An investigation on using MDCs for an efficient desalination process as pretreatment of reverse osmosis

Avin Habibi, Madjid Abbaspour, Amir H. Javid and Amir H. Hassani

ABSTRACT

Microbial desalination cell (MDC) is a new bio-electrochemical technique which converts chemical energy into electrical energy, and at the same time desalinates water and treats wastewater. In this study, MDC performance and water biofouling conditions were tested as an efficient pretreatment desalination process of reverse osmosis (RO). The experiments were designed in a three-chamber reactor to compare the performance of batch and continuous fed modes, using bio-cathode and synthetic wastewater in four different hydraulic retention times and 17 and 35 g/L NaCl concentrations. According to the results, maximum salt removal of about 52.3% was obtained in the continuously fed MDC at 35 g/L NaCl concentration. The maximum salt removal at 17 g/L NaCl was also observed in continuous mode. The anolyte pH in both batch and continuous modes dropped from 7 to 6.32, 6.47 and 6.37, 6.48 in 17 and 35 g/L NaCl concentrations respectively. The chemical oxygen demand (COD) removal values in the continuous mode were 61 and 65% in the anolyte and catholyte respectively, higher than those of fed-batch MDC. The biofouling of the middle chamber solution was confirmed by conducting bio-microbial tests. Our results suggest that the type of hydraulic flow can improve the performance of MDC in different concentrations of NaCl.

Key words | batch mode, biofouling, continuous mode, microbial desalination cell, pretreatment

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INTRODUCTION

Fresh water (salinity <0.5 g/L) shortage has become one of the major challenges for societies worldwide (Elmakawy *et al.* 2014). In the next two decades, it is predicted that the average amount of fresh water per person will decrease by one-third, and two-thirds of the world's inhabitants will live under fresh water stressed conditions. Although desalination technologies are commonly used for producing clean water from the sea and brackish water around the world, most of these technologies still require electrical or thermal energy, and therefore require significant capital and energy resources. Hence, developing desalination technologies sustained by renewable energy, such as wind, solar or other green technology for generating energy, is being seriously considered (Elmakawy *et al.* 2014). Microbial desalination cell (MDC) is a new type of bio-electrochemical system that can simultaneously desalinate water and treat wastewater. This process has great potential as a green method for water desalination, as compared to more traditional methods such as reverse osmosis (RO) that requires electrical grid energy (~3.7 kwh/m^3) (Qu *et al.* 2013; Jingyu *et al.* 2017). The concept of MDC was first introduced by Cao *et al.* (2009) in small scale (3 mL salt water capacity) and it was later scaled up by Jacobson *et al.* (2011) to create a large scale MDC (~1 L salt water capacity). MDCs are derived from microbial fuel cells (MFCs), which use microorganisms as biocatalysts to convert chemical energy into electrical energy (Kim & Logan 2013a, 2013b). The most basic type of MDC consists of three chambers, the anode chamber, a desalination chamber, and the cathode chamber (Qu *et al.* 2013). Bacteria growing on an electrode in the anode chamber oxidizes organic substrate and results in the transfer of electrons,

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which causes the movement of cations from the middle chamber to the cathode chamber and the migration of anions from the middle chamber to the anode chamber, thereby generating electric current at the same time (Kim & Logan 2013a, 2013b; Qu *et al.* 2013). The migration of ions to the middle chamber is powered by the difference in potential between the anode and cathode (Elmakawy *et al.* 2014). When current is generated by bacteria on the anode and protons are released into solution, positively charged species are prevented from leaving the anode by the anion-exchange membrane (AEM) and, therefore, negatively charged species (i.e. chlorine) move from the middle chamber to the anolyte. In the cathode chamber, protons are consumed, resulting in positively charged species (i.e. Na^+) moving from the middle chamber across the cation-exchange membrane (CEM) to the catholyte. This loss of ions from the middle chamber results in water desalination and no electrical energy and water pressure is required (Jingyu *et al.* 2017). The current desalination technologies such as RO or distillation require a large amount of energy, ranging from 3.7 to 650 kwh per cubic meter of water. In addition, the amount of energy needed for RO increases in proportion to the initial salinity of the feed water (Mehanna *et al.* 2010; Elmakawy *et al.* 2014). Decreasing the conductivity of the salt water would significantly help in reducing the overall amount of energy consumed by the RO modules (Elmakawy *et al.* 2014). Thereby, MDC could be advantageous as a pretreatment process for water desalination RO. According to recent studies, MDC can be operated under different modes including fed-batch and continuous, and it can also be used as a pretreatment process prior to RO (Mehanna *et al.* 2010; Lou *et al.* 2012; Jingyu *et al.* 2017; Sevda *et al.* 2017). In the reported literature on batch-mode, the potential applications of MDC viz. seawater and brackish water desalination, hydrogen gas generation, hardness removal, power generation, chemical oxygen demand (COD) removal, and coulombic efficiency have been investigated (Lou *et al.* 2012; Jingyu *et al.* 2017; Sevda *et al.* 2017). Mehanna *et al.* (2010) investigated using air-cathode MDC as an RO pretreatment method. Based on the reported data, MDC could be used to reduce salinity and thus energy demand for downstream RO systems. In the latest reported literature on continuous MDC, four MDCs were hydraulically connected and operated under continuous flow conditions (Qu *et al.* 2013). Ebrahimi *et al.* (2017) investigated the treatment of wastewater under fed-batch vs. continuous operation with hydraulic retention times (HRTs) of 2 days and 35 g/L NaCl concentration. The reported data revealed that enrichment of anolyte under controlled continuous feeding mode would relatively improve the MDC performance. Although there are many studies on the effect of operational conditions on MDC performance, only a few studies investigated using MDC prior to RO systems, and there is no comparative study on the operational conditions using bio-cathode MDC with two different NaCl concentrations and four different HRTs. Water contamination by organic pollutants is a major problem (Ali *et al.* 2016) and contamination due to drugs and pharmaceuticals residues is increasing and alarming. Many water resources have been found to have new emerging pollutants (Basheer 2018). Although some studies have investigated the methods for removal of contamination from water bodies (Ali *et al.* 2018; Basheer & Ali 2018), there are no studies that have evaluated the bio-contamination of the middle chamber in MDCs as a new technology in desalination and water treatment. In the current study, the near future integration of MDC as one of the novel technologies with RO systems for an efficient desalination process using bio-cathode MDC as pretreatment of RO was investigated. The experiments were designed to compare the performance of two operational fed modes (batch and continuous), using bio-cathode and synthetic wastewater in four different HRTs (1, 2, 3 and 4 days) and two different NaCl concentrations (17 and 35 g/L). The performance of MDC was evaluated in terms of water electrical conductivity (EC), Coulombic efficiencies (CE), pH variations, NaCl removal (%), and changes in COD. The biofouling of the MDC middle chamber through bio-microbial tests was investigated.

MATERIALS AND METHODS

MDC setup

In this study, the design of MDC was based on three cubic-shaped Plexiglas chambers, with equal dimensions ($10 \times 10 \times 10$ cm). The Plexiglas sheets were cut to make

cubic-shaped holes with a thickness of 4 cm accordingly so that the volume of each chamber was 125 mL, with equal dimensions ($5 \times 5 \times 5$ cm). The chambers were separated with ion-exchange membranes: an anion-exchange membrane (AMI-7001, Risingsun Membrane, China) between the anode chamber and the middle chamber and a cation-exchange membrane (CMI-7001, Risingsun Membrane, China) between the cathode chamber and the middle chamber. The anode and cathode electrodes were produced by inserting a graphite rod (5 mm diameter) into the anode and cathode chambers. Prior to use, graphite rods were washed for 48 h in 1 M HCl and rinsed with deionized water to remove trace metals. The electrodes were connected to an external resistance through a copper wire under a fixed resistance of 100 Ω in a closed circuit condition. The schematic and photographs of batch and continuous mode of MDC setup are depicted in Figure 1.

Reactor start up and operation

The anode chamber was inoculated with bacterial culture from cow manure and synthetic wastewater solution (50%, V/V). The synthetic wastewater solution contained: sodium acetate 1.6 g/L, KH_2PO_4 1.5 g/L, K_2HPO_4 1.5 g/L, NH_4Cl 1.5 g/L, $MgCl_2.6H_2O$ 0.1 g/L, $CaCl_2.2H_2O$ 0.1 g/L, KCl 0.1 g/L, trace minerals 10 mL/L and sugar 3.47 g/L (Mehanna *et al.* 2010; Jingyu *et al.* 2017). The wastewater solution in the anolyte was diluted to reach anaerobic ratio with a C:N:P ratio of 250:5:1 respectively, and COD of 500 ± 20 mg/L. The bio-cathode chamber was inoculated

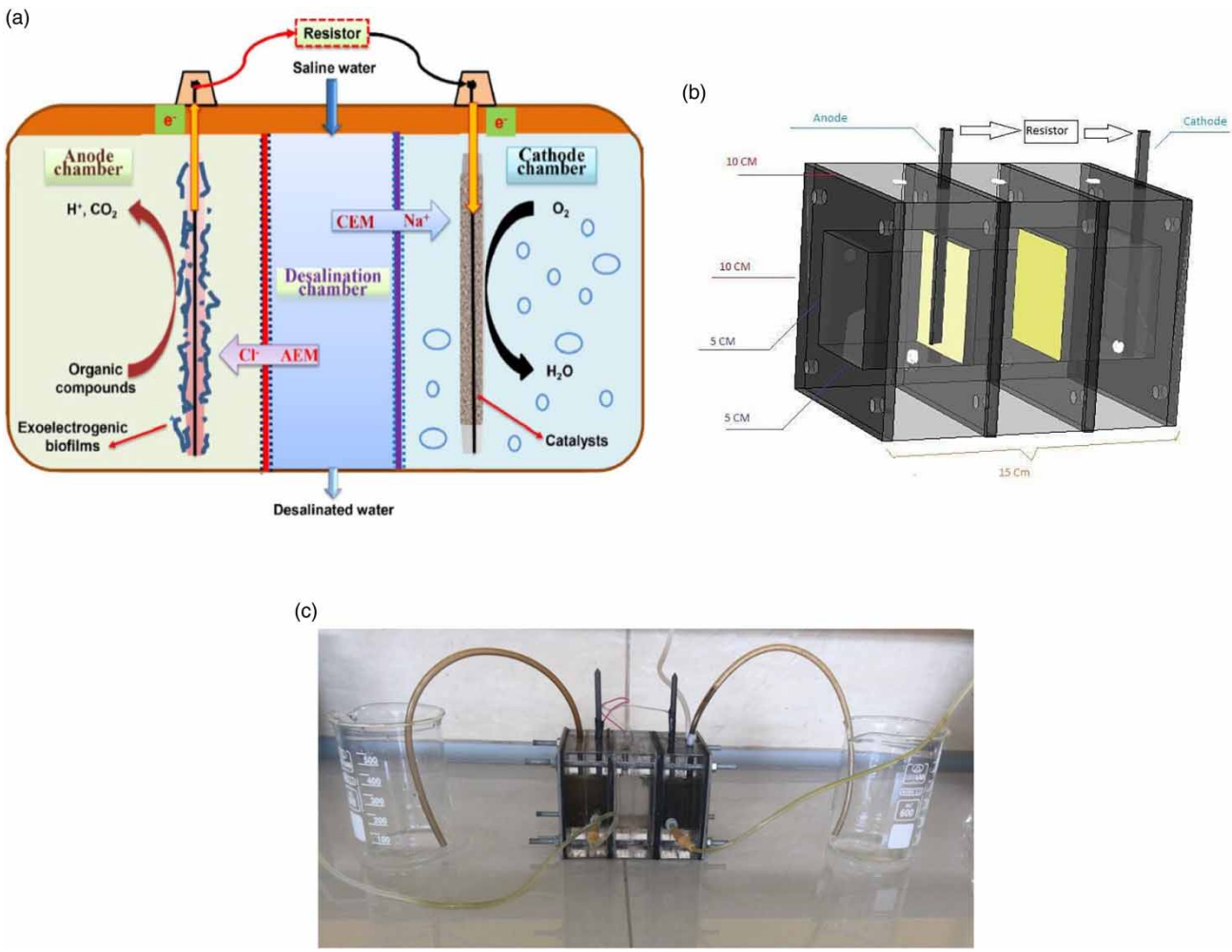

Figure 1 | Three-chamber microbial desalination cell. AEM: Anion-exchange membrane, CEM: Cation-exchange membrane; (a) and (b) schematic, (c) photograph.

with bacterial culture from activated sludge (50%, V/V). The synthetic wastewater solution contained: sodium acetate 1.6 g, KH_2PO_4 1.5 g/L, K_2HPO_4 1.5 g/L, KCl 0.1 g/L, $CaCl_2.2H_2O$ 0.1 g/L, NH_4Cl 1.5 g/L, $MgCl_2.6H_2O$ 0.1 g/L, trace mineral 10 mL/L and sugar 1.4 g/L (Mehanna *et al.* 2010; Jingyu *et al.* 2017). The wastewater solution in the catholyte was diluted to reach aerobic ratio with a C:N:P ratio of 100:5:1 respectively, and COD of 500 ± 20 mg/L The catholyte was aerated at a constant flow-rate in order to supply a dissolved oxygen level of 4 mg/L. The middle chamber was filled with the water at NaCl concentrations of 17 and 35 g/L in different tests. After establishment of biofilm on the anode and cathode electrodes, the MDC was evaluated under fed-batch mode. In the batch mode experiment, solutions were replaced after 4 days in all three chambers. In the second part of the experiments, the MDC was evaluated under fed-continuous mode with a flow rate of 0.08 mL/min giving HRTs of 1 day, 0.04 mL/min with HRTs of 2 days, 0.028 mL/min with HRTs of 3 days, 0.021 mL/min with HRTs of 4 days. All the tests were operated at ambient temperature (20–25 °C) (Ebrahimi *et al.* 2017). In the third part of the experiments, water biofouling of the middle desalination chamber was examined.

Analysis and calculation

The voltage (E) across the external resistor (R_e) in the MDC was recorded every 20 min using a multimeter. Current was calculated according to Ohm's law (I = ER). Power density was calculated as $P = EI/A$ (mw/m^2), where A (m^2) is the projected surface area of the anode (Veerman *et al.* 2008). The coulombic efficiency (CE) (%) is defined as:

$$CE = \frac{\text{Coulombs recovered}}{\text{Total coulombs in substrate}} \quad (1)$$

The CE can therefore be calculated through the following equation (Kim & Logan 2013a, 2013b):

$$CE(\%) = \frac{M \sum_{i=1}^{n} I_i t_i}{F\, b \Delta C\, V} \times 100 \quad (2)$$

where I_i is the output current of MDC at time t_i, F is Faraday's constant (96,485 c/mol), b is the number of moles of electrons produced per mol of COD (4 mol of e^-/mol of COD), ΔC is the removal of COD concentration (g/L), V is the anolyte volume (L), and M is the molecular weight of substrate (g/mol) (Sutzkover-Gutman & Hassan 2010). The desalination chamber effluent was analyzed for conductivity and salinity. Salinity was measured by a Sea Water Refractometer (Red Sea Co., USA). NaCl removal (%) was calculated for the desalination chamber. COD measurements were made by spectrophotometer (DR6000, HACH Co., USA). COD removal (%) was calculated for anode and cathode chambers. Conductivity and pH of solutions were measured using a conductivity meter (HQ 40d, HACH Co., USA) and pH meter (914 pH meter, Metrohm AG, Switzerland) respectively. Biocontamination of water in the middle desalination chamber was tested by biochemical tests for *Escherichia coli (E. coli)* identification.

RESULTS AND DISCUSSION

Desalination performance and changes of solution conductivity in the middle chamber

The MDC was operated for several months to analyze desalination performance during fed-batch and fed-continuous mode at four different HRTs (1, 2, 3 and 4 days) and two different NaCl concentrations (17 and 35 g/L) (Table 1).

According to the results, the salinity of the middle chamber in both operational modes decreased gradually when the HRTs were increased from 1 to 4 days. The maximum rates of NaCl removal in the fed-batch mode were 43 ± 0.5 and 48.8 ± 0.2%, and for the continuously fed mode the maximum rates were 47.7 ± 0.3 and 52.3 ± 0.7% in 17 and 35 g/L NaCl concentrations, respectively. The higher salt removal rate in the continuously operated

Table 1 | Desalination performance for MDC during batch and continuous fed modes

		NaCl removal (%)			
Operational mode	**NaCl concentration (g/L)**	**24 h**	**48 h**	**72 h**	**96 h**
Fed-batch	17	11.1	20.9	34.2	43
	35	13.6	22.7	36	48.8
Fed-continuous	17	11.9	20.7	37.8	47.7
	35	13.6	23.1	40	52.3

MDC could be mainly attributed to the higher power generation during each desalination cycle. On the other hand, the data obtained showed that increasing HRTs enhanced the NaCl removal rate because of the longer period of time for desalination. These rates were lower than those previously obtained by Cao *et al.* (2009) in the three-chamber MDC using ferricyanide catholyte. Despite the ability of ferricyanide to produce higher cathode potentials and achieve faster reduction kinetics, it is not suitable due to the high cost and toxic characteristics (Kokabian & Gude 2015). To account for these issues, using the bio-cathode is appropriate, as revealed in this study. The use of bio-cathodes as catalysts in MDC is becoming increasingly popular due to self-regeneration and sustainability (Husseini *et al.* 2015). In some other studies, salinity removal has been ~90% using 30–35 g/L NaCl in three-chamber MDCs (Kim & Logan 2013a, 2013b). Increasing the HRTs of salt solution from 1 to 4 days increased total NaCl removal. On the other hand, the salt removal rate was higher in the continuously fed mode as compared to the fed-batch mode and the ratio of removal was also increasing in higher concentrations of NaCl (Figures 2 and 3).

These results are similar to those obtained in previous studies using a three-chamber bio-cathode MDC (Mehanna *et al.* 2010; Qu *et al.* 2013; Ebrahimi *et al.* 2017). The changes of conductivity in the middle chamber, based on initial salt concentration and operational mode, is demonstrated in Table 2. According to the results, the salt solution conductivity decreased gradually in both operational modes over a 4-day desalination cycle. The conductivity of the middle chamber in the batch mode decreased from 61.8 ± 0.2 and 31.7 ± 0.3% to 31.5 ± 0.5 and 18.2 ± 0.8% mS·cm^{-1} at 17 and 35 g/L NaCl concentrations respectively. The corresponding data for the continuously fed mode were 61.8 ± 0.5 and 31.7 ± 0.3% to 30.2 ± 0.8% and 16.2 ± 0.8% mS·cm^{-1} respectively. The obtained data are comparable to those reported by Meng *et al.* (2014) using dewatered sludge as fuel.

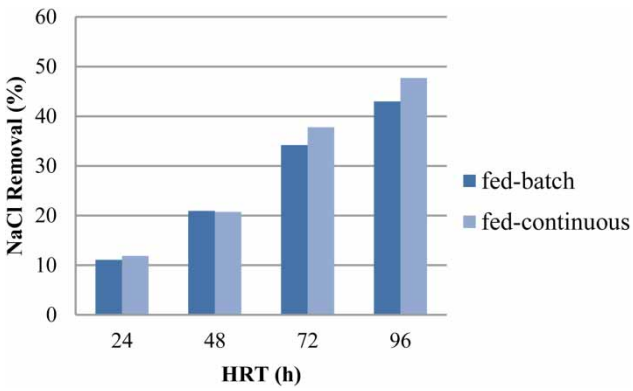

Figure 2 | Desalination performance at 17 g/L NaCl concentration.

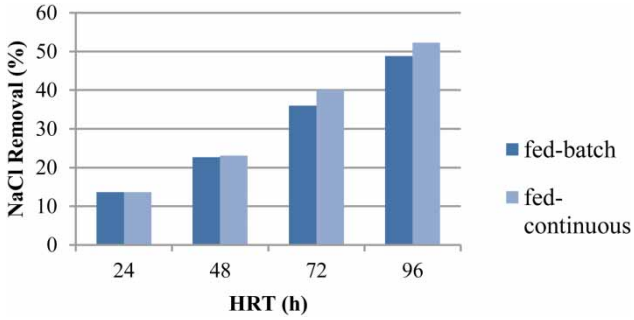

Figure 3 | Desalination performance at 35 g/L NaCl concentration.

Table 2 | Changes of conductivity in the middle chamber during batch and continuously fed modes

Operational mode	NaCl concentration (g/L)	Salt solution conductivity (mS per cm)				
		0 h	24 h	48 h	72 h	96 h
Fed-batch	17	31.7	28.1	25.2	21.1	18.2
	35	61.8	55.2	47.7	40.8	31.5
Fed-continuous	17	31.7	28.4	24.1	19.3	16.2
	35	61.8	54.1	48.5	40.1	30.2

pH and conductivity in the electrode chambers

Conductivity changes in the anode and cathode chambers were less than those in the middle chamber (Figure 4). As indicated in Figure 4, the anode chamber conductivity increased by only 16.1 and 16.9% when the conductivity of the middle chamber decreased by 42.5 and 49% during the fed-batch mode with 17 and 35 g/L NaCl concentrations respectively. The corresponding data for continuous mode were 17.6 and 17.1% in the anolyte and 48.8 and 51.1% in the middle chamber respectively. The conductivity of catholyte as shown in Figure 4 increased gradually in both operational modes and at both concentrations of NaCl. The results showed that the rate of conductivity changes in

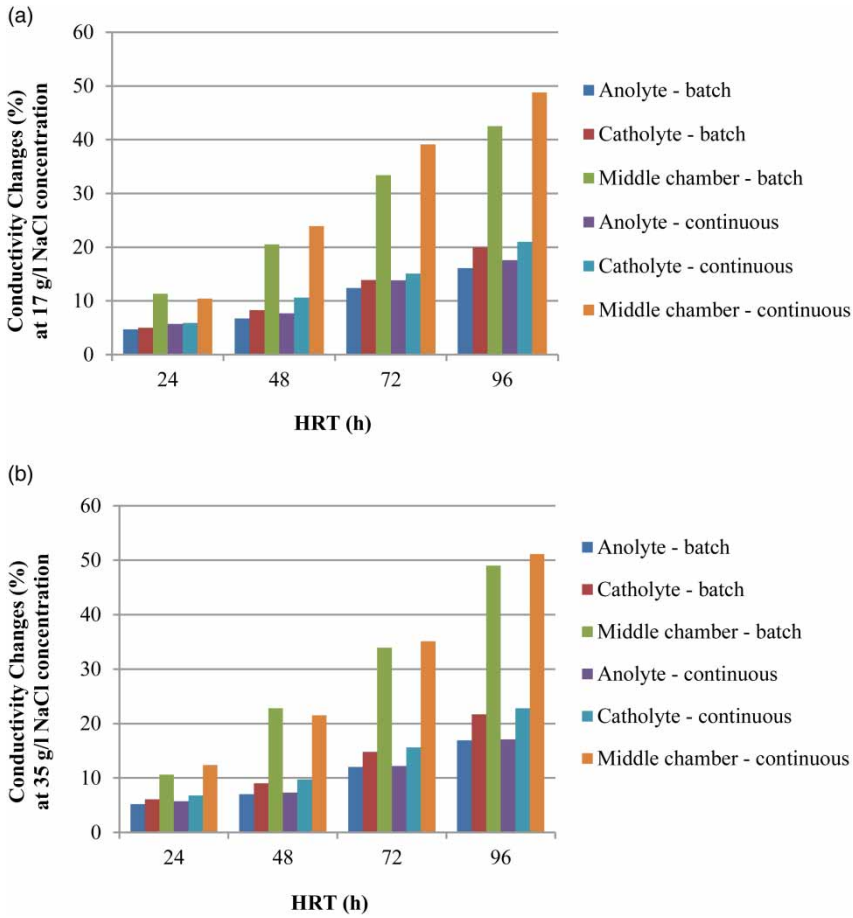

Figure 4 | Changes of conductivity in electrode chambers during fed-batch mode and continuously fed mode: (a) performance at 17 g/L NaCl concentration; (b) performance at 35 g/l NaCl concentration.

continuous mode at 35 g/l NaCl concentration were higher in comparison with that of the batch mode at 17 g/l NaCl concentration.

The anolyte pH in both batch and continuous modes dropped from 7 to 6.32, 6.47 and 6.37, 6.48 in 17 and 35 g/l NaCl concentrations respectively. This decline was attributed to the anaerobic activity of microorganisms and proton accumulation. The catholyte pH increased from 7 to 7.2 and 7.1 in batch mode with 17 and 35 g/L NaCl concentrations respectively. The corresponding pH values for the continuous mode were 7.1 and 7.08 respectively, which occurred mainly due to the consumption of protons and hydroxide accumulation in the cathode chamber, as well as oxygen reduction that led to an increase of pH within the cathode chamber (Jingyu *et al.* 2017). pH fluctuation is a common phenomenon in the MDC systems (Lou *et al.* 2012). The data obtained from this study showed that the changes of pH in the anolyte and catholyte over time were gradual and low (Figure 5). The relatively smaller changes in anolyte pH and stable performance of anode potential suggest that the migration of HCO_3^- from the middle chamber to the anode chamber during desalination increased the buffer capacity of wastewater and helped maintain microbial activity (Lou *et al.* 2012). A sudden decrease in the anode pH to less than 6 would hinder the activity of anode respiring microbes, and a high pH in the cathode would lead to the non-performance of the system (Luo *et al.* 2011). Hence, MDC is extensively buffered to control the biological environment of microbes in a stable state (Li *et al.* 2017). The pH variation in the anode and cathode chambers was comparable to previously reported MDCs (Cao *et al.* 2009; Mehanna *et al.* 2010; Li *et al.* 2017).

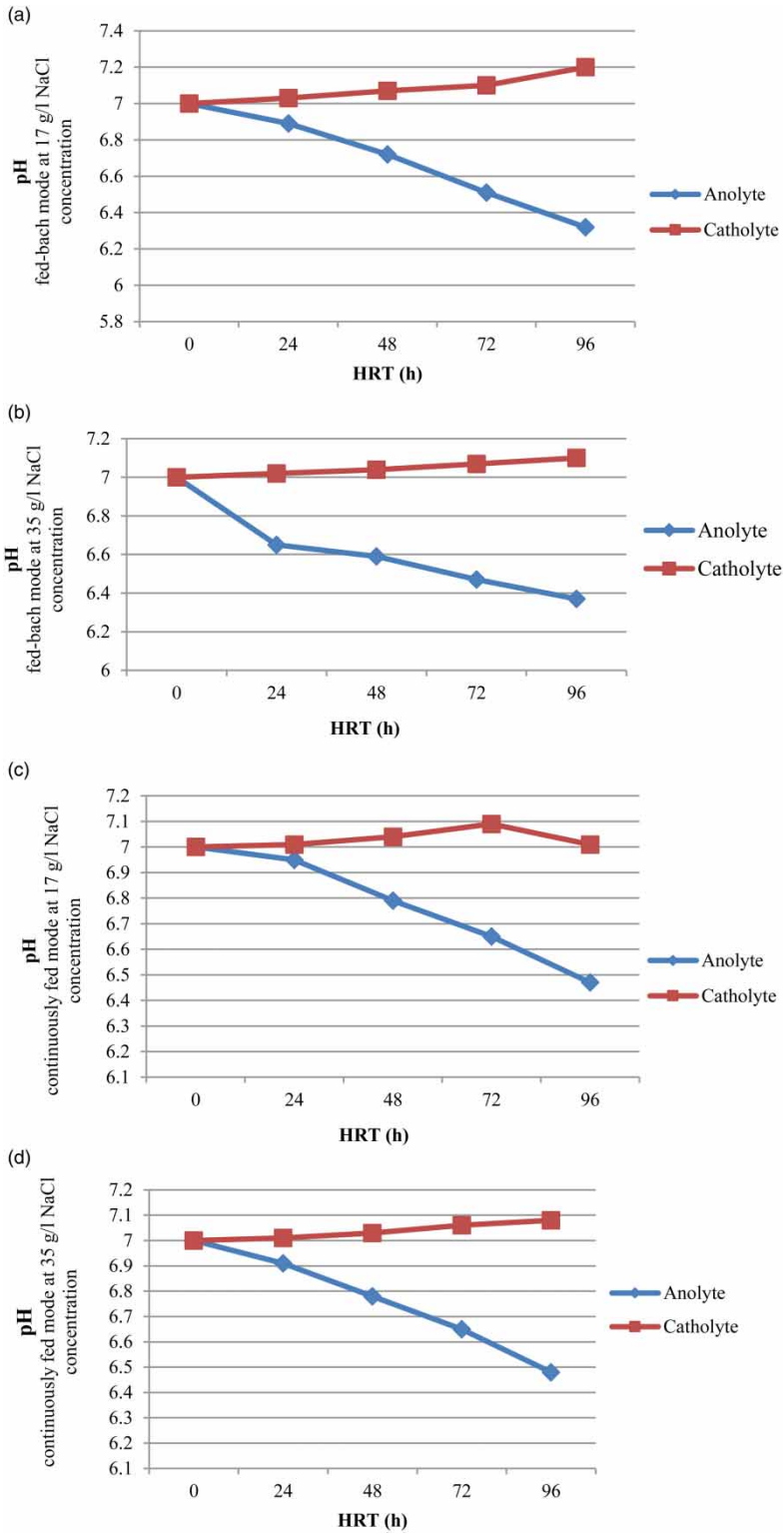

Figure 5 | Changes of pH in electrode chambers: (a) fed-batch mode at 17 g/L NaCl concentration; (b) fed-batch mode at 35 g/l NaCl concentration; (c) continuously fed mode at 17 g/L NaCl concentration; (d) continuously fed mode at 35 g/l NaCl concentration.

COD removal and coulombic efficiency

The three-chamber MDC was operated with a total COD value of 500 ± 20 mg/L in both the anolyte and catholyte solutions in order to prepare a carbon source to feed microorganisms in the electrode chambers. Figure 6 demonstrates the COD removal efficiency (%) in the anode and cathode chambers under batch and continuous mode with 17 g/l NaCl concentration. The corresponding data for the chambers with 35 g/l NaCl concentration are shown in Figure 7. As shown in both experiments, the maximum COD removal (%) in the anode chambers was 54 and 61%, feeding at HRTs of 4 days at two different NaCl concentrations (17 and 35 g/L), under fed-batch and continuous mode, respectively. These results showed that the COD removal in continuous mode improved when compared to the batch mode. On the other hand, COD removal improved at high level NaCl concentrations (35 g/L) compared to 17 g/l NaCl concentration in each operational mode. Its efficiency in the cathode chamber was also higher than that in the anode chamber. Greater COD removal efficiency under continuous mode was probably due to preventing large pH variation in the anode and cathode chambers by the continuous flow operation when compared to batch mode. The data obtained in this study indicated an effective COD removal as per previous studies carried out by Chen *et al.* (2012) using stacked MDC and Lou *et al.* (2012) using domestic wastewater as the sole substrate in the anode chamber during fed-batch operational mode. The COD removal efficiency in this study was lower than that obtained by Yuan *et al.* (2016) and Sevda *et al.* (2017). This could be because of the different operational modes in these studies. Yuan *et al.* (2016) achieved 70.6% COD removal from synthetic wastewater in their study to couple forward osmosis (FO) with MDC. In the study by Sevda *et al.* (2017) the maximum COD removal was achieved by MDC using an initial salt concentration of 20 g/L NaCl solution, and the COD removal increased slightly from 64.0 to 70.5% when the catholyte was changed from phosphate buffer solution (PBS) to acidified water.

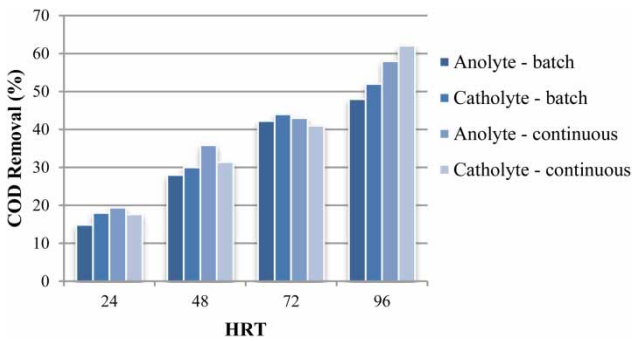

Figure 6 | COD removal efficiency (%) at 17 g/L NaCl concentration.

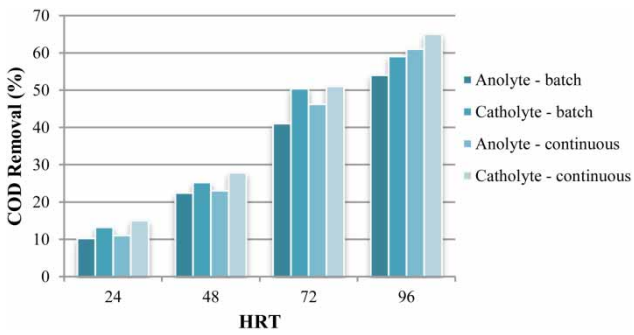

Figure 7 | COD removal efficiency (%) at 35 g/L NaCl concentration.

Coulombic efficiency (%) is the fraction of electrons transported to the anode to the total electrons released by substrate oxidation (Mahdi 2016). The oxidation of a substrate occurs with the removal of electrons with the electrons defined for each substrate based on writing out a half reaction. For complex substrates, it is more convenient to use COD as a measure of substrate concentration (Kim & Logan 2013a, 2013b). Indeed, coulombic efficiency demonstrates the amount of electrons which are obtainable from the substrate for producing the electrical current. It is defined as the ratio of the actual amount of electrons gained by the substrate to the theoretical amount of electrons obtained by the bacteria based on COD removal as outlined in the Materials and methods section above (Ebrahimi *et al.* 2017). The CE changes determined when the performance of the MDC reactor was stable. The variation of CE is shown in Figure 8. According to the presented data, the continuously fed mode had higher CE with the value of 13% which was more than the fed-batch mode with the CE value of 12 and 12.8% at NaCl concentrations of 17 and 35 g/l respectively. The data obtained showed that coulombic efficiency (%) was less than 50% which could have resulted from the fermentation of substrate, methanogenesis process or even the aerobic oxidation of the substrate in the anode (Mahdi 2016). The fermentation process in the anode occurs due to

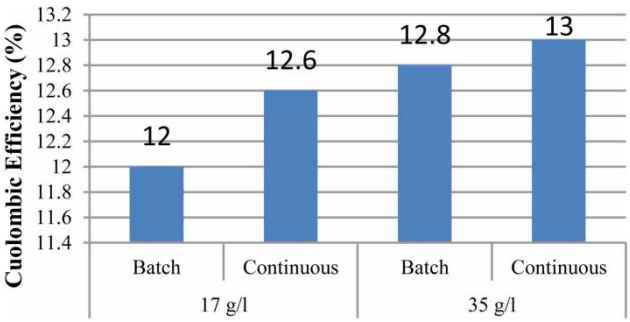

Figure 8 | Coulomic efficiency (%) during batch and continuous mode at 17 and 35 g/L NaCl concentrations.

the deficiency of NO_2, as was mentioned by Zhang *et al.* (2012). Kokabian & Gude (2015) also reported that the enrichment of microbial consortium could decrease the fermentation process and thus increase the CE.

This is in agreement with Kim & Logan (2013a, 2013b), where the authors reported that the coulombic efficiency can be minimized by the alternative electron acceptors which could be present in medium or diffused to the anode, also the bacterial growth could decrease the coulombic efficiency, so the bacteria that did not have the ability to use the electrode as the electron acceptor could utilize the substrate for fermentation (Kim & Logan 2013a, 2013b).

Membrane and middle chamber biofouling

Membrane biofouling is one of the main obstacles to an efficient desalination process (Sutzkover-Gutman & Hassan 2010). Biofouling is an important phenomenon that occurs as a result of a microbial adhesion or absorption on the source of the membrane, developing a polymeric layer of a biofilm (transparent exopolymer particles (TEP)) (Elmakawy *et al.* 2014). Such a polymeric biofilm is hard to regulate and eradicate, leading to reduced flux and an overall decline in productivity. Biofouling of the ion-exchange membranes facing the middle chamber (seawater) could generate potential concerns of deteriorating quality of the desalination water and cause biofouling of the downstream desalination process (Sutzkover-Gutman & Hassan 2010). In this study, biofouling of ion-exchange membranes contacted with anolyte and catholyte (using bio-cathode) was expected. Through visual inspection, it was observed that the surface of the AEM and CEM turned black and was covered with a fungi-like deposit.

To further confirm the biofouling of the middle chamber across the AEM or the CEM in the MDC, biochemical tests were conducted. The bio-contamination of the middle chamber was observed with all the tests. Ion-exchange membranes are expected to block the transport of the microorganisms, thus the source of microbial contamination was possibly from the feeding salt water. However, previous studies have demonstrated that the AEM could transport ionic organic compounds. Those organic compounds supported the growth of microorganisms in the middle chamber, resulting in biofouling of both the AEM and CEM. Some other previous studies also revealed the transport of organic compounds into the middle chamber across all the tests (Ping 2016). Although biofouling may not significantly change the membrane properties, the presence of microorganisms in the middle chamber would deteriorate the quality of the desalinated water and also introduce organisms to cause biofouling in the downstream desalination units.

CONCLUSIONS

In this study, the experiments were designed to compare the performance of two operational fed modes using bio-cathode and synthetic wastewater in four different HRTs and two different NaCl concentrations and to investigate the biofouling of the middle chamber. The maximum salt removal and extent of COD removal were both improved in the continuously fed mode compared to the fed-batch MDC. It can be said that the type of hydraulic flow can improve the performance of MDC in both NaCl concentrations (17 and 35 g/l). However, further studies are required to evaluate the cost of these operational conditions in order to make a final decision. The biofouling on the solution of the middle chamber was identified in this study and was confirmed by conducting bio-microbial tests, so appropriate pretreatment before desalination will greatly slow down membrane and, in consequence, middle chamber biofouling, and extend the system performance. Thus, MDC could be employed as a standalone technology or be integrated with the traditional membrane-based systems such as RO to decrease salinity of the feed water and subsequently reduce energy requirements. Further development of this research could involve studying continuous operation in

other MDCs configurations. Models for diagnosing and predicting biofouling can also be developed in order to design and optimize the performance of MDC as a pretreatment method for an efficient desalination by RO.

REFERENCES

Ali, I., Al-Othman, Z. A. & Al-Warthan, A. 2016 Removal of secbumeton herbicide from water on composite nano adsorbent. *Desalin. Water Treat.* **57**, 10409–10421.

Ali, I., Alharbi, O. M. L., Alothman, Z. A., Badjah, A. Y., Al Warthan, A. & Basheer, A. A. 2018 Artificial neural network modelling of amido black dye sorption on iron composite nano material: kinetics and thermodynamics studies. *J. Mol. Liq.* **250**, 1–8.

Basheer, A. A. 2018 New generation nano-adsorbents for the removal of emerging contaminants in water. *J. Mol. Liq.* **261**, 583–593.

Basheer, A. A. & Ali, I. 2018 Stereo selective uptake and degradation of (±)-o, p-DDD pesticide stereomers in water-sediment system. *Chirality* **30**, 1088–1095.

Cao, X., Huang, X., Liang, P., Xiao, K., Zhou, Y., Hang, Z., & Logan, X. & E, B. 2009 A new method for water desalination using microbial desalination cells. *Environ. Sci. Technol.* **43**, 7148–7152.

Chen, X., Liang, P., Wei, Z., Zhang, X. & Huang, X. 2012 Sustainable water desalination and electricity generation in a separator coupled stacked microbial desalination cell with buffer free electrolyte circulation. *Bioresour. Technol.* **119**, 88–93.

Ebrahimi, A., Najafpour, G. D. & Kebria, D. Y. 2017 Effect of batch vs. continuous mode of operation on microbial desalination cell performance treating municipal wastewater. *Iran. J. Hydrogen Fuel Cell* **4**, 281–290.

Elmakawy, A., Hegab, H. & Pant, D. 2014 The near-future integration of microbial desalination cell with reverse Osmosis technology. *Energy Environ. Sci.* **7**, 3921–3933.

Husseini, G. A., Azzam, A., Khawaga, R. & Aidan, A. 2015 Microbial desalination cell technology: a review and case study. *Desalination* **359**, 1–3.

Jacobson, K. S., Drew, D. M. & He, Z. 2011 Use of a liter-scale microbial desalination cell as a platform to study bioelectrochemical desalination with salt solution or artificial seawater. *Environ. Sci. Technol.* **45** (10), 4652–4657.

Jingyu, H., Ewusi-Mensah, D. & Norgbey, E. 2017 Microbial desalination cells technology: a review of the factors affecting the process, performance and efficiendy. *Desalin. Water Treat.* **87**, 140–159.

Kim, Y. & Logan, B. E. 2013a Microbial desalination cells for energy production and desalination. *Desalination* **308**, 122–130.

Kim, Y. & Logan, B. E. 2013b Simultaneous removal organic matter and salt ion from saline wastewater in bioelectrochemical systems. *Desalination* **308**, 116–121.

Kokabian, B. & Gude, V. G. 2015 Sustainable photosynthetic biocathode in microbial desalination cells. *Chem. Eng. J.* **262**, 958–965.

Li, Y., Styczynski, J., Huang, Y., Xu, Z., McCutcheon, J. & Li, B. 2017 Energy-positive wastewater treatment and desalination in an integrated microbial desalination cell (MDC) – microbial electrolysis cell (MEC). *J. Power Sources* **356**, 529–538.

Lou, H., Xu, P., Roane, T. M., Jenkins, P. E. & Ren, Z. 2012 Microbial desalination cells for improved performance in wastewater treatment, electricity production and desalination. *Bioresour. Technol.* **105**, 60–66.

Luo, H., Jenkins, P. E. & Ren, Z. 2011 Concurrent desalination and hydrogen generation using microbial electrolysis and desalination cells. *Environ. Sci. Technol.* **45** (1), 340–344.

Mahdi, A. A. 2016 *Study of the Desalination Efficiency for Different Configurations of Microbial Desalination Cell (MDC)*. Institute of Water and Environment, Al-Azhar Ubiversity-Gaza, pp. 57–59.

Mehanna, M., Saito, T., You, J., Hichner, M., Cao, X., Huang, X. & Logan, B. E. 2010 Using microbial desalination cells to reduce water salinity prior to reverse osmosis. *Energy Environ. Sci.* **3**, 1114–1120.

Meng, F., Jiang, J., Zhao, Q., Wang, K., Zhang, G., Fan, Q., Wei, L., Ding, J. & Heng, Z. 2014 Bioelectrochemical desalination cell and electricity generation in microbial desalination cell with dewatered sludge as fuel. *Bioresour. Technol.* **157**, 120–126.

Ping, Q. 2016 *Advancing Microbial Desalination Cell Towards Practical Application*. Doctoral Dissertation, Virginia Polytechnic Institute. Available from: https://vtechworks.lib.vt.edu/handle/10919/73373.

Qu, Y., Feg, Y., Liu, J., He, W., Shi, X., Yang, Q., Lv, J. & Logan, B. E. 2013 Salt removal using multiple microbial desalination cells under continuous flow conditions. *Desalination* **318**, 17–22.

Sevda, S., Abu-reesh, I. M., Yuan, H. & He, Z. 2017 Bioelectricity generation from treatment of petroleum refinery wastewater with simultaneous seawater desalination in microbial desalination cells. *Energy Convers. Manage.* **141**, 101–107.

Sutzkover-Gutman, I. & Hassan, D. 2010 Feed water pretreatment for desalination plants. *Desalination* **264**, 289–296.

Veerman, J., Post, J. W., Saakes, M., Metz, S. J. & Harmsen, G. J. 2008 Reducing power losses caused by ionic shortcut currents in reverse electrodialysis stacks by a validated model. *J. Memb. Sci.* **310**, 418–430.

Yuan, H., Abu-Reesh, I. M. & He, Z. 2016 Mathematical modeling assisted investigation of forward osmosis as pretreatment for microbial desalination cells to achieve continuous water desalination and wastewater treatment. *Membr. Sci.* **502**, 116–123.

Zhang, G., Zhao, Q., Jiao, Y., Wang, K., Lee, D. J. & Ren, N. 2012 Biocathode microbial fuel cell for efficient electricity recovery from dairy manure. *Biosens. Bioelectron.* **31**, 537–543.

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