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Study of the performance of bench-scale electro-membranes bioreactor in leachate treatment

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ABSTRACT

In the present study, the integration of the electrochemical process with a membrane bioreactor was used as a new technology for leachate treatment. In the electro-membrane bioreactor (EMBR), aluminum electrodes were used as anodes and cathodes. The EMBR was operated at a current density of 0.5 mA/cm^2 and a solids retention time of 90 days to remove common contaminants such as ammonia-nitrogen (NH_3-N), chemical oxygen demand (COD), phosphate (PO_4^{3-}-P), and ultraviolet absorbance at 254 nm (UV_{254}). The maximum removal efficiencies of COD and NH_3-N were above 98%. The average removal efficiency of PO_4^{3-}-P by the EMBR system was 93%, which was significant compared to previous studies. The removal rate of humic substances based on UV_{254} was provided at approximately 96.95%. The trans-membrane pressure rate was acceptable for 80 days in the EMBR, which could be related to sludge size improvement and filtration resistance through the occurrence of electrocoagulation, electrophoresis, and electroosmosis mechanisms. The mean removal efficiencies in the EMBR were 90, 91.25, 96, and 87.5 % for chromium (Cr), cadmium (Cd), zinc (Zn), and iron (Fe), respectively. The slight change of mixed liquor-suspended solids (MLSS) in the leachate treatment reactor showed that the microorganisms in the new EMBR system had high adaptation. Based on the results, the EMBR is a promising technology to improve leachate treatment performance due to its excellent removal efficiency of common contaminants, metal removal, and reducing fouling.

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1. Introduction

In developed and developing countries, the economy and rapid population growth have led to the production of waste with different characteristics [1]. It is estimated that solid waste production will increase to 2.2 billion tons by 2025 [2]. As a result of the growing concerns of managing this amount of waste, various waste disposal methods have been used: incineration, recycling, open dump, composting, and landfilling [3]. Among the methods, sanitary landfill is one of the old and popular techniques due to its low operating and maintenance costs. Despite the economic benefits, landfill operation is associated with high amounts of leachate production. This leachate is caused by physical and chemical changes occurring in the waste and the entry of moisture and rainfall [1]. Common contaminants in leachate include ammonium, phosphate, suspended solids, and soluble organic matter, the values of which can vary based on parameters such as type and age of landfill, amount of rainfall, and climate change [4]. In addition to common compounds, the presence of metal ions (such as lead, cadmium, arsenic, copper, and iron), antibiotics, toxins, polycyclic aromatic hydrocarbons, and microplastics in leachate has recently been reported [2]. Improper disposal and treatment of leachate as a major source of pollution can have serious long-term effects on health and the ecosystem. Therefore, landfill leachate needs to be collected and treated before being discharged into the environment. It is very difficult to design a leachate treatment system to achieve a better disposal quality standard. However, most countries use biological processes such as conventional activated sludge, sequencing batch reactor (SBR), aeration lagoons, and upflow anaerobic sludge blanket (UASB) for leachate treatment, especially young leachate with high BOD concentration, due to their cost-effectiveness and simplicity [5]. Despite the good performance of these processes, the removal of COD from old landfill leachate is a challenging problem due to the presence of toxic and bio-refractory contaminants. To overcome this problem, researchers and wastewater engineers proposed the integration of physicochemical processes such as coagulation, electrooxidation, chemical precipitation, and membrane separation [4,6-8]. Among the above methods, adding membrane filtration to activated sludge treatment (known as membrane bioreactor process) is a promising method that can be a guarantee for removing persistent contaminants and reducing footprints [9-11]. In addition, the membrane bioreactor (MBR) system for the treatment of various solutions has advantages such as good output quality, high organic load, process stability, high biomass production, less sludge production, and low energy density [12,13]. Despite these advantages, the MBR process in a wide range of treatment applications still suffers from the fouling of membranes and maintaining long solids retention time (SRT) to improve the removal of micro-contaminants [14,15]. To solve this problem, most studies [14,15] have used membrane cleaning by backwashing, improving optimization parameters (e.g., SRT, hydraulic retention time (HRT), and mixed liquor suspended solids (MLSS) concentration), and improving wastewater properties by adding coagulant and adsorbent. Despite being efficient, these technologies can lead to increased operating costs, the risk of membrane destruction, and an increase in the volume of sludge produced [15]. To resolve the drawbacks mentioned above, electro-oxidation in MBR has recently been used to eliminate the fouling of membranes and improve the removal of micro-contaminants (12). This method, as an electro-membrane bioreactor (EMBR) process, facilitates the removal of contaminants through the mechanisms of electrophoresis, electroosmosis, and electrocoagulation by applying alternating-current (AC) or direct-current (DC). Electrocoagulation, as the main mechanism with in situ production of coagulant from the electrochemical decomposition of the anode, leads to sedimentation of the suspended particles. Applying an electric current also allows negatively charged particles such as activated sludge and polymers to move towards the positively charged electrode and stay away from membranes by electrophoretic mechanisms. Along with these mechanisms, the electroosmotic force, by removing bound water from the microbial floc, reduces the stability of the sludge to filtration and improves fouling control [16,17]. In addition, recent studies have reported that the application of an appropriate current can develop the hydraulic function of MBR to remove contaminants without a significant effect on microbial activities [18]. By examining aquaculture wastewater treatment, Song et al. (2020) explained that the application of current could improve ammonia removal by increasing the microbial population of nitrifying [19]. Hue et al. (2015) showed a seven-fold reduction in fouling rates in the EMBR compared to MBR [20]. Given these advantages, few researchers have used the EMBR process to treat real leachate in a bench-scale laboratory. The above-mentioned laboratory-scale research showed biological conversion of organic matter and ammonia, removal of phosphorus, change in floc morphology, and change in MLSS characteristics, which ultimately led to reduced membrane fouling. A full-scale study has not yet been conducted in the article databases. This means that in order to achieve a scientific approach, more experiments need to be performed on solutions of different strengths. Our research will allow leachate management enthusiasts to better understand the impact of this new process on strong solutions. Therefore, this study developed a bench-scale EMBR for landfill leachate treatment. The removal of the COD, NH_3-N, PO_4^{3-}-P, and humic substances in the new EMBR system was investigated. The fouling rate in the bioreactor was assessed by trans-membrane pressure (TMP) measurement.

2. Materials and methods

2.1. Material and reagents

In the present study, all reagents were of analytical grade and used without any further purification. Potassium dichromate ($K_2Cr_2O_7$), sulfuric acid (H_2SO_4), mercury (II) sulfate ($HgSO_4$), silver sulfate ($AgSO_4$), and potassium hydrogen phthalate (KHP, $C_8H_5KO_4$) were purchased from Merck Germany. Reagents purchased from Kimia Company, Iran were used to easily determine the anions and cations present in the leachate. Deionized water was used in the process of washing the electrodes, preparing samples, and determining the concentration of contaminants.

2.2. Characteristics of leachate

The studied bioreactors were fed with leachate collected from a landfill in Ahvaz, Iran. The average waste generated in the city of Ahvaz is close to 1000 tons, of which 95% enter directly into the landfill. 35 L landfill leachate was taken daily according to the grab sample and then added to the feed tank by mixing. The characteristics of the leachate are given in Table 1.

2.3. Experiment setup

A membrane bioreactor (28.2 × 29 × 39.2 cm) was fabricated according to Figure 1 and used to treat the leachate. Two aluminum plates, as anode and cathode with a distance of 5 cm, were submerged in the reactor. The electrodes were connected to a DC power supply using a copper wire, which was controlled by a timer to obtain a current of 0.5 mA/cm^2 and adjust the alternating DC with an operating mode of 15 min ON-45 min OFF. A hollow fiber ultrafiltration sheet (Material: Polysulfone, Porosity= 48.3%, Pore radius= 6.9 nm, Surface roughness= 2.9 Ra-nm, Layer thickness= 200 μm) with a height of 23 cm was installed vertically in the center of the bioreactor. Aeration diffusers were placed on the reactor bottom and near the module to provide oxygen for biological oxidation and the proper mixing of the solution. A pressure gauge was installed outside the reactor to indicate the output flow rate and measure the degree of membrane fouling.

2.4. Operating conditions

The submerged EMBR was incubated with activated sludge obtained from the Ahvaz wastewater treatment plant. During the start-up phase, the reactor was fed with leachate at a flow rate of 11 ml/min to expand the high concentration of biomass. Some glucose was added to the reactor to ensure rapid biomass production. During this period, the hydraulic time of 48 h was considered to stabilize the treatment efficiency. During the first 30 days (startup phase), the sludge concentration was slowly increased from 2200 to 9300 mg/L. Thereafter, the HRT was gradually reduced to 18 h to evaluate the reactor's performance under the same operating condition. After 30 days of reactor start-up, the EMBR system was continued at room temperature for 60 days by measuring the parameters and their removal efficiency. Fouling behavior was determined by measuring the output flow from the membranes. No backwash of the membrane module was performed during the operation. However, in the case of the fouling of the membranes, the membrane module was withdrawn from the bioreactor, and its physical and chemical washing was performed based on the occurrence of 40 kPa trans-membrane pressure with distilled water (for 20 minutes) and sodium hypochlorite solution (for 8 hours). In addition, the EMBR at complete SRT operated to reduce sludge loss from the process.

2.5. Analytical methods

The chemical oxygen demand (COD) was determined by the colorimetric method in the presence of potassium dichromate, and its absorption was performed using a UV-vis spectrophotometer (DR 6000, Hack, USA) at 600 nm (standard method 5220B). The total suspended solids were determined by 100 mL of sample extracted from the sludge mixture and stored for drying at 105 °C for 24 h. Ammonia nitrogen (NH_3-N) and orthophosphate (PO_4^{3-}-P) tests were performed to evaluate the process using standard methods 4030 and 4020, and a UV-vis spectrophotometer at 254 nm was used to measure UV_{254} (a simple and reliable parameter for monitoring and control of the removal of organic matter). The temperature and pH were determined by a multiparametric probe. Heavy metal concentrations were measured by inductively coupled plasma atomic emission spectroscopy (ICP-AES, DV 5300 model, Perkin-Elmer, USA). Transmembrane pressure (TMP) changes during the operation were monitored by a pressure gauge.

Table 1. The properties of the collected landfill leachate.

Parameter	Value mean	Unit
COD	3100	mg/L
BOD_5	900	mg/L
NH_3-N	450	mg/L
PO_4^{3-}-P	25	mg/L
UV_{254}	2.1	nm
EC	3.25	mS/cm
pH	7.6	
Temperature	26.5	°C

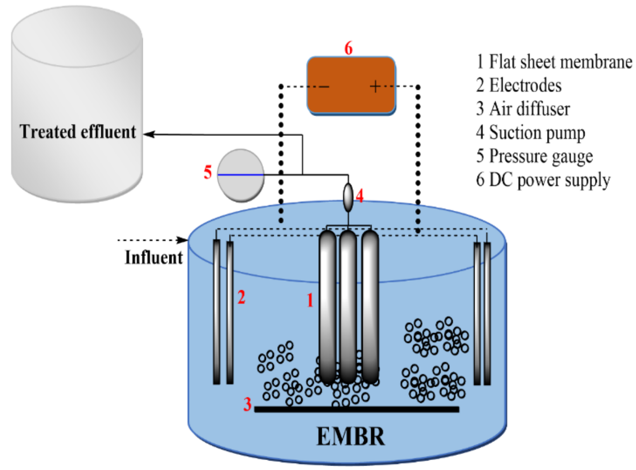

Fig. 1. Schematic of electro-membrane bioreactor.

3. Results and discussion

The performance of the EMBR system has been studied by changing parameters such as COD, NH_3-N, PO_4^{3-}-P, UV_{254}, and MLSS, and its results are discussed in the following sections.

3.1. COD removal

Fig. 2a shows the COD removal trend in the EMBR system. As can be seen, the efficiency of COD removal was low at the beginning (startup period), which may be due to the lack of adaptation of the sludge used with the leachate. However, the removal efficiency of COD was enhanced from 50.23 to 98% on the 60th day by increasing the operating time. This may be related to the improvement of microbial activity during the relatively low increase in pH and temperature in the EMBR reactor. Hemmati et al. (2012) found that temperature had a significant effect on the changes in biomass properties and their stability against xenobiotic contaminants [21]. In the present study, temperature and pH increased between 2-4 units due to current density. As the operating time continued, a significant reduction in the COD removal efficiency in the system was observed on the 65th day due to the deactivation of the electrodes. Both electrodes were withdrawn from the reactor and physically washed to improve the process. Chemical cleaning was also performed to remove the biofilms; then, the electrodes were returned to the reactor. On the 83rd day, the efficiency decreased from 98.58% to 90%, which could be due to fouling problems caused by the presence of toxic substances from the leachate. At this time, the membranes were withdrawn from the reactor, and physicochemical washing was performed simultaneously to remove the adhesive materials. This result shows that the EMBR reactor encountered membrane fouling only once during 90 days of operation, at which time the efficiency was significantly higher. This efficiency improvement in the EMBR may be related to the predominant effect of the electrocoagulation process in combination with biological and filtration processes. In the electrocoagulation system, Al^{3+} is produced by the electrooxidation of the aluminum anode and forms aluminum hydroxide ($Al(OH)_3$) as a floc sweep to absorb pollutants. At the same time, the occurrence of electrooxidation in the reactor leads to the production of hydroxyl radicals as a strong oxidant for the degradation of organic pollutants. Similar results were reported by Akkaya et al. (2020) for leachate treatment by EMBR [22]. In this study, the COD removal efficiency in the EMBR system was 10-20% higher than the conventional MBR system. They explained this efficiency improvement based on the interaction between Al ions and negative colloids. Manica et al. (2021) investigated the effect of electrocoagulation on EMBR performance in wastewater treatment and reported that the COD removal efficiency improved from 87% to 95% by applying the current density. This increase in efficiency can be related to good bacterial activity, membrane filtration, and the effect of oxidation processes on the biological availability of organic matter [23]. By studying the effect of electrical density on the microbial population in wastewater treatment, Zeyoudi et al. (2015) found that by applying a current, the removal efficiency of organic matter improved through the production of the hydroxyl radical as a strong oxidizer of stable pollutants [24].

3.2. NH_3-N and PO_4^{3-} removal

The histogram of NH_3-N removal efficiency during the 90-day operation period is shown in Fig. 2b. Similar to COD removal, the NH_3-N removal trend was low at the beginning of the startup period; however, with the adaptation of the microorganism, the efficiency improved significantly during the operation period. Efficiency changes in the second and third 30 days were between 85 and 99%. This efficiency was higher than the solutions treated by the MBR and EMBR in previous studies [25,26]. This increase in NH_3-N removal efficiency may be related to the synergistic effect of the electrocoagulation process, biodegradation with nitrifying bacteria, and electrodegradation of NH_3-N on the anode surface. These results emphasize that applying a current density of 0.5 mA/cm^2 in the EMBR system has no significant effect on nitrifying bacteria. Li et al. (2001) reported that applying currents above 2.5 mA/cm^2 had a limiting effect on their metabolism [21]. Similar results were observed by Ensano et al. (2018) for the control of drug compounds in a submerged EMBR [28]. In this study, the concentration of NH_3-N decreased from 32.53 to 11.12 mg/L by applying a current density of 0.5 mA/cm^2. The increase in efficiency was due to the influence of electrocoagulation, oxidation, and biological processes (activity of nitrifying bacteria) on the conversion of NH_4 to NO_3. In the EMBR process, NO_3-N can be rapidly converted to N_2 gas due to the activity of denitrifying bacteria. Treviño-Reséndez and Nacheva (2021) investigated the removal of petroleum compounds by the EMBR process. They reported that the production of hydroxyl radicals at the anode and solution surfaces led to the conversion of stable and toxic contaminants into nitrifying bacteria into biodegradable products [29]. Dawas-Massalha (2014) reported that applying high SRT to the reactor increased the growth of slow-growing bacteria such as nitrifying bacteria, resulting in improved NH_3-N degradation [30]. In our study, high SRT (60 days) was considered to improve bacterial activity and high growth. The usefulness of integrating electrooxidation with the MBR was clearly seen in the PO_4^{3-}-P removal efficiency (Figure 2c). The mean PO_4^{3-}-P removal efficiency in the present study was higher than the MBR system used in other studies [15,25]. This efficiency improvement can be explained as follows. When current is applied to the MBR, the Al^{3+} ion is released from the anode surface (Eq. 1) and then reacts with the hydroxide ion (OH^-) generated at the cathode surface to produce $Al(OH)_3$ (Eq. 2 and 3). The produced $Al(OH)_3$ with the characteristic of high surface area absorbs soluble phosphorus. At the same time, the excess Al^{3+} reacts directly with the phosphorus ions to form $AlPO_4$ (Eq. 4). Hasan et al. (2014) observed similar results for the removal of phosphorus during the treatment of municipal wastewater by a submerged EMBR [31]. Ensano et al. (2019) also found 100% removal efficiencies. In this study, the concentration of PO_4^{3-}-P decreased from 6.61 to 0 mg/L by applying a current density of 0.5 mA/cm^2. This efficiency improvement was explained based on the formation of phosphorus ($Al_6(OH_{15})PO_4(s)$ depositable species through the reaction of PO_4^{3-} with the active species of Al ($Al_6OH_{15}^{3+}$) in solution [28]. Ibeid and Elektorowicz (2021) found that in addition to the effect of the phosphorus aluminum complex mechanism in solution, PO_4^{3-}-P moved towards the positively charged electrode through electrostatic absorption [32].

$$Al \rightarrow Al^{3+} + 3e^- \quad (1)$$

$$3H_2O + 3e^- \rightarrow 1.5\ H_2 + 3OH^- \quad (2)$$

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

$$Al^{3+} + PO_4^{3-} \rightarrow AlPO_4 \quad (4)$$

3.3. *UV_{254} and metal removal*

The removal efficiency of humic substances measured on the basis of UV_{254} is shown in Figure 2d. Applying a current field in a bioreactor increases the removal of humic substances by an average of over 90%, which is higher than the MBR used in other studies [28]. This efficiency improvement can be useful to minimize the formation of disinfection products in the effluent of wastewater treatment plants containing chlorination. The increase in efficiency can be explained based on the production of $Al(OH)_3$ as a floc sweep. The application of current also leads to the charge of the particles, and thus, the probability of moving to the electrodes instead of the membranes increases. To confirm the discussion, particle aggregation and its effect on electrode inactivation were observed on day 65 of the present study. Ensano et al. (2019) investigated the effect of the EMBR process on humic removal efficiency. They reported that applying current to MBR increased UV_{254} removal efficiency from 74.24% to 90.68%. This efficiency improvement was explained based on the combined effect of the electro-oxidation process with biodegradation and filtration [28]. Millanar-Marfa et al. (2018) observed similar results in higher UV_{254} removal in the EMBR system than the conventional MBR. The efficiency increased from 86.8% in the conventional MBR to 92.36% in the EMBR [33]. The removal efficiency of heavy metals from leachate by the EMBR system during the 80th day of operation is shown in Figure 3a. It is clear from the graph that the combined system of electro-oxidation with the MBR has a high potential for metal removal. The concentrations of Cr, Cd, Zn and Fe decreased from 0.5, 0.05, 0.4, and 4.3 mg/L to 0.05, 0.004, 0.016, and 0.53 mg/L, respectively. This efficiency rate was higher than the conventional MBR process used in previous studies [34]. The efficiency improvement can be explained based on the formation of $Al(OH)_3$ sweep floc through the reaction of hydroxide ions and Al^{3+}. In addition, increasing the pH in the EMBR by producing hydroxide ions from the cathode electrode can be beneficial for metal precipitation from the leachate. Similar results were observed by Keerthi and Balasubramanian (2015) for the removal of metals by the MBR-electrocoagulation process [34].

3.4. *Evaluation of trans-membrane pressure*

The changes in trans-membrane pressure during the EMBR operation with an MLSS of 9450 mg/L are shown in Figure 3b. In the early days of operation, the TMP was less than 15 kPa. However, with the further increase in the operating period to the 65th day, the TMP value reached 20kPa, which was due to the withdrawing of the electrodes and the movement of sludge towards the membranes. Ensano et al. (2018) found that applying a current field to the MBR allowed the negatively charged sludge to move toward the positively charged electrode and stay away from the membranes via the electrophoretic [28]. As the treatment time extended to the 84th day, the TMP rate reached above 40 kPa. These results indicate that the membranes in the EMBR are washed once, and the process can be used for long-term leachate treatment. The reduction of membrane fouling can be explained based on the effect of current density on the interaction between $Al(OH)_3$ with solids to form sludge with different properties and low TMP. Borea et al. (2019) found similar results and reported that applying current density improved the electrophoresis mechanism by moving a negatively charged sludge toward the anode instead of the membrane surface [26]. Millanar-

Marfa et al. (2018) investigated fouling in the MBR and EMBR treatment systems. They found that by increasing the treatment time from 0 to 1200 h, the clogging rate in the MBR increased relative to the EMBR so that in this time range, six times the TMP MBR reached 40 kPa compared to two times the EMBR [33]. Ibeid et al. (2013) observed similar results in reducing the TMP rate in EMBR compared to conventional MBR. Reduction of clogging rate in this study was explained by degradation of organic flocs, control, and restriction of floc movement and reduction of floc settling on the membrane surface [35].

3.5. MLSS concentration changes

The development of biomass concentration in the process was monitored by evaluating the MLSS concentration, and the results are shown in Figure 4. As can be seen at the beginning of the startup period, the MLSS concentration was between 2250 and 2590 mg/L. After day 10, MLSS growth in the reactor significantly improved so that it reached 9200 mg/L at the end of the startup period. This amount of biomass concentration was almost constant until the 60th day, after which a significant decrease in the MLSS concentration was observed after the 61st day. This may be due to the accumulation of some sludge on the electrodes due to the mechanism of electrophoresis. At this time, the electrodes were removed from the reactor for washing and reuse. After this day, the biomass concentration increased slowly and then remained constant. This increase may be due to the oxidation of the anode electrode and the release of Al^{3+} ions inside the bioreactor for electro-oxidation of pollutants and the formation of $Al(OH)_3$ floc. Similar results were observed by Borea et al. [36] and Bani-Melhem and Smith [25] for changes in biomass concentration in the EMBR process.

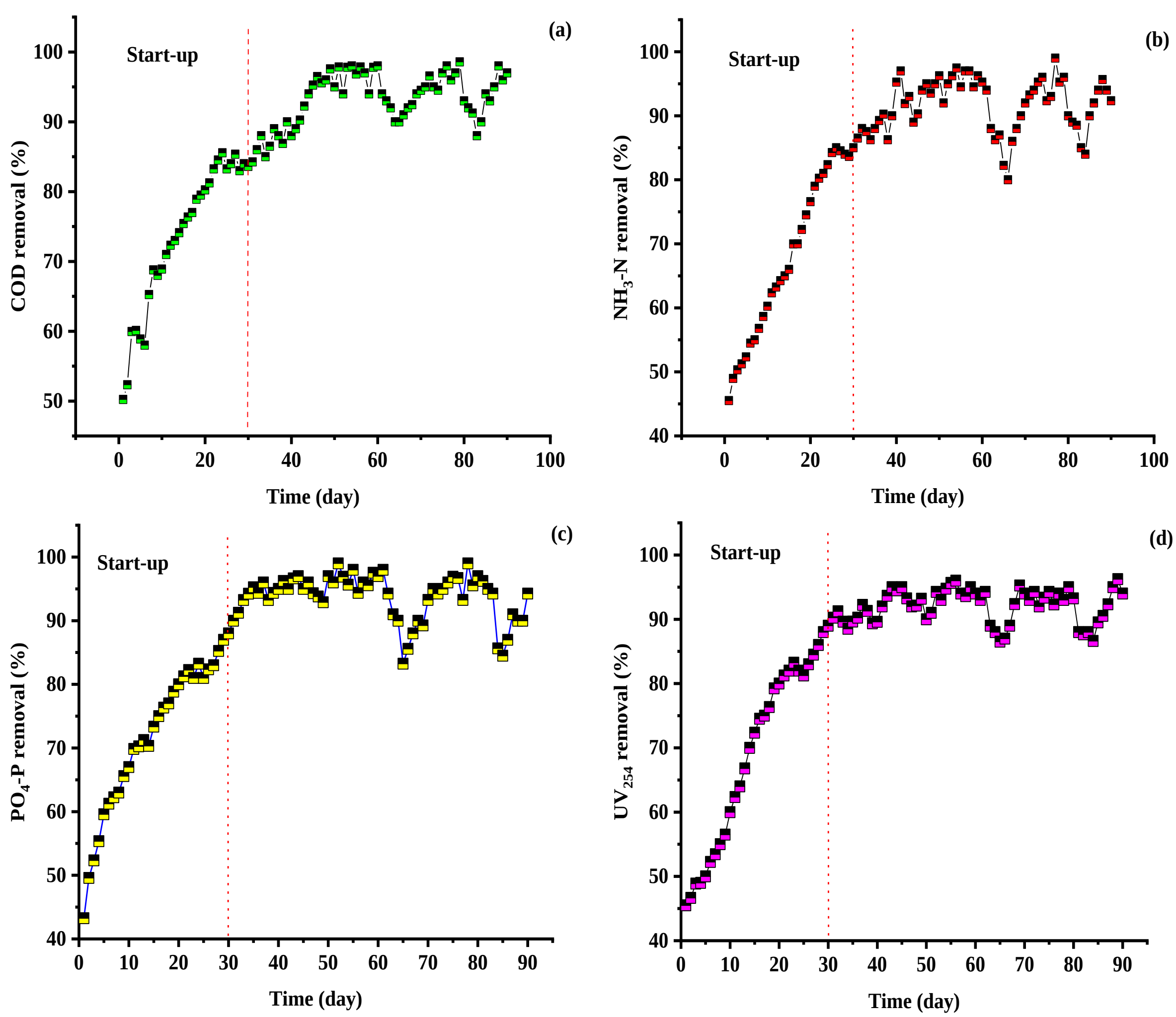

Fig. 2. Removal of COD (**a**), NH_3-N (**b**), PO_4^{3-}-P (**c**) and UV_{254} (**d**) in the bench-scale EMBR.

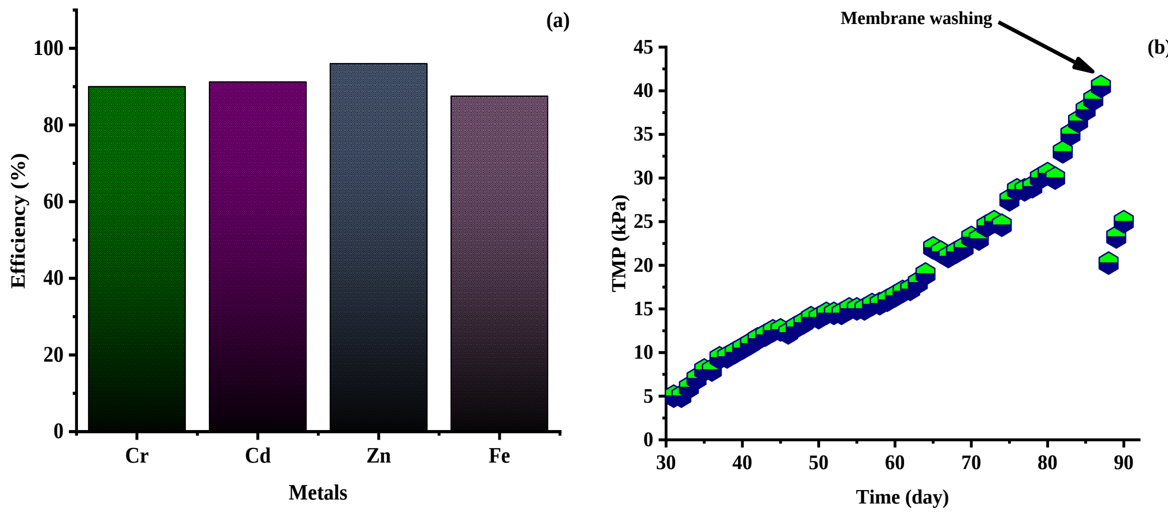

Fig. 3. Metal removal (**a**) and TMP changes (**b**) during the EMBR operation.

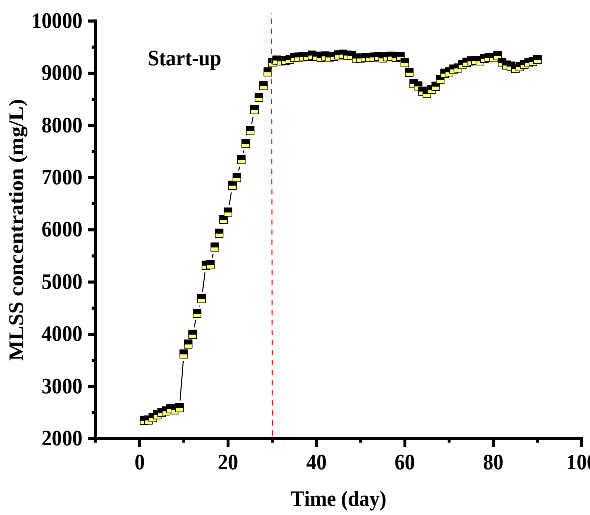

Fig. 4. The variations in sludge concentrations.

3.6. Comparison of EMBR performance for removal of contaminants

In order to evaluate the performance of the EMBR process, a comparison of the efficiency of the present process and other biological processes in leachate and wastewater treatment with different physicochemical properties is shown in Table 2. According to this table, the removal efficiency of common contaminants such as COD, NH_3-N, and PO_4^{3-}-P by the EMBR process is approximately between 80 to 99%, which is significant compared to the efficiency of the conventional MBR process. When an electric current is applied to the MBR process, the internal production of $AL(OH)_3$ as a sweeper is increased and is then led to the control of membrane clogging through 1) decomposition of organic floc, 2) control and restriction of floc movement, and 3) increase in clots settling on the surface of the membranes. In addition, electrochemical processes often increase the degradation efficiency of pollutants by inducing three mechanisms, namely electrocoagulation with direct/indirect oxidation, electrophoresis (negatively charged organic flocs move to positively charged electrodes), and electroosmosis (application of current facilitates removal of water adhering to activated sludge). The physicochemical properties of wastewater vary depending on the type of production source and mostly contain a wide range of anions such as chloride, sulfate, phosphate, and carbonate ions. The anion oxidation anode in the EMBR produces strong oxidants such as free chlorine, hypochlorous acid, hypochlorite ions, chlorate ions, persulfate radicals, percarbonate radicals, and perphosphate radicals. These oxidants can degrade organic compounds depending on the physicochemical properties of the contaminant and the solution chemistry (e.g., pH and temperature). Table 2 also shows that the EMBR process has a higher removal efficiency than the SBR and MBR process. This event may be due to the occurrence of electrostatic absorption mechanisms, direct adsorption, and precipitation. In these mechanisms, the adsorption between organic compounds and coagulant floc is based on the presence of negative and positive charge levels in the floc. In addition, the cake formed on the surface of the membranes prevents the passage of metal ions and organic matter into the effluent.

Table 2. Comparison of the EMBR performance with other processes in leachate and wastewater treatment.

Type of solution	Type of method	Characteristics	Operational conditions	Efficiency removal	Reference
Gray water	EMBR MBR	pH= 7.5, COD= 463 mg/L, NH_3-N= 11.3 mg/L, PO_4^{3-}-P= 0.53 mg/L	HRT= 24 d, SRT= no sludge waste, DO= 1.63 mg/L, Current= 12 V	COD = 88.9% NH_3-N = 77.8% PO_4^{3-}-P = 94.3% COD = 86% NH_3-N = 97.4% PO_4^{3-}-P = 65%	[37]
Tannery wastewater	EMBR MBR	pH = 7.4, COD = 1600 mg/L, Color = Black	HRT= 10–11 h, CD = 15 mA/cm^2	COD = 90.2% Color = 92.75% COD = 72.69% Color = 75.82%	[38]
Municipal wastewater	EMBR	COD =316 mg/L, NH_3–N =43 mg/L, PO_4^{3-}-P = 4.1 mg/L	HRT= 11 h, CD = 12 mA/cm^2, SRT = 10 d	COD = 92% NH_3–N =99% PO_4^{3-}-P = 99%	[31]
Leachate	SBR-EC	pH= 8.34-8.44, COD= 3530-6420 mg/L, NH_3-N = 958-1403 mg/L, Color= 3800-4510 PtCO	HRT= 1 d, SRT= 30 d	COD ≈ 84.89 NH_3-N = 94.25% Color = 85.81% As = 34.8% Zn= 41.3% Cd=95% Cu= 95.3	[39]
Leachate	MBR	pH= 8.4, COD= 3350 mg/L, NH_3-N= 693 mg/L, PO_4-P = 38.6 mg/L	SRT= 140 d	NH_3-N = 83%	[40]
Leachate	MBR SBR	pH= 7.2, COD= 2200 mg/L, NH_3-N= 210 mg/L, PO_4^{3-}-P = 38.6 mg/L	SRT= 140 d	COD = 84.54% NH_3–N =98.85% PO_4^{3-}-P = 98% COD = 77.27% NH_3–N =99% PO_4^{3-}-P = 82.12%	[41]
Leachate	EMBR	pH= 7.6, COD= 3100 mg/L, NH_3-N = 450 mg/L, PO_4^{3-}-P $^-$= 10.8 mg/L	HRT= 18 h, SRT= 60 d, CD= 0.5 mA/cm^2	COD ≈ 98.5 NH_4^+ = 99% PO_4^{3-}-P = 99% Color ≈ 95% Metal >85%	This study

4. Conclusions

This study evaluated a novel EMBR system for leachate treatment. The COD and NH_3-N removal efficiencies were above 98% in the EMBR treatment system. The PO_4^{3-}-P removal was significantly higher than MBR, and the maximum efficiency was 98%. This efficiency improvement was due to the effect of current density on the release of Al^{3+} from the anode surface and the subsequent formation of $Al(OH)_3$ for phosphorus adsorption. Following the application of a current density of 0.5 mA/cm^2, the removal efficiency of UV_{254} reached to values more than 96%, which may be related to the occurrence of electrocoagulation, electrooxidation, membrane filtration, and biodegradation mechanisms. The high removal of heavy metals from leachate by the EMBR system occurred by increasing solution pH, MLSS, and $Al(OH)_3$ concentrations. The membranes were withdrawn from the system for washing only once during the 90-day operation period. According to these results, leachate treatment by the EMBR was satisfactory in terms of pollution removal and fouling control.

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