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Pushing the hydraulic retention time envelope in Modified Ludzack Ettinger systems

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ABSTRACT

This study demonstrated the feasibility of operating a biological nutrient-removal employing Modified Ludzack Ettinger (MLE), at a reduced hydraulic retention time (HRT) of 5.5 h. The MLE process not only achieved full nitrification and 75–80% total nitrogen removal, at temperatures as low as 12 °C but also reduced the net observed biomass yield by 28% relative to the full-scale plant to 0.31 g VSS g^{-1} COD_r. Biomass settleability drastically improved relative to the conventional activated sludge (CAS) full-scale plant as reflected by sludge volume index (SVI) of 97 mL g^{-1} versus 202 mL g^{-1}. Respirometric studies indicated that the heterotrophic biokinetic coefficients for the conventional activated sludge and the MLE systems were very similar and consistent with literature values for primary effluent. However, biomass-specific nitrification rates in the MLE system of 0.14 g NH_4-N g^{-1} VSS d^{-1} at 20 °C were 55% higher than in the CAS, with both systems having comparable nitrification temperature correction coefficients of 1.084 for (CAS) and 1.092 in (MLE). This study confirmed through both pilot testing and modeling that biological nutrient removal (BNR) systems treating low strength municipal wastewater can be designed for overall HRT as low as 5 h in cold climates. Furthermore, the process achieved significant biomass yield reduction due to anoxic consumption of organic matter.

1. Introduction

Increased urbanization has resulted in an effort to reduce the overall nutrient load on receiving water bodies. According to the Ontario Ministry of Environment (MOE), wastewater treatment facilities in Canada may be required to meet discharge limitations for nitrogen and phosphorus as stringent as <6.0 mg TN L^{-1} and <1.0 mg TP L^{-1}, respectively. In preparation for the eventual discharge limits, municipalities and state regulatory agencies are investigating various nutrient removal alternatives.

Nutrient removal, primarily nitrogen and phosphorus from municipal wastewater, has become the most important concern for the wastewater treatment plants in the past three decades. Phosphorus and nitrogen can be removed from wastewater by both biological and physical chemical means. Biological means of nutrient removal are generally preferred, as they result in lower waste sludge production, produce a sludge that is more amenable to land application, and biological processes are more "environmental friendly" than chemical processes [1]. Processes using biological mechanisms for phosphorus and nitrogen removal are generally referred to as biological nutrient removal (BNR).

Oldham and Rabinowitz [1] reported that the first full-scale BNR plant in Canada was designed to treat municipal wastewater at a temperature as low as 10 °C, an average dry weather flow of 22.5 ML d^{-1}, solids retention time (SRT) of 22 d in summer and 35 d in winter, and a hydraulic retention time (HRT) of 21 h. Over the past 20 years there has been significant improvement in process understanding that plants are designed and successfully operated at SRTs of 9–13 d during the winter and 5–7 d during summer. At the same time, design HRTs have been reduced to 7–9 h for typical North American wastewaters. Such design figures are directly comparable to those used in conventional activated sludge processes for organic carbon removal plus nitrification. As listed in Table 1, the lower limit of HRTs reported in the literature for BNR systems ranged from (7.7–10 h) using several processes such as: Phoredox [2](A/O), Bardenpho [2], Johannesburg [2] (JHB) and anaerobic–anoxic–oxic [3] (A^2O). On the other hand, the upper limit of HRTs ranged from 17.5 h to 23 h in processes like University of Cape Town [4] (UCT), JHB [5] and Dephanox [6].

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Table 1
HRTs and SRTs for selected BNR processes.

Process	HRT (h)	SRT (d)	Reference
A^2O	10	10	You et al. [3]
Dephanox	23	10	Sorm et al. [6]
JHB	17.8	20	Bortone et al. [5]
JHB	8.5	5	Burke et al. [2]
Phoredox	6.4	3	Burke et al. [2]
Bardenpho	7.7	6	Burke et al. [2]
UCT	17.5	20	Siebritz et al. [4]

The other driver for BNR systems are biosolids production. Excess biomass produced within the process must be disposed of and may account for 60% of total plant operating costs [7]. There is therefore a considerable impetus to develop novel methods for reducing the amount of biomass produced. Recent attempts have been made to couple anaerobic–anoxic and aerobic zones in one reactor for efficient nutrient removal and reduced sludge production (Feng et al. [8], Chae et al. [9], Monti et al. [10]). There are controversial reports of the impact of anoxic conditions on biomass yields, although IWA models such as ASM1 and ASM2 use similar aerobic and anoxic yields. Using identical aerobic and anoxic biomass yields, both Ferrer et al. [11] and Brdjanovic et al. [12] successfully modeled wastewater treatment plants in Spain and Netherlands respectively. On the other hand, Sperandio et al. [13], Muller et al. [14] and Chuang and Ouyang [15] have reported anoxic yields lower than aerobic yields by 8–19%. Several factors impact biomass yield, most notably fraction of readily biodegradable COD, C:N:P ratios, variation of nutrient ratios, inert COD, and operating SRTs.

The primary objective of this paper is to investigate the feasibility of operating an example of BNR processes (i.e. the MLE) at HRTs less than the lower limit of 8 h reported in the literature. The secondary objective of this work is to conduct a side-by-side comparison of biomass yields in high-rate MLE systems and conventional activated sludge processes. This paper will also present detailed wastewater characterization and biokinetics of heterotrophic and autotrophic microorganisms as derived from respirometric studies, and their utility in proper process model calibration. It should be noted that the municipal wastewater treatment plant at which this pilot study was conducted, is currently operating at about 85–90% of its rated capacity and will be undergoing an expansion. Furthermore, the quantities of waste activated sludge (WAS) generated are a major concern at this plant, that utilizes only dewatering and incineration for biosolids management.

2. Materials and methods

2.1. Pilot plant description and operational conditions

The pilot plant process schematic is depicted in Fig. 1a, with operating anoxic, aerobic, and secondary clarifier volumes of 1.3 m^3, 4.0 m^3, and 2.3 m^3 respectively was set-up to treat the primary effluent of the Greenway Pollution Control Plant (GPCP; Fig. 1b), London, ON. The Modified Ludzack Ettinger (MLE) pilot plant was operated at a constant average daily flow (ADF) of $20.8 \pm 0.7\ m^3\ d^{-1}$. The corresponding HRTs, based on the average influent flow, were 1.5 h for the anoxic tank, 4.6 h in the aeration tank and 2.6 h in the secondary clarifier. The internal mixed liquor recirculation rate (IR) from the aeration tank to the anoxic tank was set at $75\ m^3\ d^{-1}$ ($3.75Q_{inf}$) while the return activated sludge flow was set at $43.0\ m^3\ d^{-1}$ ($2.0Q_{inf}$) due to the relatively low RAS suspended solids concentrations of $4700\ mg\ L^{-1}$ about 1.4 times the bioreactor MLSS as opposed to the typical 2–3 times MLSS concentrations. GPCP employs primary clarification followed by conventional activated sludge (CAS) process (Fig. 1b), with an aerobic HRT of 6.5 h at the ADF of 80,600 $m^3\ d^{-1}$ and a SRT of 3.6 d during the pilot testing period. Thus, the bioreactor HRT for the pilot was 6.1 h (versus 6.5 h in the full-scale) and the settling time was 2.6 h (versus 3.3 h in the full-scale). The conventional activated sludge plant is designed for nitrification and has capacity of 93,000 $m^3\ d^{-1}$ (ADF). Alum is added upstream of the primary clarifiers for partial P removal. The full-scale CAS does not utilize any anoxic zones and/or intermittent aeration. Dissolved oxygen concentration in the aeration tank is maintained at 3–5 $mg\ L^{-1}$ by four variable speed 100-hp blowers (2 duty and 2 standby) and a system of medium-bubble diffusers.

2.2. Sampling and monitoring

For the pilot plant, grab samples of the influent (full-scale primary effluent), anoxic tank effluent, aeration tank effluent, clarifier effluent, return activated sludge and internal recirculation were collected five times a week throughout the pilot plant study. All streams, with the exception of return activated sludge and internal recirculation were analyzed for TSS, VSS, total BOD_5 (TBOD), soluble BOD_5 (BOD), total COD (TCOD), soluble COD (SCOD), total Kjeldahl nitrogen (TKN), soluble TKN (STKN), ammonia, nitrates, nitrites, total phosphates (TP), soluble phosphate (SP) and alkalinity in accordance with standard methods [16]. Return activated sludge and internal recirculation were analyzed only for TSS and VSS. In addition to the regular weekly samples, 24-h composite samples of the primary effluent and clarifier effluent were collected twice a week and analyzed for TCOD, SCOD, BOD, SBOD, TSS, VSS, TKN, STKN, ammonia, and TP. Sludge volume index (SVI) and zone setting velocity (ZSV) for both pilot and full-scale plants were performed regularly. The dissolved oxygen (DO) concentration in the aeration tank of the pilot was monitored throughout the pilot plant study. For a comparison of effluent quality, samples of secondary clarifier effluent from the Greenway full-scale WWTP were also collected regularly and analyzed for TBOD, TSS and VSS.

2.3. Respirometric studies

The MLE and full-scale samples were collected on May 31, June 4, and June 11, 2009. The experimental set-ups at an initial (S_0/X_0) ratio of 4 mg COD mg^{-1} VSS and temperature of 20 °C maintained by the water bath were adopted in all the respirometric runs.

During the three months pilot testing program, nitrification tests, and heterotrophic biomass kinetics were determined by conducting 6 respirometric studies (3 on the MLE pilot and 3 on the CAS) at 20 °C. Oxygen uptake rate (OUR) was measured via respirometric methods to determine the various COD fractions and kinetic coefficients of the primary effluent from Greenway Pollution Control Plant. A Challenge respirometer (Challenge Environmental Systems, Inc., Springdale, AR) was adapted to measure the OUR and facilitate wastewater characterization and biokinetics determination. The respirometric system consists of an 8-cell flow measuring module, 8 reaction bottles, an interface module, a magnetic stirring base, water bath and cover, a temperature control module, computer and Windows™ based software. Filtered wastewater samples via 47 mm diameter and 0.45 μm pore size membrane filters were mixed with pre-aerated and acclimatized sludge at an initial substrate to biomass ratio (S_0/X_0) of 4 mg COD mg^{-1} VSS. A stock solution of allylthiourea (ATU) was added to the mixture in 500 mL reaction bottles to inhibit nitrification at an initial concentration of 20 $mg\ L^{-1}$. KOH tubes were inserted into each vessel to absorb carbon dioxide. Wastewater samples and sludge in the reaction bottles were mixed by magnetic stirrers and all the bottles were put in a 20 °C water bath maintained by a temperature control module.

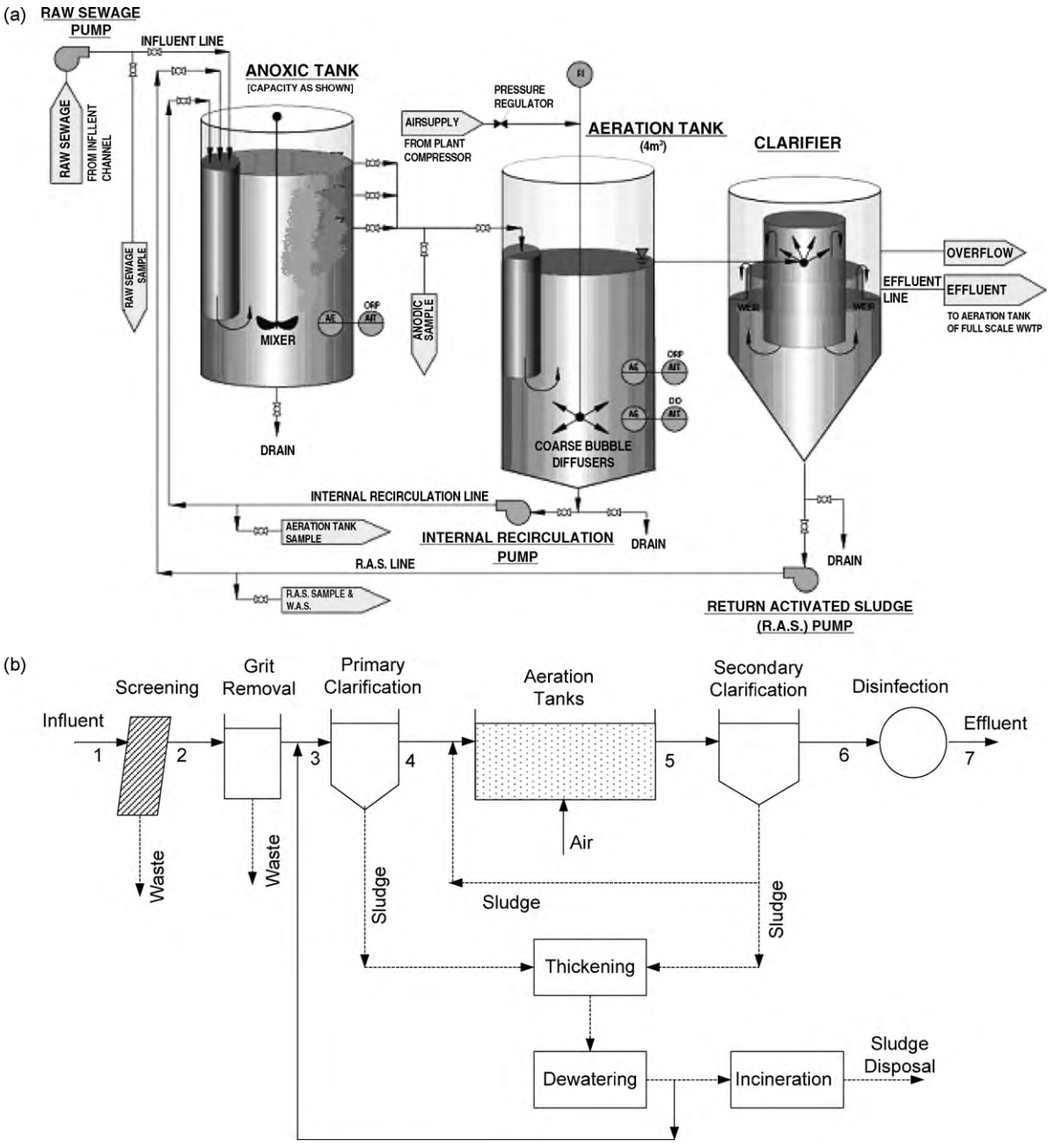

Fig. 1. (a) Process flow schematic of MLE system. (b) Process flow diagram of the activated sludge process at Greenway pollution control plant.

Respirometer samples were collected every 2 h in the first 10 h and then every day for the remaining period during the 2-d run. TCOD, SCOD, TSS and VSS were measured for the samples collected routinely.

2.4. Determination of readily biodegradable substrate (S_S) and biomass yield coefficient (Y_H)

Respirometric tests with filtered wastewater and activated sludge at initial S_0/X_0 ratio of 4 mg COD mg^{-1} VSS were conducted to determine the readily biodegradable substrate (COD) concentration (S_S) and biomass yield coefficient (Y_H). A control blank with deionized water and the same amount of activated sludge as the above test samples, was run simultaneously in the experiment. Temporal variations of total and soluble COD as well as TSS and VSS were monitored. OUR decreased rapidly and dropped to a lower level when S_S was depleted. S_S can be calculated from the equivalent oxygen consumption in the test sample after subtracting the oxygen consumption of the blank in accordance with Eq. (1) [17].

$$S_S = \frac{\Delta O_2}{1 - Y_H} \tag{1}$$

The biomass yield coefficient, Y_H, can be calculated using Eq. (2) by plotting net oxygen consumption versus SCOD reduction [18].

$$Y_H = 1 - \frac{\Delta O_2}{\Delta SCOD} \tag{2}$$

2.5. Determination of maximum growth rate (μ_{max}) and lysis rate constant (b_H)

The maximum heterotrophic growth rate (μ_{max}) is also determined by means of respirometer test on the basis of initial OUR

value at S_0/X_0 ratio of 4 mg COD mg^{-1} VSS. The method developed by Kappeler and Gujer [19] was used. For the observed net OUR profile (after subtracting the blank), the following linear expression (Eq. (3)) can be derived, with a slope of $\mu_{max} - b_H$.

$$\ln \frac{OUR}{OUR_{initial}} = (\mu_{max} - b_H)t \quad (3)$$

The method to calculate the first-order lysis rate constant (b_H) involves plotting the change of OUR with time in a respirometer test with only seed sludge and completely devoid of substrate, on the basis of Eq. (4) [18].

$$\ln OUR = [\ln(1 - f_e)b_H X_H] - b_H t \quad (4)$$

where X_H is the heterotrophic biomass in the raw sludge measured as COD and the coefficient for the production of inert COD from endogenous respiration (f_e) was set at a determined value of 0.2 g COD g^{-1} COD [20].

2.6. Determination of influent heterotrophic biomass concentration (X_{H0})

A respirometer run on influent wastewater to the MLE process (i.e. primary effluent) only was conducted to determine the concentration of heterotrophic biomass in the raw wastewater (X_{H0}). If b_H, Y_H and f_e are known, X_{H0} can be calculated according to Eq. (5) [19].

$$OUR_{initial} = \frac{1 - Y_H}{Y_H}\mu_{max} X_{H_0} + (1 - f_e)b_H X_{H_0} \quad (5)$$

2.7. Nitrification tests

Six nitrification tests were performed on the MLE and full-scale CAS sludges. Three tests were conducted at 20 °C, while three were conducted at 12 °C, the minimum observed influent water temperature. The samples were collected on a weekly basis. The nitrification test comprised of 500 mL bottles for each test. The initial VSS concentrations were set at 1 g L^{-1}, 2 g L^{-1}, 3 g L^{-1} and 4 g L^{-1}. Ammonium chloride was used as the source of ammonia, with 0.75 mL of a 103 g L^{-1} NH_4Cl solution added to the 500 mL liquid sample to affect an increase in initial ammonia concentration to around 40 mg N L^{-1}. Alkalinity was adjusted by adding 0.24 g sodium bicarbonate ($NaHCO_3$) to increase alkalinity by 285 mg $CaCO_3$ L^{-1}. The sludge was aerated using laboratory air at 2 L min^{-1}.

2.8. Process modeling

The MLE process was modeled using BioWin to calibrate and validate results obtained from the pilot study. BioWin is primarily developed using kinetics and stoichiometric relationship used in ASM1 [21] for municipal activated sludge wastewater treatment plants. The model predicts the removal of organic carbon compounds, and N, with simultaneous consumption of oxygen and nitrate as electron acceptors. Furthermore, the model aims at a good prediction of the sludge production. Chemical oxygen demand (COD) was adopted as the measure of organic matter. In the model, the wide variety of organic carbon compounds and nitrogenous compounds are subdivided into a limited number of fractions based on biodegradability and solubility considerations. This model was employed to predict nutrient removal efficiency and effluent quality of the pilot-BNR and full-scale CAS system.

Table 2
Steady-state performance of the MLE and full-scale CAS.

Parameter (mg L^{-1})	Primary effluent quality	Effluent quality	
		Pilot plant	Full-scale plant
	Average ± SD	Average ± SD	Average ± SD
TSS	115 ± 62	24.5 ± 10	4.4 ± 3
VSS	108 ± 62	18.4 ± 8	3 ± 2
TCOD	235 ± 28	42 ± 15	–
SCOD	93 ± 28	17.5 ± 8	–
TBOD	134 ± 38	13.7 ± 9.8	2 ± 1
SBOD	60.4 ± 35	3.0 ± 0.8	–
NH_4-N	15.6 ± 3.3	0.5 ± 0.1	1.3 ± 1
NO_3-N	0.5 ± 0.3	3.9 ± 2.7	–
TKN	25.9 ± 3.3	2.4 ± 0.8	–
STKN	17.3 ± 2.3	1.3 ± 0.2	–
TP	4.7 ± 0.6	1.9 ± 0.6	–
SP (PO_4-P)	2.0 ± 0.7	1.5 ± 0.8	–
Alkalinity	288 ± 21	214 ± 15	–
T (°C)	19 ± 7		
Anoxic MLSS	3285 ± 390		
Anoxic MLVSS	2480 ± 250		
Aerobic MLSS	3300 ± 360		
Aerobic MLVSS	2470 ± 210		

3. Results and discussion

3.1. MLE system performance: effluent quality

Table 2 lists the steady-state performance of the MLE and full-scale CAS. The primary effluent was characterized by an average COD, BOD, TSS, TKN, and TP characteristics of 235 mg L^{-1}, 134 mg L^{-1}, 115 mg L^{-1}, 26 mg L^{-1}, and 4.7 mg L^{-1} respectively corresponding to COD:N:P ratios of 100:11:2. The primary effluent characteristics are typical of low strength municipal wastewater (MWW) [28a]. As apparent from Table 2, the MLE final effluent was characterized by average TBOD and SBOD concentrations of 13.7 mg L^{-1} and 3.0 mg L^{-1}, respectively. Moreover, the average MLE effluent STKN, NH_4-N and NO_3-N concentrations were 1.3 mg L^{-1}, 0.5 mg L^{-1} and 3.9 mg L^{-1}. Full-scale effluent was characterized by average TBOD, TSS, VSS, and NH_4-N concentrations of 2 mg L^{-1}, 4.4 mg L^{-1}, 3 mg L^{-1}, and 1.3 mg L^{-1}, respectively. It is apparent that while the MLE effluent SBOD of 3 mg L^{-1} closely matches the full-scale effluent TBOD of 2 mg L^{-1}, effluent TSS and VSS (see Fig. 2a) are substantially different. It must be asserted that the higher TSS concentrations in the pilot-effluent relative to the full-scale plant are not reflective of poor sludge settling characteristics, as reflected by an SVI of 97 mL g^{-1} discussed later, but are primarily due to problems with the 2.56 m^2, 0.9 m deep secondary clarifier. Based on a correlation (not shown) with R^2 of 0.93, the volatile fraction of the pilot MLE biomass was 75%.

Fig. 2b, illustrating the temporal variation of COD and BOD, shows a stable effluent quality for the last month of operation. Fig. 2c shows the diurnal variation of effluent nitrogenous compounds, i.e. TKN, soluble TKN (STKN), ammonia, and nitrates. The stability of nitrification, as reflected by constant ammonia, and STKN concentrations during the steady-state period, is evident from Fig. 2c as well as the coefficient of variation (COV = standard deviation/mean) of only 20%, and 15% (Table 2), respectively. On the other hand, effluent nitrate concentrations fluctuated between <2 mg NO_3-N L^{-1} and 6 mg NO_3-N L^{-1}, but hovered mostly around the 2–4 mg NO_3-N L^{-1}. It is noteworthy that the full-scale CAS achieved full nitrification, as reflected by average effluent ammonia concentrations of 1.3 NH_4-N L^{-1} (Table 2) at the SRT of 3.6 days. However, nitrification in the full-scale CAS varied more widely than in the pilot MLE, as reflected by COV of 77%. Nitrification kinetic studies on the MLE and full-scale CAS indicated that the biomass specific nitrification rate, an indicator of nitrifiers population in MLE, was 56% higher than in the full-scale. In general,

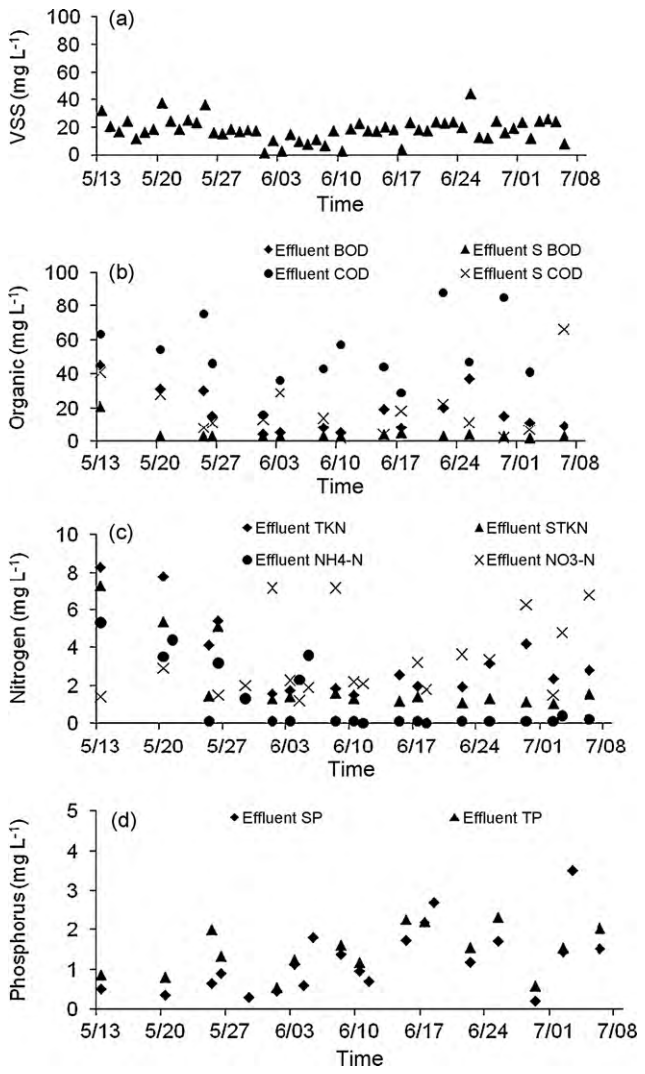

Fig. 2. Temporal variation of (a) VSS, (b) organics, (c) nitrogen, (d) phosphorus in the pilot MLE.

overall biological nitrogen removal in the MLE plant was about 75–80%. The comparatively higher nitrification rate of the MLE process relative to the full-scale treatment plant was primarily due to the strict anoxic and aerobic zones and reduced readily biodegradable organic matter in the aerobic reactor. The nitrifiers are the slow growers ($\mu_{max} = 0.6\,d^{-1}$) and are outcompeted by the aerobic heterotrophs ($\mu_{max} = 7.3\,d^{-1}$) in the presence of readily biodegradable substrate, well documented in literature [22–25]. Furthermore, the full-scale WWTP was configured with an aeration tank and no anoxic reactor. In the absence of anoxic organic consumption, the influent COD of $235 \pm 28\,mg\,L^{-1}$ and BOD of $134 \pm 38\,mg\,L^{-1}$ were primarily degraded or consumed in the aeration tank and significantly inhibited the growth of the nitrifiers in the nutrient removal process. Inhibition of nitrification in the BNR processes at soluble carbonaceous BOD of >20 mg L^{-1} was also reported by Parker et al. [26] during nitrogen and phosphorus removal from municipal wastewater using a trickling filter. Boller et al. [27] reported a 30–50% loss of nitrification at an organic loading rate of 5.8 g COD $m^{-2}\,d^{-1}$ compared to the nitrification rate at an organic loading rate of 0.75 g COD $m^{-2}\,d^{-1}$ in the biofilm reactors (trickling filters, rotating biological contractors, and biofilters) at an NH_4-N loading of 0.7 kg NH_4-N $m^{-3}\,d^{-1}$.

Fig. 2d confirms that sometimes, effluent soluble phosphorus concentration was as low as 0.2 mg L^{-1}, though averaged 1.5 mg L^{-1} for the last month of operation. Enhanced biological phosphorus removal (EBPR) in the MLE was unstable due to several reasons: fluctuating anoxic tank and final effluent nitrates concentrations, which reflect the variations in C:N ratios, and thus the availability of readily biodegradable organic matter (rbCOD) for EBPR; the absence of an anaerobic bioreactor dedicated for P release and volatile fatty acids uptake, both of which are conducive to enhanced biological phosphorus removal. In the MLE process, EBPR would occur only when nitrates are low and rbCOD is available; the influent COD-to-P ratio was less than the minimum of 40 required for P removal in 20% of the tests; and finally, the addition of alum upstream of the primary clarifier precipitates orthophosphates and hinders the process of phosphorus release compromising bio-P.

The experimental data from the MLE pilot plant clearly indicate that despite operating at temperatures as low as 12 °C (Table 2), the MLE pilot plant achieved full nitrification and almost 80% nitrogen removal from this low strength MWW at anoxic and aerobic HRTs of 1.5 h and 4.6 h, respectively, corresponding to an overall bioreactor HRTs of 6.1 h. The aerobic mass fraction of 0.75 resulting in an aerobic SRT of 7 d, proved sufficient for complete nitrification even at low temperatures. As apparent from Table 2, the pilot MLE process operated at a COD volumetric loading rate of 0.92 kg COD $m^{-3}\,d^{-1}$, as compared with the full-scale CAS of 0.87 kg COD $m^{-3}\,d^{-1}$ while the Respective BOD loadings were 0.52 kg BOD_5 $m^{-3}\,d^{-1}$ and 0.5 kg BOD_5 $m^{-3}\,d^{-1}$ at the middle of the 0.3–0.7 kg BOD_5 $m^{-3}\,d^{-1}$ reported for conventional plug flow activated sludge systems [28b]. Based on only the aerobic HRT in the pilot, which controls the rate-limiting process of nitrification, volumetric COD and BOD loading rates are respectively 1.25 kg COD $m^{-3}\,d^{-1}$, and 0.7 kg BOD_5 $m^{-3}\,d^{-1}$. While the aerobic HRT of 6.5 hours in the full-scale CAS is within the typical 4–8 h range [28b], the aerobic HRT in the pilot MLE of 4.5 h is evidently close to the minimum HRT. Thus, both the volumetric organic loading rate and aerobic HRT in the pilot MLE are respectively at the top and bottom of the typical operational range. Furthermore, the average bioreactor MLSS concentration of 3300 mg L^{-1} is typical and does not represent high biomass that could result in excessive solids loading to the clarifier, as based on the projected ADF after expansion to 127 MLD and secondary clarification area of 5000 m^2, the solids loading rate with 100% return activated sludge flow is 168 kg $m^{-2}\,d^{-1}$, within the typical 120–192 kg $m^{-2}\,d^{-1}$ for BNR systems [28c].

3.2. Nutrient mass balances

Table 3 presents the detailed mass distributions of COD, NH_4-N, NO_3-N, and PO_4-P, calculated using comprehensive pseudo-steady state experimental data on MLE process influent and effluents from anoxic and aerobic reactors, where positive values reflect removal and negative values reflect formation. Particulate COD, nitrogen, and phosphorus were calculated from the difference between total and soluble influent and effluent concentrations. Thus, due to the various transformations of the particulate and soluble fractions in the process, overall mass balance are pertinent for total COD, total nitrogen, and total phosphorus only. Nitrogen (%) closure has been calculated using influent and effluent TKN, effluent nitrates, and nitrate denitrified in the system. Phosphorus mass balance was based on influent and effluent total P and P in waste sludge.

Denitrification and phosphorus release were observed in the anoxic reactor. As apparent from Table 3, 0.17 kg NO_3-N d^{-1} removal and 0.28 g PO_4-P d^{-1} release were observed in the anoxic reactor. Approximately 0.87 kg COD was consumed for denitrification in the anoxic reactor, estimated using Eq. (1) [28d] and the observed process yield of 0.31 g VSS g^{-1} COD elaborated upon later, which shows that approximately 5.1 g COD were consumed to denitrify 1.0 g NO_3-N. It is well established in the literature

Table 3
Nutrient balances.

	Mass in influent ($kg\,d^{-1}$)	Anoxic mass consumed ($kg\,d^{-1}$)	Aerobic mass utilized ($kg\,d^{-1}$)	Mass in effluent ($kg\,d^{-1}$)	Mass in wastage sludge ($kg\,d^{-1}$)	Percent closure (%)
TCOD	4.9 ± 0.6	1.43	3.11	0.87 ± 0.30	2.3	98[a]
PCOD	3.0	n/a	3.0	0.51	–	
SCOD	1.9 ± 0.6	1.43	0.11	0.36 ± 0.17	–	
NH_4-N	0.32 ± 0.05	0.10 ± 0.07	0.21 ± 0.12	0.01 ± 0.01	–	
NO_3-N	0.01 ± 0.04	0.17 ± 0.05	−0.25 ± 0.11	0.08 ± 0.05	–	
STKN	0.36 ± 0.05	0.1[d]	0.21[d]	0.03 ± 0.004	–	
PTKN	0.18	n/a	0.09	0.02	–	
TKN	0.54 ± 0.07	0.1[e]	0.30[e]	0.05 ± 0.017	0.16	85[b]
SP	0.04 ± 0.02	−0.28 ± 0.23	0.14 ± 0.23	0.03 ± 0.01	–	
PP	0.06	–	–	0.00	–	
TP	0.1 ± 0.01	–	–	0.04 ± 0.01	0.032	72[c]

Standard deviation for 20 samples; particulate COD, nitrogen, and phosphorus were estimated from the difference between total and soluble fractions present in the influent and effluent stream of the reactor.

[a] COD %closure = (1.43+2.53+0.87)/4.9.

[b] TKN %closure = (0.17+0.08+0.05+0.16)/(0.54).

[c] TP %closure = (0.04+0.032)/(0.1).

[d] STKN is equivalent to the NH_4-N utilized in the anoxic–aerobic reactor

[e] TKN utilized in the anoxic reactor is equivalent to the NH_4-N utilized in the bioreactor; TKN consumed in the aerobic reactor is the amount of nitrogen utilized for cell synthesis, estimated considering aerobic COD consumption, process yield, and nitrogen content of the waste biomass (approximately 10% of the dry weight of biomass).

that using a typical anoxic yield of 0.54 g $COD_{biomass}\,g^{-1}$ $COD_{substrate}$ and a 15% anoxic reduction [21] 6.2 g COD is consumed to denitrify 1 g NO_3-N during anoxic biological nitrogen removal from municipal wastewater [28d,29]. Comparatively the reduced COD consumption for denitrification observed in this study was primarily due to the reduced biomass yield (Y) of 0.31 g VSS g^{-1} COD which was attributed to the SRT and anoxic COD consumption. Approximately 0.56 g SCOD was sequestered during the anoxic P release of 0.28 PO_4-P, estimated considering 2 g COD consumption per gram P release [30]. Even though denitrification was the main reaction in the anoxic reactor, approximately 0.1 kg NH_4-N d^{-1} consumption was observed, of which 0.03 kg NH_4-N d^{-1}, calculated based on COD consumption, process yield, and average N content of the biomass of 0.1 g N g^{-1} VSS, was utilized for cell synthesis.

$$\text{COD consumed during denitrification} = \frac{2.86}{1 - 1.42 \times Y_{observed}} \quad (6)$$

The main reactions in the aerobic reactor were nitrification, oxidation of accumulated polyhydroxyalkonates and organic matter. Approximately 0.21 kg NH_4-N d^{-1} was nitrified and 0.25 kg NO_3-N d^{-1} was formed in the aerobic reactor. The discrepancy between ammonium nitrified and nitrate generated of approximately 0.04 kg N d^{-1}, is primarily due to the nitrification of soluble organic nitrogen. Moreover, approximately 0.1 g N d^{-1} would have been consumed for biomass synthesis during oxidation of 3.11 g COD d^{-1}. This finding also supports utilization of 0.09 g N d^{-1} particulate nitrogen in the aerobic reactor. The anoxic phosphorus release of 0.28 ± 0.23 kg PO_4-P d^{-1} (Table 3) coupled with aerobic phosphorus uptake of 0.14 ± 0.23 kg PO_4-P d^{-1} (well above P for aerobic biomass synthesis) of ~0.02 kg PO_4-P d^{-1} may reflect potential biological phosphorus removal activities. However, the wide variations in anoxic–aerobic reactor PO_4-P concentrations as reflected by the standard deviation indicate unstable bio-P activities in the pilot MLE process. The effluent TP of 0.04 kg P d^{-1} and phosphorus content of the waste biomass of 0.03 kg P d^{-1}, estimated from the observed biomass P content of 2% of the dry weight of biomass (as VSS), balance 72% of the influent TP, with approximately 0.04 kg P d^{-1} or 28% of the influent TP unaccounted in the process, i.e. leave in the process effluent. Approximately, 20–50% variation of influent, effluent, and reactor SP, TP had a significant impact on overall phosphorus mass balance.

3.3. Biomass characteristics and sludge yield

The comparison of the sludge volume index (SVI) in the MLE pilot with the full-scale CAS is depicted in Fig. 3. During the steady-state operation, SVI in the MLE averaged at 97 mL g^{-1} versus 202 mL g^{-1} in full-scale. The weekly measurements of sludge settling velocities in the two plants revealed that while the settling velocity in full-scale plant varied from 4 to 11 m d^{-1}, it stabilized around 34 m d^{-1} for MLE sludge. It must be asserted, therefore, that the higher observed VSS concentrations in the MLE effluent relative to the full-scale CAS are not a reflection of poor sludge settling characteristics but rather of the MLE secondary clarifier's size and design.

Sludge wastage from the MLE during the 33 d of steady-state operation average 0.35 $m^3\,d^{-1}$ or 1.7% of the influent flow, as compared with 2913 $m^3\,d^{-1}$ or 3.6% of the influent flow in full-scale. The calculated MLE system overall SRT, estimated considering biomass waste and effluent biomass, was 9.45 ± 2.4 d.

Fig. 4a shows the observed yield of 0.31 g VSS g^{-1} COD removed in the pilot-scale MLE system. The yield was calculated as the sum of the net change in reactor biomass, biomass waste, and effluent solids divided by the total COD consumed in the process. The observed yield in this study was approximately 72% of the conventional activated sludge process as employed at Greenway Pollution Control plant. The observed yield of the Greenway wastewater treatment plant (derived from the cumulative data depicted in Fig. 4b) inclusive of secondary effluent TSS was 0.43 g VSS g^{-1} COD.

The reduced observed yield in the MLE pilot was primarily due to the anoxic consumption of about 30% of the influent COD (Table 3).

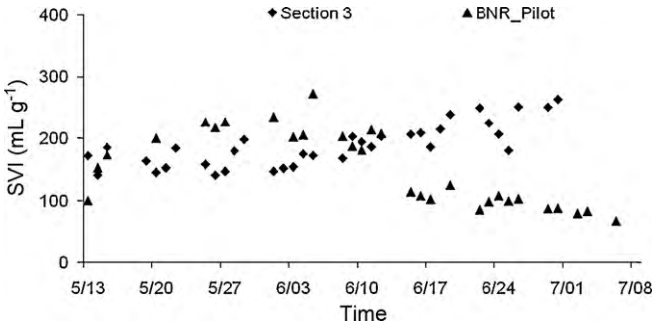

Fig. 3. Temporal variation in sludge volume index (SVI).

Table 4
Breakdown of the various biomass production rates.

	Aerobic	Anoxic	Nitrification
Stoichiometric yield	0.63 (g COD g^{-1} COD)	0.54 (g COD g^{-1} COD)	0.24 (g COD g^{-1} N)
Process SRT	7.1 (d)	2.3 (d)	7.1 (d)
Estimated process yield[a]	0.28 (g VSS g^{-1} COD)	0.32 (g VSS g^{-1} COD)	0.11 (g VSS g^{-1} N)
Consumption	3.11 (g COD d^{-1})	1.43 (g COD d^{-1})	0.21 (g N d^{-1})
Biomass produced (g VSS d^{-1})	0.88	0.46	0.02

[a] Estimated process yield ($Y_{true}/(1 + b_H \times \theta_c)$); where Y_{true} is stoichiometric yields, b_H is the lysis rate constant (0.08 d^{-1}), and θ_c is the solids retention time (d); overall yield = (biomass produced in the aerobic and anoxic conditions)/total organic consumed = (0.46 + 0.88 + 0.02)/(3.11 + 1.43) = 0.30 g VSS g^{-1} COD.

The detailed breakdown of the various biomass production rates is depicted in Table 4. Typical stoichiometric yield coefficients for aerobic, anoxic, and nitrification processes are 0.63 g COD g^{-1} COD, 0.54 g COD g^{-1} COD, and 0.24 g COD g^{-1} N respectively and decay coefficient is 0.08 d^{-1} [28e] and nonbiodegradable fraction of VSS (f_d) of 0.15. As apparent from Table 4 the anoxic reactor generated 0.47 kg VSS d^{-1}, while consuming 1.43 kg COD d^{-1} while the aerobic reactor generated 0.96 kg VSS d^{-1} of heterotrophic biomass and 0.02 kg VSS d^{-1} during the oxidation of 3.11 kg COD d^{-1} and 0.25 kg N d^{-1}. The overall process yield of 0.30 g VSS g^{-1} COD, estimated from typical MLE yields, decay coefficient, and process SRTs, is in close agreement with the observed yield of 0.31 g VSS g^{-1} COD (Fig. 4a).

4. Determination of biokinetic parameters

4.1. Parameters derived by linearization

The respirometric study carried out on the filtered samples for pilot MLE and full-scale CAS in runs 1–3 and 4–6, respectively, are used for the calculation of COD fractions and kinetic coefficients. ΔSCOD can be calculated from the difference between SCOD at a specific time t and initial SCOD, and ΔO_2 could be calculated from the difference between the cumulative oxygen uptake at time t and the cumulative oxygen uptake of the control due to hydrolyzed substrate and endogenous metabolism at the same time. The biomass yield coefficient, Y_H, was calculated according to Eq. (2) with the slope equal to $(1 - Y_H)$ as displayed in Fig. 5a for the three MLE respirometric studies. S_S could be calculated using Eq. (1) when the time of initial S_S substrate exhaustion was determined from the typical net OUR profile (after adjusting for the blanks) and considering the dilution in the calculation. Eq. (3) was adopted to get the slope of $\mu_{max} - b_H$ based on the initial OUR value at the recommended S_0/X_0 of 4 mgCOD mg^{-1} VSS [18] as shown in Fig. 5b. Eq. (4) was used to calculate b_H based on activated sludge (the control) which experienced no growth but only endogenous respiration only as shown in Fig. 5c. And then μ_{max} and b_H were calculated for the samples. X_{H0} was determined by Eq. (5) based on the un-filtered wastewater run. X_{H0} could be calculated since Y_H, μ_{max} and b_H were already determined and initial OUR could be read from the OUR profile for the wastewater only sample. Similarly, the kinetic coefficients Y_H, b_H, μ_{max} and X_{H0} were calculated for full-scale plant as shown in Table 5.

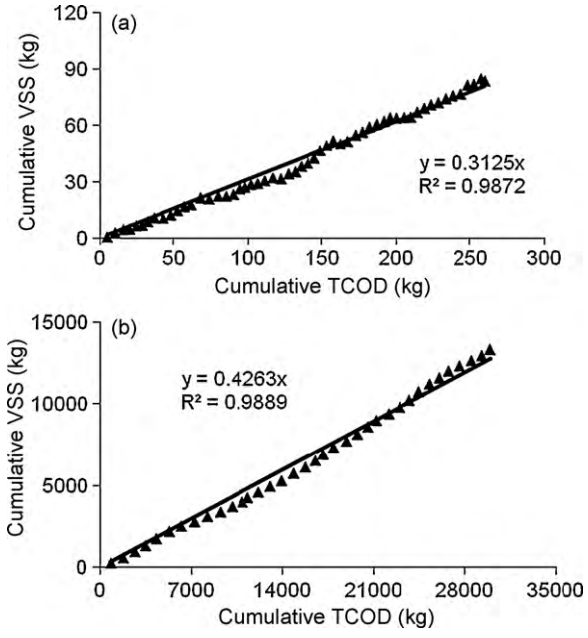

Fig. 4. Biomass yields (a) pilot MLE and (b) Greenway wastewater treatment plant.

4.2. Parameters derived from GPSX simulation

The substrate saturation coefficient (K_S), hydrolysis rate constant (K_h) and slowly biodegradable substrates (X_S) were determined by GPS-X (Hydromantic Inc., Ontario, Canada) optimization

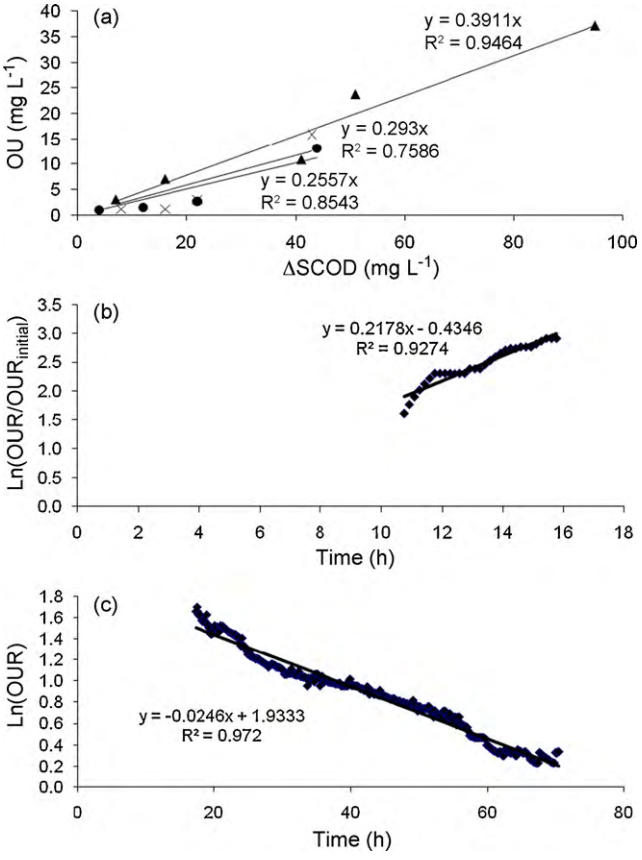

Fig. 5. MLE parameters derived by linearization (a) yield coefficient Y_H, (b) $\mu_{max} - b_H$ (filtered wastewater sample) and (c) b_H (sludge only sample).

Table 5
-Summary of the respirometric studies.

Parameter (unit)	MLE	Full-scale	ASM2
b_H (d^{-1})	0.45 ± 0.06	0.47 ± 0.08	0.4
Y_H (mg COD mg^{-1} COD)	0.69 ± 0.07	0.63 ± 0.05	0.63
μ_{max} (d^{-1})	6.4 ± 0.9	6.5 ± 0.8	6
S_S (mg COD L^{-1})	22 ± 4	21 ± 2	30
S_I (mg COD L^{-1})	15	15	30
X_S (mg COD L^{-1})	102	102	125
X_{H0} (mg COD L^{-1})	7.9 ± 1.6	7.6 ± 1.5	30
X_I (mg COD L^{-1})	40	40	25
K_S (mg COD L^{-1})	5.8 ± 0.8	6 ± 0.5	4
K_h (d^{-1})	2.9 ± 0.1	3 ± 0.3	3

module. K_S dominates oxygen respiration especially in the range with increasing substrate limitation [19]. Therefore K_S can be estimated by a graphical comparison of the measured respiration with the simulated one based on batch test with filtered wastewater and sludge, especially the declining phase of the OUR profile, as shown in Fig. 6a. Alternatively, K_S can also be estimated from the simulated OUR for the unfiltered wastewater with biomass. Both the hydrolysis rate coefficient (K_h) and the slowly biodegradable substrate (X_S) affect oxygen respiration only if growth is limited by substrate and therefore respiration is dominated by hydrolyzed substrate. The hydrolysis rate constant (K_h) and the slowly biodegradable substrate (X_S) can be determined by a graphical comparison of the measured respiration with the simulated one based on batch test just with unfiltered wastewater as illustrated in Fig. 6b. Measured and simulated OUR were compared to reveal the values of K_S, K_h and X_S. Similarly, the kinetic coefficient K_S, K_h and X_S were calculated for full-scale plant as shown in Table 5.

4.3. Summarized COD fractionation and kinetic coefficients

Based on all the above calculations and simulations, a summary of all these respirometric runs for MLE and full-scale is presented in Table 5.

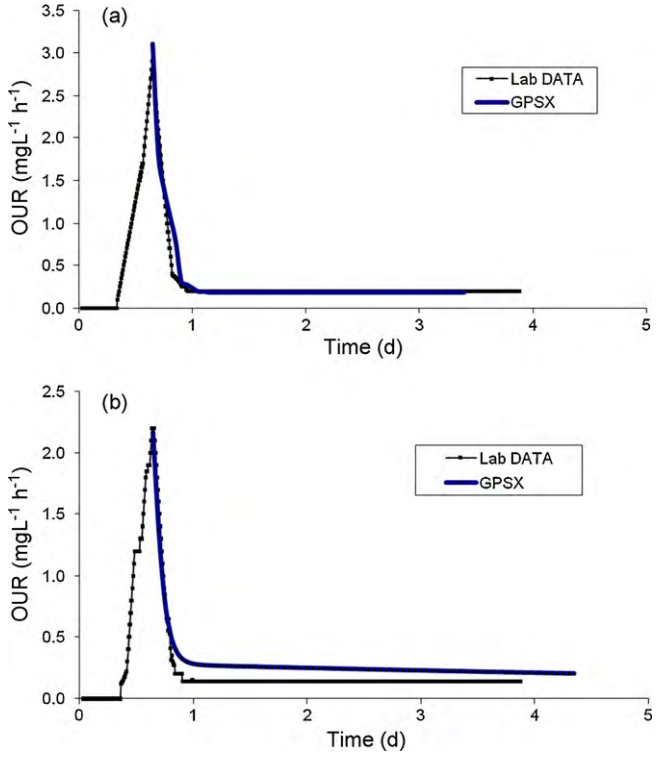

Fig. 6. Parameters derived from GPSX simulations (a) filtered sample and (b) unfiltered sample.

Table 6
Summary of the nitrification rates for the six runs.

	MLE (g NH_4 g^{-1} VSS d^{-1})	Full-scale (g NH_4 g^{-1} VSS d^{-1})
Run 1 (20 °C)	0.24	0.08
Run 2 (20 °C)	0.14	0.07
Run 3 (20 °C)	0.14	0.13
Average ± SD	0.14 ± 0.06	0.09 ± 0.03
Run 4 (12 °C)	0.06	0.05
Run 5 (12 °C)	0.05	0.04
Run 6 (12 °C)	0.05	0.02
Average ± SD	0.053 ± 0.01	0.037 ± 0.02

Table 5 indicates that essentially all biokinetic constants for heterotrophic biomass in the two systems, i.e. the MLE and full-scale plant were very close and well within the range for primary effluents (ASM2) [21]. Furthermore, the kinetic coefficients listed above are very close to the typical values for municipal wastewater with b_H, Y_H, μ_{max}, and K_h of 0.6 d^{-1}, 0.63 mg $COD_{biomass}$ mg^{-1} $COD_{substrate}$, 6.0 d^{-1}, and 3.0 d^{-1}, respectively. The typical K_S value for screened primary effluent is 5.0 mg COD L^{-1}, which is in close agreement to that of the MLE, and full-scale of 5-6 mg COD L^{-1}. Furthermore S_S, S_I, X_S, X_{H0}, and X_I, constituted 9.6%, 6.5%, 44.3%, 3.4%, and 17.4% of the total primary effluent COD of 230 mg L^{-1}, on average, as compared with the typical ranges of 10–20%, 5–10%, 30–60%, 5–15%, and 10–15%, respectively ASM2 [21]. It is thus apparent that the various COD fractions of the GPCP full-scale primary effluent are either well within or close to the typical ranges for municipal wastewater. The rest of the COD is attributed to volatile fatty acids.

4.4. Nitrification tests

Six nitrification tests were performed on the MLE and full-scale sludges. Three tests were done at 20 °C, while three were done at 12 °C. Monitoring of the temporal variation of ammonia concentrations in each of the three batches allowed for the determination of the maximum (initial) nitrification rates, which were then plotted against the initial biomass concentrations (as VSS) to determine biomass-specific nitrification rates.

Table 6 summarizing the nitrification rates for MLE and full-scale, indicates that the nitrification kinetics normalized to biomass, a measure of the population of nitrifiers in the system, at 20 °C in the MLE process was on average 56% higher than in full-scale. Similarly, at the minimum water temperature of 12 °C, the average nitrification rate in the MLE was 44% higher than in full-scale. The Arrhenius temperature dependence coefficients (θ) for full-scale and the MLE are 1.084 and 1.092, respectively, consistent with the typical 1.1 reported for nitrification (ASM2) [21]. Since the MLE was operated on average at 20% higher VSS concentrations, it is thus evident that the maximum nitrification capacity per unit reactor volume in the MLE is 88% and 73% higher than full-scale at 20 °C and 12 °C, respectively. Due to the 30% anoxic COD consumption in the MLE process, the ratio of BOD_5-to-TKN entering the aerobic processes, on average, was 5.2:1 and 3.6:1 in the CAS and MLE respectively. According to Metcalf and Eddy [28] the corresponding fractions of nitrifying organisms in the biomass are 0.052 and 0.072 translating to a 40% increase in biomass-specific nitrification rate in the MLE relative to the CAS.

5. Process modeling

To predict the performance of the MLE system (as a proposed system for Greenway upgrade) under the projected rated capacity, the MLE pilot was first modeled using Biowin, at 2 temperatures of 20 °C (overall average) and 12 °C (minimum observed tempera-

Table 7
Summary of modeling results.

Parameter (mg L^{-1})	Measured		MLE calibration		Predicted MLE full-scale	
	Measured	Range	Scenario 120 °C	Scenario 212 °C	Scenario 320 °C	Scenario 412 °C
TSS	24.5	7.5–38.3	25.2	26.3	10.5	10.6
VSS	18.4	3.0–37.4	18.4	18.1	7.6	7.8
TCOD	42	28–60	39.1	39.6	24.5	24.8
SCOD	17.5	13–25	14.2	15.1	14.2	14.2
TBOD	13.7	5–21	8.4	8.6	3.7	4.0
SBOD	3.0	2.9–3.1	0.6	0.6	0.5	0.6
TKN	2.4	1.7–3.2	3.9	4.3	2.9	3.6
STKN	1.3	1–1.5	2.4	2.8	2.3	3.0
NH_4-N	0.5	0–1.5	0.60	1.1	0.5	1.1
NO_3-N	3.9	1.6–7.0	4.9	0.00	7.2	5.8
TP	1.9	1.2–2.3	3.5	2.3	3.3	3.2
SP	1.5	0.6–2.5	3.1	1.3	3.1	3.0
MLSS	3305	2930–3690	3075	3150	3700	3900
WAS (m^3 d^{-1})	0.45	0.2–0.5	0.3	0.3	1312	1317
RAS SS	4666	4200–5480	4342	4628	7362	7705
Alkalinity	214	197–235	211	222	203	210
Anoxic NH_4-N	1.9	1.1–2.8	3.4	3.9	3.7	4.2
Anoxic NO_3-N	0.7	0.2–1.0	2.1	0	3.9	2.9
Anoxic NO_2-N	–	–	0.2	0.03	0.26	0.32
Anoxic SP	5.3	2.1–9.4	2.9	2.5	2.8	2.8
Aeration NH_4-N	0.6	0–1.3	0.6	1.1	0.5	1.1
Aeration NO_3-N	1.2	0.4–2.2	4.9	0	7.2	5.8
Aeration NO_2-N	–	–	0.23	2.61	0.13	0.38
Aeration SP	4.3	1.6–7.8	3.1	1.3	3.1	3.0

ture) denoted as Scenarios 1 and 2, respectively. In the predictive Scenarios 3 and 4 (at 20 °C and 12 °C, respectively), the flow rate was increased to 127,000 m^3 d^{-1}, and the process units were scaled up as follows: anoxic volume = 4630 m^3, aeration volume = 21,826 m^3, clarifier area = 5000 m^2, with the anoxic reactor sized based on the operating volume of existing rectangular clarifiers that would be converted to bioreactors at Greenway plant.

Table 7 summarizes the model output. Due to the unstable EBPR in the MLE system alluded to above, the modeling was focused on BOD, COD, TSS, and nitrogenous compounds. Scenarios 1 and 2 illustrate the good match between the model predictions and experimental data for the MLE pilot, validating the accuracy of model input, i.e. influent characteristics and modified kinetics. The predicted effluent characteristics were mostly in the range of measured averages and standard deviations. The model predicted average effluent TSS concentrations in the 25–26 mg L^{-1} range, with 73% volatile fraction, i.e. effluent VSS of 18 mg L^{-1}, are in line with the measured VSS. Both effluent ammonia and nitrates were predicted within 0.1 mg L^{-1} and 1 mg L^{-1} of the measured average. Alkalinity predictions were 99% accurate, and since alkalinity is consumed by nitrification and produced by denitrification, the accurate effluent alkalinity prediction confirms that the model nitrification and denitrification rates match the experimental data. Noting that the accuracy of measured reactor biomass is crucial to successful modeling of any bioreactor system, Table 7 shows that for the MLE calibration runs (Scenarios 1 and 2), the average predicted MLSS of 3075 mg L^{-1} is 93% of the measured average of 3305 mg L^{-1}. Model-predicted RAS solids of 4600 mg L^{-1} match very well the measured value of 4666 mg L^{-1}. The accuracy of model calibration is not only reflected by successful prediction of effluent parameters but also by the soluble components in the bioreactor, i.e. ammonia, nitrates and phosphates. Given the influent variations and the operating temperature ranges coupled with the intricacy of comparing composite effluent data with grab samples data, the model calibration was deemed satisfactory if predictions were within the 10th to 90th percentile ranges illustrated in Table 7. The accuracy of modeling the overall system was emphasized above. Furthermore, as apparent from Table 7 the model aeration tank ammonia and phosphates concentrations are within the range of experimental data. However, the model over predicted the anoxic tank ammonia and nitrate concentrations despite matching the soluble phosphorous concentrations clearly suggesting that the model under predicted denitrification, which might be due to changing influent rbCOD concentrations not accounted for by the average characteristics. Diurnal variations of influent characteristics were not assessed in this study. The over prediction of ammonia in the anoxic tank confirms the findings of the steady state mass balance analysis discussed above, emphasizing potentially other mechanisms for anoxic ammonia removal than biomass synthesis. The lower measured anoxic nitrate concentrations relative to the model rationalize the higher observed P release in the anoxic nitrates concentrations by 1.4 mg L^{-1}, corresponding to an additional consumption of about 7.1 mg rbCOD L^{-1}. Based on the typical 2 mg VFA mg^{-1} P, this additional 7.1 mg rbCOD L^{-1} would contribute to the release of about 3 mg PO_4-P L^{-1} in the anoxic tank, closely matching the 2.6 g PO_4-P L^{-1} on an average basis. As evident from Table 7, in the MLE predictive runs (Scenarios 3 and 4), the system was able to handle the required rated capacity with a low final effluent TSS and VSS concentrations of 11 mg L^{-1} and 8 mg L^{-1}, respectively. It is noteworthy that the clarification area used in Scenarios 3 and 4 was set to the area available for expansion in the full-scale plant of 5000 m^2. Full-scale clarifier HRT was 3.2 h, 20% larger than the pilot MLE clarifier. As expected, the large full-scale clarifier design also impacted return activated sludge SS, increasing RAS SS on average by about 60% to 7400–7700 mg L^{-1}. Moreover, the model predicted effluent ammonia concentrations of approximately 1.0 mg L^{-1} for both low and high temperatures, while the TKN was around 3.6 mg L^{-1}. The effluent TBOD and SBOD in the MLE plant were 4.0 mg L^{-1} and 0.6 mg L^{-1}, respectively.

The results of the calibrated model clearly confirm that for this low strength MWW, the MLE system can not only handle 127,000 m^3 d^{-1} at an overall bioreactor HRT of only 5 h, but also achieve full nitrification and about 75% nitrogen removal, at MLSS concentrations of 3700–3900 mg L^{-1}, close to the upper range of typical MLE plants. At the projected bioreactor HRT of 5 hours, overall volumetric COD and BOD loading rates are 1.1 kg COD m^{-3} d^{-1} and 0.63 kg BOD_5 m^{-3} d^{-1}, respectively, corresponding to 1.52 kg COD m^{-3} d^{-1} and 0.85 kg BOD_5 m^{-3} d^{-1}, based on the aerobic bioreactor volume, well above the maximum of

0.7 kg BOD_5 m^{-3} d^{-1} for conventional activated sludge systems [28b]. Furthermore, the aerobic HRT of 3.7 h is well below the typical for BNR systems (Table 1). The detailed mass balances conducted on the pilot MLE system with an overall bioreactor HRT of 6.1 h indicated that approximately 30% of the organic matter is oxidized under anoxic conditions, and thus the MLE system would achieve about 20% reduction in the overall oxygen demand. Furthermore, for this specific case, the improvement in sludge settleability and significant reduction in waste activated sludge quantities are additional benefits that render the MLE as an attractive upgrade option for the plant despite the lack of explicit effluent total nitrogen requirement.

6. Summary and conclusions

Based on the pilot experimental data and the process modeling on Biowin, the following conclusions can be drawn:

- For low strength MWW with contaminant concentrations similar to the primary effluent used in this study, MLE plants can in fact be designed for overall bioreactor HRT of 5–6 h, instead of the widely accepted minimum of 8 h, without compromising low temperature nitrification (at 12 °C) and still maintain effluent ammonia concentrations of 0.5–1.5 mg L^{-1}.
- The MLE process improved sludge settleability drastically relative to the full-scale CAS, reducing the overall SVI from 202 mL g^{-1} to 97 mL g^{-1}, and increasing ZSV by 30%.
- The MLE process operating at an SRT of 9.5 days produced a net observed yield of 0.31 g VSS g^{-1} COD, which is 28% lower than the 0.43 g VSS g^{-1} COD observed in the full-scale CAS at an SRT of 3.5 d.
- Respirometric work indicated that while the heterotrophic biokinetic parameters in the BNR and CAS were very similar at 20 °C and 12 °C, the biomass specific nitrification rates in the MLE at 20 °C and 12 °C averaged 0.14 g NH_4 g^{-1} VSS d^{-1} and 0.053 g NH_4 g^{-1} VSS d^{-1}, respectively, as compared to 0.09 g NH_4 g^{-1} VSS d^{-1} and 0.037 g NH_4 g^{-1} VSS d^{-1} for the CAS at 20 °C and 12 °C respectively.
- Despite the lack of an explicit anaerobic zone in the MLE pilot, the system affected about 70% removal of phosphorus occurring without any chemical addition, with phosphorus release occurring in the anoxic tank.

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