Oleszkiewicz 2010; Zhou et al. 2010). In terms of P uptake, when only a slight decrease in the PUR was observed with increasing NO$_3$ concentration (16% of overall inhibition, from 0.038 to 0.032 mmol/gTS*h; Figure 2(e)). No increase in PUR and a continuous increase in denitrification (evident in Figure 2(f)) could be a result of some NO$_3$ being utilised by denitrifying glycogen accumulating organisms (denitrifying GAOs) in the biofilm. The slight decrease in PUR is consistent with the findings of Yuan & Oleszkiewicz (2010), who observed an increased anoxic PUR and a decreased aerobic PUR with increasing NO$_3$ concentrations in the bulk water. It is also possible that when the NO$_3$ concentration increased in the bulk water, some PAOs were able to switch from using O$_2$ as electron acceptor to use of NO$_3$. With more PAOs using NO$_3$ as electron acceptor, the OUR and PUR may have decreased. Whether the elevated level of O$_2$ inhibited the activity of denitrifying PAOs remains unclear, and should be the subject of further research.

More than half of the stored reducing power was used for denitrification at 8 mg/L of DO

In the absence of a soluble carbon, the observed P uptake and denitrification activities were driven by internal carbon storage (e.g., PHAs) in the EBPR-r biofilm. To compare the reduction kinetics of NO$_3$ and O$_2$ for the biofilm, the OUR and NO$_x$ removal rates (Figure 2(b) and 2(f)) were transformed into a common unit (electron accepting rate; Figure 2(c) and 2(g)), and were also expressed as a percentage of the electrons used for internal carbon oxidation (Figure 2(d) and 2(h)).

At a bulk DO concentration of 8 mg/L and an NO$_3$ concentration of 0 mg-N/L, the electrons stored in the biofilm were predominately used to reduce O$_2$ at a maximum electron accepting rate of 0.46 mmol e$^-$/gTS*h (Figure 2(g)). With increasing bulk water NO$_3$ concentration the electron reduction rate for O$_2$ decreased only slightly, whereas the electron reduction rate for NO$_3$ increased dramatically. At NO$_3$ concentration of approximately 8 mg-N/L, the biofilm appeared to be transferring electrons at a similar rate to both O$_2$ and NO$_3$. At a bulk water NO$_3$ concentration exceeding 8 mg-N/L, NO$_3$ became the dominant electron acceptor (>50%; Figure 2(h)). Thus, an NO$_3$ concentration of >8 mg-N/L in the wastewater is favourable for efficient NO$_3$ reduction.

At an NO$_3$ concentration of 10 mg-N/L and a DO concentration of 0 mg/L, the electrons stored in the biofilm were solely used to reduce NO$_3$ at a maximum electron accepting rate of 0.98 mmol e$^-$/gTS*h (Figure 2(c)). When the DO concentration increased from 0 to 8 mg/L, the proportion of electrons accepted by O$_2$ gradually increased from 0 to 35% (Figure 2(d)). It is noteworthy that even at such a high bulk water DO concentration the biofilm was able to channel approximately 65% of the electrons to NO$_3$ reduction. This unique ability of the biofilm to reduce NO$_3$ in the presence of DO is critical for the EBPR-r process, and is probably a consequence of the presence of an oxygen gradient across the biofilm, as discussed above.

The biofilm structure is essential for denitrification in the presence of O$_2$

To determine whether the cells in the biofilm could continue to denitrify when the DO gradient was disrupted, the biofilm was removed from the carriers and physically disturbed to form suspended aggregates (mean size 185 ± 11 µm). The aerobic and anoxic ratio difference of P uptake (PUR$_{aer}$/PUR$_{anx}$) and denitrification activities of the suspended biomass (MLSS of 1.03 ± 0.05 g/L) were compared with that of the intact biofilm.

Similar to the intact biofilm, the suspended biomass showed the highest PUR (0.53 mmol-P/gMLSS*h) when both O$_2$ and NO$_3$ were provided as electron acceptors (Table 2). When NO$_3$ was the sole electron acceptor the PUR decreased by 38%, which is consistent with the observed decline in the previous experiments with the intact biofilm (~50% in Table 1). Thus, the decreased PUR observed with NO$_3$ was not caused by the biofilm structure, but rather appeared to be influenced by the type of electron acceptor.
As with the intact biofilm, the highest level of denitrification by the suspended culture was observed in the absence of DO (Table 2). However, when O2 (DO > 6 mg/L) was introduced into the bulk water, a six-fold decrease in the denitrification rate (from 1.00 to 0.16 mmol-N/gMLSS*h) was observed. This was remarkably different from the intact biofilm, for which the respective decline in the denitrification rate was only 1.5-fold (Table 1). One plausible explanation for the decrease in denitrification activity after the disruption of oxygen gradient is the inhibition of nitrate reductase, which is a membrane-bound enzyme that catalyses the reduction of NO3 to NO2, and is sensitive to O2 (Ogunseitan 2005). Under anoxic conditions, the rate of reduction of NO3 (2.23 mmol-N/gMLSS*h) far exceeded that of NO2, resulting in the accumulation of the NO2 observed in this study (an accumulation rate of 2.39 mmol-N/gMLSS*h, Figure 3(c)). Only when NO3 became limiting was an overall reduction in NOx observed. Accumulation of NO2 is a common observation during denitrification and has been extensively discussed in the literature (Ahn et al. 2001; Zhou et al. 2010). Under aerobic conditions the nitrate reductase enzyme was exposed to O2, resulting in inhibition of the enzyme and a significant decrease in the NO3 reduction rate (Figure 3(a)). These results demonstrate that maintenance of the biofilm structure for bacterial growth is critical for the EBPR-r process to achieve satisfactory rates of denitrification, particularly when strict anoxic conditions cannot be maintained.

### The dependency of denitrification on P

The observed denitrification in suspended biomass could be performed by either PAOs or other non-PAO bacteria, including GAO. By definition, GAOs do not require storage of P under either aerobic or anoxic conditions (Oehmen et al. 2006). To investigate the denitrification activities of non-PAO organisms, the suspended biomass experiment was conducted aerobically and anoxically with no P in the bulk water.

Under anoxic conditions the absence of phosphate decreased the NOx removal rate only by 22%, from 1.00 mmol-N/gMLSS*h in the presence of P to 0.78 mmol-N/gMLSS*h in the absence of P (Table 2). This confirmed the role of non-PAO organisms (e.g., denitrifying GAOs) in

<table>
<thead>
<tr>
<th>Rates</th>
<th>Aerobic</th>
<th>Anoxic</th>
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<tbody>
<tr>
<td>With P</td>
<td>0.53 0.33</td>
<td>0.16 1.00</td>
</tr>
<tr>
<td>NO3 removal rate (mmol-N/gMLSS*h):</td>
<td>2.23</td>
<td>n.a.</td>
</tr>
<tr>
<td>NO2 formation rate (mmol-N/gMLSS*h):</td>
<td>2.39</td>
<td>n.a.</td>
</tr>
<tr>
<td>No P</td>
<td>n.a. n.a.</td>
<td>n.a.</td>
</tr>
<tr>
<td>NO3 removal rate (mmol-N/gMLSS*h):</td>
<td>0.00 0.78</td>
<td>n.a. 1.58</td>
</tr>
<tr>
<td>NO2 formation rate (mmol-N/gMLSS*h):</td>
<td>n.a. 1.89</td>
<td>n.a.</td>
</tr>
</tbody>
</table>

Table 2 | The result for the dispersed biofilm under four conditions tested
biofilms carrying out denitrification. However, exposing the suspended biomass to aerobic conditions in the absence of phosphate resulted in a complete inhibition of denitrification (Figure 3(b)), indicating that the non-PAO denitrifiers in the EBPR-r culture were unable to denitriﬁe when exposed to O2. It is plausible that they had reduced afﬁnity for NO3 than did the PAOs, or that they were more sensitive to O2. Alternatively, it is possible that the denitrifying GAOs predominately occupied the inner parts of the bioﬁlm, where the penetration of O2 was reduced (Lemaire et al. 2008), as has been reported for the granules enriched in the SNDPR process.

Implications of the study

The EBPR-r process is a novel post-denitriﬁcation process that enables P recovery. The success of this strategy depends on whether denitriﬁcation can be efﬁciently driven by the reducing power stored in the bioﬁlm. The lack of soluble carbon and ammonia in the inﬂuent of this process could result in an elevated bulk water DO, which might affect the denitriﬁcation process. Our previous study suggested that the EBPR-r process can facilitate denitriﬁcation without the need to maintain a strictly anoxic environment in the bulk water (Wong et al. 2013). This was conﬁrmed in the current study, where 60% of the reducing power stored in the bioﬁlm was found to be expended on denitriﬁcation, even when the bulk water DO concentration was near saturation (8 mg/L). However, the denitriﬁcatory ability of the EBPR-r process was remarkably compromised when the bioﬁlm structure was physically disturbed, implying that maintenance of the bioﬁlm structure is critical for the success of EBPR-r as a post-denitriﬁcation strategy when oxygen intrusion occurs.

CONCLUSIONS

The results of this study suggest that:

- the EBPR-r bioﬁlm facilitated P and N removal in a process that was not sensitive to oxygen intrusion;
- at an NO3 concentration of 10 mg-N/L, increasing the DO concentration (from 0 to 8 mg/L) increased the PUR by 43% and decreased the denitriﬁcation rate by 31%;
- at a DO concentration of 8 mg/L, increasing the NO3 concentration (from 0 to 50 mg-N/L) increased the denitriﬁcation rate (from 0 to 0.096 mmol-N/gTS* h).

In summary, this study highlights the importance of the EBPR-r bioﬁlm structure in enabling denitriﬁcation to take place at the same time as P removal for recovery. The data also suggest some operational boundaries (e.g., speciﬁc DO and NO3 concentrations in the inﬂuent) necessary for the EBPR-r bioﬁlm to reduce P and N to acceptable levels in the efﬂuent.

REFERENCES


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