Desalination and Water Treatment
www.deswater.com

34 (2011) 141–149
October

1944-3994/1944-3986 © 2011 Desalination Publications. All rights reserved
doi: 10/5004/dwt.2011.2882

Performance evaluation of microfiltration with electrocoagulation and chemical coagulation pretreatment

G. Sharma, H.K Shon*, R. Aryal, S. Phuntsho

School of Civil and Environmental Engineering, University of Technology,
Sydney (UTS), P.O. Box 123, Broadway, NSW 2007, Australia
Tel. +61 2 9514 2629; Fax: +61 2 9514 2633; email: hkshon@eng.uts.edu.au

Received 3 September 2010; Accepted 3 January 2011

ABSTRACT

One of the significant parameters to be considered for evaluating the process and economic viability of crosssflow microfiltration (MF) is flux stability. The MF economics are dependent on the flux decay through the membrane caused by membrane fouling. This work aims to evaluate the performance of MF by electro and chemical coagulation as pretreatments. The performance of MF was found to be sensitive to pH of feed solution, coagulant dosing and generation time. Acrylonitrile butadiene styrene (ABS) MF membrane of pore size 0.4 μm was used in this study. Without pretreatment normalised flux declined by 94% after 160 min of MF operation using model wastewater. However with pretreatments, the MF flux was significantly improved. The optimum performance for MF with both electro and chemical coagulation pretreatments occurred at isoelectric point where the highest removal of organic and turbidity was observed. With chemical coagulation under optimum conditions (30 mg/l alum dose and pH 6.5), MF did not experience any flux decline. MF performed better with chemical coagulation compared to electrocoagulation (EC). Also organic matter removal was found to be more for chemical coagulation than for EC.

Keywords: Pretreatment; Crossflow; Electrocoagulation; Flux decline; Microfiltration; Zeta potential

1. Introduction

Membrane filtration has become an indispensible technology for waste and wastewater purification. However, membrane fouling has remained one of the most serious challenges [1]. Membrane fouling results in the reduction of permeate flux, increase in energy consumption ultimately affecting plant economy. The performance of membrane is influenced by the characteristics of feed water and how its constituents interact with the membrane under certain operating conditions [2]. Crossflow or tangential filtration is a process in which the formation of a filter cake is either limited or, under certain conditions, almost completely suppressed by a flow of the suspension parallel to the filtration surface. Since this system is pressurized, water is forced through the filter. Particles deposited on the filter medium are swept away by the cross-flow velocity action, which produces shear and lift forces on the particles as they become attached to the filter medium whereas for the case of conventional cake filtration, the suspension flows at right angles to the filter medium under the applied pressure.

Due to the increasingly stringent regulations, the rising costs of liquid waste disposal and the growing needs for innovative sources of water supply, crossflow membrane separation processes are gaining a considerable prominence in many sectors of the water industry.

---

*Corresponding author.

Presented at the AMS6/IMSTEC10, The 6th conference of the Aseanian Membrane Society in conjunction with the 7th International Membrane Science and Technology Conference, Sydney Australia, November 22–26, 2010

The efficiency of crossflow filtration is primarily a function of the operating parameters. This efficiency is measured by the filtrate flow rate (flux) and its quality. Crossflow velocity, transmembrane pressure, temperature, pore size of the membrane and concentration of suspended solids in the feed were reported to affect the performance of crossflow MF [3,4]. The various other industrial applications where crossflow MF is applied includes upgrading of cleaning solutions in food and metalworking industries; clarification of fruit juice, wine, cider and vinegar, removal of beer yeast; clarification of whey before other membrane filtration steps, like ultrafiltration or electrodialysis; separation of fat and bacterial removal from milk and filtration of fermentation broths [5].

Flux stability is a significant parameter that must be taken into consideration for evaluating the process and economic viability of crossflow MF applications. The overall economics of MF is primarily dependent on flux decay and any subsequent membrane cleaning or replacement. Several techniques have been implemented to prevent the particles reaching the membrane, which include the use of abrasives, filtration aids, coagulants, and electrofiltration. Different techniques have been used to increase the flux rate [6,7]. The flux decline is caused by the continuous infiltration of fine particulate matter into the concentration polarization layer or by the compaction of the layer. One of the major drawbacks of membrane processes is the membrane fouling. Effective pretreatment is an efficient way of reducing membrane fouling. Pretreatment also reduces the need for frequent chemical cleaning, which is a major factor impacting membrane life. Pretreatment offers great potential for improving the efficiency of membrane processes.

Coagulation is a term used to describe the process of aggregation of colloidal particles into large aggregates. Aggregation of particles is carried out by two distinct mechanisms: particle transport to affect interparticle contact, and particle destabilization to permit attachment when contact occurs. Coagulation is defined as the term applied to the overall process of particle aggregation, including both particle destabilization and transport, while flocculation is the term used to describe only the transport mechanism [8]. Coagulants, when used in crossflow membrane filtration, aim to reduce or even to eliminate internal clogging of the membrane by depositing the colloidal particles and helping form sufficiently large aggregates to facilitate obtaining higher flux rates. Shon et al. [9] found that the permeate flux of ultrafiltraion can be significantly increased by flocculation as a pretreatment. Peuchot and Ben Aim [10] investigated the improvement of crossflow MF performance with flocculation. They concluded that incorporating flocculation arrangement with crossflow MF would increase the particle size, which would tend to limit the increase of the deposit resistance and that the permeate flux was much higher at the optimum dose of the coagulant. In another study by Ben Aim et al. [11], in-line flocculation-crossflow MF was investigated. They found that a significant improvement in the permeate flux was achieved with such an arrangement. The degree of improvement depended on the coagulation dose, flocculation time and crossflow velocity.

EC pretreatment is an alternative to conventional chemical coagulation using Fe or Al electrodes prior to MF. In EC, the coagulant (Fe or Al) is generated by electrolytic oxidation of an anode. The advantages of EC over conventional chemical coagulation include (i) no alkalinity consumption, (ii) no change in bulk pH, (iii) the direct handling of corrosive chemicals is nearly eliminated and (iv) can be easily adapted for use in portable water treatment units especially during emergencies [12]. EC had been investigated to be economically feasible. The cost analysis for the treatment of poultry slaughterhouse wastewater was assessed [13]. Total operating cost was calculated for a plant with 480 m^3 of wastewater per day. The direct and indirect costs items including electrical, sacrificial electrodes, labour, sludge handling, maintenance and depreciation cost was considered in the calculation of total cost where total operating cost was found to be between 0.3 and 0.4 $\$/m^3$ for iron electrodes (figure in US dollars). Also a comparative study on the economic and cost analysis for EC with chemical coagulation was performed [14] for textile wastewater where it was found that the treatment cost of EC at optimum conditions was 3.2 times cheaper than that of chemical coagulation.

EC had been reported to be efficient and feasible for removing large range of contaminant [15–17]. But very limited information is available regarding EC as a MF pretreatment process to improve membrane performance and reduce fouling. EC is a complex and interdependent process. A sacrificial metal anode is used to produce coagulating agent to dose the polluted water and electrolytic gases (mainly hydrogen at the cathode) are generated. Electrochemistry, coagulation and hydrodynamics form the basis of EC [18]. The most widely used electrode materials in EC process are aluminium and iron, sometimes steel. The electrical current causes the dissolution of metal into wastewater. The metal ions, at an appropriate pH value, can form wide ranges of coagulated species and metal hydroxides that destabilize and aggregate the suspended particles or precipitate and adsorb dissolved contaminants. In the case of aluminium, main reactions are as:

$$\text{Anode: Al} \rightarrow \text{Al}^{3+} + 3e^- \tag{1}$$

$$\text{Cathode: } 3H_2O + 3e^- \rightarrow 3/2H_2 + 3OH^- \quad (2)$$

The generated Al^{3+} and OH^- react with each other to form $Al(OH)_3$

$$Al^{3+} + 3H_2O \rightarrow Al(OH)_3 + 3H^+ \quad (3)$$

The objectives of this study is to evaluate the performance of crossflow MF with chemical coagulation and EC as pretreatments. This research also aims in finding out the operating conditions favouring removal of organic matter and turbidity by EC and chemical coagulation. MF performance was evaluated using flux decline and organic removal at constant transmembrane pressure and crossflow velocity.

2. Materials and methods

2.1. Synthetic wastewater (SWW)

SWW was prepared in the laboratory using kaolin and humic acid (Table 1). Stock solution was prepared using 2 g humic acid sodium salt and 40 g kaolin which were then diluted with normal tap water to give a fairly constant turbidity between 80 to 85 NTU.

2.2. MF Membrane

Flat sheet MF membrane was used throughout the study; properties of which are given in Table 2.

2.3. Electrocoagulation setup

The EC reactor used in this study consisted of a 5 l pyrex glass beaker with two aluminum electrodes (17 cm x 9 cm x 0.2 cm) in a monopolar configuration and the spacing between the electrodes was 2 cm. The source of power supply included DC power converter (Q1770, Dick Smith Electronics, Australia). Suspensions were stirred using a magnetic stirrer adjusted to an optimal rate (250 rpm) so as to obtain the highest efficiency of turbidity removal. Experiments were performed at different generation times (5–30 min) and at wide pH range (3–11) for determining the optimum conditions. Prior to each test, 5 l of synthetic wastewater was used in the EC cell which was carried out by varying the process operating parameters. The current was adjusted by varying the voltage. When not in use, the aluminium electrodes were immersed in acid bath (4% HCl) and prior to each experiment, they were carefully cleaned using steel wool to remove any aluminium oxide that may have formed on the surface. The desired aluminium concentration can be achieved by operating the unit under variable generation time (t) mode in accordance with Faraday's law:

$$m_{Al} = (27 \times I \times t)/Z \times F$$

where m_{Al} is the mass of Al generated (g), I is the current , Z is the number of electrons transferred in the reaction at the electrode and F is the Faraday's constant (96,486 C/eq).

Table 1
Properties of SWW

Turbidity	Total organic carbon (TOC)	UV Abs (254 nm)	Conductivity
80 NTU	5.5 mg/l	0.150	0.0805 ms/cm

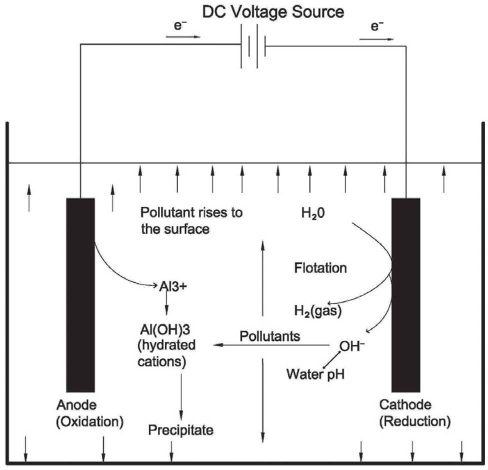

Fig. 1. Schematic diagram of a bench-scale two-electrode EC cell [4].

Table 2
General characteristics of MF membrane used in this study

Pore size	pH resistance range	Temperature range	Pressure range	Material
0.4 μm	2~11	2~38°C	0~ 60 cm Hg	Acrylonitrile butadiene styrene (ABS)

Faraday's law is mostly used for calculating the amount of aluminium released as a good correlation between the theoretically calculated and the experimentally determined amount of aluminium that went into solution was found in the previous study [12].

2.4. Chemical coagulation setup

Conventional chemical coagulation tests were conducted using standard jar test. Aluminium sulphate was added to the wastewater at room temperature. The sample was stirred rapidly for 3 min at 130 rpm, followed by 20 min of slow mixing at 30 rpm, and 60 min of settling. Jar tests were performed at different dosings of aluminium sulphate (10–60 mg/l) and under wide pH range (2–11) to determine the optimum conditions. The pH of the solution was adjusted with 0.1 N HCl and 0.1 N NaOH.

2.5. Microfiltration setup

A crossflow membrane filtration unit (Nitto Denko Corp., Japan) was used to study the effect of pretreatment on membrane performance. The schematic diagram of the crossflow MF experimental setup is shown in Fig. 2. Synthetic wastewater, with and without pretreatment, was pumped into a flat sheet membrane module (effective membrane area of 0.007 m^2). The operating transmembrane pressure and crossflow velocity were controlled at 10 kPa and 0.5 l/m respectively by means of bypass and regulating valves. The filtrate was accumulated. The mass of filtrate was determined by using an electronic balance and transferred to a computer. New membranes were used in each experiment to avoid the effect of residual fouling and to compare the results obtained under different conditions.

2.6. Analytical measurements

A turbidimeter (HACH 2100P, USA) was used to measure the turbidity for all samples. 25 ml of supernatant was taken for turbidity measurement after 60 min settling period. Measurements for the turbidity were taken three times and the mean value was recorded.

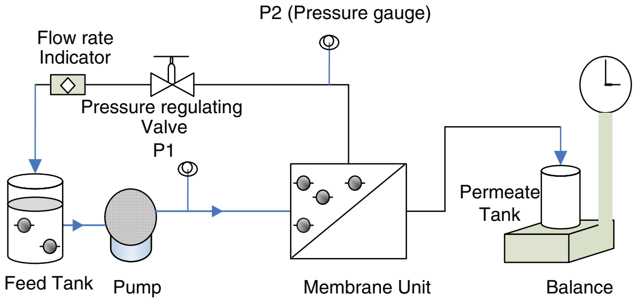

Fig. 2. Schematic diagram of the crossflow MF unit.

Dissolved organic (humic acid) removal was measured in terms of dissolved organic carbon (DOC) and UV absorbance at 254 nm. Total organic carbon (TOC) was measured by using the Dohrmann Phoenix 8000 UV-persulfate TOC analyzer equipped with an autosampler. All samples were filtered through a 0.45 µm filter prior to the TOC measurement. Thus, the TOC values obtained are, in fact, DOC values. UV absorbance was measured with a UV-vis spectrophotometer (Shimadzu, N 595, Kyoto, Japan) in a 1 cm quartz cell. Here too, the samples were filtered through a 0.45 µm filter prior to measurement. The zeta potential of supernatants was measured with Zetasizer Nano Series-Zs (Malvern, UK). The current that flows through the cell and the voltage across the electrodes were measured by using multimeter (Jaycar Electronics, N287).

3. Results and discussion

3.1. Effect of EC generation time on MF flux

Fig. 3 shows the normalised permeate flux profile for 60 min operated at different EC generation times (0–30 min) at pH 8.0. Normalisation was done by dividing the obtained flux by pure water flux (J/J_0). Here, J_0 is the pure water flux and J is the filtrate flux at a given time. The current density was kept constant at 12 A/m^2 for all the experiments.

In all cases, flux decline was observed with time. The flux decline was rapid at an initial stage (0–20 min) followed by more gradual decline. The decline trend was proportional to EC generation time. At lower EC time low flux was observed whereas at higher EC time

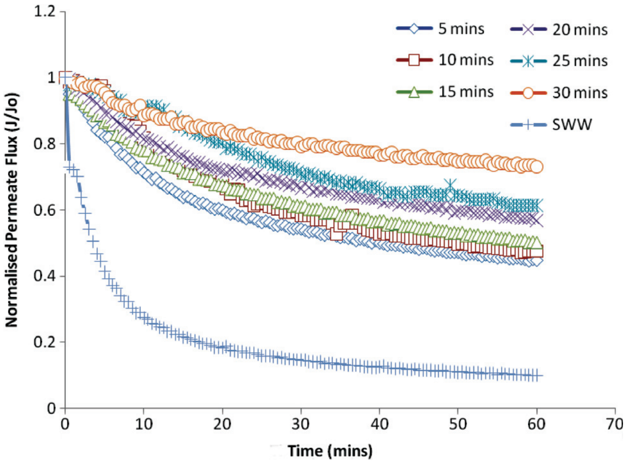

Fig. 3. Normalised permeate flux at different generation times of EC (current density: 12 A/m^2, transmembrane pressure 10 kPa, cross flow velocity 0.5 l/min, pH 8).

higher flux was observed. Without EC pretreatment, the normalized flux for wastewater decreased by as much as 90%. With operation of 5 min of EC, the flux decline was 55% (flux increase of 35% compared to without pretreatment). As EC time was increased, the flux through MF gradually increased. The flux was found narrowly changed after 30 min. This suggests that the optimum EC time for the best flux was 30 min. At this point we also observed highest turbidity and organic removal (DOC and UV abs). Similar results were obtained using polyester MF membrane for the electrocoagulated kaolin suspension [19] where the initial flux increased by more than 490 percent with optimum EC generation time.

Increase of EC time increases the aluminium ion released to the system. As the experiment was conducted at pH 8, it was believed that the released ion converted to $Al(OH)_3$ floc quickly. Thus the formed floc caused the particles and organics to agglomerate, float, settle and remove from the wastewater.

3.2. Effect of chemical coagulant dose on MF flux

Fig. 4 shows the effect of alum doses in MF flux variation. Experiments were conducted with alum dosing from 0 to 60 mg/l. In absence of alum, the flux through MF decreased by almost 90%. Increase of alum dose from 10 to 30 mg/l increased the flux. The flux decline was 84% and 50% respectively at 10 and 20 mg/l doses (Flux increase of 6% and 40% respectively compared to without pretreatment). Above 30 mg/l the flux again decreased. A significant result was observed at 30 mg/l dose where MF did not experience any flux decline. Similar results were found for polyester yarn woven MF membrane [20] where flux increase by 70% with the addition of optimum dose of alum compared to without pretreatment. The result is also in agreement with the investigation was performed for flocculation as a pretreatment for ultrafiltration membrane (NTR 7410) [9]. With optimum dose of coagulant ultrafiltration membrane did not experience any flux decline during the whole operation of 6 h.

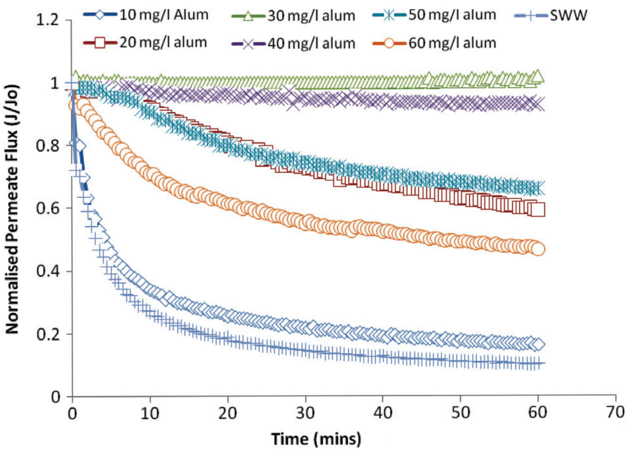

Fig. 4. Normalised permeate flux at different dosing of chemical coagulation (coagulation time: 3 min fast mixing at 130 rpm, 20 min slow mixing at 30 rpm followed by 60 min of settling, transmembrane pressure 10 kPa, cross flow velocity 0.5 l/min, pH 6.5).

The dose response curve shows that the optimum dose for chemical coagulation was 30 mg/l. Above and below the optimum dose reduced the flux. This result confirms with the optimal concentration determined by the jar-test where the maximum removal of organic matter and turbidity occurred at this dose. A possible reason for obtaining less flux below the optimum dose is less availability of the aluminium coagulant. As the dose increased above 30 mg/l, the excess floc caused alteration in the flux size that reduced the performance of colloidal removal [21].

3.3. Effect of pH on turbidity and zeta potential with EC

To examine the effect of pH on EC pretreatment for MF performance, EC experiments were conducted in the pH range of 3–11. Fig. 5 shows the turbidity removal efficiency and zeta potential for wastewater in the range of pH 3–11. It is seen that turbidity removal efficiency increased gradually from 52% at pH 3.0 to as high as 88% at pH 8 and decreased to 46% at pH 11. The highest turbidity removal efficiency (88%) was found at pH 8. One of the possible reasons for getting higher turbidity removal is the formation of stable floc of Al. At lower pH the alum dissolves itself. However, at pH 8 the alum works well but once it crosses it, the floc becomes unstable and begins to come apart. This is supported by zeta potential result which shows isoelectric point at pH 8. The pHs below 8 gave positive values of the zeta potential, whereas pH above 8 produced negative zeta potentials. This behavior could be explained in terms of the charge of the particles, which were positive due to the adsorption of cations such as Al^{3+},

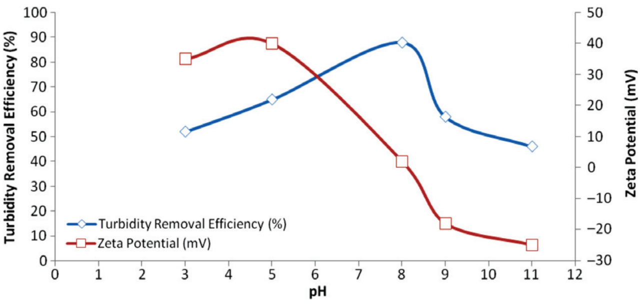

Fig. 5. Turbidity removal efficiency and zeta potential after EC treatment at different pH (current density: 12 A/m^2; EC time: 30 min).

$Al(OH)_2^+$, $Al(OH)^{2+}$ and negative due to the adsorption of anions ($Al(OH)_4$). The zero isoelectric potential shows that no net charge is formed at this pH. This suggests that the aluminum hydroxide is precipitated and suitable for flocculation that helped to remove maximum turbidity at pH 8.

3.4. Effect of pH on organic matter removal with EC

Organic matter removal in terms of DOC and UV absorbance at 254 nm at different pH was carried out. Fig. 6 shows the DOC removal efficiency and normalised UV absorbance at 254 nm in the pH range 3–11. Normalisation for UV abs was done by dividing the obtained UV abs value after EC treatment with value without treatment. DOC removal increased from 40% at pH 3 to 68% at pH 8 and gradually decreased to 48% at pH 11. As can be seen from the graph, the maximum DOC removal was found to be 68% at pH 8. Normalised UV abs also followed the same trend gradually decreasing from pH 3. and then again increasing after pH 8. Normalised UV 254 absorbance was minimum at pH 8. The intensity corresponded to 78% removal at pH 8. The performance of MF was affected by the organic matter removal by EC pretreatment due to the adsorption of organic matter on the membrane surface.

3.5. Effect of pH on turbidity and zeta potential with chemical coagulation

To study the effect of pH on MF performance by chemical coagulation pretreatment, coagulation was performed at different pH in the wide range of 3–11. Fig. 7 shows turbidity removal efficiency and zeta potential respectively for wastewater for pH range 3–11. It is seen from the figure that the turbidity removal efficiency was only 52% at pH 3 and increased steadily with increasing pH up to 94% at pH 6.5 and then gradually decreased to 60% at pH 11. The zeta potential curve went through two isoelectric points at pH 3 and pH 6.5. The zeta potential remained at it's near-zero values within the wide pH range of 5–8 where the turbidity removal efficiencies were about 90%.

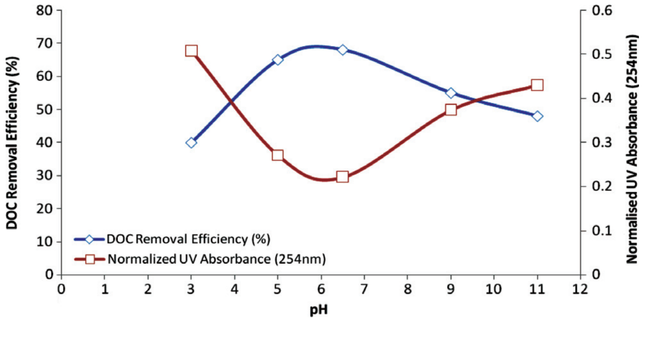

Fig. 6. DOC removal efficiency and normalized UV absorbance after EC treatment at different pH (current density: 12 A/m²; EC time: 30 min).

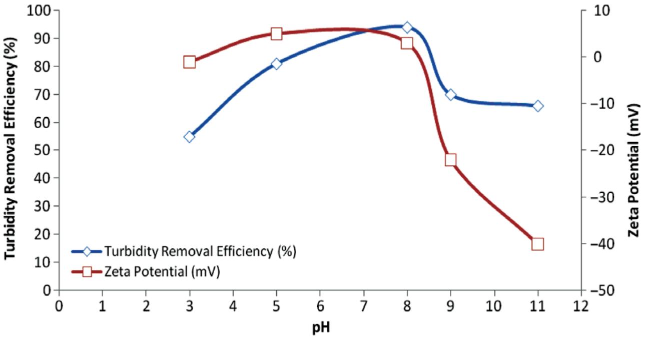

Fig. 7. Turbidity removal efficiency and zeta potential after chemical coagulation experiments at different pH (alum dosing: 30 mg/l ; coagulation time: 3 min fast mixing at 130 rpm, 20 min slow mixing at 30 rpm followed by 60 min of settling).

As the zeta potential shows two isoelectric points, it shows that neutralization occurred at two points. At low pH almost all Al compounds are in cationic monomeric species Al^{3+} and $Al(OH)_2^+$. Kaolin clay contains net negative charge on its surface. So the clay surface and other particles in the solution attract the aluminium ion on its surface and neutralizes. The first point of zero charge is mainly due to adsorption of Al^{3+} ion on remaining kaolin. The first isolectric point is not due to the aluminum itself but due to the adsorption of aluminium with the particles of the solution. As the pH increases (pH 5–8), hydrolysis occurs forming various monomeric species aluminum hydroxyl such as $Al(OH)_2^+$, $Al(OH)_2^{+2}$, and polymeric species such as $Al_6(OH)_{15}^{3+}$, $Al_7(OH)_{17}^{4+}$, $Al_{13}(OH)_{34}^{5+}$ predominate, which finally transform into $Al(OH)_3$ [22,23]. These hydroxyl species being positive in charge, tends to adsorb on the solution resulting in very low zeta potential. As the pH crosses the neutral region (pH 7), Al starts to form a stable hydroxide ($Al(OH)_3$) that starts to precipitate. But at higher pH the floc breaks and form $Al(OH)_4$ ions.

3.6. Effect of pH on organic matter removal with chemical coagulation

Organic matter removal in terms of DOC and UV absorbance at 254 nm at different pH was carried out to investigate the effect of organic removal on MF performance. Fig. 8 shows the DOC removal efficiency and normalised UV absorbance at 254 nm respectively in the pH range 3–11. There was a gradual increase of DOC removal from 55% at pH 3 to 58% at pH 11 showing the maximum DOC removal (78%) at pH 6.5. UV abs (254 nm) also showed the highest removal at pH 6.5 (86%).

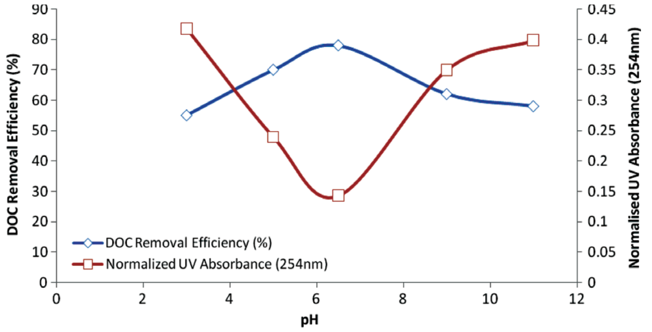

Fig. 8. DOC removal efficiency and normalised UV absorbance after EC experiments at different pH (alum dosing: 30 mg/l ; coagulation time: 3 min fast mixing at 130 rpm, 20 min slow mixing at 30 rpm followed by 60 min of settling).

As DOC and UV abs followed similar trend, it can be concluded that the MF membrane performed better under the conditions when organic matter removal was highest.

Three possible mechanisms for DOC removal in a conventional alum coagulation/sedimentation process have been proposed. They are i) charge neutralization, ii) adsorption and iii) entrapment [24,25]. According to the equilibrium concentrations of hydroxo aluminium (III) complexes in a solution in contact with $Al(OH)_3$, aluminium from Al^{3+} at pH 4.5 converts to $Al_7(OH)_{17}^{\ 4+}$ at pH 6 and becomes $Al(OH)_3$ at the isoelectric point of around pH 6.5, i.e., the positive charge of aluminium increases as pH increases from 4.5 to 6.5, whereas visible floc particles of aluminium hydroxide precipitate form after the isoelectric point. It is to say that with the increasing pH from 4.5 to 6.5, contribution of the charge neutralization to DOC removal increased, whereas after pH 6.5 contribution of the charge neutralization decreased and contributions of the adsorption and entrapment predominated for DOC removal [26].

3.7. Validation of pH on MF flux

The performance of MF was evaluated in terms of normalised permeate flux (J/J_0) through the membrane [27], turbidity, DOC, UV abs at 254 nm and zeta potential at different pH in order to validate our above experiments. Figs. 10 and 11 show the flux trend in EC and chemical coagulation. During the experiments, the operating condition (cross flow velocity and pressure) and feed water characteristics (kaolin and humic acid) remained same except pH.

Fig. 9 shows the flux through the membrane depended on pH of the feed solution. At pH 3, the flux decline was 92%, only 2% increase in flux compared to without pretreatment. MF performed best at pH 8 where flux increased by 55% compared to without pretreatment. The flux data is provided in Table 3.

At lower pH (< 7) the coagulation is very poor. As pH was increased the flux through MF increased. Increase of pH helped to form the floc as well as to dissolve the humic acid. The optimum flux was observed at pH 8. This result is confirmed from our previous experiments that this maximum flux occurred at isoelectric point for EC where there was optimum removal of turbidity and organic matter. Above pH 8, the decline in flux was again observed. As mentioned above, the decline was possibly due to breakage of floc after pH 8.

The profile of normalised flux after chemical coagulation pretreatment at different pH is shown in Fig. 10. Similar to EC, the chemical coagulation also followed

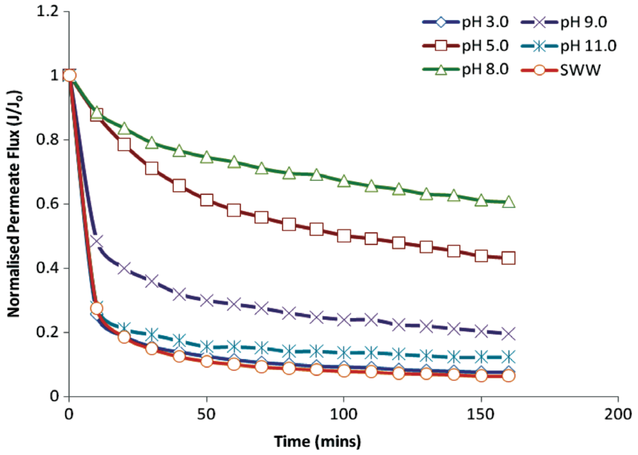

Fig. 9. Normalised permeate flux through MF for EC using Aluminium plates (transmembrane pressure 10 kPa, cross flow velocity 0.5 l/min).

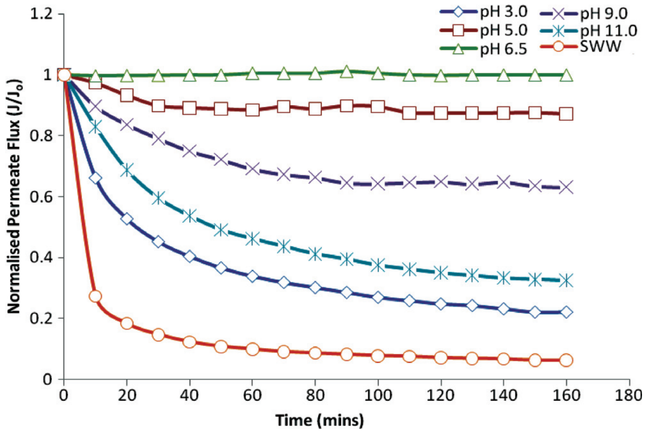

Fig. 10. Normalized permeate flux through MF for chemical coagulation (transmembrane pressure 10 kPa, cross flow velocity 0.5 l/min).

Table 3
Normalised flux for EC and chemical coagulation at optimum dosing

Normalised flux (160 min) (J/J_0)		pH	Flux decline (%)		Flux increase (%)	
EC	CC*		EC	CC	EC	CC
0.08	0.22	3	92	78	2	16
0.43	0.87	5	57	13	37	81
–	1	6.5	–	0	–	94
0.61	–	8	39	–	55	–
0.20	0.63	9	80	37	14	57
0.12	0.32	11	88	68	6	26

*CC- Chemical coagulation.

similar trend. Flux decline was 78% and 13% at pH 3 and pH 5 respectively. At pH 6.5, the MF did not experience any flux decline. Initially the flux decline was sharp for the first 20 min for all the experiments. After 20 min, the decline was more gradual. Compared to EC, chemical coagulation showed higher flux. Previously we observed two isoelectric points at pH 3 and pH 6.5. Between these two pH the isoelectric point was very minimal compared to EC. This shows that floc formation is relatively higher in chemical coagulation compared to EC. The enriched (higher) floc formation in chemical coagualtion caused higher removal of colloidal (kaolin as well as humic acid) materials from the wastewater. This possibly caused good flux from the beginning in chemical coagulation. Above pH 6.5, flux declined. The normalised flux data is provided in Table 3. The reason for getting low flux above pH 6.5 is the same as with EC.

4. Conclusions

A bench-scale study was undertaken to investigate the performance of MF through electro and chemical coagulation pretreatments. The results lead to the following conclusions:

1. Without pretreatment, MF experienced significant flux decline. Normalised flux declined by 94% after 160 min of operation.
2. The optimum removal of organic matter and turbidity for both electro and chemical coagulation occured at their isoelectric points. Both DOC removal and UV abs. removal for chemical coagulaiton (68% and 78% respectively) was more than for EC (78% and 85% respectively). Also turbidity removal was 6% higher for chemical coagualtion (94%) compared to EC (88%).
3. MF flux was found to be sensitive to pH of feed solution. Maximum flux after both electro and chemical coagulation was obtained at their respective isolectric points. Below and above the isoelectric points flux was significantly declined.
4. MF performed better with chemical coagulation pretreatment than that with EC. For chemical coagualtion pretreatment; under optimum conditions (pH 6.5, 30 mg/l alum), MF did not experience any flux decline while with electrocoagulation under optimum conditions (pH 8.0, generation time 30 min), normalised flux increased by 55% compared to without pretreatment.

Acknowledgements

This research was funded by ARC and UTS internal grants. The authors also appreciate the kind membrane support from Pure-Envitech Co. Ltd. South Korea.

References

[1] G. Amy, Fundamental understanding of organic matter fouling of membranes, Desalination, 231 (2008) 44–51.
[2] Georges Belfort, R.H. Davis and A.L. Zydney, The behaviour of suspensions and macromolecular solutions in crossflow microfiltration, J. membr. sci., 96 (1994) 1–58.
[3] S.M. Forman, E.R. DeBernardez, R.S. Feldberg and R.W. Swartz, Crossflow filtration for the separation of inclusion bodies from soluble proteins in recombinant Escherichia coli cell lysate, J. Membr. Sci., 48 (1990) 263–279.
[4] Mohammad et al., Fundamental, present and future perspectives of electrocoagulation, J. Hazard Mater., B114 (2004) 199–210.
[5] Siegfried Ripperger and Justus Altmann, Crossflow microfiltration – state of the art, Sep. Purif. Technol., 26 (2002) 19–31.
[6] M. Mulder, Basic Principles of Membrane Technology, Kluwer Academic Publishers, Dordrecht, The Netherlands, 1991.
[7] V. Gekas and B. Hallstrom, Microfiltration membranes, cross-flow transport and fouling studies, Desalination, 77 (1990) 195–218.
[8] C.R. O'Melia, Coagulation and flocculation, in W.J. Weber (Ed.), Physicochemical Processes for Water Quality Control, Wiley Interscience, New York, USA, 1972.
[9] H.K. Shon, S. Vigneswarana, H.H. Ngo and R. Ben Aim, Is semi-flocculation effective as pretreatment to ultrafiltration in wastewater treatment? Water Res., 39 (2005) 147–153.
[10] M.M. Peuchot and R. Ben Aim, Improvement of crossflow microfiltration performance with flocculation, J. Membr. Sci., 68 (1992) 241–248.
[11] R. Ben Aim, M.M. Peuchot, S. Vigneswaran, K. Yamamoto and S. Boonthanon, A new process for water reuse; in-line flocculation-crossflow filtration, in T. Panswad, C. Polprasert and K. Yamamoto (Eds.), Water Pollution Control in ASIA, Bangkok, Thailand, 1988.
[12] Ashima Baggaa, Shankararaman Chellam and Dennis A. Clifford, Evaluation of iron chemical coagulation and electrocoagulation pretreatment for surface water microfiltration, J. Membr. Sci., 309 (2008) 82–93.
[13] Mahmut Bayramoglu, Mehmet Kobya, Murat Eyvaz and Elif Senturk, Technical and economic analysis of electrocoagulation for the treatment of poultry slaughterhouse wastewater, Sep. Purif. Technol., 51 (2006) 404–408.

[14] Mahmut Bayramoglu, Murat Eyvaz and Mehmet Kobya, Treatment of the textile wastewater by electrocoagulation Economical evaluation, Chem. Eng. J., 128 (2007) 155–161.

[15] G. Chen, Electrochemical technologies in wastewater treatment, Sep. Purif. Technol., 38(1) (2004) 11–41.

[16] E.A. Vik, D.A. Carlson, A.S. Eikum and E.T. Gjessing, Electrocoagulation of potable water, Water Res., 18 (1984) 1355–1360.

[17] M. Kobya, E. Demirbas and O.T. Can, Treatment of levafix orange textile dye solution by electrocoagulation, J. Hazard Mater., B132 (2006) 183–188.

[18] P.K. Holt, G.W. Barton and M. Wark, A quantitative comparison between chemical dosing and electrocoagulation, Colloid Surface A: Physicochem. Eng. Aspects, 211 (2002) 233–248.

[19] M.H. Al-Malack, A.A. Bukhari and N.S. Abuzaid, Crossflow microfiltration of electrocoagulated kaolin suspension: fouling mechanism, J. Membr. Sci., 243 (2004) 143–153.

[20] Muhammad H. Al-Malack and G.K. Anderson, Coagulation-crossflow microfiltration of domestic wastewater, J. Membr. Sci., 121 (1996) 59–70.

[21] M. Mietton Peuchot and R. Ben Aim, Improvement of crossflow microfiltration performances with flocculation, J. Membr. Sci., 68 (1992) 241–248.

[22] A comparative study of electrocoagulation and coagulation of aqueous suspensions of kaolinite powders Mehtap Gulsun Kılıc, Cetin Hosten, J. Hazard. Mater., 176 (2010) 735–740.

[23] B. Merzouk, B. Gourich, A. Sekki, K. Madani and M. Chibane, Removal turbidity and separation of heavy metals using electrocoagulation-electroflotation technique a case study, J. Hazard. Mater., 164 (2009) 215–222.

[24] R. Fabris, C. Chow and M. Drikas, The impact of coagulation type on NOM removal, in: Proceedings of the Technology Transfer Workshop on Natural Organic Matter in Drinking Water, Adelaide, Australia, November 27, 2002.

[25] E.M. Vrijenhoek, A.E. Childress, M. Elimelech and T.S. Tanaka, Removing particles and THM precursors by enhanced coagulation, AWWA 90 (1998) 139–150.

[26] Jian-Jun Qin, Maung Htun Oo, Kiran A. Kekre, Frans Knops and Peter Miller, Impact of coagulation pH on enhanced removal of natural organic matter in treatment of reservoir water, Sep. Purif. Technol., 49 (2006) 295–298.

[27] H.K. Shon, S. Vigneswaran, In S. Kim, J. Cho and H.H. Ngo, Effect of pretreatment on the fouling of membranes: application in biologically treated sewage effluent, J. Membr. Sci., 234 (2004) 111–120.