The effect of drastic temperature changes on the performance of MBR treating municipal wastewater
Abdullah Al-Amri, Mohd Razman Salim and Azmi Aris

ABSTRACT

A study has been carried out to define the effect of drastic temperature changes on the performance of lab-scale hollow-fibre MBR in treating municipal wastewater at a flux of 10 L m⁻² h⁻¹ (LMH). The objectives of the study were to estimate the activated sludge properties, the removal efficiencies of COD and NH₃-N and the membrane fouling tendency under critical conditions of drastic temperature changes (23, 33, 42 & 33 °C) and MLSS concentration ranged between 6,382 and 8,680 mg/L. The study exhibited that the biomass reduction, the low sludge settleability and the supernatant turbidity were results of temperature increase. The temperature increase led to increase in SMP carbohydrate and protein, and to decrease in EPS carbohydrate and protein. The BRE of COD dropped from 80% at 23 °C to 47% at 42 °C, while the FRE was relatively constant at about 90%. Both removal efficiencies of NH₃-N trended from about 100% at 33 °C to less than 50% at 42 °C. TMP and BWP ascended critically with temperature increase up to 336 and 304 mbar respectively by the end of the experiment. The values of suspended solids (SS) and the turbidity in the final effluent were negligible. The DO in the mixed liquor was varying with temperature change, while the pH was within the range of 6.7–8.3.

Key words | activated sludge, COD removal, hollow fibre, membrane bioreactor, NH₃-N removal, temperature

INTRODUCTION

The thermophilic membrane bioreactor (MBR) process has many advantages. High biomass concentrations, high loading rates and producing good-quality effluent (Ramaekers et al. 2001; Huuhilo et al. 2002) are some of them. For the treatment of recirculated newsprint whitewater, Tardiff & Hall (1997) compared several alternatives and found that MBR is the most reliable under high temperatures.

Despite many researchers having proved the feasibility of the thermophilic process it still has troublesome problems. Temperature increase causes poor sludge settling properties (Huuhilo et al. 2002) and insufficient solid–liquid separation (Krishna & van Loosdrecht 1999). MBR is an advanced water treatment technology. The integration of MBR with the thermophilic aerobic process effectively solves the difficulty of solid–liquid separation (Zhang & Yang 2008). However, such application still requires more research.

The main aim of this research was to study the effect of temperature drastic changes (increasing and decreasing) on the performance of MBR in treating municipal wastewater. It was mainly to study activated sludge properties, COD and ammonia nitrogen biological and final removal efficiencies, and membrane fouling tendency in terms of transmembrane pressure (TMP) and backwash pressure (BWP) at temperature drastic changes (23, 33, 42 and 33 °C). The biological removal efficiency (BRE) is the removal efficiency obtained by biological activities before the membrane barrier and the final removal efficiency (FRE) is the removal efficiency obtained by both biological activities and membrane barrier filterability.

EXPERIMENTAL MATERIALS AND ANALYTICAL METHODS

The lab-scale MBR was set up as shown in Figure 1. The system consisted of a 10 L feed tank and 3.8 L reactor with a submerged membrane module. The module comprised a modified polyethersulfone (PES) hollow fibre membrane with outer/internal diameters of 0.9 and 0.55 mm respectively, and pore size of 0.2 μm with a total membrane area of...
The filtration driving force is created by negative suction pressure inside the hollow fibres and permeate was withdrawn using a peristaltic pump. The volumetric loading rate was 9 L/d. The operating mode was 2 h run and 30 min pause (14 min and 15 s suction and 45 s backwash). The sampling and data collection were carried out five days a week and each parameter was analyzed twice weekly. The lab-scale MBR was seeded with a biomass obtained from the returns activated sludge (RAS) line of Al-Ansab Wastewater Treatment Plant at Muscat City in Oman. The biomass was aerated for 48 h to enable acclimatization; thereafter the system was turned on progressively in order to feed the biomass daily with 7.3 L sewage from Pulai Utama Wastewater Treatment plant mixed with 850 ml synthetic wastewater from each medium, as shown in Table 1. Aeration of the membrane was conducted by coarse bubble which supplied oxygen for the biomass and air for membrane air scouring. The acclimatization and biomass development study was run for two weeks while the main study was run for eighteen days at temperatures 23, 33, 42 and 33 °C. Laboratory experiments associated with this study were carried out in the Environmental Engineering Laboratory, Faculty of Civil Engineering, Universiti Teknologi Malaysia. The operating period was 14 h/day and the operating flux was 10 LMH.

The experimental analyses were carried out within 3 h after sample collection according to standard methods (APHA 2005). The activated sludge was regularly tested for mixed liquor suspended solids (MLSS), mixed liquor volatile suspended solids (MLVSS) concentrations, sludge volume index (SVI) and dynamic viscosity using DV-I + viscometer. The activated sludge also was tested for soluble microbial products (SMP) and extracellular polymeric substances (EPS) according to the heating method (Morgan et al. 1990). The feed sewage, activated sludge and permeate solutions were analyzed for chemical oxygen demand (COD) and ammonia nitrogen (NH₃-N) using Hach DR5000 spectrophotometer, whereas for the soluble COD (CODsol) and soluble NH₃-N (NH₃-Nsol) of the supernatant was filtered with a 0.45 μm filter.

Table 1 | Chemical components of synthetic wastewater (Medium A & B and Trace elements)

<table>
<thead>
<tr>
<th>Chemical</th>
<th>Molarity (mmol/L)</th>
<th>Molecular weight (mg/mmol)</th>
<th>Volume (L)</th>
<th>Amount (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Medium A:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CH₃COONa</td>
<td>63</td>
<td>136.67</td>
<td>10</td>
<td>86.1</td>
</tr>
<tr>
<td>MgSO₄·7H₂O</td>
<td>3.6</td>
<td>246.47</td>
<td>10</td>
<td>8.87</td>
</tr>
<tr>
<td>KCl</td>
<td>4.7</td>
<td>74.56</td>
<td>10</td>
<td>3.5</td>
</tr>
<tr>
<td>Medium B:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH₄Cl</td>
<td>35.4</td>
<td>64.26</td>
<td>10</td>
<td>22.75</td>
</tr>
<tr>
<td>K₂HPO₄</td>
<td>4.2</td>
<td>174.18</td>
<td>10</td>
<td>7.32</td>
</tr>
<tr>
<td>KH₂PO₄</td>
<td>2.1</td>
<td>136.09</td>
<td>10</td>
<td>2.86</td>
</tr>
<tr>
<td>Trace element</td>
<td>10</td>
<td></td>
<td>10</td>
<td>93 mL</td>
</tr>
</tbody>
</table>

Trace solution (in 2 L of demi-water): EDTA 100 g, ZnSO₄·7H₂O 4.4 g, CaCl₂·2H₂O 16.36 g, MnCl₂·4H₂O 10.12 g, FeSO₄·7H₂O 9.98 g, Na₂MoO₄·2H₂O 3.02 g, CuSO₄·5H₂O 3.14 g, CoCl₂·6H₂O 3.22 g.

Source: Majone et al. (1999).
before testing using Hach DR5000 spectrophotometer. The influent and effluent were tested for suspended solids (SS) and turbidity using the standard method and Hanna microprocessor turbidity meter, respectively. The dissolved oxygen (DO) concentration and temperature of the influent, activated sludge and effluent were measured using portable DO YSI-55 DO meter, while pH was measured using Consort C535 multi parameter analyzer. The system was kept monitored during the whole study, where the transmembrane pressure (TMP) and the backwash pressure (BWP) were read twice daily.

RESULTS AND DISCUSSION

Temperature fluctuating

The temperature (T) of the mixed liquor was changed drastically. While the flux was constant at 10 LMH during the whole study, the temperature was raised from 23 to 33°C and then to 42°C to be finally calmed dawn to 33°C. The system was operated at the mentioned temperatures 8, 3, 3 and 2 days respectively (Figure 2). The parameters tested and monitored were MLSS, MLVSS, SVI, viscosity, DO, pH, EPS, SMP, removal efficiency (COD and NH3-N) and membrane operating pressures (TMP and BWP).

Activated sludge

Biomass growth

As a normal result, the biomass growth ascended during the first week (under room temperature (23°C)) from 7,500 to 8,680 mg/L MLSS depending on the food to microorganisms ratio (Marin et al. 2008). As the temperature rose, the biomass started descending to 8,200, 7,760 and then to 6,582 mg/L MLSS at temperatures 33, 42 (increasing) and 33°C (decreasing) respectively and correspondingly with the MLVSS values (Figure 3). This biomass reduction at high temperatures has been reported before in various studies (Surucu et al. 1976; Couillard et al. 1989; Tardiff & Hall 1997; Vogelaar 2002). The biomass reduction was the highest at the final phase when the temperature decreased to 33°C after increasing to 42°C. This surprising reduction in temperature caused the hardest biomass shock and from the results the highest biomass reduction. Thus, the biomass reduction is attributed to the biomass shock caused by the drastic temperature changes. Moreover, the filtration time for the MLSS and MLVSS tests was increasing ravidly with the time. It was 1, 1.2, 7 and 13 min at 23, 33, 42 and 33°C respectively. The filtration time increase could be ascribed to defloculation, high solubility, increase of supernatant turbidity and decrease of particle size correspondingly with the drastic temperature changes.

Except during the second phase (33°C) in which the volumetric loading rate was 1.12 COD kg/m³ d⁻¹, the volumetric loading rate was relatively constant, i.e. 1.7 COD kg/m³ d⁻¹ at all other phases. Excluding the second phase (33°C) in which the influent COD (In COD) was lower, the F/M ratio was increasing with the temperature increase due to the biomass reduction. It was 0.33, 0.23, 0.36 and 0.46 at the phases of 23, 33, 42 and 33°C respectively. The MLVSS/MLSS ratio was also relatively increasing with the temperature increase indicating the lower effect of the temperature on the MLVSS values. This relative increase in MLVSS/MLSS
ratio could be referred to the relative increase of VSS concentration in the influent which was 110 and 162 mg/L during the first and second phases respectively. However, the SS concentration in the effluent was negligible.

**Mixed liquor viscosity and sludge volume index (SVI)**

Figure 4 shows the mixed liquor viscosity and SVI changes under the four various temperatures of this study. While the SVI was increasing with the time and temperature changes, the mixed liquor viscosity was decreasing. Viscosity decrease was a result of biomass reduction and temperature increase. Thus, it started with 15 cP at 23 °C and then declined down to 8 cP at the final phase of 33 °C.

From 32 °C the supernatant of mixed liquor by settling for 30 min became more and more turbid and the sludge settlement became poorer with increasing temperature, the fact has been reported by Zhang et al. (2006). SVI increased with the temperature whereas it was 86, 104, 107 and 143 mL/g at 23, 33, 42 and 33 °C respectively. Due to the poor floc formation under thermophilic conditions, the thermophilic aerobic processes suffer from poor sludge settling properties (Rozich & Bordacs 2002), since the floc is a microbial aggregate, which must be large and dense enough to settle in solution (Zhang et al. 2006).

**Extracellular polymeric substances (EPS) and soluble microbial products (SMP)**

On the basis of the study of Mikkelsen & Keiding (2002) who suggested that SMP and EPS are the most important factors in floc structure, Vogelaar (2002) proposed that SMP production and EPS reduction correlate with the temperature and that a decrease of EPS inhibits floc formation. Recently, Zhang et al. (2006) reported that when the temperature increased from 40 to 45 °C, the EPS in the sludge was relatively steady while the SMP comparatively increased. In contrast, Zhang & Yang (2008) pointed to the decrease in total EPS with temperature increase. The total EPS in their study was relatively steady until 55 °C and beyond that it descended obviously. However, Tripathi (2000) found no statistically significant trend of the total EPS as a function of temperature in thermophilic activated sludge.

The results of this study show the increase of SMP carbohydrate and protein as well as the decrease of EPS carbohydrate and protein and EPS protein to carbohydrate ratio with temperature increase. The opposite thing occurred when the temperature was lessened from 42 to 33 °C (Figure 5). The SMP showed some reduction and the EPS obviously increased. Thus, the metabolic activity of microbes accelerates when temperature increases and so does the secretion of SMP (Zhang et al. 2006). The decrease of EPS protein and carbohydrate with temperature increase could justify the slow and/or inhibited floc formation under high temperature activated sludge, since it could be the main component of floc structure. Therefore, the increase of SVI discussed earlier and the supernatant turbidity, were mainly affected by SMP secretion at higher temperatures (Zhang et al. 2006).

**Dissolved oxygen and pH**

Notwithstanding, that the system was aerated continuously with a constant rate of 8 L/min, the dissolved oxygen (DO) in the reactor was affected by the temperature increase. Moreover, the pH value was affected by the DO concentration varying. During the whole experiment, the DO was higher than 1.5 mg/L and the pH was ranging between
Beyond the sudden temperature increase from 23 to 33 °C, the DO dropped from 3.5 to 1.6 mg/L as a result of biomass shock. That DO drop was an attempt by microorganisms to survive what cost them a large amount of energy. Then, the DO increased gradually with the biomass reduction illustrating the failure of the biomass to adapt faster to the new conditions. The DO concentrations during temperature increases were increasing from 1.6 mg/L at 33 °C to 4 mg/L at 42 °C and then to 5.5 mg/L beyond temperature decrease to 33 °C. Thus, the DO concentration increases with temperature increase due to the microorganism death (less richness and diversity) caused by temperature shock. As expected, the ML pH (mixed liquor pH) was varying correspondingly with the DO concentration ascendingly and descendingly, despite the almost constant In pH (influent pH) which was around 7 during the whole study. The highest ML pH (8) was during the last two phases because of the biomass decline and DO increase. Unlike the beginning (at 23 and 33 °C), the Ef pH (effluent pH) was relatively higher than the ML pH as a result of lengthening of membrane bore diameters by the end of the study due to the temperature increase at 42 °C.

Removal efficiency

COD removal efficiency

Thermophilic activated sludge processes produce lower COD removal than mesophilic processes (Zhang & Yang 2011). This fact is due to the negative effect of temperature increase on activated sludge properties, which appears in the lower removal efficiency. Figure 7, shows values of both COD removal efficiencies, biological (Bio RE) and final (Fin RE). At temperatures 23, 33, 42 and 33 °C, the Bio RE was 80.2, 69.2, 47.2 and 48.3% while the Fin RE was 92.5, 85.3, 89.4, and 88.8% respectively. The average value of In COD was 621 mg/L and the CODsol was 491 mg/L. The filtered supernatant COD (ML COD) was 100, 96, 300 and 300 mg/L, while the effluent COD was 38, 46, 60 and 65 mg/L at 23, 33, 42 and 33 °C respectively.

The results imply that temperature increase has a significant influence on the Bio RE but the influence on Fin RE is slight. This trend in Bio RE with increasing temperature has been reported earlier (Murthy 1998; Liu & Fang 2002; Suvilampi 2003). It is expected that high temperature biomass is unable to oxidize the same variety of complex soluble components of which the mesophilic biomass is capable (Vogelaar 2002). This inability could be ascribed to the reduction in microbial richness and diversity caused by sudden changes in operational conditions (LaPara et al. 2000) and to the decay of bacteria, which releases SMP and EPS into the solution, which increases CODsol in the supernatant. Moreover, this deterioration can be attributed to the short HRT (insufficient contact time) of the system, since activated sludge systems need long HRT at high temperatures. However, the Fin RE was comparatively steady revealing the excellent filtering ability of hollow fiber membrane to macromolecule organic compounds.

NH₃-N removal efficiency

Since most previous studies related to this subject have been interested in the applications of MBR in treating industrial wastewater, the reports of nitrification and denitrification at high temperatures were very rare. Accordingly, high and constant nitrification has not been achieved under high temperature aerobic conditions.
Zhang et al. (2006) also documented that the nitrification process decreases with increasing temperature. This fact could be justified by the reduction in microbial richness and diversity caused by the drastic temperature changes. The inability of high temperature wastewater treatment systems to separate sludge from effluent liquid combined with low sludge yields will result in biomass washout and low quality effluent (Kurian & Nakhla 2006).

Figure 8, demonstrates Ammonia nitrogen (NH$_3$-N) values of Bio RE and Fin RE. Although, NH$_3$-N values of Soluble Influent (S In) were lying around 60 mg/L, the Effluent (Ef) values were negligible and 0.25 mg/L at 23 and 33°C respectively. When the temperature increased from 33 to 42°C and then reduced again to 33°C, the values of NH$_3$-N in the Ef increased to 39 and 42 mg/L respectively. Both NH$_3$-N removal efficiencies Bio and Fin were very high (99.6 and 100%) at 23 and 33°C, but when temperature increased to 42°C and then reduced to 33°C, both removals dropped down more than 50%. The low nitrification process can be attributed to the same factors (discussed earlier) causing low sludge settlement and high COD$_{sol}$ in the supernatant. The strange fact is that at 42 and 33°C, the Fin RE was lower than the Bio RE. This could be explained by the widening of membrane bore diameters as the temperature increases, which allows more colloidal organic compounds to flee through the membrane.

Turbidity and effluent color

Despite the filtered supernatant becoming more and more turbid with increasing temperature, the final removal efficiency of turbidity was excellent at all temperature. The excellent removal efficiency of the turbidity illustrates the superior properties of the hollow-fiber membrane. Although, the turbidity in the influent was 38, 40, 50 and 54 FTU at the four phases, the turbidity removal efficiency was 100% during the rise up of the temperature from 23 to 33°C and then up to 42°C. As an expected result of drastic temperature reduction from 42 to 33°C, the turbidity removal efficiency was slightly affected but it was still very high (99.98%). The high turbidity of the supernatant can be attributed to supernatant colloidal COD which increases in a thermophilic sludge due to it’s inability to oxidize influent colloidal material (Vogelaar et al. 2002a, b) and to the absence of higher organisms, which consume the free bacteria in mesophilic conditions (Zhang et al. 2006).

The effluent was clear at 23°C but upon temperature rise up to 33°C, its color became pale-yellow for a while (3 h) and it became clear again by the end of the day. Moreover, at temperatures of 42 and 33°C, the effluent color became yellowish and that was until the end of the study. This colored effluent can be explained by the high mixed liquor solubility, which causes fleeing of contaminants through the membrane in the thermophilic MBR processes (Kuriana et al. 2005).

Membrane performance and fouling

Generally in this study, both TMP and BWP were increasing with temperature increase indicating the membrane fouling tendency with mixed liquor temperature increase. The lowest fouling rate was observed during the first phase while the highest was during the last one (Figure 9).

TMP was flocculating at the first phase of 23°C ascending and descending since the fouling was still reversible
(external). The main factors behind this type of fouling were the coagulation of sludge aggregations and cake formations between and within membrane fibers (Mohammed et al. 2008). These coagulations can be removed by the bubble course of the aeration system, thus the TMP descended many times in this phase.

As the temperature increased, many other factors took place to make the fouling irreversible (internal) rather than reversible. These factors are such as biomass deterioration, SMP secretion (Zhang et al. 2006), ML pH increase, particle size reduction and membrane pore widening. Therefore, TMP and BWP ascended obviously with drastic temperature changes. Thus, the BWP was zero during the first phase but it was 100, 250 and 305 mbar during second, third and fourth phases respectively, since the BWP reflects more the irreversible fouling. Otherwise, the TMP was 62, 132, 175 and 336 mbar during the first, second, third and forth phases respectively. TMP descents occurring during the first phase did not occurred any more in the other phases due to the increase of membrane irreversible fouling.

**CONCLUSION**

The activated sludge properties, the removal efficiencies and the membrane fouling phenomena were investigated in this study. The following conclusions were obtained:

The drastic temperature changes caused biomass shock, high solubility, high SMP secretion and EPS reduction. These factors were responsible for biomass reduction and deterioration, and mixed liquor viscosity descent. As a result, the sludge settleability became poorer, the supernatant colloidal COD increased and the mixed liquor turbidity increased as well.

Both removal efficiencies biological and final of COD and NH$_3$-N were affected by drastic temperature increase particularly beyond 40 °C, due to the rapid reduction in microbial richness and diversity. Notwithstanding, the COD final removal efficiency remained higher up to about 90% and the turbidity removal efficiency up to 100%, illustrating the excellent ability of the membrane for retention.

The membrane bore widening, serious SMP release, turbid supernatant and poor sludge settling properties had the membrane fouling irreversibly rather than reversibly with temperature increase. Hence, both TMP and BWP ascended critically with time when the temperature started rising up. Despite the critical membrane fouling, the hollow fiber membrane had relatively longer service life.

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