PERFORMANCE OF MEMBRANE BIOLOGICAL REACTOR FOR TOBACCO WASTEWATER TREATMENT

by

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Declaration of Authorship

I declare that this thesis is my own work and, to the best of my knowledge and belief, it does not contain material published or written by a third party, except where permission has been obtained and/or appropriately cited through full and accurate referencing.

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Dedication

To my idols in this life, my parents…

Abstract

As technologies develop and populations grow, the demand for water sources increases. To keep up with such demand, the need for innovative methods for water regeneration becomes necessary. Amongst these methods is the utilization of Membrane Biological Reactors (MBRs) for wastewater treatment. This work discusses MBR technology for tobacco wastewater treatment and the factors that affect its performance. Currently, tobacco wastewater treatment methods reveal high time requirement, complexity, and cost. The study revealed that MBRs can effectively remove 80-90% of Chemical Oxygen Demand (COD), 88-94% of Biological Oxygen Demand (BOD_5), and 88-91% of total organic carbon (TOC) content from wastewater produced by the tobacco industry while operating at a pH of 6 – 8 with a hydraulic retention time (HRT) of 1 day. The process was favorable in treating tobacco wastewater at temperatures ranging between 20 to 40 ºC. Additionally, the study revealed that the design is able to treat wastewater with a COD concentration of 2000 ppm given that the HRT is to be increased. Moreover, the design was successful in decreasing the turbidity by 97% and TDS by over 85% even though it incorporates microfiltration. Furthermore, the results showed better membrane performance with a low fouling propensity, which indicates the suitability of ceramic membranes for installment within the system. Finally, the study compared the effect of membrane pore size by observing the difference in performance between 0.1 and 0.3 μm nominal pore sizes. This change in nominal pore size had minimal effect on the performance of the MBR. Overall, the system showed great potential and applicability to be adapted for the treatment of tobacco wastewater.

Keywords: **MBR; wastewater treatment; tobacco; membrane performance**

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List of Abbreviations

ADMBR	Aerobic Dynamic Membrane Biological Reactor
AEM	Anion Exchange Membrane
AFMBR	Aerobic Fluidized Membrane Bioreactor
AMBR	Aerobic Membrane Biological Reactor
AnMBR	Anaerobic Membrane Biological Reactor
BOD	Biological Oxygen Demand
BOD_5	Biological Oxygen Demand Measured over 5 Days
CAS	Conventional Activated Sludge
CEM	Cation Exchange Membrane
COD	Chemical Oxygen Demand
DS	Draw Solution
EPS	Extramolecular Polymer Substance
F/M	Food to Microorganism
FO	Forward Osmosis
GO	Graphene Oxide
HRT	Hydraulic Retention Time
MBR	Membrane Biological Reactor
MC	Membrane Crystallization
MCRT	Mean Cell Residence Time
MD	Membrane Distillation
MF	Microfiltration
MFC	Microbial Fuel Cell

MLSS	Mixed Liquor Suspended Solids
MPBR	Membrane Photobioreactor
NF	Nanofiltration
NSAID	Non-Steroidal Anti-Inflammatory Drugs
ODMP	Osmotically Driven Membrane Process
OLR	Organic Loading Rate
OMBR	Osmotic Membrane Biological Reactor
P&ID	Piping and Instrumentation Diagram
PAC	Powdered Activated Carbon
PEM	Proton Exchange Membrane
PRO	Pressure Retarded Osmosis
PTFE	Polytetrafluoroethane
PV	Pervaporation
PVDF	Polyvinylidene Fluoride
PVP	Polyvinylpyrrolidone
RAKWA	Ras Al Khaimah Wastewater Agency
RO	Reverse Osmosis
SC	Sludge Concentration
SCOD	Soluble Chemical Oxygen Demand
SMP	Soluble Microbial Product
SRT	Sludge Retention Time
TDS	Total Dissolved Solids
TMP	Transmembrane Pressure
TOC	Total Organic Carbon

TPV	Thermopervaporation
TSS	Total Suspended Solids
UF	Ultrafiltration
WHO	World Health Organization

Chapter 1. Introduction

1.1. Overview

The demand for water has increased by 600% since 1918, which corresponds to a 1.8% incremental annual increase [1]. This could be traced back to the exponential population growth and industrial expansion to satisfy the current development requirements. Recently, at global level 4600 km^3 of water has been in demand annually and it is expected to reach 6000 km^3 per annum by 2050; however, the greater concerns are associated with the availability of sources. With the current demand, the supply of water is close to a maximum, and yet there are about 1.9 billion people living in potentially severe water-scarce conditions. Due to the excessive consumption for industrial development and the coincident water pollution, it is expected that by 2050, 3.2 billion people will be endangered by water scarcity [1]. Thus, at this alarming rate, it is now more important than ever to develop and employ processes and techniques that can treat the wastewater, industrially or domestically produced, to supply freshwater that can meet the ginormous demand.

For many decades, wastewater has been treated by employing conventional methods [2]. In this scenario, raw sewage is pre-treated through screens to remove solids such as rags, sticks, and debris. Then, the water undergoes primary treatment to remove suspended particles through flocculation and sedimentation. Later, the effluent is introduced to the secondary treatment to remove the present organic matter. Beyond this stage, the water is safe for discharge into water reservoir, or to be used for irrigation. If a higher level of purity is desired, the discharge can undergo tertiary treatment, which involves membrane technology such as reverse osmosis (RO) [3].

Although the conventional treatment method yields water with desired purity, it is slow and requires large areas [2]. Therefore, it is of necessity to develop technologies that is rapid and efficient for wastewater treatment while utilizing relatively small land space. Amongst these technologies are membrane biological reactors (MBRs), which involve submerged membranes that strict the passage of contaminants present in wastewater [4] [5]. In essence, they replace the secondary treatment stage and eliminate the need for a tertiary stage as they validate microfiltration or ultrafiltration. In this case, as that of the conventional treatment, raw sewage undergoes pretreatment and primary treatment

stages to remove suspended solids. Then the effluent is sent to an aeration tank to assist in the growth of the microorganisms producing activated sludge, which enhances the filtration process. Then, the effluent passes a membrane where the separation process takes place, achieving high removal levels of chemical oxygen demand (COD) and biological oxygen demand (BOD).

Since the development of MBRs in 1969, numerous studies were conducted investigating their feasibility. Tadkaew et al. [6] investigated MBRs' ability to remove trace organics utilizing a submerged membrane within the bioreactor. A synthetic wastewater comprised of glucose, peptone, KH_2PO_4, $MgSO_2$, $FeSO_4$, and sodium acetate was used. Fourty organic compounds were added including pesticides, endocrine disrupting chemicals, steroid hormones, and pharmaceutically active chemicals. The study revealed 85% - 98% removal efficiency of hydrophobic trace organic compounds whereas a removal efficiency below 20% was observed for the hydrophilic trace organic compounds [6].

MBRs potential caught the pharmaceutical industry's attention. In a study conducted by Sipma et al. [7], MBRs capability of removing pharmaceuticals was compared to conventional activated sludge (CAS) systems. The study revealed that MBR has similar removal efficiency to CAS when it comes to easily removed pharmaceuticals such as ibuprofen. Additionally, it was shown that the removal efficiency of pharmaceuticals was better in MBRs. Overall, MBRs showed better removal operations compared to CAS except for few chemicals such as sotalol and hydrochlorothiazide [7].

Tambosi et al. [8] tested MBRs performance for the removal of non-steroidal anti-inflammatory drugs (NSAIDs) and antibiotics. Utilizing a submerged membrane, their experiments revealed high removal efficiency of NSAIDs ranging between 86% to 100% with a sludge retention time (SRT) of 15 days. Meanwhile, antibiotics removal efficiency ranged between 55% to 86%. When the SRT was increased to 30 days, the treatment process resulted in the removal of 89% to 100% of NSAIDs while the antibiotics removal efficiency ranged between 64% to 94% [8].

In a study by Tan et al. [9], the performance of MBRs for the removal of microorganisms present in saline wastewater was reviewed. According to the authors, the removal efficiencies of COD and ammoniacal nitrogen in a conventional MBR were

93% and near 100%, respectively. In addition, the study suggests seeding the wastewater with halophilic microorganisms. The removal efficiencies of total organic carbon (TOC) and ammoniacal nitrogen were found to be 98% and 95%, respectively. This is due to enhanced bioactivity provided by the microorganisms [9].

Membrane Biological Reactors (MBRs) provide a fast and reliable method for water regeneration [10] [11]. They offer a sustainable, low foot-print process that can produce water with high quality while discarding the need for large secondary clarifiers and disinfection processes [12]. MBRs combine the primary wastewater treatment stage with a membrane technology to achieve their desired product. Meanwhile, the only drawback that prevents decision makers from fully adopting MBRs is the membrane fouling, an essential issue in membrane applications [13] [14]. However, with the recent scientific developments and the better understanding of membrane preservation strategies, MBRs become a leading solution to an urgent crisis [15] [16] [17] [18].

1.2. Thesis Objectives

Tobacco manufacturing is a water-intensive process. According to a study conducted by the World Health Organization (WHO) in 2022, tobacco production globally consumes 22 billion tonnes of water annually [19]. The water is mainly utilized for tobacco pulp processing, dry iced expanded tobacco treatment, and dye production for packaging while the discharged wastewater contains toxic substances namely nicotine, heavy metals, and volatile organic hydrocarbons [20]. The current tobacco wastewater treatment methods reveal high time requirement, complexity and cost [21] [22]. Therefore, this study aims to test the performance of MBR technology in tobacco wastewater treatment as a sustainable treatment method by achieving the following:

- Investigating MBRs effectiveness for tobacco wastewater treatment.
- Identifying parameters that enhance MBR's performance.
- Studying the effects of membrane fouling on the effluent and explore optimum control strategies.
- Studying the effect of altering operating conditions on the effluent.

1.3. Thesis Classification

Following this section, this thesis presents the following: Chapter 3 provides a background on membrane technology, types of membranes, and fouling phenomena.

Additionally, it discusses MBRs, reveals their significance, and showcases recent development in MBRs. Chapter 4 defines the scope of work and its contribution to academia and the industry. Chapter 5 highlights the methodology and the design utilized to perform the experimental work. Chapter 6 illustrates the results obtained, justifies them, and shows their significance by comparing them to the literature. Finally, Chapter 7 concludes this thesis while providing recommendations on how to implement the design better and improve its efficacy.

Chapter 2. Background and Literature Review

2.1. Membrane Technology

A membrane is a permeable or semi-permeable material that defies the passage of certain species [12]. It yields two products; the permeate, which contains the species that have passed across the membrane, and the retentate, which comprises the species that the membrane rejected. Two factors govern the performance of a membrane, namely flux and retention. The flux is the flowrate of the fluid that passes through the membrane per unit area of the membrane. The retention is the fraction of solute in the feed that has been separated by the membrane. It can be estimated by a parameter called selectivity, which evaluates the flowrates of different species across the membrane. Therefore, an ideal membrane is optimized by maximizing both the flux and the retention [12].

2.1.1. Types of membrane separation processes

The uniqueness of membranes rises from creating a phase for the separation process. This phase can be solid, liquid, or gaseous and acts as an interface between the permeate and the retentate. The phase is electrically charged or neutral and can be porous or non-porous depending on the properties of the species to be separated [12]. To achieve the separation, a driving force is implemented across the membrane in the form of a pressure, concentration, or temperature gradient. Some membrane applications like electrodialysis require an electrical potential to drive the separation process [23].

Pressure driven membrane processes

Pressure driven membrane processes are the most prominent separation methods in membrane technology. Their significance arises from the application of hydraulic pressure to fasten the separation process [23]. Pressure driven membranes are microporous in nature, and accordingly, they are classified by the size of their pores. Figure 2-1 shows the different types of pressure driven membrane and their applications relevant to their pore size distributions. The membranes classes are:

- Reverse osmosis (RO): Their pore size ranges from 0.0001 - 0.001 microns [23]. They are prominently used in water desalination as the pores are smaller than sodium chloride, but larger than water.

- Nanofiltration (NF): Their pore size ranges from 0.0002 – 0.002 microns [24]. Their pores reject divalent molecules such as calcium carbonate, which validates their purpose for water softening. Additionally, they are vital in removing heavy toxic metals such as mercury, arsenic, and lead.
- Ultrafiltration (UF): Their pore size ranges from 0.005 – 0.1 microns [25]. Their pores prevent the passage of microorganisms including bacteria, viruses, and parasites.
- Microfiltration (MF): Their pore size ranges from 0.1 – 10 microns [26]. They are mainly used in food processing industries as they validate the removal of water borne parasites such as Protozoa and Giardia.
- Pervaporation: They utilize a non-porous membrane, which allows the passage of species that are soluble in the membrane. They are widely used in the extraction of water and small volatile molecules.

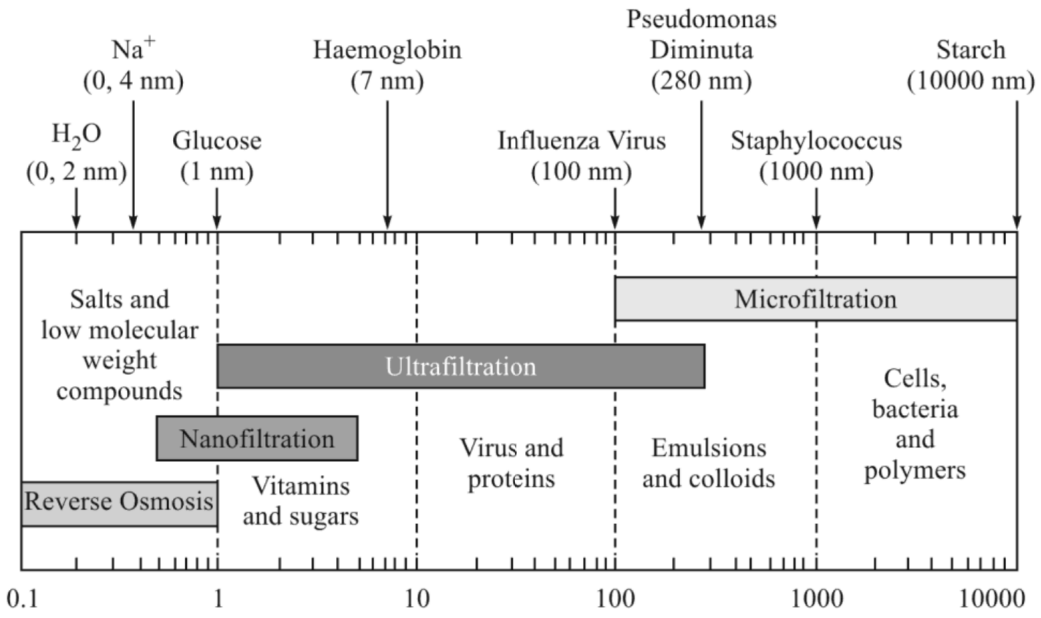

Figure 2-1: Pore size distribution of pressure driven membranes [23].

Concentration driven membrane processes

Concentration driven membrane processes, commonly known as osmotically driven membrane processes (ODMPs), rely on the difference in concentrations across a membrane to generate a flux [27]. Their utilization is commonly observed in processing water for treatment and power generation under the terms forward osmosis (FO) and pressure retarded osmosis (PRO) respectively. Both processes stimulate mass flow by

employing the osmotic pressure difference between a dilute feed solution and a concentrated draw solution, or brine, as schematically shown in Figure 2-2.

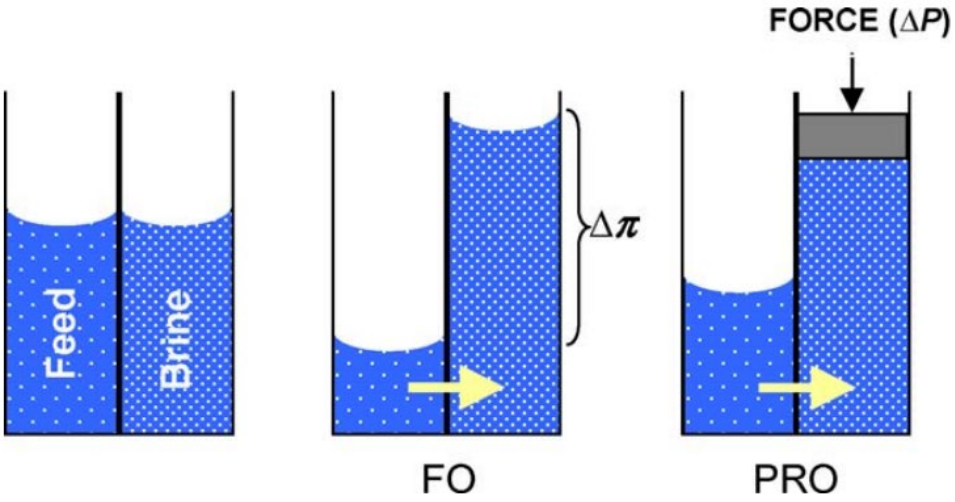

Figure 2-2: Solvent flows in FO and PRO [27].

The flux for the ODMPs can be calculated using the following equation:

$$J_w = A(\sigma\Delta\pi - \Delta P) \quad (1)$$

where $J_w$is the water flux, A is the water permeability constant of the membrane, σ is the reflection coefficient, $\Delta\pi$ is the osmotic pressure difference, and ΔP is the applied pressure [28]. In FO, a non-porous semipermeable membrane separates the feed solution from the draw solution. As the water transports across it, the contaminants are rejected by the membrane. The draw solution is then reconcentrated and recycled to continue the separation process. PRO applies the same principle; however, a turbine is installed near the increased pressure flow to produce hydroelectricity.

The issue arising with ODMPs is the salt build up at the membrane's surface revealing the concentration polarization phenomenon. This phenomenon creates a new concentration gradient that leads to a reverse flux, which hinders the separation process by lowering the water flux and blocking the membrane. However, studies have shown that ODMPs performance relies heavily on membrane selection [27]. It was concluded that highly dense, thin, minimally porous, and highly mechanically supported membranes optimize the performance of ODMPs as they achieve high solute rejection, low crystallization polarization, high solvent flux, and high sustain to hydraulic pressure.

Temperature driven membrane processes

Temperature (thermal) driven membrane processes separate volatile substances from the initial solution by means of evaporation through a membrane followed by condensation on the other side of the membrane [29]. The separation process is driven by maintaining the feed at a high temperature while the permeate is at a lower temperature. Amongst the processes that utilize such principle are membrane distillation (MD), membrane crystallization (MC), pervaporation (PV), and thermopervaporation (TPV). MD is mainly used for water treatment and water desalination. It offers almost complete removal of non-volatile substances, independence from salinity levels for water transport, and results in the elimination of disinfection stages. MC promotes a controlled pathway for nucleation. The porous membrane allows the passage of the desired species while the rate of supersaturation is well-controlled on the other side of the membrane [30]. PV and TPV use a dense non-porous membrane, which favors the absorption of selective species [29]. They are mainly used for the extraction of volatile organic compounds.

Electrically driven membrane processes

Electrically driven membrane processes rely on electric potential to drive the separation process [31]. Therefore, their applications are limited to the separation of charged substances based on the segregation of positively and negatively charged ions. For instance, they are used to remove salts from seawater, and they are currently being widely studied to be employed in hydraulic fracturing. The membrane allows the passage of cations or anions, each attracted to the oppositely charged terminal of the electric potential source as shown in Figure 2-3.

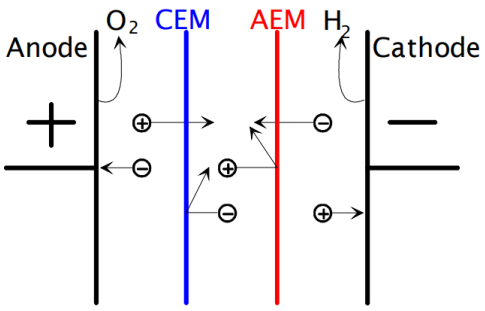

Figure 2-3: Schematic of an electrically driven membrane process. CEM: cation exchange membrane, AEM: anion exchange membrane [31].

2.1.2. Types of synthetic membranes

Membranes and filters operate in a similar manner; however, the latter are commonly defined as media that separate suspended particulates larger than 1 to 10 microns. Because membranes serve wider applications, they can be classified into five types. These are briefly described below.

Microporous membranes

These are similar in structure to filters. Microporous membranes are rigid structures with randomly distributed pores [23]. However, the pores are extremely small with diameters ranging from 0.01 to 10 microns. Their operation principle is simple. Particles that have a diameter larger than the membrane pores are rejected, thus achieving the separation target. There are 2 types of microporous membranes: isotropic and anisotropic. In isotropic membranes, the pores have the same size while anisotropic membranes' pores differ in size throughout the membrane's structure.

Asymmetric membranes

Also known as skinned membranes. Asymmetric membranes are characterized by a thin skin on the surface of the membrane with a thickness of 0.1 to 1 micron [23] [32]. The skin is supported by a layer consisting of voids. The separation process is not only dependent on the membrane's pores size but is also governed by the nature of the membrane material. There are 2 types of asymmetric membranes: integrally and non-integrally skinned. In integrally skinned, the skin layer is porous and produced from a phase inversion process, while the skin layers in non-integrally skinned, membranes are homogenous and deposited from solution.

Thin film composite

Primarily developed for RO and NF applications. Thin film composites are constructed from a thin dense polymer skin supported by a microporous film [33]. The pores of the microporous film have a diameter smaller than the thickness of the skin layer for enhanced adhesion. Thin film composites were revolutionary for RO applications as they provided better performance in terms of pH stability, pressure, temperature, and salt rejection compared to cellulose acetate membranes that were used priorly; however, they have a higher tendency to biofouling [23].

Electrically charged membranes

Electrically charged membranes are constructed from gels carrying a positive or negative charge. The positively charged membranes are called anion exchange membranes (AEMs) whereas the negatively charged membranes are called cation exchange membranes (CEMs) [23] [34]. Each charged membrane attracts either oppositely or negatively charged ions. Charged membranes can be porous or non-porous, but mostly, they consist of very fine micropores.

Inorganic membranes

Also known as ceramic membranes. Inorganic membranes are designed to operate at high temperatures and in severe conditions. Inorganic membranes can operate at temperatures beyond 1000ºC and are resistant to corrosive liquids and gases [23] [35]. Their structure is usually asymmetrical with 2 or 3 different porosity levels. A mesoporous layer is applied below the active microporous layer to decrease the surface roughness.

2.2. Membrane Fouling

Membranes reveal efficient solutions to many industrial problems. However, membrane fouling is the primary challenging issue that restricts their applications and hinders their extension in some situations. Fouling depends on many factors including membrane's nature, influent properties, operating conditions, and membrane cleaning methods.

2.2.1. Fouling phenomena

Fouling is the blockage of solute molecules to the membrane's pores, which impedes the flow of the solvent. The decrease of the permeation flux or the increase in transmembrane pressure (TMP) [13] can be considered as indicators or measures of fouling. In general, TMP is a better indicator as most processes are conducted at constant flux.

During fouling, TMP is observed to follow two patterns: a two-stage jump and a three-stage jump. The two-stage jump pattern illustrates at first a slight increase in TMP caused by the adsorption of particles into the membrane's pores while retaining the local flux below the critical flux. When the local flux exceeds the critical flux, a sharp

jump is observed in the TMP. This indicates that the pores of the membrane have been blocked and the membrane requires cleaning. The second TMP pattern, the three-stage jump, shows a rapid and small jump in TMP due to the rapid obstruction of pores in the first stage. The second and third jumps follow the same trend as the two-stage TMP jump. In the second stage, the TMP increases at a steadier-slower rate until the critical flux is exceeded for which the third stage is observed by the sharp jump in TMP. Figure 2-4 illustrates the TMP jump patterns.

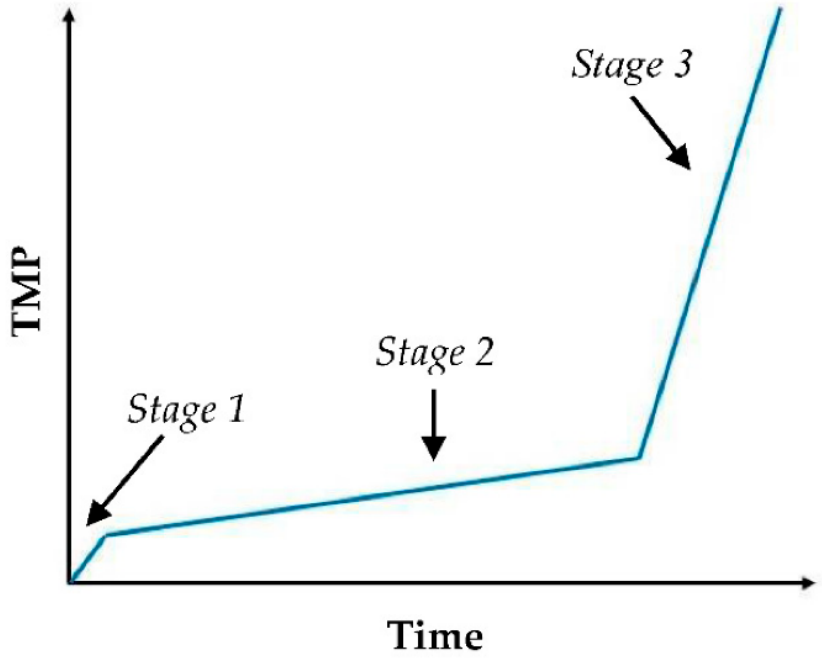

Figure 2-4: TMP's three-stage jump pattern [36].

2.2.2. Fouling rate

Membranes are fouled on stages. First, the smallest pores are blocked. Then, molecules start adhering to the inner surfaces of the pores. Later, flowing molecules start aggregating over the molecules inside the pores. Finally, a cake layer is formed. These stages are illustrated in Figure 2-5. Characterizing each stage is difficult; hence, the overall fouling propensity is quantified using the fouling rate. The fouling rate can be expressed as the TMP per time. The operating flux and the fouling rate are proportional i.e., as the operating flux increases, the fouling rate increases until the critical flux is achieved where the fouling rate abruptly increases.

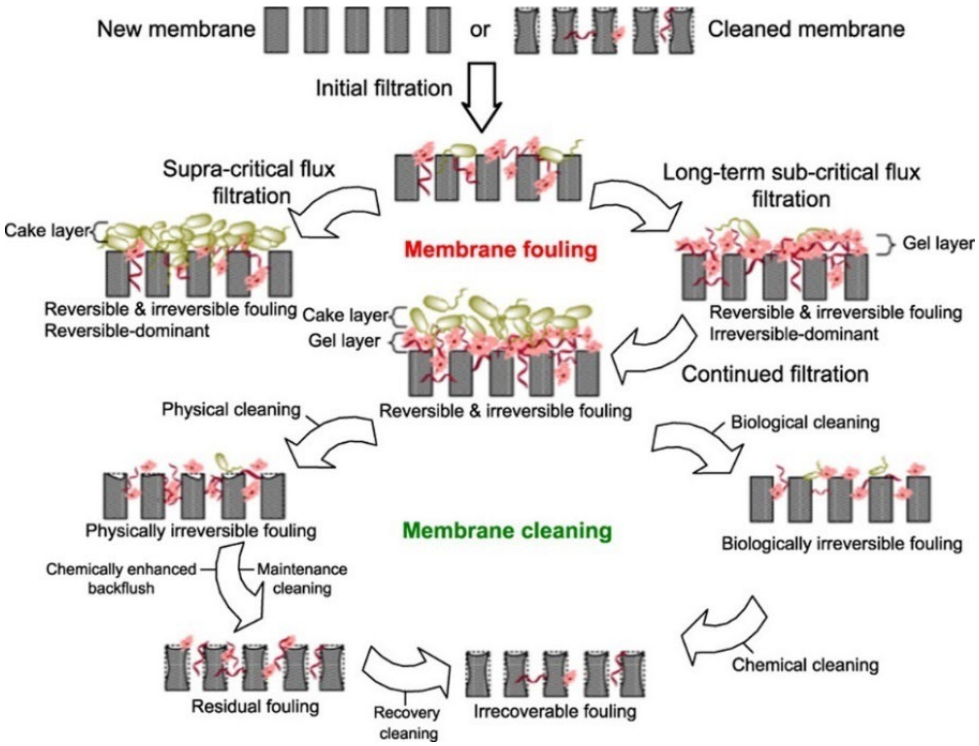

Figure 2-5: Membrane fouling and cleaning stages [14].

2.2.3. Classification of fouling

Fouling can be classified based on the flux recovery after cleaning, fouling location, and solid deposition pattern [13]. This is shown in Figure 2-6 [37]. Flux recovery after cleaning describes the act of restoring the flux after a single cleaning method [14]. It can be categorized as reversible, irreversible, or irrecoverable fouling. As the term indicates, the flux of reversible fouling can be readily recovered after a simple physical cleaning stage whether it is backwash, pressure relaxation, or air scouring. When a strong matrix of the fouling layer is formed, the reversible fouling is converted into irreversible fouling. Irreversible fouling requires biological cleaning as physical cleaning methods are insufficient. Irrecoverable fouling refers to the case when the membrane is fouled on a long-term operation and its permeability cannot be recovered.

Fouling location describes the shape format of the fouling on the membrane. The form can be either clogging, cake layer, or internal pore. Poorly designed membrane modules can lead to clogging. As the particles accumulate inside the pores, the flow is hindered and the TMP is increased. A cake layer is formed when the particles adhere to each other on the membrane's surface, which enhances the permeate purity as it forms a secondary filter; however, it also decreases the flux and impedes the separation process.

The third form, or the internal pore fouling, describes the adhesion of particles on the internal pores' walls. Nonetheless, cake layer formation is the most dominant in describing fouling [13].

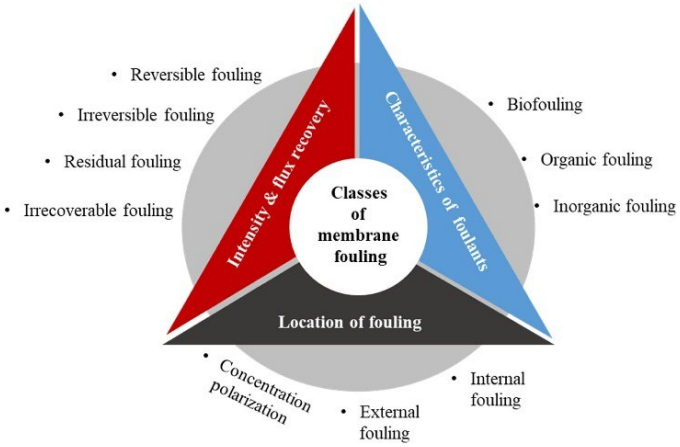

Figure 2-6: Classification of fouling [37].

Solid deposition pattern describes how solids adhere to the membrane. In principle, this classification is very similar to fouling location as it consists of cake layer formation, pore narrowing, and pore plugging. Pore narrowing refers to the adhesion of particles on the inner pore walls as their diameters are narrower than the pore's whereas pore plugging suggests the blockage of the pores as the particles have a wider diameter [13].

2.2.4. Factors affecting membrane fouling

There are three main factors that contribute greatly to fouling in MBRs, namely membrane characteristics, operating conditions, and microbial characteristics. These are summarized in Figure 2-7 [37].

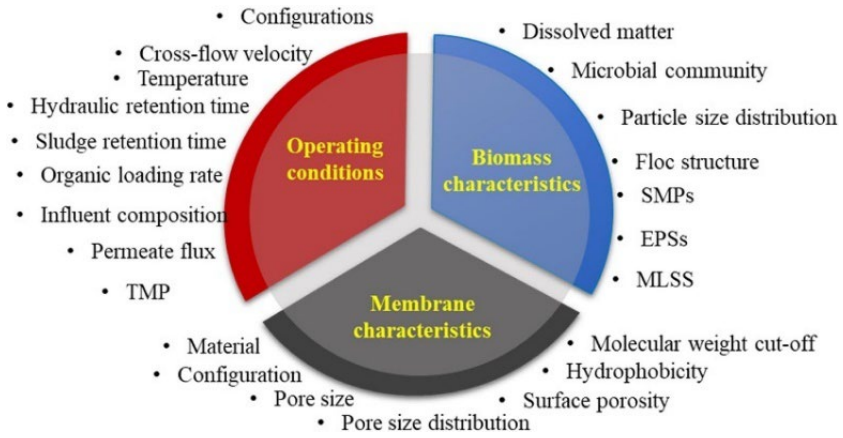

Figure 2-7: Factors affecting membrane fouling [37].

Membrane characteristics

Membrane pores are designed slightly larger than the size of molecules desired to pass through. Usually, hydrophilic membranes are preferred over hydrophobic membranes for such applications as wastewater treatment. This is because hydrophilic membranes tend to strongly interact with the constituents of wastewater, which in turn enhances the separation rate. However, a high fouling rate is observed in addition. Furthermore, the membrane type plays a significant role in reducing fouling. Most commercial membranes are polymeric, which means that they cannot withstand the severe conditions accompanying wastewater such as high temperatures or extreme pH values. As mentioned earlier, inorganic membranes are better for such foreseen conditions; however, they are costly, and their configurations are limited. For wastewater treatment, higher packing density is preferred while inorganic membranes provide low packing density modules [38] [39].

Operating conditions

One of the most important parameters for wastewater treatment is the hydraulic retention time (HRT). It is defined as the ratio of the size of the wastewater treatment reactor to the inlet volumetric flowrate, or in simpler terms, it is the time the wastewater spends inside the treatment reactor. Higher HRT means a longer contact time between the present microorganisms and the substances in the wastewater. It is also related to the food to microorganism ratio (F/M). As HRT increases, F/M decreases, which basically means less flow rate and thus less food available to the microorganisms. Therefore, less fouling is anticipated. In addition to HRT, the mean cell residence time (MCRT) contributes to fouling. MCRT refers to the time, which the microorganisms, or activated sludge, spend in the wastewater treatment clarifier. MCRT has a direct relation to the mixed liquor suspended solids (MLSS), which is defined as the concentration of suspended solids inside the reactor containing the activated sludge. Higher MLSS concentration could be detrimental to the membrane as it describes a slimy fluid that can easily block the membrane. It is important to mention that increasing MCRT decreases the presence of substances in the reactor, which in turn lowers the fouling propensity. Therefore, it is necessary to optimize the reactor to a design that increases MCRT while avoiding the severity of a high MLSS concentration [40].

Microbial characteristics

Microbial presence in wastewater can be in the form of microorganisms, biomass, or dissolved biopolymers. They can be quantified by evaluating the MLSS concentration in the bioreactor. As mentioned previously, high MLSS concentration increases fouling propensity. The biomass starts accumulating on the membrane's surface increasing the cake layer thickness. However, the layer is restricted to a certain thickness due to the shear forces applied on the membrane by the aeration source [13].

2.3. Membrane Biological Reactors (MBRs)

Membrane biological reactors (MBRs) combine the main two units of the conventional secondary wastewater treatment in one membrane separation process [11]. The activated sludge reactor is operated at similar conditions to the conventional reactor in secondary treatment; however, it eliminates the need for the secondary clarifier. There are two main MBR configurations as shown in Figure 2-8. The first one is the side-stream configuration introduced by Smith et al. in 1969 [41], where an ultrafiltration membrane is placed outside the reactor to separate the treated water from the activated sludge. This was the first time such technology has been used. In such a case, the MBR produced water with an incredibly high purity; however, it consumed high energy, and the membrane fouling at that time was difficult to treat.

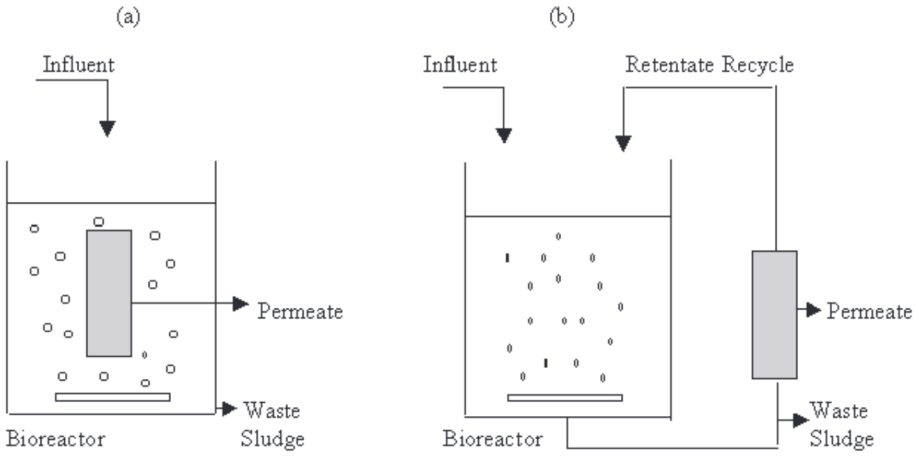

Figure 2-8: Configuration of MBRs: (a) submerged, (b) side-stream [11].

To overcome the deficiencies in MBRs, in 1989 Yamamoto et al. developed the second configuration involving a submerged membrane [42]. They installed a hollow fiber membrane inside the bioreactor, which eliminated the need for a pressurizing pump to

carry the effluent of the aeration tank to the membrane. Instead, a suction pump is installed to the membrane, removing the water in an intermit regime. Ever since, numerous studies were conducted to enhance MBRs performance by decreasing capital and operating costs and minimizing membrane fouling [13].

2.3.1. Types of MBRs

Aerobic membrane bioreactors (AMBRs)

An aerobic membrane bioreactor (AMBR) couples an aerated activated sludge tank with a membrane. AMBRs remove suspended solids by membranal rejection. Additionally, they remove dissolved chemical substances by microbial activity. For instance, carbon is removed by microbial metabolization. Ammonia is removed through nitrification and denitrification processes utilizing Nitrosomonas bacteria. Other substances including phosphate are removed similarly by microbial consumption. All these processes require oxygen; hence, the reactor is aerated [5]. AMBRs reveal great product consistency, reduced footprint, and almost complete suspended solids removal. The water produced is suitable for irrigation or can be treated to become potable. The main drawback of AMBRs is the accompanied membrane fouling.

Dvorak et al [43] have investigated AMBRs for pharmaceutical and chemical wastewater treatment. The influent contained 1898 ppm COD and was treated over an HRT of 14 days and an SRT ranging between 30 – 51 days. The experiment revealed a COD removal of 80%. It is worth noticing that the industrial wastewater was mixed with municipal wastewater to lower the initial COD and facilitate the treatment process. Overall, the system performed effectively with a maximum of 30% industrial wastewater in the influent. Ahmadi et al [44] have employed AMBRs for the treatment of oil refinery wastewater. The influent wastewater was characterized by a COD in the range of 195 – 590 ppm. In the study, the HRT was varied between 12 - 24 hr, and the MLSS was varied between 6.5 – 8.5 g/l to assess their influence on the performance. Overall, the system was capable of removing 74 – 97% of COD while the HRT had the most prominent effect on the treatment process.

Anaerobic membrane bioreactors (AnMBRs)

Anaerobic membrane bioreactors (AnMBRs) are known for their ability to provide a similar treatment quality as AMBRs, but with considerable energy conservation [45].

An AnMBR combines an anaerobic bioreactor with a membrane. The membrane can be submerged in the bioreactor or installed in a side-stream configuration; however, most designs recommend submerging the membrane in a tank external to the anaerobic reactor, and the water can be pumped to the membrane chamber. Thus, the cleaning process of the membrane will not alter the anaerobic condition of the wastewater.

Although AnMBRs are energy efficient, their deficiencies appear in the provision of an anaerobic environment [45]. Anaerobic conditions require a temperature between 30 and 60 ^{o}C [45]. Thus, AnMBRs applications were restricted to regions with mesophilic atmospheric temperatures to avoid the energy requirement for heating the wastewater. Furthermore, a long solid retention time (SRT) is required, which may result in membrane fouling due to longer use of the membrane. Increasing SRT results in an increase in biomass concentration, which decreases the permeate flux. Additionally, it increases the concentrations of soluble microbial products and extracellular polymeric substances, which in turn contribute significantly to membrane fouling. Nonetheless, AnMBRs were able to achieve the removal of more than 90% of chemical oxygen demand (COD) [46].

Wijekoon et al [47] have investigated AnMBRs for treating high strength molasses-based wastewater. The wastewater fed was at different organic loading rates from 2 – 22 kg COD/m^{3}d while the system operated a temperature of 55 ^{o}C and an HRT of 16 h. The study revealed that the COD removal was enhanced when the organic loading rate increased from 5 to 8 kg COD/m^{3}d by 3%; however, increasing the loading rate further decreases the removal rate. Ultimately, the COD removal peaked at 81% and then decreased to 61% at the highest loading rate. Similarly, the BOD_5 removal reached a high of 96% then decreased to 89%.

Lin et al [48] have studied AnMBRs for municipal wastewater treatment. The influent COD was in the range of 342 – 527 ppm and at a pH of 7 and a temperature of 30 ^{o}C. The study revealed a COD removal of 90% and a high life-cycle cost attributed mainly to membrane expenditures. Additionally, An et al [49]investigated AnMBRs for municipal wastewater treatment. The wastewater fed consisted of COD of 259.5 ppm, ammoniacal nitrogen of 27.5, and total phosphorous of 4.2. The AnMBR yielded a COD removal of 70%, 0% removal of ammoniacal nitrogen, and 24% removal of total phosphorous.

2.3.2. Factors affecting MBRs performance

The efficiency of Wastewater treatment processes is affected by the inlet wastewater properties and the treatment process conditions. For the wastewater treatment process, MBRs are mainly affected by the chemical properties of micropollutants and the hydraulic retention time (HRT).

Biokinetics

Biokinetics refer to the rate, by which microbial organisms utilize the substrates present in the wastewater to promote growth and treatment of the influent [13]. In biokinetics, the microorganisms serve as catalysts to produce biomass while reproducing themselves simultaneously. Their significance lies behind their capability to predict and estimate biological performance and treatments. Also, the reaction rate is important to determine the volume of the bioreactor required to achieve the desired biomass concentration.

Biokinetics are mainly related to the microbial growth rate and rate of substrate utilization. The microbial growth is achieved via metabolizing biodegradable substrates present in the wastewater; however, the wastewater influent also consists of substrates that are not biodegradable, which are to be removed by other techniques such as flocculation or filtration. Additionally, microorganisms tend to decay themselves during growth. Therefore, the net growth rate can be evaluated as the following:

$$R_{g,net} = R_{growth} - R_{decay} \tag{2}$$

$$R_{g,net} = \frac{dX}{dt} = \frac{\mu_m S X}{K_s + S} - k_d X \tag{3}$$

where $R_{g,net}$ is the net growth rate ($\frac{g\ VSS}{m^3.\ day}$), X is the biomass concentration ($\frac{g\ VSS}{m^3}$), S is the biodegradable substrate concentration ($\frac{g\ COD}{m^3}$), μ_m is the maximum specific growth rate (day^{-1}), K_s is the half saturation constant for biodegradable substrate ($\frac{g\ COD}{m^3}$), and k_d is the decay coefficient ($\frac{g\ VSS}{g\ VSS.\ day}$). Table 2-1 summarizes the biokinetic coefficients of different MBR plants for different bacteria.

Table 2-1: Biokinetic coefficient of different bacteria in different MBRs [13].

MBR characteristics	Biomass type	Yield coefficient	Maximum specific growth rate	Half-saturation coefficient	Decay coefficient	Ref
Submerged hollow fiber ultrafiltration MBR worked at HRT=9.5 h, T=14.7˚C, MLSS concentration =6.6 g/l.	Heterotrophic bacteria	0.4887 $(\frac{\mathrm{mgVSS}}{\mathrm{mgCOD}})$	0.0141 (h^{-1})	7.467 $(\frac{\mathrm{mgO_2}}{\mathrm{L}})$	0.0521 (day^{-1})	[50]
	Ammonium-oxidizing bacteria	1.191 $(\frac{\mathrm{mgVSS}}{\mathrm{mgN}})$	0.1612 (h^{-1})	0.2204 $(\frac{\mathrm{mgN}}{\mathrm{L}})$	NM	
	Nitrite-oxidizing bacteria	0.6473 $(\frac{\mathrm{mgVSS}}{\mathrm{mgN}})$	0.0786 (h^{-1})	0.324 $(\frac{\mathrm{mgN}}{\mathrm{L}})$	NM	
Submerged hollow fiber ultrafiltration MBR HRT=9.5 h, MLSS concentration =3.3 g/l.	Heterotrophic bacteria	0.4609 $(\frac{\mathrm{mgVSS}}{\mathrm{mgCOD}})$	0.01917 (h^{-1})	16.47 $(\frac{\mathrm{mgO_2}}{\mathrm{L}})$	Total bacteria decay =0.03043 (day^{-1})	[51]
	nitrifying bacteria	1.0389 $(\frac{\mathrm{mgO_2}}{\mathrm{mgN}})$	0.27193 (h^{-1})	0.9329 $(\frac{\mathrm{mgN}}{\mathrm{L}})$		
	nitrite-oxidizing bacteria	0.77913 $(\frac{\mathrm{mgO_2}}{\mathrm{mgN}})$	0.11244(h^{-1})	0.4364 $(\frac{\mathrm{mgN}}{\mathrm{L}})$		
Immersed MBR operated at HRT=13h, MLSS concentration=5 g/l under steady state condition.	Serratia liquefaciens and Aeromonas hydrophila (predominant bacteria)	0.567 $(\frac{\mathrm{mg}}{\mathrm{mg}})$	0.0233 (h^{-1})	326.14 $(\frac{\mathrm{mgCOD}}{\mathrm{L}})$	0.062 (day^{-1})	[52]
Microfiltration MBR operated at HRT= 33h , T=25˚C.	heterotrophic biomass	0.756 $(\frac{\mathrm{mgCOD}}{\mathrm{mgCOD}})$	3.687 $(\frac{\mathrm{mg}}{\mathrm{g.h}})$	NM	0.353 (day^{-1})	[53]
Moving bed biofilm reactor combined with MBR operated at HRT=26.47h, T=15˚C, and MLSS concentration=2.9 g/l.	heterotrophic biomass	0.5041 $(\frac{\mathrm{mgVSS}}{\mathrm{mgCOD}})$	0.00484 (h^{-1})	0.96 $(\frac{\mathrm{mgO_2}}{\mathrm{L}})$	Total bacteria decay =0.04844 (day^{-1})	[54]
	autotrophic biomass	0.77718 $(\frac{\mathrm{mgO_2}}{\mathrm{mgN}})$	0.02632 (h^{-1})	0.76 $(\frac{\mathrm{mg}N}{\mathrm{L}})$		
MBR plant with hollow fiber membrane	heterotrophic biomass	0.703 $(\frac{\mathrm{gVSS}}{\mathrm{gCOD}})$	NM	NM	0.02 (day^{-1})	[55]

operated at HRT=8h, T=27°C, and MLSS concentration= 1.3 g/l.						
Bioreactor plant by using glucose as substrate at T=20°C.	shewanella baltica KB30	0.6681 $(\frac{mgVSS}{mgCOD})$	0.0840 (h^{-1})	1.608 $(\frac{mgO_2}{L})$	NM	[56]

Hydraulic retention time (HRT)

HRT refers to the time spent by the wastewater inside the bioreactor. As mentioned earlier, shorter HRT reduces membrane fouling propensity, which enhances MBRs' operations. In a study conducted by Isma et al. [28], synthetic wastewater was treated using an AMBR. The process was set at a solid retention time of 30 days while the HRT was varied as 4, 8, and 12 hours. The study revealed that 12 h HRT had the highest removal efficiency for COD, BOD_5, total suspended solids (TSS), NH_3-N, and PO_4^{3-} as 95%, 93%, 99%, 97%, and 81%. Regardless of the value, all HRT estimates showed a removal efficiency above 90% except for PO_4^{3-}, which revealed 62% and 68% removal percentages for 4 and 8 h HRT respectively [15].

Chemical properties of micropollutants

The chemical nature of micropollutants contributes greatly to their removal. The micropollutants found in wastewater are hydrophobic, which means that they repel from water. Hydrophobic pollutants tend to adsorb on the sludge, suspended solids, and particulate matter during the treatment process. This aids the treatment process as the priorly mentioned matter either settle in the clarifier or are rejected by the membrane [57].

2.3.3. Recent developments in MBRs

Integrated microbial fuel cell – membrane biological reactor (MFC-MBR) system

Microbial fuel cells (MFCs) are devices that generate an electric current from bacteria's oxidation of different organic compounds [58]. The fuel cell is set up incorporating an anode where the organic substrates are oxidized producing protons and electrons. The

electrons transport through an external circuit towards the cathode generating a current while the protons transport through a proton exchange membrane (PEM) to the cathode. At the cathode, the protons are oxidized by oxygen producing water.

MFCs have potential in wastewater treatment processes as they convert biomass into an energy source; however, they are still under development, and many cost and technical aspects are hindering their progress. One of the approaches that enhances MFCs performance is to integrate them into an existing wastewater treatment process. Arising from that principle is the MFC-MBR system where the bioreactor is used as the cathode chamber. In a study conducted by Wang et al. [30], an MFC excluding a PEM was set up and submerged into the aeration tank of an MBR [16]. The study revealed that the MFC did not affect the MBR performance. The aeration tank favoured the production of a biofilm on the cathode, which enhances oxygen's function as an oxidizer. All of which contribute greatly to the generation of sufficient current. However, most importantly is the cost used to manufacture the MFC. The high cost of a PEM causes the undesirability of the MFC. The model developed by Wang et al. [30], shown in Figure 2-9, does not require a PEM, which makes the integration cost effective. Therefore, MFC-MBR systems reveal a high potential for both the generation of sustainable energy and the treatment of wastewater.

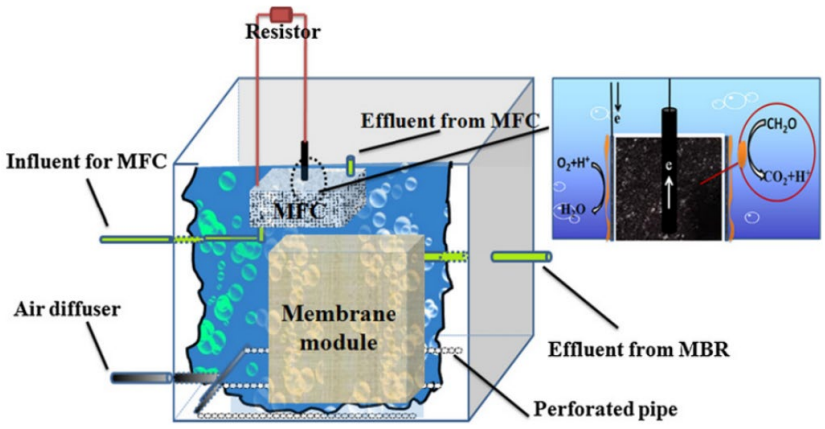

Figure 2-9: Schematic of the MFC-MBR system developed by Wang et al. [16].

Osmotic membrane biological reactor (OMBR)

Osmotic membrane biological reactors (OMBRs) differ from MBRs in relying on forward osmosis (FO) to drive the separation process. They incorporate a non-porous membrane and a draw solution (DS) to provide the concentration gradient required for

the flow of wastewater. The process is illustrated in Figure 2-10. FO membranes provide enhanced membrane rejection, lower fouling, and efficient energy consumption; however, they introduce an interruption known as salinity build-up, which is the accumulation of salts on the membrane. This interruption results in the appearance of the concentration polarization phenomenon, which incites a reverse flux due to the emerging concentration gradient on the membrane's surface [18].

OMBRs show great results throughout the literature. They were capable of removing 98% of the total organic carbon (TOC) with a concentration of less than 5 mg/L [18]. Over 80% nitrogen removal is achieved due to the nature of the FO membranes and the decent ammonia and nitrite oxidizing bacteria activity. Additionally, OMBRs revealed almost complete removal of phosphate due to the efficiency of FO membranes in rejecting calcium, which precipitates phosphate. In contrast, MBRs can only contribute to the removal of 60% of the phosphate present. Therefore, by observing the performance of OMBRs, it is noticed that they produce water with higher purity when compared to conventional MBRs; however, their shortcomings arise from the low water flux accompanied by FO and the high salt accumulation. Salt accumulation not only hinders the separation process by creating a new reversal concentration gradient, but it also significantly decreases the microbial activity in the activated sludge [18].

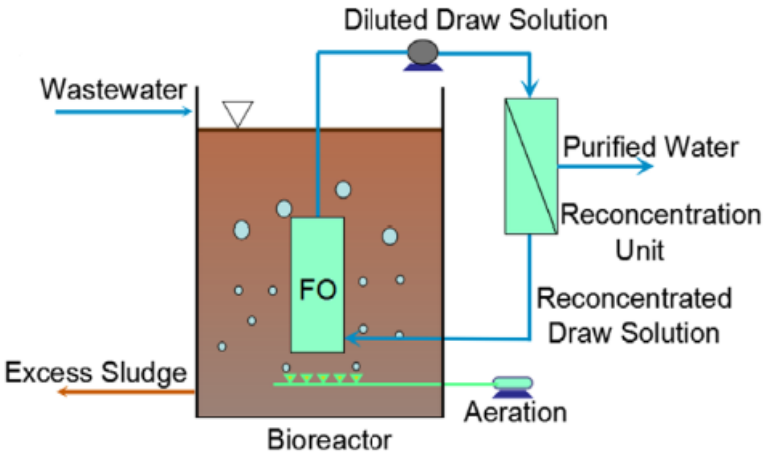

Figure 2-10: Schematic diagram of OMBR [18].

Other MBR systems

Other investigations were conducted on MBRs and their hybrid systems including aerobic dynamic membrane bioreactors (ADMBR), aerobic fluidized membrane

bioreactors (AFMBR), and membrane photobioreactors (MBPR). Table 2-2 summarizes these investigations involving the source of wastewater used.

Table 2-2: Operational and performance of different MBR configurations with different feedstock [13].

Configuration	Feedstock	Operating condition	Performance	Ref
AnMBR (external)	Synthetic molasses wastewater	• T= 35–37 ˚C, HRT=30h in pilot scale • Silver (as nanoparticles) coated PVDF • Backwashing the membrane with the air and produced gas from an anaerobic baffled reactor	• COD removal=82% • 91% reduction in EPS formation on the nanocomposite PVDF membrane surface.	[59]
AnMBR (submerged)	Synthetic wastewater	• T= 35.7 ˚C, HRT=10 days in lab scale • Flat sheet membrane with 0.014m^2 filtration area • 2 kg COD/m3/d as organic loading rate (OLR).	• COD removal=99.5% • Provide shorter start-up period, higher biogas production, and better COD removal in comparison to external one.	[60]
AnMBR (external)	Pharmaceutical wastewater	• T= 14-38 ˚C, HRT=36 h in pilot scale • Hollow-fiber membrane with 1m^2 surface area • 3.48 kg COD/m3/d as OLR	• TCOD removal=88-92.5% • 77-171 L/d for biogas production	[61]
Aerobic MBR	Pharmaceutical wastewater	• Reactor was operated until clogging (around one month) • Hollow-fiber membrane with 0.125m^2 surface area • Using natural quorum quencher (i.e., endophytic Penicillium restrictum) to control biofouling and increase removal efficiency of antibiotics.	• COD removal=82% • Longer operation before clogging happened in comparison to the reactor without the addition of P.restrictum (35days and 24 days respectively) • 4.5% increase in removal efficiency of antibiotics.	[62]
Aerobic MBR	Municipal wastewater	• SRT= 50 days, HRT=10 h in pilot scale • PVDF flat sheet membrane with 0.1m2 surface area • Average TSS of 9.6 g/l and 10.1 g/l without and with the addition of powdered activated carbon (PAC) to evaluate the effect of fouling.	• COD removal=89% • 19% increase in the critical flux by using PAC in comparison to sludge without PAC.	[63]
Aerobic Dynamic Membrane Bioreactor (ADMBR)	Municipal wastewater	• SRT= 106day, HRT=0.87 day, and MLSS= 5 g/l • Hollow-fiber membrane with 0.015m2 surface area • 0.66 kg COD/m3/d as OLR	• COD removal=92.8% • Faster clogging and decreasing COD removal efficiency were experienced by increasing sludge concentration.	[64]
Aerobic Fluidized Membrane Bioreactor (AFMBR)	Synthetic wastewater	• T= 37.1˚C, HRT=12h in a lab scale. • PVDF tubular membrane • 0.6 kg COD/m3/d as OLR • Using polymeric materials as fluidized agents	• COD removal=87.6% • Major foulant material on the membrane surface corresponds to EPS which fluidization wasn't sufficient enough to eliminate pore blockage. • The low energy consumption and stability of PVDF provide an excellent surface for biofilm formation.	[65]

Membrane photobioreactor (MPBR)	High ammonia nitrogen wastewater	• T= 25-30 ˚C, HRT=24 h, and SRT= 30 d • Using novel hollow fiber membrane with enhanced antifouling properties and hydrophilicity (i.e., polyvinylpyrrolidone (PVP)-graphene oxide (GO)/PVDF)	• COD removal=93% • Increase permeability and flux rate in filtration in comparison to normal PVDF • The great potential of PVP-GO/PVDF for high-density chlorella cultivation and effective treatment of high nitrogen ammonia wastewater	[17]
MFC-MBR	Synthetic wastewater	• Submerging anode and cathode in the submerged aerobic MBR at both sides of the membrane module • Using PVDF hollow fiber membrane with 0.2 m^2 surface area.	• 4.4% increase in COD removal efficiency when compared to control MBR • Maximum power density and average voltage were 2.18w/m^3 and 0.15 v respectively. • Improvement in filterability and dewaterability of the sludge • Alleviate the membrane fouling	[66]
MFC-MBR	Synthetic medium	• T=30 ˚C • Integration of anode as microfiltration membrane in side stream crossflow AnMBR configuration • Using stainless steel filtration membrane	• Achieving 4-fold higher current density in comparison to experiment without filtration (6 A/m^2 for 0.5 um filter grade) • Improving the current density was mainly due to permeate flow	[67]

Chapter 3. Methodology

3.1. Allowable Discharge Limits

For the purpose of this study, the wastewater was collected from a local tobacco processing firm, located in Ras Al Khaimah (RAK), UAE. The company is specialized in producing flavoured tobacco molasses. The competent authority responsible for issuing wastewater discharge standards in that Emirate is RAK Wastewater Agency (RAKWA). Table 3-1 presents the allowable discharge limits for such type of wastewater according to RAKWA [68].

Table 3-1: Allowable wastewater discharge limits issued by RAKWA [68].

#	Parameter	Unit	Limits
1	pH	-	6 to 8
2	Conductivity	µS/cm	≤5000
3	Total Dissolved Solids (TDS)	ppm	≤2500
4	Salinity	ppt	≤1
5	Temperature	^{o}C	-
6	Total Suspended Solids (TSS)	ppm	≤350
7	Chemical Oxygen Demand (COD)	ppm	≤700
8	Biological Oxygen Demand (BOD_5)	ppm	≤300
9	Ammonical Nitrogen	ppm	≤50
10	Total Phosphorus	ppm	≤20
11	Sulphate	ppm	≤300
12	Chloride	ppm	≤2000
13	Turbidity	NTU	-
14	Total Alkalinity	ppm	≤600
15	Nitrate Nitrogen	ppm	≤10
16	Fat, Oil & Grease	ppm	≤50

3.2. Wastewater Characteristics

The wastewater collected from Al Mokhtar Tobacco shows significant disagreement with the allowable limits. The effluent had a black appearance, and it was analyzed by RAKWA. Table 3-2 summarizes the laboratory analysis [69].

Table 3-2: Local tobacco processing firm wastewater laboratory analysis [69].

	Parameter	Teste Method	Unit	Amount
1	pH	APHA 4500-H	-	3.2
2	Conductivity	APHA 2510-B	µS/cm	2640
3	TDS	HACH	mg/L	1345
4	Temperature	-	^{o}C	21
5	TSS	APHA 2450-D	mg/L	150
6	COD	HACH	mg/L	3440
7	Soluble COD (SCOD)	HACH	mg/L	3170
8	BOD_5	APHA 5210-B	mg/L	1119
9	Ammonical Nitrogen	HACH	mg/L	2.51
10	Nitrate Nitrogen	HACH	mg/L	0.986
11	Total Phosphorus	HACH	mg/L	1.31
12	Sulphate	HACH	mg/L	89.7
13	Chloride	HACH	mg/L	287
14	Turbidity	HACH	NTU	121

From Table 3-2, it is observed that the major parameters of concern are pH, COD, SCOD, and BOD_5.

The current treatment process implemented by Al Mokhtar Tobacco Manufacturing and Trading consists of an aerated clarifier for which the wastewater is fed and biologically treated. Alum is added to the clarifier to facilitate flocculation. The effluent of the clarifier is sent to a tank where the pH is neutralized. Then, the mixed liquor is separated using a filter press to produce water within the allowable discharge limits. Although this process produces water that complies with RAKWA's regulations, it is not sustainable as it generates two waste products namely the water and the sludge-suspended solids mixture.

By employing the MBR, it is targeted to produce water sufficient enough to be recycled and used in processing while the sludge waste is minimized by recycling it into the biological reactor to enhance the treatment process further.

3.3. Apparatus

The MBR lab unit used in this work is designed by CERAFILTEC (Germany) for short-term simple filtration purposes. It incorporates a ceramic membrane with an area of 0.01 m^2, a valveless rotary piston pump, an air diffuser, and an aeration tank. The unit operates automatically, with adjustable settings, via a control system designed using Siemens LOGO!8 with TDE software. The control unit records the transmembrane pressure (TMP) changes with time. Additionally, the volumetric flowrate can be adjusted using the control panel by changing the frequency of the pump. The operating conditions and the filtration specifications for this unit are represented in Table 3-3.

Table 3-3: The operating conditions and the filtration specifications for CERAFILTEC MBR lab unit.

Parameter/ Instrument	Specification
Medium temperature	5 – 40 ºC
Ambient temperature	5 – 40 ºC
Maximum recommended sludge concentration	15 g/L
Maximum relative humidity	80%
Maximum diameter of suspended solids inflow	1 mm
Type of pump	valveless rotary piston pump
CPF pump-head	FMI007
Fuse rating	2 x T800 mA/ 250 V (230 V_{AC})
Speed range	18 – 1800 rpm; 0 – 50 HZ; 0 – 100%
Flow range	0.002 – 10.8 L/ h
Maximum differential pressure	6.9 bar
Power consumption	maximum 50 W
Connections	polytetrafluoroethane (PTFE) tubes 1.6 mm (ID); 3.2 mm (OD); with 2 fittings UNF ¼-28 (male)

The sludge utilized for the treatment process comprised of a mixture of 70% *Pseudomonas* bacteria and 30% *Bacillus* supplied under the commercial name RoeTech 302, manufactured by Roebic Technology Inc. (USA) [70]. The two main factors

utilized for the selection of this specific mixture are their commercial feasibility and formulation that can degrade hydrocarbons and oils, which makes this selection favourable for industrial wastewater treatment. These organisms are of the most abundant strains present in wastewater treatment systems [71]. The membrane used is a SiC membrane with a nominal pore size of 0.3 µm. SiC membranes are of great significance due to their high chemical stability and ability to withstand high temperatures [72]. Therefore, they became widely spread in wastewater treatment applications and in the food and pharmaceutical processing industries.

3.4. Experimental Procedure

The wastewater sample collected from a local tobacco plant consists of many suspended solids that might foul the membrane rapidly. Therefore, the sample underwent a pre-treatment stage. This involved the following steps:

1. The wastewater sample was introduced to a settling tank, for one day, to remove any suspended solids; hence, remove the major source of colour.
2. Utilizing a dosing pump, the sample was diluted with distilled water and neutralized using NaOH, then transferred into the MBR unit.
3. The rotary piston pump sucked the sample across the membrane into the product tank.
4. Water analysis was carried out.
5. The previous steps were repeated under different conditions, and the performance was evaluated.

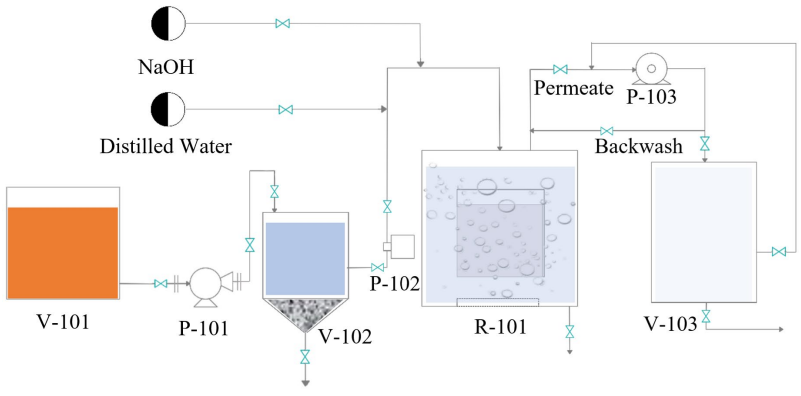

Figure 3-1: Process P&ID for the MBR apparatus. P-101: pump; P-102: dosing pump; P-103: suction pump; R-101: aerated biological reactor; V-101: storage tank; V-102: settling tank; V-103: storage tank.

Figure 3-1 presents the piping and instrumentation diagram (P&ID) for the process mentioned above.

3.5. Water Analysis

To ensure that the treatment process has achieved its objective, analyses were carried out on the effluent of the MBR with emphasis on COD, BOD_5, and TOC. In addition, several parameters were evaluated. The parameters considered and instruments used for their analyses are shown in Table 3-4.

Table 3-4: Water parameters under investigation.

#	Parameter	Definition	Instrument
1	COD	Measures the oxygen used for the oxidation of organic and oxidizable inorganic substances.	Aqualytic PC Compact COD Vario Photometer utilizing Lovibond COD Vario Vials
2	BOD_5	Measures the oxygen required by aerobic microorganisms to oxidize organic matter.	OxiTop® Measuring System
3	TOC	Measured the total organic carbon.	Suez Innovox TOC AutoSampler
4	pH	Measures the acidity of water.	Hanna edge® Multiparameter pH Meter – HI2020
5	Conductivity	Measures the ability of water to conduct current.	Hanna edge® Multiparameter EC/TDS/Salinity Meter – HI2030
6	TDS	Measures the solids dissolved in water.	Hanna edge® Multiparameter EC/TDS/Salinity Meter – HI2030
8	Turbidity	Measures the transparency of water.	Hanna LP 2000 Turbidity Meter

10	Ammonical Nitrogen	Measures ammonia and ammonium ions concentrations.	APEL PD-303S Spectrophotometer
11	Total Phosphorus	Measures phosphorus concentration.	APEL PD-303S Spectrophotometer
12	Sulphate	Measures the concentration of sulphate ions.	APEL PD-303S Spectrophotometer
13	Chloride	Measures the concentration of chloride ions.	APEL PD-303S Spectrophotometer

3.6. Effect of Operating Parameters

After testing the MBR's ability to treat tobacco industrial wastewater, its effectiveness will be evaluated for operation under different parameters including:

- pH: The wastewater sample acidity will be altered to reveal its influence over the range of pH of 5 – 9.
- Sludge Concentration: This can be performed by different dilutions of the wastewater sample over the range of 2000 – 4000 ppm.
- Temperature: The wastewater sample will be heated to evaluate the technology's adaptation temperatures between 30 – 50 ºC.
- Membrane pore size: Different membrane ratings 0.1 and 0.3 µm will be tested.

Chapter 4. Results & Discussion

4.1. Pre-Treatment

The first observation made on the wastewater sample collected was its dark orange colour. The colour is attributed to the commercial food dye used in the production of molasses. The issue with the concentrated colour is its high capability to foul the membrane. To remove the dye, the sample was filtered through sand. Sand filtration is one of the most common wastewater treatment methods. The sample infiltrates through the sand where the sand extracts any microorganisms, suspended solids, and some dissolved solids. To carry out the lab scale experiment, sand was placed in a cone supported by a filtration paper, and the sample was poured into the cone. The results were unpleasant as after 3 rounds of filtration, the product had changed in colour from orange to greenish black.

Another method attempted for removing colour is the use Xanthan gum. Xanthan gum is known to interact electrostatically with dyes via adsorption and has many applications due to its low cost of operation [73]. However, the method failed as it has been observed that the colour changed to a colour similar to the one observed in the effluent of sand filtration.

Alternatively, the wastewater sample was contacted with ion exchange resins. In this case, an adsorption column filled with resins was constructed, and the sample was introduced into the column. It was observed that the effluent was highly clear. However, ion exchange resins are expensive for which a more cost-effective adsorbent needed to be used.

To compromise between the cost and effectiveness, activated carbon was used to extract the dye. The wastewater sample was mixed with activated carbon and left for a day to settle the carbon particles. The collected sample was colourless; however, it was hazy.

Although contacting the wastewater with activated carbon or resin produced wastewater that is almost colourless, both substances can alter the COD content of the water, which diverts from the scope of the work that is investigating the performance of MBRs. Therefore, the settling of the wastewater content was investigated. After 1 day of settling, the enriched colour of the wastewater was significantly decreased indicating that the colour was due to suspended solids. Thus, settling was utilized as the

method for removal of the colour due to its cost effectiveness and its ability to preserve the COD content of the wastewater.

4.1.1. Evaluation of the COD shock load

Following the colour removal, the COD levels should be reduced to a level below the shock load to sustain the treatment process. Industrial wastewater is usually discharged to sewage systems to be treated by municipal's sewer systems after pre-treatment. However, this can disturb the treatment process as the received wastewater contains loadings, which require adjustments to the process [74]. In essence, such highly contaminated flows can decrease the activity of the microorganisms required for biodegradation. Therefore, it is of importance to investigate the load at which the treatment can be carried out efficiently.

As per the initial analysis conducted on the wastewater, the main parameter that can shock the system is COD. In order to decrease its value to a feasible limit, it was mixed with distilled water yielding samples with COD values of around 500, 1000, 1500, and 2000 ppm. The system operated at 20 ^{o}C and a pH of 7 while the wastewater was fed to the bioreactor with a concentrated sludge of 3000 ppm in a 1:1 volume ratio. Each analysis was conducted over a period of 24 hours. As shown in Figure 4-1, there has been a significant reduction in COD removal while increasing the COD loading. When operating at a COD loading of 500 ppm, a significant COD removal of around 89% was achieved. Increasing the COD loading to 1000, 1500, and 2000 revealed a reduction in COD removal to around 77%, 67%, and 55% respectively. The reason for such a decrease could be due to the microorganisms present in the system which experienced high biodegradation demand [74], i.e. high F/M ratio. Thus, the DO levels in the system decrease, which disturbs the treatment process. Nonetheless, the system regains stability with time. As shown in Figure 4-1, increasing the HRT to 48 hours enhances the biodiversity of the system, which in turn facilitates the biodegradation process [75]. Therefore, increasing the HRT is required when the influent is expected to experience a high F/M ratio. For the purposes of this study, the experiments were conducted at a COD concentration of 540 ppm as a conservative and comparative value to literature while the pH was adjusted using NaOH solution. Corresponding to this COD value is a TOC of 154 ppm and a BOD_5 of 175 ppm.

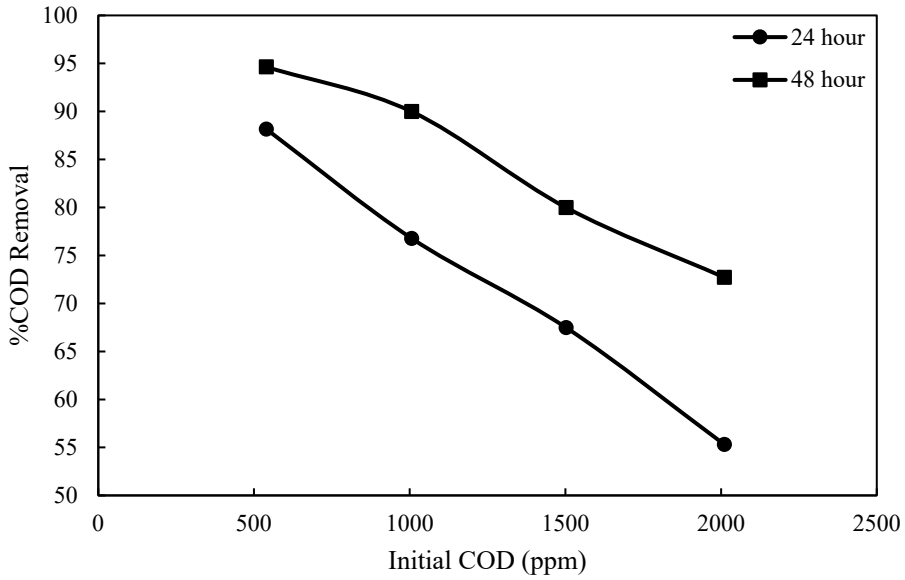

Figure 4-1: Effect of COD load on the treatment process over periods of 24 and 48 hours at 20 ºC and a pH of 7.

4.2. Effect of pH

The study of the effect of the pH was conducted on pH values of 5, 6, 7, 8, and 9 while the pH of 7 was taken as a reference state. The system operated at 20 ºC while the wastewater was fed to the bioreactor with a concentrated sludge of 3000 ppm in a 1:1 volume ratio. A sample was collected and analyzed every 3 hours for an operation of 12 hours. Then, a final sample at 24 hours was collected to conclude the experiment. Figures 4-2 and 4-3 show the effect of pH on COD and TOC removals at a moderate range of pH, namely 5, 6, and 7. Within the first 3 hours, there has been a sharp decrease in the COD content for all pH values. As shown, the COD removal for the operation at a pH of 6 and 7 were comparable with values of 46.11% and 44.07%, respectively. Meanwhile, the operation at a pH of 5 resulted in a much lower COD removal of 23.70% within the first 3 hours. This can be caused by the low activity of the bacterial mixture at that pH. During the following 6 hours, steady COD removal was observed for the operation at the pH of 6 and 7 whereas the operation at the pH of 5 showed an increase in the COD. This is due to the decrease in microbial activity due to starvation caused by the fast decrease in F/M ratio. In the following 3 hours, the COD resumes to decrease sharply achieving a COD removal of 20.19%, 74.26%, and 71.11% for the pH of 5, 6, and 7 respectively. After 24 hours of operation, the COD removal for the operation at a pH of 5 was found to be 47.04%; 83.5% at a pH of 6; and 88.15 at a pH of 7. Ultimately, these findings indicate that the operation at a pH of 5 is not favourable

for this system. Similarly, the TOC analysis revealed a similar trend to the COD analysis results, as shown in Figure 4-3. After operating for 24 hours, the TOC removal yielded values of 65.81%, 88.84%, and 90% for the operations at a pH of 5, 6, and 7, respectively.

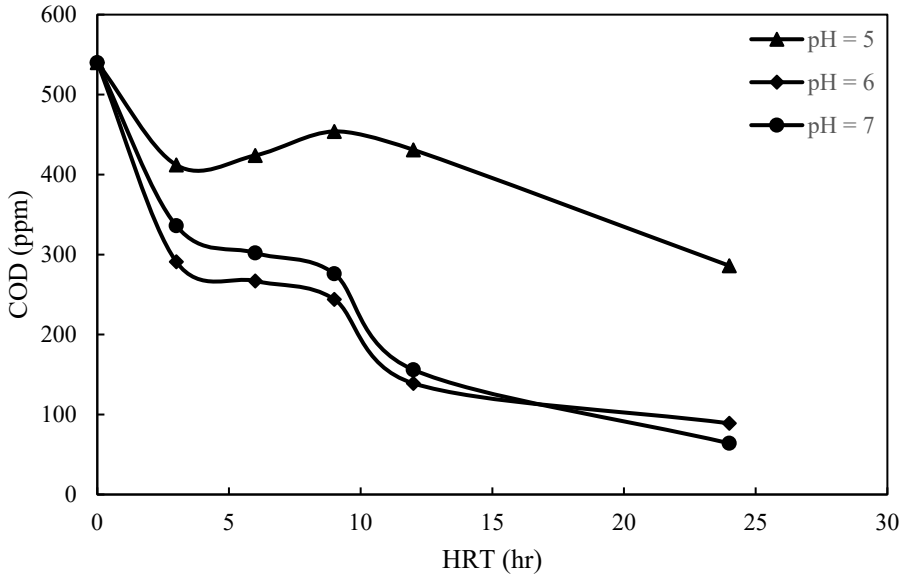

Figure 4-2: Effect of pH, moderate range, on COD removal from tobacco wastewater at 20 ºC and a sludge concentration of 3000 ppm.

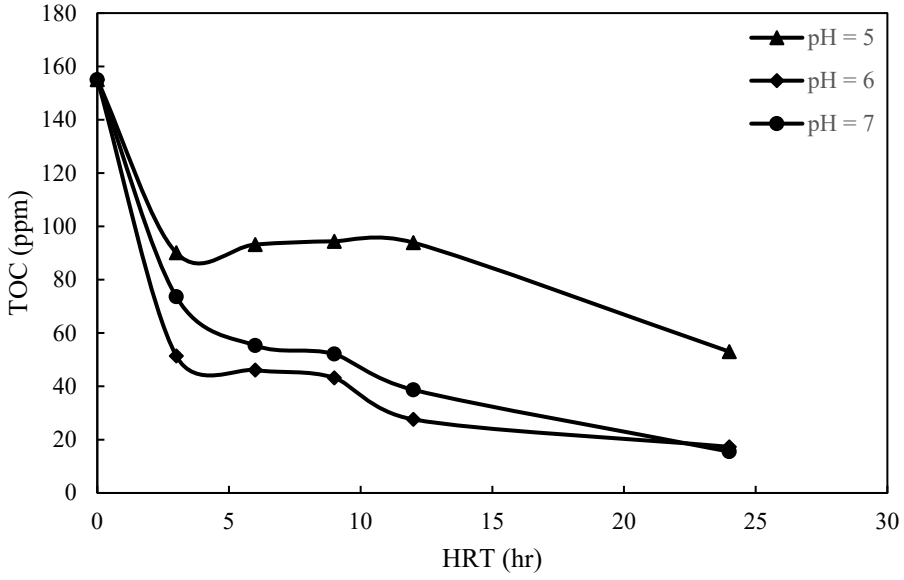

Figure 4-3: Effect of pH, moderate range, on TOC removal from tobacco wastewater at 20 ºC and a sludge concentration of 3000 ppm.

As for operating at high pH, the trend observed in the case of operating at a pH of 8 was similar to the case of pH of 6 and 7, which revealed a COD removal of 89.63% after 24 hours of operation. Meanwhile, operating at a pH of 9 revealed a COD removal

of 72.96% after 24 hours (Figure 4-4). As for the TOC analysis, a similar trend for the COD removal has been observed yielding a TOC removal of 91.10% and 84.26% for operating at pH of 8 and 9, respectively (Figure 4-5).

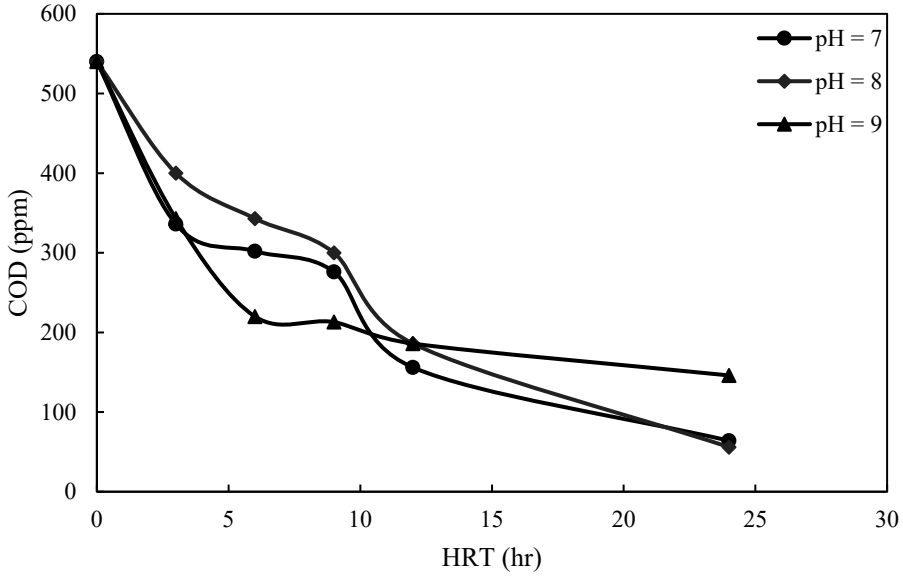

Figure 4-4: Effect of pH, high range, on COD removal from tobacco wastewater at 20 °C and a sludge concentration of 3000 ppm.

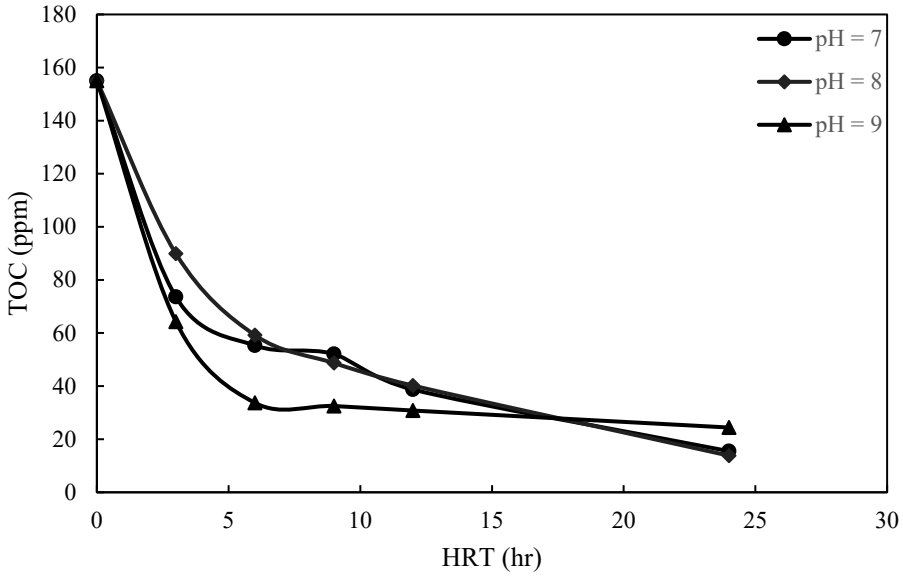

Figure 4-5: Effect of pH, high range, on TOC removal from tobacco wastewater at 20 °C and a sludge concentration of 3000 ppm.

Observing operation at pH 6, 7, and 8, within the first 12 hours, the COD removal is higher with lower pH; however, after 24 hours, the COD removal is greater at higher pH. This reveals an optimum operating condition with time. Therefore, the pH can be controlled to 6 for the first 12 hours. Then, it can be increased to 8 to increase the rate

of COD removal. As for the BOD_5 (Table 4-1), the treatment has shown removal of 54.3%, 88.6%, 91.4%, 94.3%, and 60% after operating for 24 hours at a pH of 5 through 9, respectively. Therefore, these results suggest that there is an operation window between pH of 6 and 8.

Table 4-1: Effect of pH on BOD_5 removal from tobacco wastewater at 20 ^oC and a sludge concentration of 3000 ppm.

pH	BOD_5 (ppm)
5	80
6	20
7	15
8	10
9	70

The results obtained are consistent with what has been reported by Sanguanpak et al. [76]. They investigated the effect of operating pH on the biodegradation performance in MBRs for landfill leachate treatment. Ultimately, their results revealed that the microbial activity was not affected by varying the pH between 6.5, 7.5, and 8.5. Meanwhile, as the mixed liquor pH decreased to 5.5 and below, the microbial activity has worsen revealing low COD removal. Thus, low pH can interfere with the degradation mechanism of some bacterial species [77] [78] including *Bacillus* and *Pseudomonas*. Bhattacharyya et al. [79] reported that heterotrophic bacteria favour the operation at pH in the range of 7.2 – 8.5 [80] [81]. Overall, the system has revealed great potential for the treatment of high-strength industrial wastewater. In a study conducted by Bilad et al. [82], a submerged MBR was used to treat molasses wastewater. The study has yielded a COD removal ranging between 80 – 84%. Although these removal rates are relatively low compared to municipal wastewater treatment using MBRs, it is important to note that high-strength industrial wastewater exhibit substances that inhibit microbial growth; hence, reducing the biodegradation efficiency [83].

4.3. Effect of Sludge Concentration

To investigate the effect of sludge concentration (SC) on the treatment process, 2000, 2500, 3000, 3500, and 4000 ppm concentrated sludge mixtures were prepared and fed

in a 1:1 volume ratio to the wastewater at pH 7 and 20 ºC. In this case, the wastewater treated with the 3000 ppm sludge concentration is set as the reference. As shown in Figure 4-6, the initial decrease in the COD was higher in the case of low concentrated sludge. This indicates that once the sludge concentration approaches 3000 ppm, the system becomes saturated with biomass, which contributes to the increase in COD content due to insufficient levels of oxygen [84]. Ultimately, the COD removal is less over a period of 24 hours in comparison to the sample treated with 3000 ppm. This is shown in Figure 4-6 where the COD removal yielded 59.62% and 80.37% for the cases of treating the wastewater with 2000 and 2500 ppm concentrated sludges respectively. These results can be put into perspective when compared to the 88.15% COD removal observed in the case of treatment with 3000 ppm concentrated sludge. Thus, reducing the concentration of the sludge will extend the time required for treatment. Similar results were observed when the TOC sample was analyzed. Figure 4-7 reveals 74.13% and 88.55% TOC removals for the cases of treating the wastewater with 2000 and 2500 ppm concentrated sludges, respectively, in comparison to 90% TOC removal for the 3000 ppm concentrated sludge.

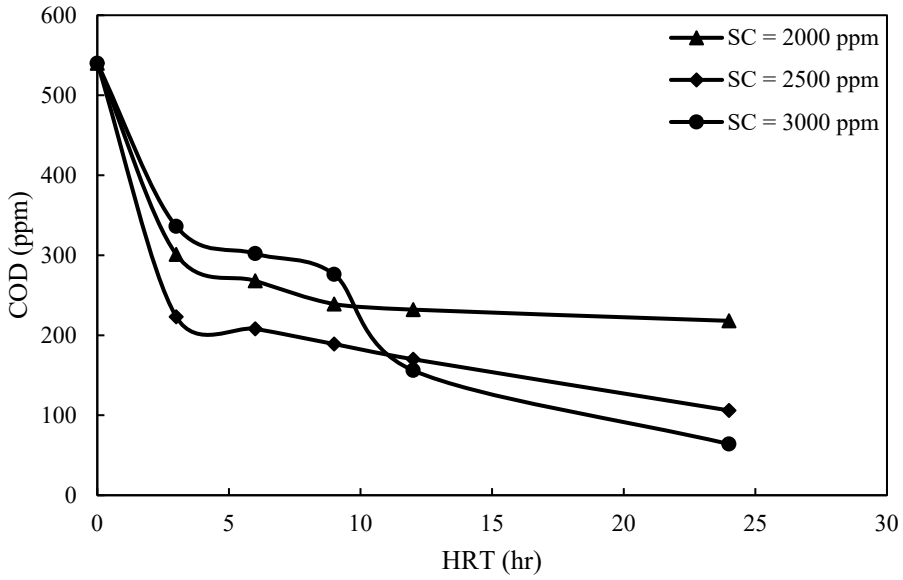

Figure 4-6: Effect of sludge concentrations, 2000, 2500, and 3000 ppm, on COD removal from tobacco wastewater at 20 ºC and pH 7.

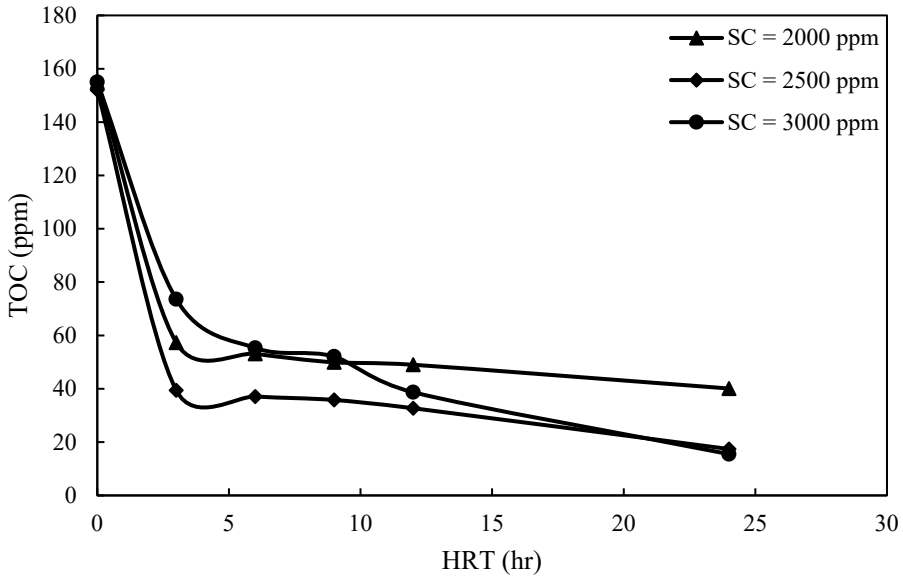

Figure 4-7: Effect of sludge concentrations, 2000, 2500, and 3000 ppm, on TOC removal from tobacco wastewater at 20 °C and pH 7.

As for operation at higher sludge concentration, the results affirm the interference of sludge saturation on the COD content. As shown in Figure 4-8, the rate of COD removal decreases with the increase of sludge concentration. This is due to the biomass deterioration as the oxygen supply was lower than the required intake to sustain the treatment process [84].

The highest COD removal was observed for the case of 3000 ppm concentrated sludge in comparison to the samples at 3500 and 4000 ppm concentrated sludge, which yielded 83.52% and 71.84% respectively (Figure 4-8). Similarly, the TOC analysis revealed a trend that is in accordance with the COD analysis yielding TOC removals of 86.39% and 75.03% when using 3500 and 4000 ppm concentrated sludge, respectively (Figure 4-9).

As for the BOD_5, the results showed removals of 82.6%, 82.6%, 91.4%, 88.6%, and 77.1% after 24 hours of operation at sludge concentrations of 2000, 2500, 3000, 3500, and 4000, respectively (Table 4-2).

In general, increasing the sludge concentration should enhance the treatment efficiency [85]. However, as evidenced by the results of this work, excessive sludge concentration can lead to elongating the treatment process to achieve the desired treatment. This can be attributed to the low oxygen levels supplied to the MBR that are required for aerobic degradation [84].

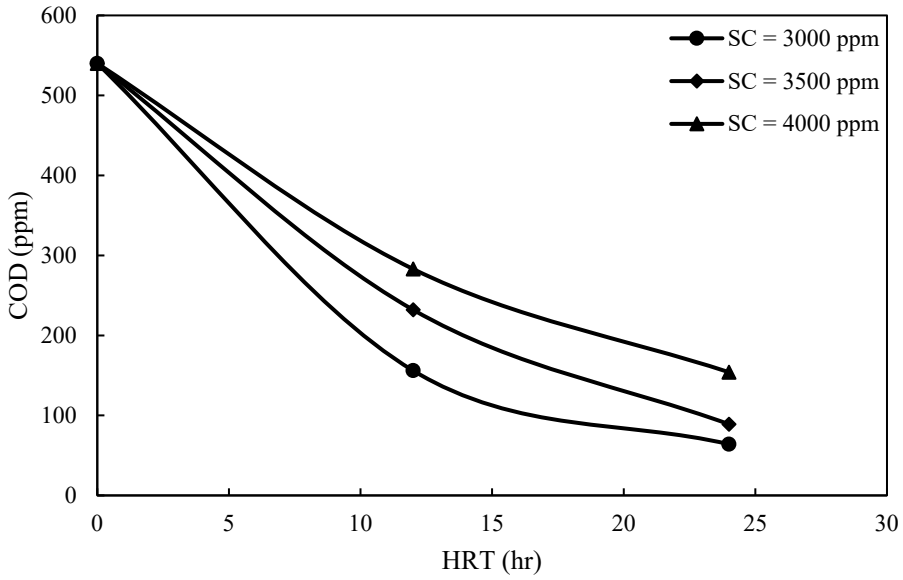

Figure 4-8: Effect of sludge concentrations, 3000, 3500, 4000 ppm, on COD removal from tobacco wastewater at 20 °C and pH 7.

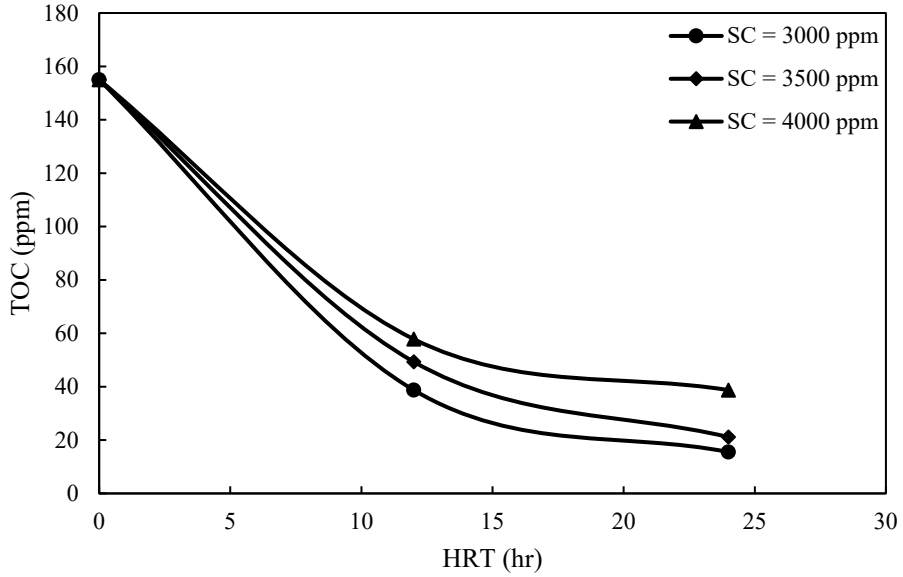

Figure 4-9: Effect of sludge concentrations, 3000, 3500, and 4000 ppm, on TOC removal from tobacco wastewater at 20 °C and pH 7.

Table 4-2: Effect of sludge concentration on BOD_5 removal from tobacco wastewater at 20 °C and pH 7.

Sludge Concentration (ppm)	BOD_5 (ppm)
2000	30
2500	30
3000	15
3500	20

4000	40

4.4. Effect of Temperature

To study the potential of the design to treat the wastewater in different climates, the temperature was varied to 30, 40, and 50 ºC. The wastewater was fed to the bioreactor with a concentrated sludge of 3000 ppm in a 1:1 volume ratio at a pH of 7. The mixture was heated to achieve the desired temperatures while the temperature was held constant within a ±1 ºC tolerance. As shown in Figure 4-10, the results at 30 and 40 ºC showed almost identical COD removal compared to that at 20 ºC within the first 5 hours. However, the removal rate showed enhancement later on which could be due to the enhanced kinetics of the system [86]. According to Atlas [87], the temperature effect on the treatment process is due to deforming the physical and chemical composition of the wastewater constituents and the metabolic activity of the microorganisms. For instance, elevating the wastewater temperature decreases the density of hydrocarbons present, which in turn increases their volatility. Thus, the biodegradation process is enhanced by increasing its rate. However, the temperature can only be raised to a certain value. In the case of hydrocarbons, this value usually falls in the range of 30 – 40 ºC. Exceeding the aforementioned range contributes to the increase in hydrocarbons toxicity. Furthermore, elevating the temperature decreases the dissolved oxygen level, which is required for aerobic degradation [88]. This decrease in treatment efficacy has been observed when operating at 50 ºC. Figure 4-10 reveals an increase in COD content, which indicates that the bacteria has deteriorated which may contribute to the increase in COD content. Therefore, the study reveals that this system can treat water in temperatures ranging between 20 and 40 ºC. In climates that exceed such temperatures, eg. summer afternoons in the Arabian Peninsula, cooling measurements need to be taken into consideration.

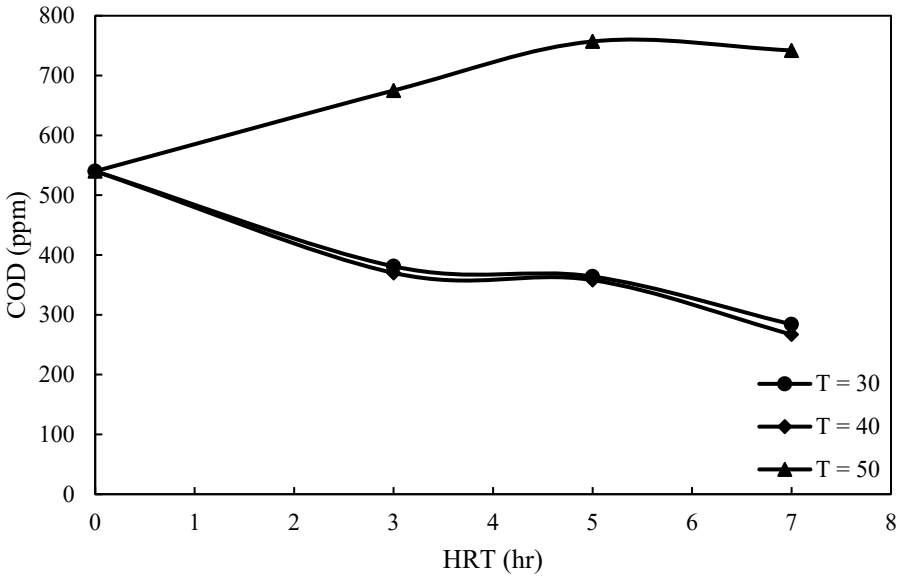

Figure 4-10: Effect of temperature in degrees Celsius on COD removal from tobacco wastewater using 3000 ppm sludge concentration at pH 7.

4.5. Other Parametric Consideration

As the MBR has shown a significant ability for the removal of COD, BOD_5, and TOC, it was a necessity to evaluate the effluent characteristics in terms of other parameters in order to assess the extent of the treatment process. The wastewater was fed to the bioreactor with a concentrated sludge of 3000 ppm in a 1:1 volume ratio and set to a pH of 7 and a temperature of 20 ^{o}C. The results displayed in Table 4-3 indicate that the system was successful in reducing the conductivity and TDS to about 88.7%. The presence of TDS in the pre-treated water is mainly attributed to the sodium ions added in the form of NaOH into the system to neutralize the mixed liquor. Sodium ions accumulate on the surface of the membrane due to the reduced electrical layer on the membrane. Although microfiltration is not the most effective technique for the removal of TDS, significant results were found in the literature. Chandrasekhar et al. [89] have employed an MBR to treat the effluent wastewater from the dairy industry. In their study, a TDS removal of 67% has been achieved. Garg and Chaudhry [90] have successfully obtained around 97% TDS removal utilizing an MBR for the treatment of wastewater of the food industry. As for the turbidity, Figure 4-11 showed that it has decreased by 97% which is expected as settling and microfiltration are integrated into the system [91]. As for ammoniacal nitrogen, total phosphorus, sulphate, and chloride, their concentrations were non-detectable after the pre-treatment stage due to their initial low concentration in the raw wastewater. Thus, the design was remarkably successful

in treating the major sources of contamination in the wastewater. With such treatment, it can be said that upon the incorporation of a disinfection stage, the effluent can be recycled to be used for processing rather than discharge into the sewers.

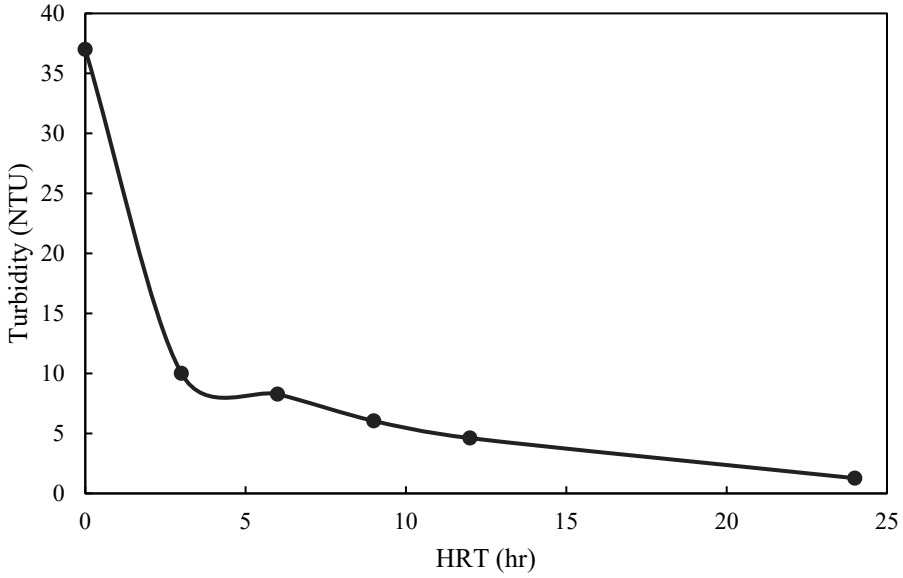

Figure 4-11: Turbidity removal over 24 hour HRT.

Table 4-3: Summary of analyses conducted on the wastewater effluent at 24 hour HRT.

Parameter	Unit	Amount after pre-treatment	Amount after biological treatment
pH	-	7	7.14
Conductivity	µS/cm	1384	153
TDS	ppm	693	78.3
Turbidity	NTU	37	1.27
Ammoniacal Nitrogen	ppm N	0	0
Total Phosphorus	ppm	0	0
Sulphate	ppm S	0	0
Chloride	ppm	0	0

4.6. Evaluation of the Membrane Performance

To evaluate the performance of the membrane and its adequacy for the treatment of tobacco wastewater, the TMP was recorded during the operation. The wastewater was

fed to the bioreactor with a concentrated sludge of 3000 ppm in a 1:1 volume ratio, set to a pH of 7 and temperature of 20 ºC. Figure 4-12 shows that TMP has increased rapidly in the first 20 minutes of the operation to 4.5 kPa. Exceeding that point, the increase in TMP became slower at a rate of about 0.01 kPa/ min. Obseving the effluent flowrate, it has decreased after the 20 minutes mark indicating the hinderance caused by fouling. Therefore, to maximize the effluent and ensure the stability of the filtration process, backwash was introduced for 1 minute every 15 minutes of operation, which was sufficient to maintain the effluent flowrate.

Since the flowrate was successfully retained it can be concluded that only reversible fouling takes place at the proposed operating conditions, which is favourable as only physical cleaning methods are required. Nonetheless, it is recommended that chemical cleaning should take place every 2 weeks to maintain the safety and functionality of the membrane [92]. Overall, the membrane has performed extremely well indicating its robustness. Minimizing the colour, amount of suspended solids, and sludge concentration has contributed sufficiently to its steady performance and low fouling propensity.

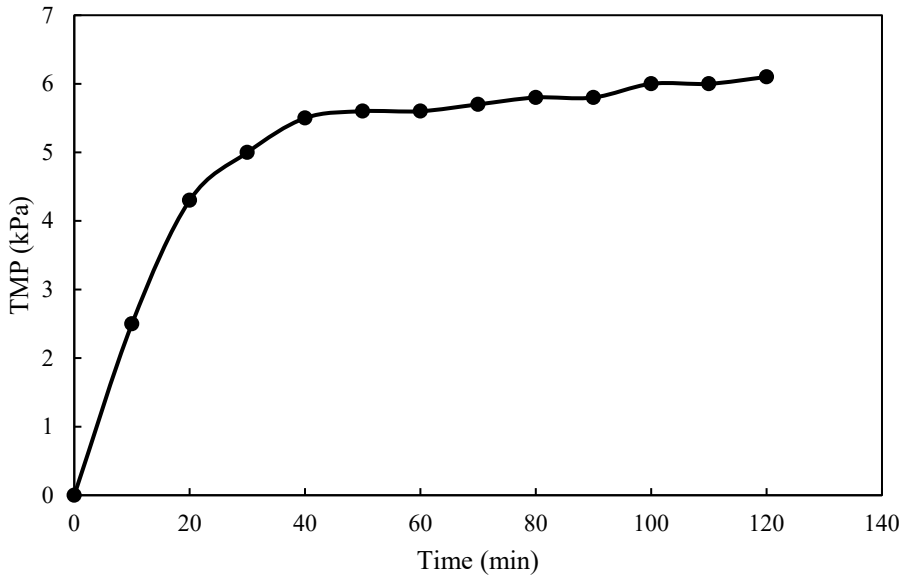

Figure 4-12: Performance of the membrane evaluated in TMP over the operation period.

4.7. Effect of Membrane Pore Size

To study the effect of the membrane pore size on the MBR performance, an alumina flat sheet membrane with a 0.1 µm nominal pore size was used to carry out the treatment process. The wastewater was fed to the bioreactor with a concentrated sludge of 3000

ppm in a 1:1 volume ratio, set to a pH of 7 and temperature of 20 ºC. The alumina membrane revealed a slight improvement, but very similar, removal rates of COD, BOD_5, and TOC compared to the SiC membrane. As shown in Figure 4-13, after 24 hours of operation, the system was capable of removing 89.2% of the COD contained in the influent to the biological reactor (Figure 4-13). Furthermore, it was capable of achieving the removal of 90.4% and 91.4% of TOC and BOD_5 respectively (Figure 4-14 and Table 4-4). The similarities in the results are due to the similar operating conditions and microorganisms' diversity of both membrane systems. In essence, the alternation of the membrane pore size affects the filtration regime but does not influence microbial activity [93]. Huang et al. [93] have investigated the effect of ceramic membrane pore size on MBR performance by utilizing mean pore sizes of 0.5, 0.63, 0.8, and 1.02 μm. As per their study, the membrane pore size had minimal effect on COD removal.

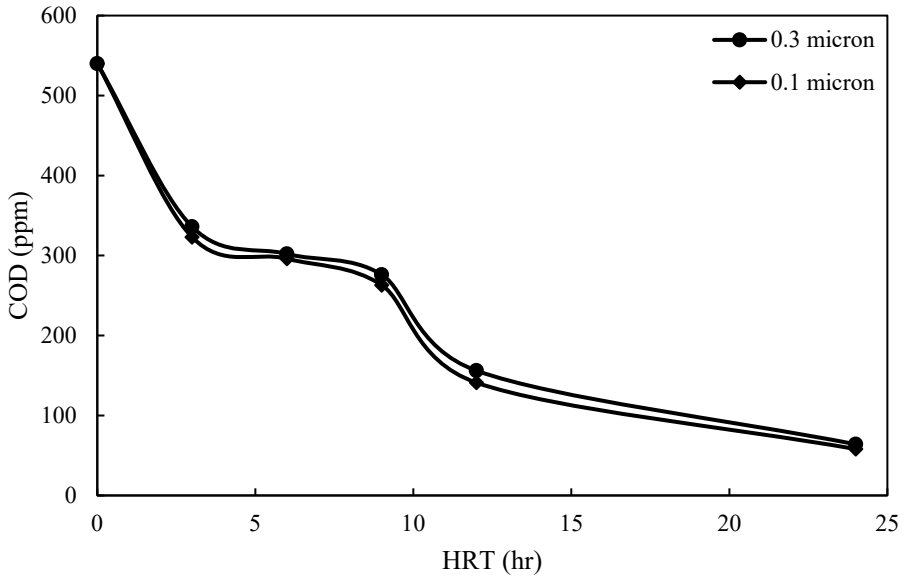

Figure 4-13: Effect of membrane nominal pore size on COD removal from tobacco wastewater at a temperature of 20 ºC, pH 7, and sludge concentration of 3000 ppm.

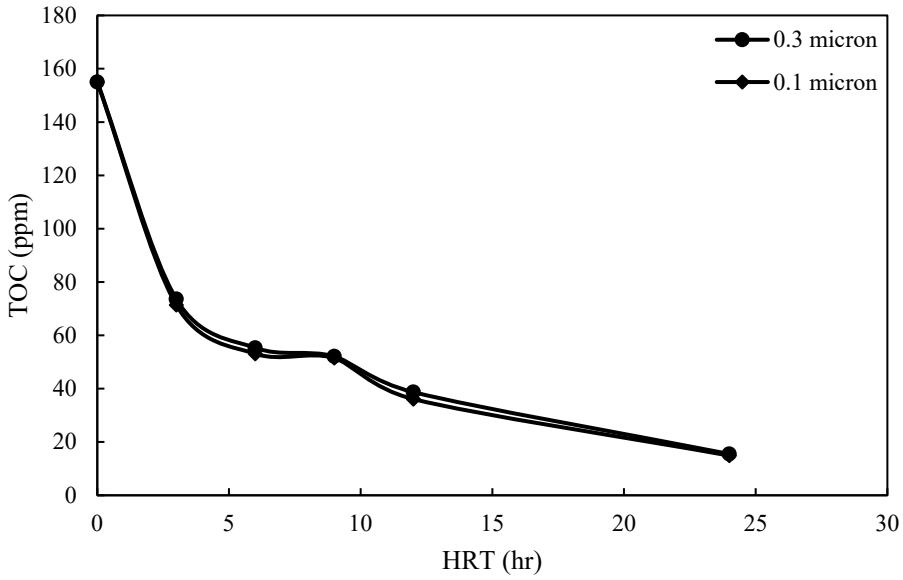

Figure 4-14: Effect of membrane nominal pore size on TOC removal from tobacco wastewater at a temperature of 20 °C, pH 7, and sludge concentration of 3000 ppm.

Table 4-4: Effect of membrane nominal pore size on BOD_5 removal from tobacco wastewater at a temperature of 20 °C, pH 7, and sludge concentration of 3000 ppm.

Nominal Pore Size	Initial BOD_5 (ppm)	Final BOD_5 (ppm)
0.1 µm	175	15
0.3 µm	175	15

Additionally, the effect of reducing the membrane pore size on TDS removal and turbidity reduction was investigated. Similarly, the alumina membrane enhanced the TDS removal slightly with removal of 88.9% while the turbidity was reduced to 97.3% of the initial turbidity (Table 4-5). The decrease in TDS and turbidity could be due to the reduction in pore size, which reduces the amount of solids in the permeate. Nonetheless, the reduction in the nominal pore size of the ceramic membrane from 0.3 to 0.1 µm did not influence the performance by a high margin.

Table 4-5: Effect of membrane nominal pore size on conductivity, TDS, and turbidity removal from tobacco wastewater at a temperature of 20 °C, pH 7, and sludge concentration of 3000 ppm.

Parameter	Unit	Nominal Pore Size	Amount after Biological Treatment
Conductivity	µS/cm	0.1 µm	153
		0.3 µm	156.7
TDS	ppm	0.1 µm	76.5

		0.3 µm	78.3
Turbidity	NTU	0.1 µm	1
		0.3 µm	1.27

Observing the fouling propensity and TMP, Figure 4-15 shows that reducing the nominal pore size exerted an increase in TMP. Similar to the SiC membrane, the alumina membrane exerted an initial rapid increase in the TMP followed by a slower steadier increase. Fortunately, the higher TMP did not influence the flowrate indicating good overall performance that is comparable to the alumina membrane. According to Huang et al. [93], decreasing the membrane pore size decreases the fouling propensity and the TMP. However, that is set to a limit. As per their study, exceeding the 0.63 µm nominal pore size initiated an increase in the TMP. This is consistent with the results of this work for the SiC and alumina membranes. Therefore, alternating the nominal pore size, within the studied limits, does not have a great impact on the overall performance of the MBR system.

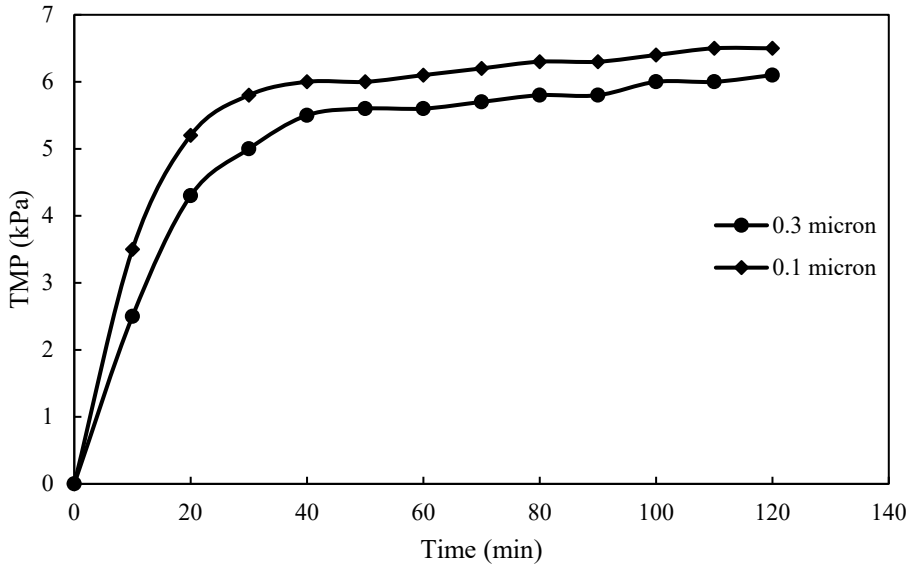

Figure 4-15: Effect of membrane nominal pore size evaluated in TMP over the operation period.

Chapter 5. Conclusions & Recommendations

The membrane biological reactor (MBR) can be employed for the treatment of wastewater generated by the tobacco industry with an emphasis on the removal of COD, BOD_5, and TOC. The MBR performance was dependent on the pH of the wastewater, sludge concentration, and operating temperature. The MBRs can treat wastewater within the range of pH 6 – 8 while pH 8 being the optimum treatment condition. The MBR is capable of treating wastewater at a temperature ranging between 20 – 40 ^oC . Increasing the temperature enhances the biodegradation activity; however, exceeding 40 ^oC can result in the deterioration of the treatment process. The system has shown great performance in removing other pollutants from the water such as phosphorus, sulphate, ammoniacal nitrogen, and chloride, which were removed during the pre-treatment of the influent. As for TDS and turbidity, the system achieved high removal rates exceeding 85% even though it incorporates microfiltration only. The membrane has performed significantly well revealing low fouling propensity and a steady effluent flow.

Nonetheless, the process can be enhanced further to reduce the operating cost and enhance the effluent quality. For instance, instead of introducing tobacco wastewater directly into the settling tank, it can be mixed with domestic water produced from the plant. This process will reduce the high COD content by diluting the industrial wastewater and introducing microorganisms that can initiate biodegradation. Additionally, a disinfection stage can be set up to further enhance the quality of the product water to achieve a level of quality that can permit the water to be recycled and used for processing instead of direct discharge to the sewage systems.

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Vita

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