Available online at www.sciencedirect.com
ScienceDirect
Journal of Membrane Science 283 (2006) 164–171
journal of MEMBRANE SCIENCE
www.elsevier.com/locate/memsci

# Effect of high salinity on activated sludge characteristics and membrane permeability in an immersed membrane bioreactor

E. Reid [a], Xingrong Liu [b], S.J. Judd [a,*]

[a] *School of Water Sciences, Cranfield University, Bedford, Bedfordshire, UK*
[b] *School of Public Health, Lanzhou University, 730000 Lanzhou, PR China*

Received 19 April 2006; received in revised form 13 June 2006; accepted 14 June 2006
Available online 18 June 2006

**Abstract**

The influence of high salinity on the characteristics of the activated sludge and performance of a pilot-scale immersed membrane bioreactor (iMBR) has been studied. The bioreactor was subjected to salinity shocks of up to 5 g/L, and the response with respect to membrane permeability monitored. Key physical and chemical parameters were measured included mixed liquor suspended solids (MLSS), viscosity, capillary suction time (CST), turbidity, and the soluble microbial product (SMP) and extracted extracellular polymeric substances (EPS) of the mixed liquor. The interrelationships between these parameters and the membrane permeability were then assessed. Results indicate that high salinity greatly affects the physical and biochemical properties of activated sludge, increasing SMP and EPS concentrations, as well as decreasing membrane permeability. Both SMP and EPS were correlated with physical parameters of the activated sludge such as particle size, CST and turbidity. Furthermore, permeability was found to be negatively correlated with SMP carbohydrate at the two different flux rate imposed, corroborating previous reports linking SMP carbohydrate to iMBR membrane fouling.


*Keywords:* Salinity; Activated sludge; Membrane bioreactor; Permeability; Fouling

## 1. Introduction

Membrane bioreactors (MBRs), the combination of membrane separation and the activated sludge process, are now widely used in wastewater treatment, and their advantages are well recognised [1]. Their performance is also known to be limited by fouling, which constrains both operating flux and, ultimately, membrane life [2] from colloidal/macromolecular free and bound species derived from the biomass [3–5], respectively referred to as soluble microbial products (SMP) and extracelluar polymeric substances (EPS) [6]. These materials include bacterially produced polymeric, lysis and hydrolysis products [7], and comprise mainly polysaccharides, nucleic acids, and proteins [8]. Their chemical composition and molecular weight depend on the feedwater composition, the physiology of microorganisms, the nutritional status and the imposed environment created by the biological system operation [9]. They are not only responsible for adhesion of aggregates to maintain the floc structure but are also key membrane foulants in MBR systems [10–12], and have been the subject of many MBR fouling studies [8,10,13–15].

High or changing salinity presents a challenge to biotreatment processes in general when coastal sewers are subjected to infiltration by seawater or when industrial effluents receive discharges from individual high-salinity processes. In areas lacking fresh water seawater may also be used for sewage flushing, again imposing a saline shock load. Moderate to high salinities are known to produce inhibitory or toxic effects on bacteria not adapted to high salinity; high salt concentrations (>1%) have been shown to cause plasmolysis and/or loss of activity of cells [16–18]. Moreover, the microbial community is also altered by salinity changes. Additionally, salinity significantly affects the physical and biochemical properties of the activated sludge, leading to changes in surface charge, hydrophobicity, filterability, settlement and bioflocculation.

Whilst some studies have been conducted which have identified the onerous impacts of salinity shocks on conventional biotreatment with reference to COD removal [19,20], none have

* Corresponding author. Tel.: +44 1234 754842; fax: +44 1234 751671.
*E-mail address:* s.j.judd@cranfield.ac.uk (S.J. Judd).

0376-7388/$ – see front matter © 2006 Elsevier B.V. All rights reserved.
doi:10.1016/j.memsci.2006.06.021

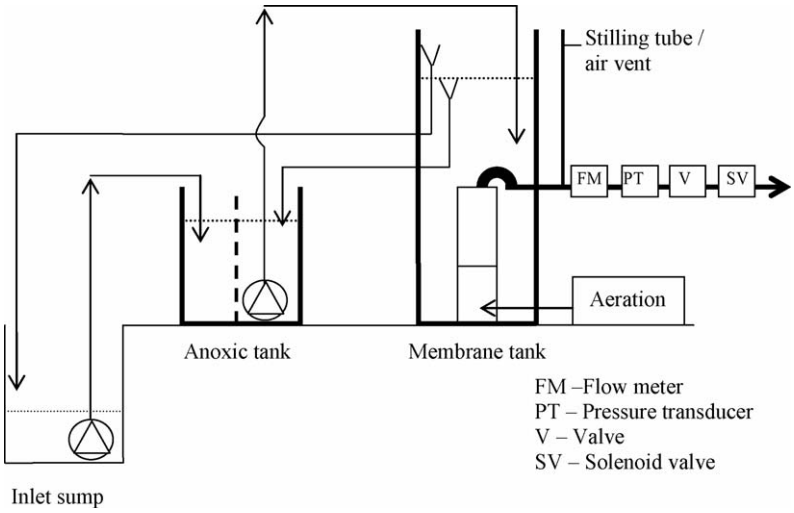


Fig. 1. Pilot plant schematic diagram.

either characterised the organic matter arising in the increased effluent COD levels or have concerned MBRs. In this study, pilot scale trials were carried out to treat low-salinity wastewater with an immersed membrane bioreactor (iMBR) and the effect of shock-loads of saline wastewater on both the characteristics of the activated sludge and performance of the iMBR observed. Key physical and chemical parameters such as mixed liquor suspended solids (MLSS), viscosity, capillary suction time (CST), turbidity, and SMP and EPS concentrations of the activated sludge were examined. The interaction and correlation between those parameters were studied, along with their possible correlation with both salinity shocking and membrane fouling.

## 2. Materials and methods

### 2.1. MBR system description

Experiments were conducted on a pilot scale aerobic iMBR (Fig. 1) fed with municipal sewage. The iMBR system consisted of seven flat pate Kubota membrane panels placed in the centre of the bioreactor with a working volume of 3.2 $m^3$. The membrane was of chlorinated polyethylene, and the seven panels provided a total filtration area of 5.6 $m^2$. A pressure transducer was installed to monitor the pressure loss across the membrane. The liquid level in the membrane tank was maintained by a bellmouth which provided the recycle back to the anoxic tank and this hydraulic head provided the gravity pressure to push the permeate through the membranes. Another bellmouth situated higher in the membrane tank provided an emergency overflow which discharged into the inlet sump.

Trials were performed at Trowbridge Sewage Treatment Works at two different fluxes. The seeding activated sludge was collected from another larger iMBR on the same site. Prior to saltwater simulations trials, the pilot plant was first operated for a number of months to establish steady state. The iMBR was operated with air scouring ("relaxation") for 30 min every 24 h, the membrane aeration rate being 5.5 $Nm^3/h$ (i.e. 0.79 $Nm^3\ h^{-1}\ m^{-2}$ membrane area). The solids retention time (SRT) was fixed at 64 days, and the hydraulic retention time (HRT) was 72 h for the 8 $L\ m^{-2}\ h^{-1}$ trial and 36 h for the 16 $L\ m^{-2}\ h^{-1}$ trial. The initial MLSS level was ~9000 mg $L^{-1}$, and during the trials it was maintained at ~17,000 mg $L^{-1}$. The average chemical oxygen demand (COD), chloride ($Cl^-$) and ammonia ($NH_3$–N) influent concentrations were 1084, 348 and 32 mg $L^{-1}$, respectively. The mean pH was 7.24, and the dissolved oxygen (DO) concentration ranged between 1 and 3 mg $L^{-1}$.

Salinity was introduced by dosing the bioreactor directly with a 30 g $L^{-1}$ NaCl solution over a period of 10 min. The response was noted with respect to mixed liquor conductivity, pressure and flow, and thus permeability.

### 2.2. Analytical methods

#### 2.2.1. Analysis of SMP and EPS in mixed liquor

SMP and EPS were extracted by heat treatment [21,22] and the extracted solution analysed for total proteins and carbohydrate, the dominant components typically found in SMP and EPS [6,23]. Carbohydrate was determined according to the phenol–sulphuric acid method using glucose as the standard [3]. Protein was determined by the Folin method using bovine serum albumin as the standard [24].

#### 2.2.2. Analysis of other parameters

CST was measured by a conventional CST instrument as detailed in Standard Methods with a Whatman No. 17 chromatography grade paper [25]. The apparent viscosity was determined using a rotational viscosity meter (Model LVDVII, Brookfield, England) [26]. Turbidity was determined by a Hach 2100P Turbidmeter, and COD, MLSS and $NH_3$–N were analysed according to Standard Methods [25].

### 2.3. Statistical analysis

Statistical analysis was performed to identify major determining factors by univariate linear correlation analysis. Whilst

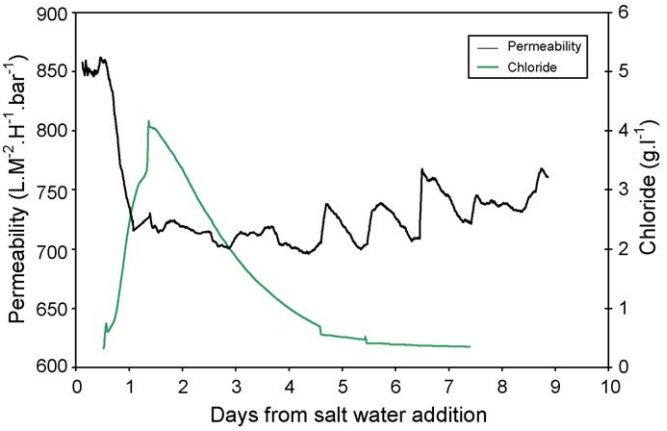


Fig. 2. Permeability during saltwater addition at a flux of 8 L m$^{-2}$ h$^{-1}$.

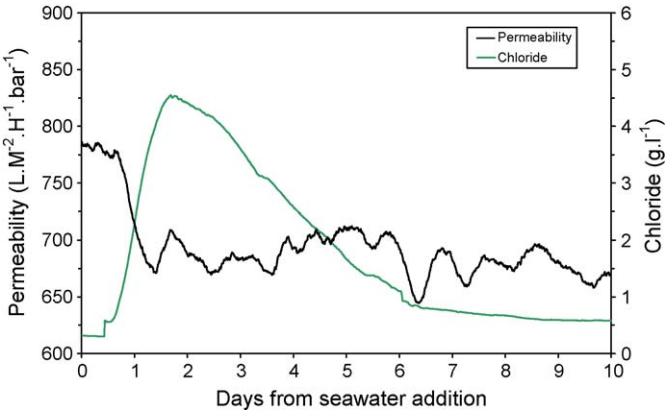


Fig. 3. Permeability during saltwater addition at a flux of 16 L m$^{-2}$ h$^{-1}$.

many relations are unlikely to be linear, analysis by means of linear correlations represents a simple way to identify significant correlations. It is also important to note that some parameters are not independent. The Pearson's product momentum correlation coefficient ($r_p$) was used for linear estimations of the strength and direction of linear correlations between two parameters. The Pearson's coefficient ($r_p$) is always between −1 and +1, where −1 indicates a perfect negative correlation, +1 a perfect positive correlation and 0 no correlation. The statistical analyses were performed with the software *Statistica* (Statsoft, Tulsa, OK, USA).

## 3. Results and discussion

### 3.1. Overall performance of the iMBR

Fig. 2 shows the profiles of permeability and chloride level for the iMBR at a flux of 8 L m$^{-2}$ h$^{-1}$. On introducing salinity the chloride level increased to a peak of ∼4 g L$^{-1}$ after 20 h before gradually returning to the initial background level after 5 days. The average permeability decreased immediately following the salt shock, falling from 850 to 725 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ over 22 h of operation and giving an average permeability decline rate of 5.8 L m$^{-2}$ h$^{-2}$ bar$^{-1}$. The permeability eventually stabilised, appearing to recover from a minimum of ∼700 at 4–5 days to ∼750 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ after 9 days and remaining at this level for the next 3 weeks of operation. Noticeable permeability recovery arose following relaxation. Profiles of permeability and chloride level at flux of 16 L m$^{-2}$ h$^{-1}$ (Fig. 3) were similar to those attained at the lower flux. In this case, however, no permeability recovery took place, the permeability declining from ∼770 to 650 L m$^{-2}$ h$^{-1}$ bar$^{-1}$ following salinity shocking.

COD and $NH_3$–N removal values are shown in Table 1 with reference to average feedwater concentration values for COD, chloride and ammonia of 1084, 348 and 32 mg L$^{-1}$, respectively, for trials operated at fluxes of 8 and 16 L m$^{-2}$ h$^{-1}$. Consistent COD and $NH_3$–N removal of ∼99% were recorded under steady state conditions. Following salt addition COD removal was affected immediately and took around one week to recover, whereas nitrification was less profoundly affected. This would appear to be in keeping with previously reported observations of inhibitory effects of salt shocking on BOD or COD removal [17,27–29], and adverse effects have also been reported on flocculation and sludge settleability [27]. Kincannon and Gaudy [28] studied the effects of shock and continuous salt loadings on the batch and continuous flow activated sludge systems and concluded that high salinity shocks caused a drastic decrease in the substrate removal efficiency, but that acclimatisation restored this capability. Burnett [30] studied the impact of intermittently introduced highly saline sewage on the activated sludge process. Results indicated that after an initial lag period the microbial population was able to adapt to the high salinity environment and the floc formation was not inhibited. In the current study, the iMBR performance displayed similar tendency to those reported previously for high-salinity media [28,30]. Delayed nitrification inhibition has also been previously reported [27], but the effect recorded in the current study is less pronounced than that on COD removal.

### 3.2. SMP and EPS in the mixed liquor

The carbohydrate and protein content of the SMP and EPS were measured prior to and following chloride addition at 8 and 16 L m$^{-2}$ h$^{-1}$. Their concentrations were found to increase rapidly at low salinities (0–1 g L$^{-1}$ NaCl), with rather less change between 1 and 4 g L$^{-1}$ salt concentration (Table 2 and Figs. 4–7). On a temporal basis the concentration values initially increased, reaching the maximum values on day 2, before gradually returning to the background concentration by day 6. Generally, the content of carbohydrate was higher than that of protein in both SMP (Fig. 5) and EPS (Fig. 7), with greater changes at lower flux values (and thus longer HRTs).

Table 1
Mean COD and ammonia removal efficiencies before and after chloride addition

| | Removal efficiency (%) | | | |
|---|---|---|---|---|
| | Steady state | 8 h | 24 h | 1 week |
| COD | 99 | 95 | 88 | 98 |
| $NH_3$–N | 99 | 99 | 95 | 98 |

Table 2
Average fraction of SMP and EPS at high and low chloride levels

| Fraction | Flux ($L\,m^{-2}\,h^{-1}$) | Average concentration | |
|---|---|---|---|
| | | Low chloride level | High chloride level |
| EPS ($g\,kg^{-1}$ MLSS) | | | |
| Protein | 8 | 41.28 | 49.20 |
| | 16 | 37.54 | 46.06 |
| Carbohydrate | 8 | 45.43 | 112.54 |
| | 16 | 54.68 | 88.84 |
| SMP ($g\,L^{-1}$) | | | |
| Protein | 8 | 6.71 | 38.90 |
| | 16 | 10.84 | 22.98 |
| Carbohydrate | 8 | 18.22 | 40.61 |
| | 16 | 31.65 | 44.78 |

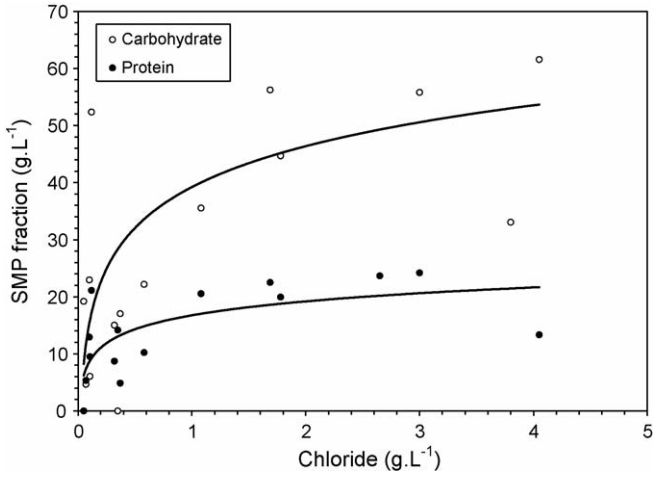


Fig. 4. The variations of SMP fractions with chloride level at $8\,L\,m^{-2}\,h^{-1}$.

SMP and EPS chemical composition are known to be highly variable and dependent upon, amongst other things, microbial diversity and physiology. The published literature generally indicates that microorganisms respond to a salt shock by aggregation of the individual cells [20,31] and acceleration of endogenous respiration [28,32], accompanied by the release of organic cellular constituents (such as SMP and EPS) by secretion and cells autolysis. Whilst no significant impact of salinity shocking on particle size distribution was noted in the current study, the impact on SMP and EPS was palpable, suggesting that physiological impacts on microorganisms took place.

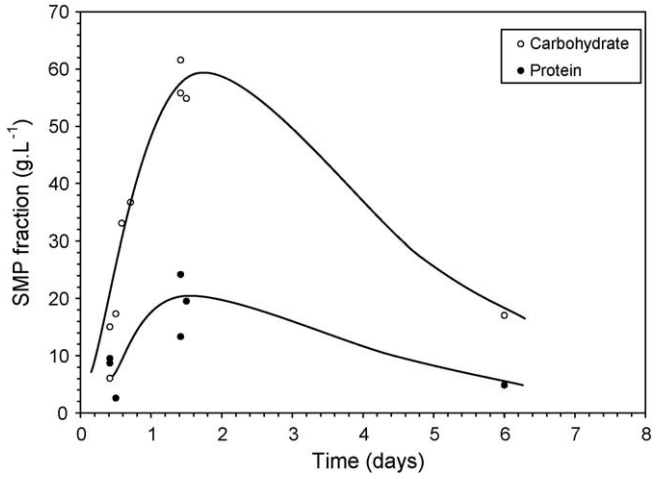


Fig. 5. The variations of SMP fraction with time at $16\,L\,m^{-2}\,h^{-1}$.

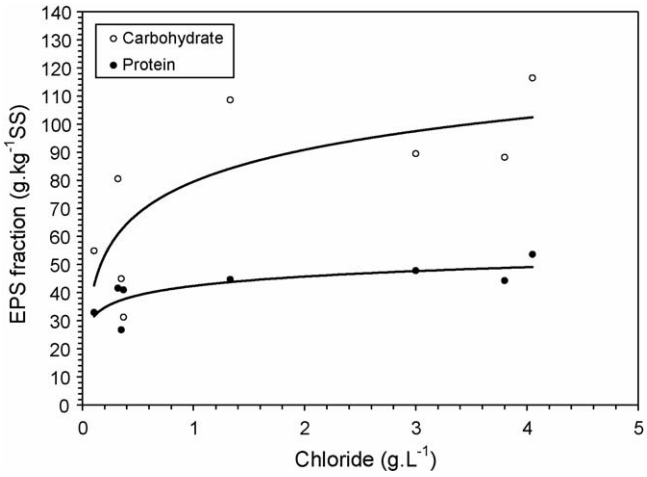


Fig. 6. The variations of EPS fraction with chloride at $8\,L\,m^{-2}\,h^{-1}$.

The higher carbohydrate than protein levels are in keeping with a previous report based on the same extraction method [22], where levels also gradually reverted to the pre-shock levels when the salinity was reduced. On the other hand, there is a linear relationship between EPS protein and carbohydrate (Fig. 8), possibly consistent with the release of cellular constituents under salinity shock conditions. No correlation between the SMP and EPS protein fraction levels was observed in the current study, indicating that salt shocking does not appear to have any significant impact on the distribution of bound and unbound protein. This may reflect that the solids hydrophobicity is unchanged by salt shocking, though this would be counter-intuitive.

### 3.3. CST correlations

Since salinity apparently affects sludge settlement, filterability and dewaterability [18,20,28,30], it may also be expected

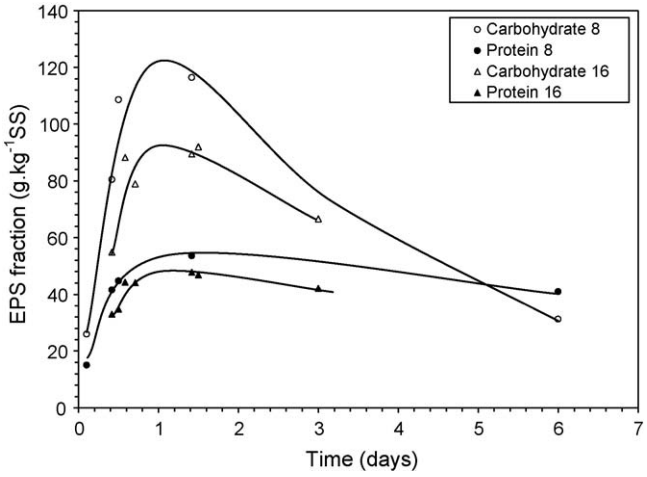


Fig. 7. The variations of the EPS fractions at the two flux values.

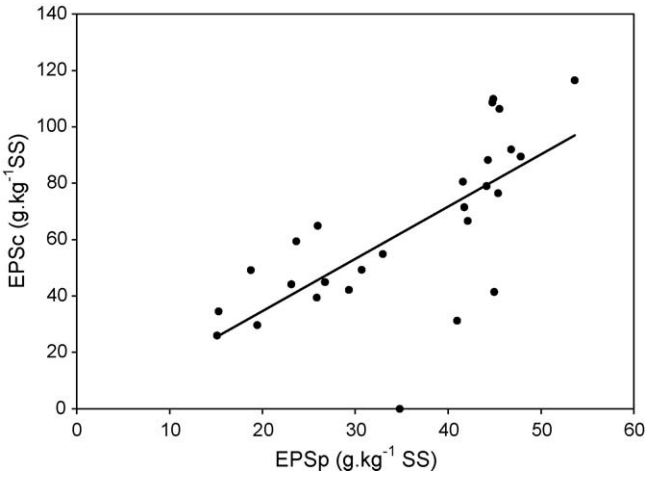


Fig. 8. Protein vs. carbohydrate fraction of EPS.

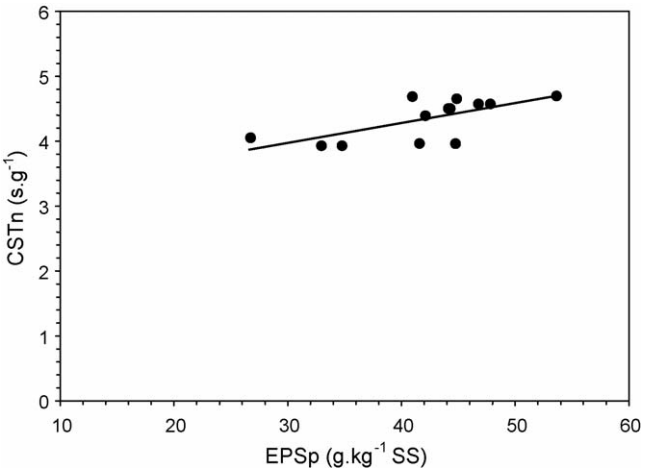


Fig. 10. CSTn va. EPS protein concentration.

to influence capillary suction time (CST). CST was found to be highly correlated with MLSS (Fig. 9). CSTn, the CST normalised against solids concentration, did not correlate with chloride concentration and increased only marginally with EPS protein (Fig. 10, $r_p = 0.67$), SMP carbohydrate (Fig. 11, $r_p = 0.66$) and MLSS ($r_p = 0.60$). These results would appear to suggest that, in this case, neither the carbohydrate nor the protein levels are a good indicator of sludge filterability. It is important to note, however, that CSTn does not necessarily correlate directly with fouling, particularly long-term fouling which may more reasonably be associated with colloidal matter [1].

### 3.4. SMP, EPS and turbidity

Turbidity provides an indication of the level of colloid material and residual suspended matter. SMP turbidity correlated positively with chloride level (Fig. 12, $r_p = 0.87$ and SMP carbohydrate ($r_p = 0.81$); EPS turbidity also positively correlated with EPS protein (Fig. 13, $r_p = 0.74$), but not with

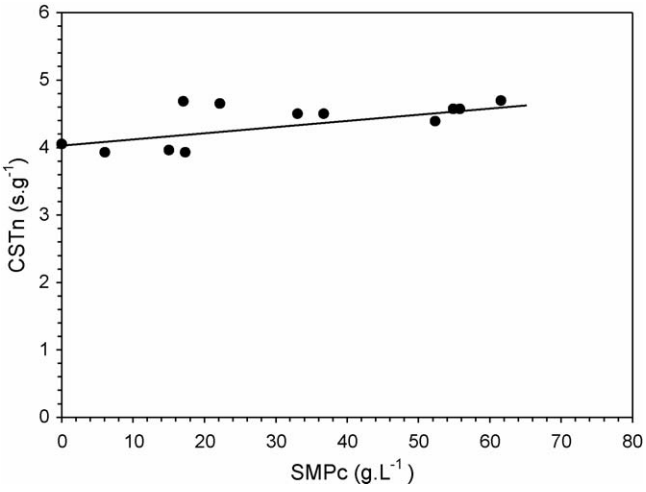


Fig. 11. CSTn vs. SMP carbohydrate concentration.

EPS carbohydrate. This is mainly due to the combined impacts of "salting-in" and protein effects. At low salt concentration ions are known to bind electrostatically to charged groups on a protein macromolecule, carrying hydration water with them

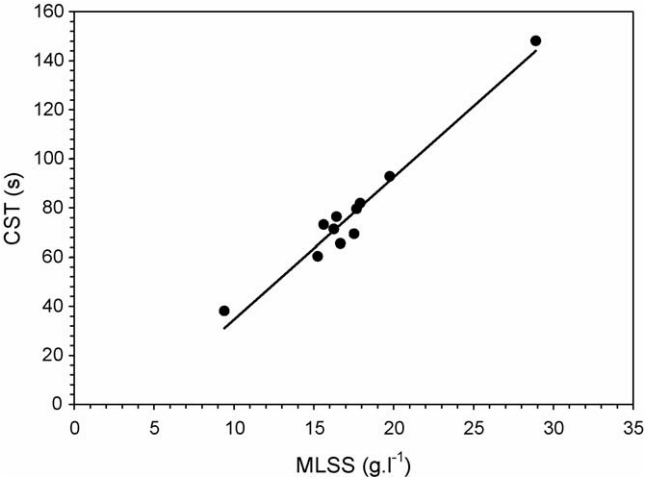


Fig. 9. CST vs. MLSS.

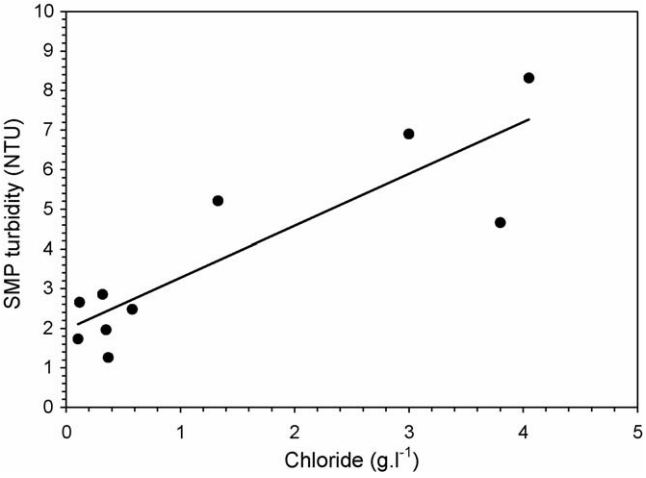


Fig. 12. The change of SMP turbidity with chloride level.

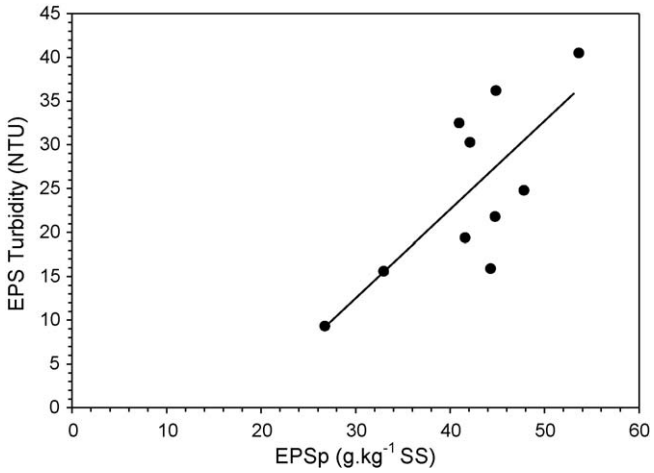


Fig. 13. The change of EPS turbidity with EPS protein.

into the vicinity of the protein and causing the "salting-in" effect [33]. This increases the protein solubility. On the other hand, protein has been shown to increase the floc negative surface charge [33], leading to strong electrostatic repulsive force between flocs. In such cases, flocs with high negative surface charge were found to be lower in mechanical strength, tending to disintegrate and so release colloidal material.

### 3.5. MLSS, viscosity, SMP and permeability

As seen in Table 1 and Figs. 14 and 15, permeability was loosely negatively correlated with MLSS ($r_p = -0.54$), but more strongly with sludge viscosity ($r_p = -0.76$). However no clear correlation of permeability with MLSS was evident from the data from this study. The carbohydrate portion of the SMP weakly correlated with permeability for the two sets of flux data (Fig. 16).

Carbohydrate constituents are present at generally higher molecular weights, and also wider molecular weight distributions, than the proteinacious materials. Previous studies have

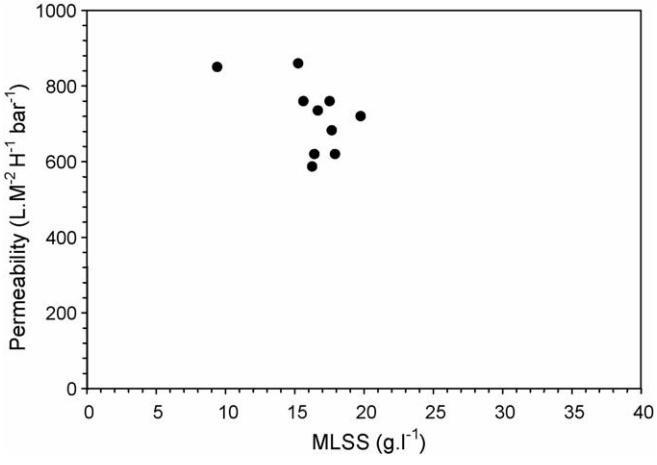


Fig. 14. The change of permeability with MLSS.

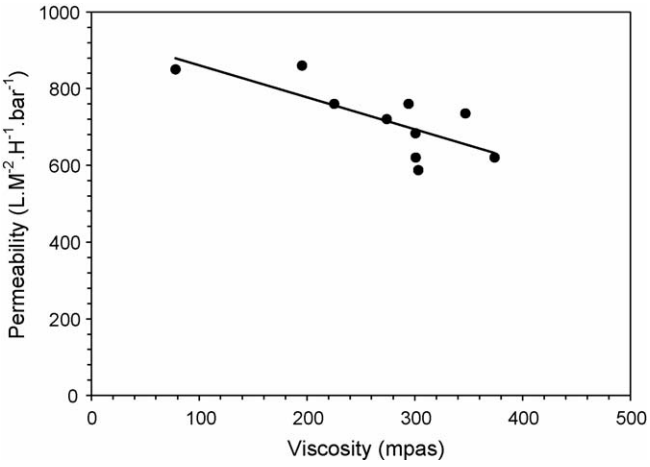


Fig. 15. The change of permeability with viscosity.

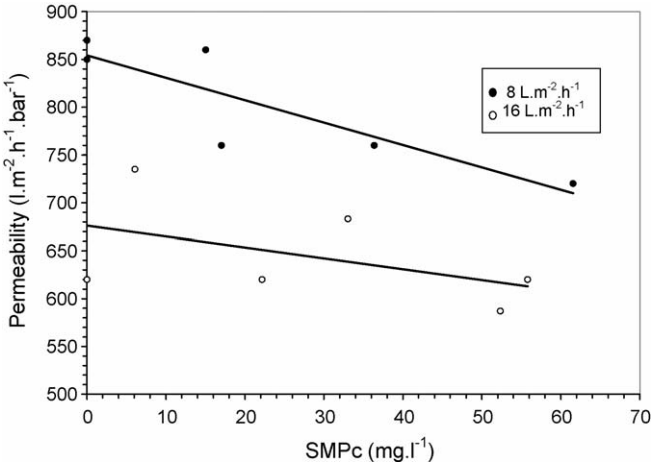


Fig. 16. The permeability decline with SMP carbohydrate at 8 and 16 L m$^{-2}$ h$^{-1}$.

shown the important relationship between MBR fouling and the sludge carbohydrate fraction [8,34–36]. Moreover, high molecular weight carbohydrates can promote the formation of "sticky" hydrogels membrane surfaces [35,36]. Since membranes are exposed directly to SMP in the bulk phase this fraction that would be expected to have a greater influence on membrane fouling than the EPS fraction. The negative correlation between permeability and SMP carbohydrate is thus consistent both with expectations and previously reported trends.

## 4. Conclusions

This study investigated the characteristics of activated sludge and membrane permeability in a system exposed to rapid changes in salinity (or "salt shocking"). The study revealed salt shock to significantly affect physical and biochemical properties of activated sludge, which then impacted upon membrane permeability. The following specific conclusions can be drawn:

1. High salinity has an initially negative influence both on membrane permeability and COD and $NH_3$–N removals. When

the salt is removed the membrane permeability stabilises and COD and $NH_3$–N removal efficiencies are restored.
2. A salt shock results in increased SMP and EPS concentrations. Both carbohydrate and protein levels in both EPS and SMP rise with increased salinity, the EPS carbohydrate fraction increasing more than EPS protein. Following a reduction in salinity both parameters eventually returned to their original levels.
3. A negative correlation was observed between permeability and SMP carbohydrate at different flux rates, corroborating previous reports which have identified SMP carbohydrate as playing a key role in membrane fouling.

The study thus reveals that effluents which are subject to salinity shocks over the range of that studied (0.1–4 g $L^{-1}$) are likely to be more challenging to an iMBR than those where salinity levels are readily stable, even at the modest fluxes studies (8 and 16 L $m^{-2}$ $h^{-1}$). Coastal-based installations suffering seawater intrusion and those treating certain industrial effluents may therefore benefit from the use of buffer tanks, or possibly pre-sedimentation which would have the additional benefit of reducing the organic load on the iMBR. Further work is needed to elucidate the precise mechanism by which salinity-promoted fouling takes place.

## Acknowledgements

The authors offer their grateful thanks to EPSRC and Copa Ltd. for funding this research and to Wessex Water for the supply of sewage and land for the pilot plant experiments. Prof. Liu is funded by the CSC, whose contribution is gratefully acknowledged.

## References

[1] S. Judd, The MBR Book, Elsevier, 2006.
[2] L. Ji, J. Zhou, Influence of aeration on microbial polymers and membrane fouling in submerged membrane bioreactors, J. Membr. Sci. 276 (2006) 168–177.
[3] J.-S. Kim, C.-H. Lee, I.-S. Chang, Effect of pump shear on the performance of a crossflow membrane bioreactor, Water Res. 35 (2001) 2137–2144.
[4] B.D. Cho, A.G. Fane, Fouling transients in nominally sub-critical flux operation of a membrane bioreactor, J. Membr. Sci. 209 (2002) 391–403.
[5] C. Nuengjamnong, J.H. Kweon, J. Cho, C. Polprasert, K.-H. Ahn, Membrane fouling caused by extracellular polymeric substances during microfiltration processes, Desalination 179 (2005) 117–124.
[6] R. Bura, M. Cheung, B. Liao, J. Finlayson, B.C. Lee, I.G. Droppo, G.G. Leppard, S.N. Liss, Composition of extracellular polymeric substances in the activated sludge floc matrix, Water Sci. Technol. 37 (1998) 325–333.
[7] P.H. Nielsen, A. Jahn, R. Palmgren, Conceptual model for production and composition of exopolymers in biofilms, Water Sci. Technol. 36 (1997) 11–19.
[8] B.-M. Wilen, B. Jin, P. Lant, The influence of key chemical constituents in activated sludge on surface and flocculating properties, Water Res. 37 (2003) 2127–2139.
[9] B.-M. Wilen, B. Jin, P. Lant, Impacts of structural characteristics on activated sludge floc stability, Water Res. 37 (2003) 3632–3645.
[10] Y. Ye, P. Le Clech, V. Chen, A.G. Fane, B. Jefferson, Fouling mechanisms of alginate solutions as model extracellular polymeric substances, Desalination 175 (2005) 7–20.
[11] I.-S. Chang, C.-H. Lee, Membrane filtration characteristics in membrane-coupled activated sludge system—the effect of physiological states of activated sludge on membrane fouling, Desalination 120 (1998) 221–233.
[12] C.S. Laspidou, B.E. Rittmann, A unified theory for extracellular polymeric substances, soluble microbial products, and active and inert biomass, Water Res. 36 (2002) 2711–2720.
[13] E.H. Bouhabila, R. Ben Aim, H. Buisson, Fouling characterisation in membrane bioreactors, Sep. Purif. Technol. 22–23 (2001) 123–132.
[14] P. Le-Clech, B. Jefferson, S.J. Judd, A comparison of submerged and side stream tubular membrane bioreactor configurations, Desalination 173 (2005) 113–122.
[15] S. Rosenberger, C. Laabs, B. Lesjean, R. Gnirss, G. Amy, M. Jekel, J.C. Schrotter, Impact of colloidal and soluble organic material on membrane performance in membrane bioreactors for municipal wastewater treatment, Water Res. 40 (2006) 710–720.
[16] A.R. Dincer, F. Kargi, Performance of rotating biological disc system treating saline wastewater, Process Biochem. 36 (2001) 901–906.
[17] F. Kargi, A.R. Dincer, Saline wastewater treatment by halophile-supplemented activated sludge culture in an aerated rotating biodisc contactor, Enzyme Microb. Technol. 22 (1998) 427–433.
[18] F. Kargi, A.R. Dincer, Effect of salt concentration on biological treatment of saline wastewater by fed-batch operation, Enzyme Microb. Technol. 19 (1996) 529–537.
[19] A. Uygur, F. Kargi, Salt inhibition on biological nutrient removal from saline wastewater in a sequencing batch reactor, Enzyme Microb. Technol. 34 (2004) 313–318.
[20] M.F. Hamoda, I.M.S. Al-Attar, Effects of high sodium chloride concentrations on activated sludge treatment, Water Sci. Technol. 31 (1995) 61–72.
[21] J.W. Morgan, C.F. Forster, L. Evison, A comparative study of the nature of biopolymers extracted from anaerobic and activated sludges, Water Res. 24 (1990) 743–750.
[22] X. Zhang, P.L. Bishop, B.K. Kinkle, Comparison of extraction methods for quantifying extracellular polymers in biofilms, Water Sci. Technol. 39 (1999) 211–218.
[23] B. Frolund, R. Palmgren, K. Keiding, P.H. Nielsen, Extraction of extracellular polymers from activated sludge using a cation exchange resin, Water Res. 30 (1996) 1749–1758.
[24] P.K. Smith, R.I. Krohn, G.T. Hermanson, A.K. Mallia, F.H. Gartner, M.D. Provenzano, E.K. Fujimoto, N.M. Goeke, B.J. Olson, D.C. Klenk, Measurement of protein using bicinchoninic acid, Anal. Biochem. 150 (1985) 76–85.
[25] APHA, Standard Methods for the Examination of Water and Wastewater, 19th ed., Baltimore, MD, 1995.
[26] B. Jin, B.-M. Wilen, P. Lant, Impacts of morphological, physical and chemical properties of sludge flocs on dewaterability of activated sludge, Chem. Eng. J. 98 (2004) 115–126.
[27] F.J. Ludzack, D.K. Noran, Tolerance of high salinities by conventional wastewater treatment processes, J. Water Poll. Cont. Fed. 37 (1965) 1404–1416.
[28] D.F. Kincannon, A.F. Gaudy, Response of biological waste treatment systems to changes in salt concentrations, Biotechnol. Bioeng. 10 (1968) 483–496.
[29] J.L. Campos, A. Mosquera-Corral, M. Sanchez, R. Mendez, J.M. Lema, Nitrification in saline wastewater with high ammonia concentration in an activated sludge unit, Water Res. 36 (2002) 2555–2560.
[30] W.E. Burnett, The effect of salinity variations on the activated sludge process, Water Sew. Works 121 (1974) 37–38.
[31] M. Doudoroff, Experiments on the adaptation of *E. coli* to sodium chloride, J. Gen. Physiol. 23 (1940) 585–591.
[32] M. Ingram, The influence of sodium chloride and temperature on the endogenous respiration of *Bacillus cereus*, J. Gen. Physiol. 23 (1940) 773–778.
[33] R. Chan, V. Chen, The effects of electrolyte concentration and pH on protein aggregation and deposition: critical flux and constant flux membrane filtration, J. Membr. Sci. 185 (2001) 177–192.

[34] P. Le-Clech, B. Jefferson, S.J. Judd, Impact of aeration, solids concentration and membrane characteristics on the hydraulic performance of a membrane bioreactor, J. Membr. Sci. 218 (2003) 117–129.
[35] B.P. Frank, G. Belfort, Polysaccharides and sticky membrane surfaces: critical ionic effects, J. Membr. Sci. 212 (2003) 205–212.
[36] C. Harscoat, M.Y. Jaffrin, R. Bouzerar, J. Courtois, Influence of fermentation conditions and microfiltration processes on membrane fouling during recovery of glucuronane polysaccharides from fermentation broths, Biotechnol. Bioeng. 65 (1999) 500–511.