Review

Energy-Efficient AnMBRs Technology for Treatment of Wastewaters: A Review

Wirginia Tomczak 1,*, and Marek Gryta 2,*

1 Faculty of Chemical Technology and Engineering, Bydgoszcz University of Science and Technology, 3 Seminarjna Street, 85-326 Bydgoszcz, Poland
2 Faculty of Chemical Technology and Engineering, West Pomeranian University of Technology in Szczecin, ul. Pułaskiego 10, 70-322 Szczecin, Poland
* Correspondence: tomczak.wirginia@gmail.com (W.T.); marek.gryta@zut.edu.pl (M.G.)

Abstract: In recent years, anaerobic membrane bioreactors (AnMBRs) technology, a combination of a biological reactor and a selective membrane process, has received increasing attention from both industrialists and researchers. Undoubtedly, this is due to the fact that AnMBRs demonstrate several unique advantages. Firstly, this paper addresses fundamentals of the AnMBRs technology and subsequently provides an overview of the current state-of-the-art in the municipal and domestic wastewaters treatment by AnMBRs. Since the operating conditions play a key role in further AnMBRs development, the impact of temperature and hydraulic retention time (HRT) on the AnMBRs performance in terms of organic matters removal is presented in detail. Although membrane technologies for wastewaters treatment are known as costly in operation, it was clearly demonstrated that the energy demand of AnMBRs may be lower than that of typical wastewater treatment plants (WWTPs). Moreover, it was indicated that AnMBRs have the potential to be a net energy producer. Consequently, this work builds on a growing body of evidence linking wastewaters treatment with the energy-efficient AnMBRs technology. Finally, the challenges and perspectives related to the full-scale implementation of AnMBRs are highlighted.

Keywords: anaerobic membrane bioreactor; biogas; ceramic membrane; energy demand; fouling; membrane process; polymeric membrane; separation; treatment; wastewater

1. Introduction

Wastewater is defined as “a combination of liquid or water-carried waste removed from residences, institutions, and commercial and industrial establishments, together with ground water, surface water and storm water” [1]. Depending on the source, it can be divided into the following types: municipal, domestic, industrial, medical, agricultural or nuclear. The amount of wastewaters produced in the world every year is equal to about 380 billion m$^3$ [2]. About 80% of them are released to the environment without being treated or reused [3,4].

Generally, untreated wastewater is a complex medium, rich in organic matters, nutrients, suspended solids (SS) and large number of both pathogenic and nonpathogenic microorganisms [5] which can cause major damage to the environment and human health. Nevertheless, wastewater characteristics depend on several interrelated factors, such as the quality of water used, conservation practice, culture of the population and the type of industry [6].

In light of recent events, it is increasingly recognized that wastewaters may be a source of clean water and a valuable resource for energy and nutrients [7–10]. Hence, the treatment of wastewaters and their reuse is becoming very important and necessary for the industries. Conventional wastewater treatment plants (WWTPs) consist of primary, secondary and advanced treatment processes with various biological, physical and chemical technologies [6,11–13]. They were designed to achieve a safe level based on wastewater use
or discharge location [14] in order to reduce water pollution and, thus, ensure human health and environmental protection [15]. Indeed, WWTPs provide the wastewaters treatment from chemical oxygen demand (COD), nitrogen (N) and phosphorous (P), as well as the production of an effluent free of pathogens [16]. Unfortunately, WWTPs contribute to the production and emission of CO₂, CH₄ and N₂O, resulting in the greenhouse effect [17–21], which was recognized as one of the major negative impacts on the environment. It was noted that each kWh of produced electricity leads to release of 0.9 kg CO₂ [22]. Worthy of note, CO₂ comes from the degradation of organics via aerobic and anaerobic processes [23] and the production of energy used for the plant operation [24]. The production of N₂O is related to nitrifying and denitrifying microorganisms, while CH₄ emission is associated with anoxic and anaerobic degradation of organic matter [20,25]. Notably, according to [26], wastewater treatment accounts for 5% of worldwide CH₄ emissions.

Moreover, within the past couple of decades, WWTPs have been designed without sufficient considerations on economic viability [27,28]. Consequently, they are highly energy intensive and classified as one of the most important energy consumers managed by municipalities [29]. It was documented that the water supply and wastewater treatment around the world accounts for more than 2% of global electricity consumption [30]. Furthermore, a report by the International Energy Agency International (IEA) [31] indicated that the amount of energy used in the water sector will more than double in the next 25 years. A careful literature review shows that, although there is no standardized approach to assessing the energy performance of WWTPs [16], several authors have attempted to determine a range of WWTPs energy demand. The term “energy demand” has come to be used to refer to the electricity consumption per volume of treated wastewater. For instance, in [32] it was indicated that WWTPs require energy in the range of 0.5–2.0 kWh/m³. In another study, Roccaro et al. [33] have pointed out that the energy consumption in classical activated sludge (CAS) systems is in the range from 0.4 to 0.7 kWh/m³ with an average value of 0.54 kWh/m³. A similar range of values, from 0.413 to 0.87 kWh/m³, was reported by the Electric Power Research Institute (EPRI) [34]. Recently, Guerra–Rodríguez et al. [35] have mentioned that the energy demand of WWTPs ranges from 0.45 to 1.25 kWh/m³. The noted differences may be attributed to the fact that the WWTPs energy requirements depend on several factors, including plant location and size, type of the treatment process and aeration system, as well as obtained effluent standard [22,27,29,36,37].

To solve these problems, an alternative approach is to apply the membrane bioreactors (MBRs). For the first time, the MBRs technology was used in the 1970s and 1980s for specific applications, such as ship bilge water and highly contaminated industrial waste [33]. Undoubtedly, MBRs offer energy-saving and environmentally friendly treatment of wastewaters. Therefore, in the industrial market, MBRs are called “the best available technology” for water reclamation [33]. Hence, in recent years, it has received increasing attention from both industrialists and researchers. Indeed, the increasing number of full-scale MBRs applications is being observed. It was documented that the global market for MBRs should grow from $2.5 billion in 2021 to $4.1 billion by 2026 [38]. With regard to the scientific community, data analysis on relevant publications in the discussed field have clearly showed that since 2011, a total of 10,611 research articles focused on MBRs were published. Moreover, the significant increase in the number of published articles was noted (Figure 1).
wastewaters consist of human waste and wastewater from household appliances [13]. Over population and economic growth [2]. Hence, the research on the treatment of municipal particularly effective in the treatment of such wastewaters as: municipal, e.g., [42,47], domestic, e.g., [48,49], pharmaceutical, e.g., [50,51], food, e.g., [52,53], textile, e.g., [54,55] and distillery, e.g., [56,57]. Among them, both municipal and domestic wastewaters are the most abundant. More specifically, municipal wastewaters originate from a domestic source and usually also contain commercial, industrial and institutional sources, while domestic wastewaters consist of human waste and wastewater from household appliances [13]. Over the last decade, the amount of municipal wastewaters produced annually in different countries has ranged from 730 million m$^3$ to 60,410 million m$^3$ [10]. Moreover, it is expected that the volume will continually increase in the coming years, mainly due to the growing population and economic growth [2]. Hence, the research on the treatment of municipal and domestic wastewaters is naturally concentrated.

This review is motivated by the progress that was made in the research and applications of AnMBRs in recent years. It is worth noting that several review articles on the MBR technology have recently been published [39,45,58–62]. Nevertheless, to the best of the authors knowledge, the present paper is the first to gain detailed insight into the status of AnMBRs as the energy-efficient technology for the treatment of municipal and domestic wastewaters. It starts with the fundamentals of the AnMBRs technology. The following sections demonstrate the application of AnMBRs for the treatment of municipal and domestic wastewaters. It starts with the fundamentals of the AnMBRs technology. The following sections demonstrate the application of AnMBRs for the treatment of municipal and domestic wastewaters. Finally, conclusions, as well as challenges and perspectives, including the framework for future research possibilities, are presented. Consequently, this

![Figure 1. The number of research papers focused on MBRs according to Science Direct. Keyword: membrane bioreactor. Data retrieved: 19 April 2022.](image-url)
and domestic wastewaters. Finally, conclusions, as well as challenges and perspectives, including the framework for future research possibilities, are presented. Consequently, this work builds on a growing body of evidence linking wastewaters treatment with the energy-efficient AnMBRs technology.

2. AnMBRs Technology

2.1. Advantages of AnMBRs

Nowadays, a common way to use microfiltration (MF) and ultrafiltration (UF) membranes for the wastewater treatment is to combine them into membrane bioreactor technology [63], which itself has proven effective in removing both organic and inorganic contaminants as well as microorganisms from various solutions and suspensions. In general, MBR is defined as a combination of a biological reactor and selective membrane process responsible for physical separation in a simplified single unit [59,64].

AnMBRs technology demonstrates several unique advantages. At first, although Kehrein et al. [65] indicated that membrane technologies for wastewaters treatment are costly in operation, an increasing number of studies have found that AnMBRs can be considered as the reliable and energy-efficient technology. For instance, Vinardell et al. [66] studied the economic feasibility of AnMBRs as a mainstream technology for the treatment of municipal wastewaters. The authors have clearly indicated that AnMBRs technology has a great potential to be an economically competitive option for full-scale implementation. It was found that AnMBRs have the potential to reduce the net energy requirements in comparison with aerobic-based processes applied in WWTPs. Questions regarding the energy demands of AnMBRs used for treating of municipal and domestic wastewaters are discussed in Sections 3.3 and 4.3, respectively.

Moreover, it was documented that AnMBRs offer total biomass retention and a high-quality produced effluent free of solids and pathogens which reduces the number of required unit operations [60]. In addition, AnMBRs are capable of producing an effluent rich in nutrients which may be used for other applications, such as agriculture irrigation [67–69]. Furthermore, AnMBRs technology ensures the high treatment capacity, small footprint and small reactor volume, minimum supervision, process flexibility toward influent changes and easy scale-up [20,56,70–72]. In addition, the AnMBRs provide low-sludge yield, resulting in sludge minimization. Indeed, it was determined that AnMBRs produce relatively less sludge by maintaining a high biomass concentration (typically 12 g/L) compared to activated sludge processes (4–6 g/L) [73]. Worthy of note, the produced sludge is stabilized and further fermentation is not required [47]. Finally, an important advantage of AnMBRs technology is high operation stability; hence, it can be applied for the treatment of wastewaters at extreme conditions, such as high salinity and poor biomass granulation [74]. Undoubtedly, the above mentioned advantages of AnMBRs provide a reduction in operating costs compared to the conventional wastewater purification plants [75]. It appears from the above discussion that anaerobic membrane-based processes have proven to be a more favorable method of wastewater treatment than conventional treatment processes [76]. The key highlight is therefore that AnMBRs technology is of emerging interest to the scientific community since it may ensure both environmental and economic benefits.

2.2. MF and UF Membranes

A membrane is defined as a selective barrier separating two phases and limiting the transport of various components [77]. A driving force may be generated by a gradient of pressure, concentration or electrical potential. A retention of microorganisms in the AnMBRs technology is achieved by the use of pressure-driven membrane processes, mainly MF and UF. Table 1 summarizes the basic characteristics of MF and UF membranes.

Generally, membranes with the pore size in the range from 0.01 μm to 1.0 μm are used for the wastewater treatments. Shin and Bae [78] have noted that in commercial applications, generally 2 m long membrane modules are used in order to provide the proper ratio of water productivity to the reactor footprint. It is worthy to mentioned that
among the most significant required characteristics of the membrane used in the wastewater treatment applications are: high resistance (acidic, basic, chemical and mechanical) over 5 years of operation [58]. It should be noted that although MF and UF are processes which provide constant treated water quality [79], information on the impact of the long-term operation of AnMBRs on the membrane characteristics and performance is rarely reported.

Table 1. The characteristics of MF and UF membranes used for the treatment of wastewaters in AnMBRs technology. Based on [63,76,80,81].

| Membrane        | MF                                      | UF                                      |
|-----------------|-----------------------------------------|-----------------------------------------|
| Separation mechanism | sieving                                | sieving                                |
| Pore size [µm]  | 0.1–1.0                                  | 0.01–0.1                                |
| Type            | porous, symmetric or asymmetric          | microporous, asymmetric                 |
| Transmembrane pressure [bar] | 0.1–2.0                                | 0.1–5.0                                |
| Example applications | clarification, pretreatment            | removal of macromolecules              |

Microfiltration and ultrafiltration are low-pressure processes operated through a sieving mechanism (Table 1). According to Darcy’s law, the water flux in MF and UF processes is proportional with the applied pressure. MF is one of the oldest pressure-driven membrane processes [82] which provides separation of contaminants with the size range 0.1–1 µm. Hence, it is widely used to separate suspended solids, bacteria and organic colloids from liquids. In turn, UF membranes have smaller pores (0.01–0.1 µm); thus, they have a wider separation range than MF and are widely used for the removal of macromolecules. Overall, MF and UF membranes provide for reducing the contaminants load in wastewaters and, additionally, may be successfully used for a feed pretreatment prior to the nanofiltration (NF) and reverse osmosis (RO) processes [83].

From the economics point of view, the selection of both membrane configuration and material is a key aspect. Indeed, it was reported that membrane cost accounts for about 46.4–72.3% of total capital cost for full-scale AnMBRs used for treating municipal wastewater [84]. As recognized in the literature, various configurations of membrane modules have been used in AnMBRs. The hollow fiber modules consist of a large number of membranes [85] where the water flows from the inside to the outside of the membranes under pressure or under vacuum [86]. Among the most significant advantages of the hollow fiber membranes are: a cost-effective fabrication process, a high area per unit volume, as well as capability to withstand various process conditions, such as backwashing [58,87]. Worthy of note, the hollow fiber membranes are most commonly used in submerged MBRs [88]. Flat sheet membranes are mostly immersed directly in mixed liquor with permeate drawn through the membranes using vacuum pumps [86]. They are characterized by good stability, as well as the ease of both cleaning and replacement [88]. Nevertheless, according to Anjum et al. [89], they seem to be more sensitive to the fouling phenomenon than hollow fiber membranes. Finally, tubular membranes are encased in pressure vessels and are predominantly used for external AnMBRs configurations [86]. They provide low fouling, easy cleaning and handling, as well as the ability to replace or plug a damaged membrane [88]. Notwithstanding, they are characterized by low packing density, high dead volume and high capital cost [58,88]. Regarding the characterization of tubular and hollow fiber modules, the outer and inner diameters, as well as the length, are the significant dimensions. In turn, for flat sheet modules, thickness, length and width are the key parameters [58].

The choice of membrane material depends both on the feed type and the overall cost [39]. From the literature survey, it became obvious that both polymeric and ceramic membranes are nowadays being used for the wastewaters treatment in AnMBRs. Therefore, it is important to contrast them in order to understand the advantages and disadvantages of both [90]. Polymeric membranes, also known as organic membranes, are the first choice for wastewater treatment in AnMBRs. Currently, the industry market based of polymeric
membranes is huge. The most commonly used are membranes fabricated from a number of inexpensive materials, such as: polyvinylidene fluoride (PVDF), polyethersulfone (PES) and polypropylene (PE). It is necessary to mention that PVDF and PES membranes present 75% of the total products on the market [91]. Although PVDF membranes are available in the whole range of pore sizes, PES and PE membranes are mostly available with the nominal pore sizes of 0.03 µm and 0.2–0.4 µm, respectively [92]. It important to point out that polymeric membranes play a significant role in AnMBRs technology since they offer several advantages such as great manufacturability, low cost for large-scale manufacture, high flexibility, good selectivity, easy processability and low energy requirements [93–97]. Nonetheless, a main drawback of polymeric membranes is the reduced chemical stability to high and low pH, which may limit the possibilities of module chemical cleaning [98].

Inorganic membranes have also gained interest. Generally, they include both ceramic and carbon-based membranes. Interestingly, Arumugham et al. [99], in a recently published review paper, indicated that 11.3% of the total annual growth rate for the ceramic membrane industry is forecasted in 2019–2027, corresponding to USD 10,893.5 million. This is due to the fact that ceramic membranes demonstrate higher thermal, chemical and mechanical stability compared to polymeric membranes [100–102], thus at least doubling the lifetime [103]. Hence, from an economical point of view, the use of ceramic membranes at the commercial scale is more acceptable than in the treatment of raw and heavily contaminated feeds [104]. In addition, they are excellent for processes performed under conditions requiring extremely high temperatures and cleaning in-place with the use of strong chemical agents [104]. An important point which should be noted is that due to the above-mentioned advantages, ceramic membranes can be backwashed without affecting their longevity adversely. Moreover, ceramic membranes are less susceptible to the deterioration by bacteria [104,105]. In addition, it was well documented that inorganic membranes are characterized by higher porosity and better separation properties [36]. Importantly, due to the hydrophilic nature, ceramic membranes provide a high permeate flux for the separation of an aqueous based liquid system [106]. In general, ceramic membranes have an asymmetrical structure [106] and consist of a microporous support layer and a meso- or microporous active layer [104]. Among the most commonly used materials for the manufacture of inorganic membranes used for the wastewaters treatment are alumina (Al₂O₃), silica (SiO₂), titania (TiO₂), zirconia (ZrO₂) and silicon carbide (SiC) [107–109]. It is important to note that the application of ceramic membranes in AnMBRs for the treatment of wastewaters is rarely studied. This is clearly related to their high capital cost. Several authors have pointed out that raw materials used for the production of ceramic membranes are expensive and their manufacture is complex [85,106,107,110]. For instance, Rani and Kumar [107] have mentioned that the average price of Al and Zr-based membranes is in the range of 500–3000 $/m², while the cost of polymeric membranes ranges from 20 to 200 $/m². Therefore, it can be indicated that with regard to the treatment of wastewaters in AnMBRs, the reduction of the cost of ceramic membranes will allow them to be economically profitable and more advantageous compared to polymeric ones [101]. Moreover, according to Ng et al. [100], the limited application of ceramic membranes is also due to the relatively poor control in pore size distribution.

2.3. Configurations of AnMBRs

The first generation of MBRs systems was based on the external (side-stream) configuration [87]. In this configuration, the membrane unit is operated independently of the reactor [111]. Indeed, wastewaters are pumped through the recirculation loop consisting of membranes placed outside the bioreactor leading to the complete retention of biomass [64,112,113] (Figure 2a). Consequently, the concentrated sludge is recycled back to the reactor, while the permeate is discharged. Usually, to provide the driving force, a crossflow velocity from 2 to 4 m/s along the membrane surface is used [64,74,114]. Undoubtedly, the advantage of this AnMBRs configuration is easy membrane replacement and the possibility of the module cleaning without interrupting the reactor operation [115].
The submerged (immersed) MBR system was introduced by Yamamoto et al. [116] in 1989, which led to a significant increase in the number of MBRs installations [43]. This technology is based on a membrane directly submerged inside the reactor (Figure 2b), which, consequently, comes into direct contact with the biomass [61]. In the submerged configuration, the membrane can be configured as vertical or horizontal hollow fibers, vertical flat plates or tubes [87]. The driving force is achieved by pressurizing the bioreactor or creating negative pressure on the permeate side [64].

Finally, in the external submerged system, the membrane is separated from the bioreactor and immersed in a tank filled with the biomass (Figure 2c) [61]. In this configuration, membranes are operated under a vacuum. Worthy of note, the low cross-flow velocity across membrane surface in submerged and external submerged reactors is related to the low permeate flux of hollow fiber and flat sheet membranes [117].

![Diagram of AnMBR configurations](image)

**Figure 2.** The configurations of AnMBRs: (a) External; (b) Submerged; (c) External submerged. Adapted from [45,61,64,72,118,119].

Comparison of the internal and external AnMBRs configurations is presented in Table 2. Importantly, the placement of the membrane module has the significant impact on the energy consumption in the long-term AnMBRs operation [120]. It was reported that the energy requirement for the operation of submerged AnMBRs is significantly lower than that for AnMBRs with an external membrane module [48,59,86,121]. For instance, Martin et al. [48] have presented a direct comparison of external and immersed membrane operation in both flocculated and upflow anaerobic sludge blanket (UASB) AnMBRs used for treating domestic wastewaters. It was found that the specific energy demand for the immersed configuration was equal to 0.3 kWh/m$^3$ of permeate produced, while for the pumped crossflow configuration it reached 3.7 kWh/m$^3$. This is related to the
fact that separating the membrane and bioreactor requires the use of recirculation loop installation [112,117], as well as the high cross flow velocity and transmembrane pressure (TMP) to maintain flux [72,122]. It must be recognized that the shear stress caused by such process conditions may have a negative effect on the microbial activity [123]. In addition, external AnMBRs require more space and the tanks more costly [61]. On the contrary, immersing the membranes into the bioreactor allows to eliminate the energy required for liquid recirculation [39,112]. It appears from the above discussion that the application of internal membranes allows to decrease the process costs. The key highlight is therefore that submerged AnMBRs stands out as a viable alternative to external AnMBRs for wastewaters treatment, and so they are more commonly applied [92,124].

2.4. Fouling

It is well known that the the economic feasibility of full-scale AnMBRs applications depends on the permeate flux levels that can be achieved [71]. Hence, it can be clearly indicated that the fouling phenomenon is one of the most important challenges. In overall, fouling is caused by the accumulation of the wastewater components on the membrane surface and/or its pores. It was generally accepted that it leads to a reduction in the permeate flux under constant transmembrane pressure (TMP) or to an increase in TMP under constant flux. Regarding the WWTPs, most of them are operated under constant permeate flux (ranging between 10 and 25 L/m²h [125]), thus, the membrane fouling is recognized as an increase in TMP [58]. Roughly speaking, it leads to several critical obstacle limiting the AnMBRs operation. Indeed, it was reported that the fouling causes low productivity, frequent membrane cleaning, increase in the energy requirement, reduced the membrane lifespan as well as an increased system downtime [59,88,121,126]. As reported by Shin and Bae [78] the energy demand for the fouling control accounts for more than 70% of the total energy consumption of AnMBRs technology.

From the viewpoint of the foulant components, membrane fouling can be classified as inorganic, organic and biofouling [127]. Inorganic fouling, also known as scaling, is caused by deposition of inorganic salts and crystals on the membrane surface. Organic fouling refers to the deposition of organic molecules which generally form a colloidal layer on the membrane surface [46]. In turn, biofouling corresponds to the interactions between biomass and the membrane [88]. With regards to the fouling mechanisms, membrane blocking include: complete pore blocking, intermediate pore blocking, pore blocking and surface deposition and cake formation. Complete pore blocking (Figure 3a) is due to particles which seal off pore entrances. Standard blocking (Figure 3b) is related to the blocking of the internal structure of the membrane by particles smaller than the membrane pore size. In intermediate blocking (Figure 3c), particles seal off pores and accumulate on the top of previously deposited particles. In turn, for the cake layer (Figure 3d), particles larger than the membrane pore size form a cake layer, permeability of which is affected by particle size, flux and electrostatic interactions [125]. Charfi et al. [128] used the model proposed by Hermia [129] to analyze the flux decline for MF and UF membranes used in AnMBRs. The above-mentioned authors found that when the membrane pore size is equal to or larger than 0.5 µm, the pores constriction is dominant. In turn, for membranes with pores smaller than 0.5 µm, the fouling occurs mainly due to the surface clogging. Although according to Dereli et al. [71] the cake layer is the most important fouling mechanism in AnMBRs,
existing research has shown that fouling is a very complex phenomenon. Hence, a single fouling mechanism may not be suitable for describing the phenomena occurring in the AnMBRs. All of the above described mechanisms may take place simultaneously [126].

**Figure 3.** The fouling mechanisms: (a) complete blocking; (b) standard blocking; (c) intermediate blocking; (d) cake layer. Based on [86,126,128,130].

The current knowledge on fouling phenomenon demonstrates that it is affected by many interrelated parameters, which can be therefore categorized into the following groups: membrane parameters (e.g., material type and pore size), process parameters (e.g., temperature and TMP) as well as feed characteristics (e.g., pH and salinity) [58,131] (Figure 4). Worthy of note, recently, many attempts have been made by researchers to investigate the influence of extracellular polymeric substances (EPS) on membrane fouling in AnMBRs [132–134]. This can be attributed to the fact that EPS is identified as a key parameter that has a negative impact on the module performance in AnMBRs. It should be kept in mind that, although great progress was made in recent years regarding investigations of the fouling phenomenon, it is still impossible to predict its degree in AnMBRs. This is mainly due to the variation in the wastewater characteristics, including the composition and pH, as well as the nature of the contaminants [135]. Nevertheless, since the costs of AnMBRs are centered on fouling, its mitigation is extensively investigated for commercial applications [39]. The most commonly used strategies of fouling control in AnMBRs technology include modification of membranes surface [136,137], feed pre-treatment [138], membranes cleaning [139], optimization of operational parameters [140] and a combination of the anaerobic–aerobic process [141].

**Figure 4.** The factors affecting fouling in MBRs. HRT—hydraulic retention time, SRT—solid retention time, TMP—transmembrane pressure, OLR—organic loading rate, MLSS—mixed liquor suspended solid, EPS—extracellular polymeric substance.
Readers interested in more information on the fouling phenomenon in AnMBRs technology are referred to the reviews which have been recently published [45,60,89,117,119,127].

3. Treatment of Municipal Wastewaters

3.1. Characteristics of Municipal Wastewaters

As was mentioned before, municipal wastewaters are the most abundant type of wastewater. Therefore, they are one of the largest sources of pollution, by volume and main contributor to the water pollution problems. Among the main contaminants in municipal wastewaters are organic micropollutants, which are of significant concern for water reuse purposes [45]. In turn, among the potential nutrient sources are carbon (i.e., carbohydrates and proteins), nitrogen (i.e., ammonia), phosphorus (i.e., dissolved orthophosphate), potassium and sulfur compounds [142]. Additionally, municipal wastewaters may be contaminated by heavy metals [143–145]. Since wastewaters may differ significantly in terms of physical and chemical properties [146], assessments of their characteristics are crucial before going ahead with the treatment process.

Generally, characteristics of municipal wastewaters is represented by the following parameters: total and soluble chemical oxygen demand (tCOD and sCOD, respectively), biochemical oxygen demand (BOD), total suspended solids (TSS), volatile fatty acids (VFA), volatile suspended solids (VSS), total nitrogen (TN), ammonium nitrogen, total phosphorous (TP), pH and alkalinity. Both BOD and COD are very useful in the pollution control of wastewaters since they measure the number of organic compounds in the sample. Indeed, COD is defined as an amount of oxygen which is required to oxidize the organic material present in wastewater, while BOD determines the amount of oxygen required by microorganisms to oxidize the biodegradable organic compounds [147]. Wastewaters can be classified in terms of COD and BOD concentrations as follows (COD and BOD concentration, respectively): weak (<400 mg/L and <200 mg/L), medium (400 mg/L and 350 mg/L), strong (1000 mg/L and 500 mg/L) and very strong (>1500 mg/L and >750 mg/L) [148].

Presented in the literature, physico-chemical characteristics of the municipal wastewaters treated in AnMBRs are summarized in Table 3. It was found that the studies regarding the application of AnMBRs for treating the municipal wastewaters have been performed for wastewaters with the wide range of COD and BOD. It was noted that COD and BOD varied from about 373 to 1729 mg/L and from about 160 mg/L to 694 mg/L, respectively. Therefore, according to the classification presented above, it can be clearly indicated that literature provides studies on the weak, medium as well as strong municipal wastewaters.

It was determined that municipal wastewaters treated in AnMBRs contained from 70 ± 9 to 964 ± 707 mg/L TSS. Also, the wide range of VFA and VSS concentrations were reported. Indeed, it was noted that VFA and VSS varied from 3.9 ± 2.5 to 100 ± 80 mg/L and from 51.2 ± 8.13 to 675 ± 651 mg/L, respectively. The concentration of TN was between 24.3 ± 5.3 to 175 ± 106 mg/L and TP from 2.1 ± 0.3 to 11.5 ± 0.6 mg/L, respectively. In turn, the concentration of ammonium nitrogen between 26.2 ± 2.5 and 75 ± 16 mg/L was reported. Generally, the pH of wastewaters was neutral to slightly alkaline. It should be pointed out that these findings are in line with the data presented in [2,7]. Qadir et al. [2], based on 53 municipal wastewaters quality datasets from across the world, estimated the average concentration of TN and TP equal to 43.7 mg/L and 7.8 mg/L, respectively. In turn, Liu et al. [7] indicated that the TP concentration in municipal wastewaters is in the range 6–15 mg/L, while the average concentration of ammonium nitrogen is 40 mg/L.
Table 3. The characteristics of real municipal wastewaters treated in AnMBRs.

| City, Country                | tCOD [mg/L] | sCOD [mg/L] | tBOD [mg/L] | TSS [mg/L] | VFA [mg/L] | VSS [mg/L] | TN [mg/L] | Ammonium Nitrogen [mg/L] | TP [mg/L] | pH | Alkalinity [mg CaCO₃/L] | Ref. |
|-----------------------------|-------------|-------------|-------------|------------|------------|------------|-----------|--------------------------|-----------|----|-------------------------|------|
| Elmhurst, U.S.              | NI          | 84 ± 21     | NI          | 120 ± 60   | NI         | 65 ± 35    | NI        | 27.3 ± 13.5              | NI        | 7.5 | ± 0.1                  | [40] |
| Valencia, Spain             | 445 ± 95    | 73 ± 25     | NI          | 186 ± 61   | 11 ± 7     | 150 ± 54   | NI        | 27.0 ± 8.1               | NI        | NI | 292.5 ± 37.2           | [44] |
| Valencia, Spain             | 650 ± 147   | NI          | NI          | 313 ± 45   | NI         | 257 ± 46   | NI        | 35 ± 3                   | NI        | NI | NI                      | [47] |
| Jumilla, Spain              | 1729 ± 914  | 372 ± 149   | NI          | 964 ± 707  | 100 ± 80   | 675 ± 651  | NI        | 56 ± 12                  | NI        | 8.2 | ± 0.3                  | [67] |
| Falconara Marittima, Italy  | 373 ± 148   | NI          | NI          | 232 ± 110  | NI         | 38.0 ± 7.0 | 31 ± 6    | 5.1 ± 1.5                | 7.7 ± 0.2 | NI | 494.7 ± 40.4           | [75] |
| Villavaquerin, Spain        | 892 ± 271   | 501 ± 165   | 573 ± 233   | 123 ± 35   | NI         | 110 ± 30   | 92 ± 12   | 71 ± 14                  | NI        | NI | 450–550                 | [120] |
| Villavaquerin, Spain        | 978 ± 210   | 610 ± 146   | 474 ± 203   | 83.00 ± 8.63 | NI   | 51.2 ± 8.13 | 92 ± 10   | 75 ± 16                  | NI        | 7.2 | ± 0.6                  | 494.7 ± 40.4 | [123] |
| Tagajo, Japan               | 422 ± 74    | NI          | 182 ± 67    | 197 ± 74   | NI         | 37.3 ± 1.3 | NI        | 2.1 ± 0.3                | 7.08 ± 0.08 | NI | NI                      | [149] |
| Tagajo, Japan               | 411.9 ± 87.8 | NI          | 159.8 ± 54.1 | 182.6 ± 71.3 | NI   | 37.3 ± 1.3 | NI        | 2.1 ± 0.3                | 7.14 ± 0.07 | NI | NI                      | [150] |
| Tagajo, Japan               | 408.0 ± 88.3 | NI          | 172.1 ± 60.4 | 194.2 ± 98.6 | NI   | 24.3 ± 5.3 | NI        | 8.5 ± 3.4                | 7.3 ± 0.3  | NI | NI                      | [151] |
| Ksour-Ensef, Tunisia         | 364.4 ± 94.8 | 129.3 ± 30.9 | 165.7 ± 46.5 | 228.4 ± 46.9 | NI   | 216.3 ± 45.6 | 44.5 ± 3.3 | 26.2 ± 2.5               | 6.4 ± 1.1  | 7.41 | ± 0.25                 | 201.8 ± 18.8 | [152] |
| Mexico City, Mexico         | 510 ± 87    | 104 ± 13    | 305 ± 37    | 342 ± 75   | 3.9 ± 2.5  | 79.5 ± 2.5 | 52.8 ± 5.4 | 42.8 ± 3.4               | 10.2 ± 2.5 | NI | 453.2 ± 34.6           | [153] |
| NI                          | 362.6 ± 46.0 | 150.1 ± 24.8 | -           | 177.2 ± 34.9 | NI   | 166.1 ± 32.8 | -         | 26.5 ± 4.8               | - 6.91 ± 0.17 | 177.1 ± 39.1  | [156] |

NI—no information; tCOD—total chemical oxygen demand; sCOD—soluble chemical oxygen demand; tBOD—total biochemical oxygen demand; TSS—total suspended solids; VFA—volatile fatty acids; VSS—volatile suspended solids; TN—total nitrogen; TP—total phosphorus.
3.2. Performance of COD Removal

Many attempts have been made by researchers to investigate the overall performance of AnMBRs technology in the treatment of municipal wastewaters (Table 4). Worthy of note, the published research results have been obtained on industrial, pilot and laboratory scales. Moreover, investigations have been carried out for both synthetic and real wastewater. An important point which should be noted is that the studies on the AnMBRs laboratory-scale may provide valuable information on the treatment performance of system and membrane module, however, the hydraulic conditions (such as shear rates and pressure drops) may differ significantly from these obtained at pilot and full scale system [157].

First of all, experimental results from several studies demonstrated that AnMBRs ensure the cell-free permeate, regardless of process parameters. Moreover, the excellent ability of the submerged and external AnMBRs technology for the removing of COD and BOD from both real and synthetic municipal wastewater was documented. For instance, Kong et al. [156] investigated the treatment of real municipal wastewater in a large pilot-scale submerged AnMBR (Table 4). The reactor was equipped with a total of 12 groups of PVDF hollow-fiber membrane units with a total area of 72 m². The process was conducted for 217 days under temperature and HRT equal to 25 °C and 6 h, respectively. The treatment capacity was equal to 20 m³/day. The authors have demonstrated that the used AnMBR ensured the high-quality effluent. The COD and BOD removal efficiency was over 90% and 95%, respectively. In turn, Saddoud et al. [154] have investigated the application of an anaerobic cross-flow UF membrane bioreactor for the treatment of raw domestic wastewaters (Table 3). The reactor was operated for 170 days under temperature of 37 °C and HRT equal to 15 h. It was determined that the system provided permeate free of microorganisms. Moreover, it was found that the AnMBR allowed to obtain the COD and BOD removal of 88% and 90%, respectively (Table 4).

With regards to synthetic feeds, in [115], the laboratory-scale AnMBR with the external PVDF membrane was used for treating a medium-strength synthetic municipal wastewater for 200 days under temperature and HRT equal to 35 ± 1 °C and 0.8 days, respectively (Table 4). It was demonstrated that the system produced a clear permeate with an average turbidity of 0.6 ± 0.2 NTU. Moreover, it was noted that an average COD removal efficiency was equal to 86.5 ± 6.4%

It is clear from that the membrane acts as a physical barrier in AnMBRs providing the accumulation of particulate COD inside the reactor and in the filtration section [67]. Through the studies discussed in this section, it can be concluded that the AnMBRs technology ensures the reliability and stability in the treatment of municipal wastewaters. However, the selection of suitable values of the process parameters is crucial for the AnMBRs performance. Moreover, performing a thorough literature review shows that the available studies have some limitations. Indeed, while research into the treatment of municipal wastewaters in AnMBRs equipped with polymer membranes is huge, the studies with the use of ceramic membranes are very limited [158,159] (Table 4).

3.2.1. Impact of Temperature

Currently, most of the anaerobic treatment processes are operated at mesophilic and thermophilic temperatures. However, it is well known that temperature is a key factor affecting the performance of AnMBRs. In literature, there is a number of studies attempted to correlate the feed temperature with the efficiency of the AnMBRs used for the treatment of municipal wastewaters. For instance, Ji et al. [149] successfully applied a submerged AnMBR for the long-term treatment of real municipal wastewater (Table 3). More specifically, the system was equipped with a hollow fiber PVDF membrane with a pore size of 0.4 μm. The process was conducted under HRT of 6 h and temperature in the range from 15 to 25 °C. It was documented that the used system provided the average removal of COD and BOD higher than 90% under temperature equal to 20 °C and 25 °C. In turn, it was noted that decreasing the temperature to 15 °C led to decreasing in COD and BOD removal efficiency to 77.4% and 81.9%, respectively (Table 4).
These observations are in good agreement with the recently published study of Rong et al. [152]. The authors investigated the impact of temperature on a pilot AnMBR performance used for treating a real municipal wastewater (Table 3). The system was equipped with a hollow fiber membrane made of PVDF with an average pore size of 0.4 µm. It was noted that the average COD removal efficiencies under 25 °C, 20 °C and 15 °C were equal to 90.0 ± 2.6%, 90.7 ± 2.6% and 84.5 ± 8.0%, respectively (Table 4). It can be inferred from these studies that the AnMBRs technology ensures a satisfactory effluent quality corresponding to the COD removal higher than 90% under the temperatures equal to and higher than 20 °C. This conclusion is in accordance with the results of another studies, wherein it was documented that, generally, at lower temperatures, the COD removal efficiencies is lower than 90% for both real [67,70,120,123,149,150,160,161] and synthetic [162] wastewaters (Table 4).

Obviously, the above-mentioned observations are attributed to the reduced maximum specific growth and lower microbial activity of anaerobic microorganisms at a lower temperature, which results in a decreased conversion rate of organic matter [157,163–166]. For instance, Ozgun et al. [157] demonstrated that a decrease in temperature leads to the reduction in microbial diversity. The authors have noted that a decreasing of temperature from 25 °C to 15 °C reduced the number of species and the phylogenetic diversity. However, according to [167], mesophilic sludge can become psycho-tolerant without losing mesophilic activity if an acclimation period of low temperature is performed.

Figure 5 shows the impact of the average feed temperature on the average COD removal in AnMBRs used for treating real and synthetic municipal wastewaters. All the data presented are based on the results published in the reviewed studies. It can be clearly observed that temperature has a significant impact on the COD removal. Indeed, the average COD removal higher than 90% was reported mainly for processes conducted under a temperature equal to or higher than 20 °C. In contrast, COD removal efficiencies higher than 90% at a temperature below 20 °C have rarely been reported [123,150].

From an economic point of view, it should be considered that temperature also has a very significant effect on both the methane yield and membrane performance. In several papers it was documented that high temperature enhances the methane production [47,67,152]. For instance, in [67] it was found that at temperature equal to 12.6 ± 1.4 °C, the biogas production in AnMBR used for treating municipal wastewater was not registered. The authors pointed out that this observation may be related to the fact that, generally, a decrease in temperature results in a decrease in microbial diversity within an anaerobic digestion community. Moreover, it must be recognized that decreasing the temperature leads to an increase in the methane solubility, which, in turn, causes a loss of biogas in the effluent.

With regards to the membrane performance, it was generally accepted that temperature has a very significant impact on the permeate flux. Indeed, results obtained by Ozgun et al. [157] clearly demonstrated that in terms of filtration performance, the AnMBR technology is not technically feasible for the treatment of municipal wastewater at 15 °C. The authors observed more severe fouling at 15 °C in comparison to 25 °C. Obviously, this is consistent with the fact that higher temperature results in decreased feed viscosity [168], higher mass transfer [169], reduced thickness of a cake layer [170] and finally, higher permeate flux.

To summarize, it should be indicated that this section highlights that the temperature is one of the key parameters in the operation of AnMBRs affecting the treatment performance, methane yield and membrane performance. Based on these findings, it can be deduced that the most favorable conditions temperature in AnMBRs applied for the treatment of municipal wastewater is equal to or higher than 20 °C.

3.2.2. Impact of Hydraulic Retention Time

In the literature, a large number of studies were focused on the impact of HRT on the AnMBRs performance. Obviously, this is due to the fact that higher HRT increases the contact time between microorganisms and substrate, which may lead to effective treatment
of the feed before leaving the reactor. Generally, in AnMBRs applied for the treatment of municipal wastewaters, HRT ranged from 0.4 h to 47 days (Table 4).

Baek et al. [41] studied the performance of a lab-scale AnMBR consisting of a bioreactor with a 10 L working volume connected to an external MF membrane module (Table 4). The authors demonstrated that the used system allowed for treating dilute municipal wastewater under a variety of operating conditions. Indeed, the AnMBR ensured a cell-free permeate and an average COD reduction of 64% over the applied HRT range (from 2 to 0.5 days). Interestingly, it was found that the process performance was not affected by HRT. Similar results were observed in another study [156], wherein a large pilot-scale submerged AnMBR was used for the treatment of municipal wastewater (Table 3) at 25 °C. It was reported that under HRT of 24 h, 12 h, 8 h and 6 h, the COD removal was equal to 89.5 ± 2.4, 90.0 ± 1, 93.2 ± 1.5 and 91.1 ± 0.8, respectively (Table 4). With regards to synthetic feeds, Ho and Sung [171] have investigated the performance of the AnMBR used for the treatment of synthetic municipal wastewater under temperature of 25 ± 1 °C and HRT equal to 12 h, 8 h and 6 h. It was determined that the quality of the obtained permeate was excellent, regardless of HRT values (Table 4). The above-mentioned results are in line with the findings presented in several other papers where it was documented that HRT did not significantly affect COD and BOD removal efficiencies from both real [40,75,120,151,172,173] and synthetic [174] urban wastewaters.

Figure 5 shows the impact of the average HRT on the average COD removal in AnMBRs used for treating real and synthetic municipal wastewaters. All the data presented are based on the results published in the reviewed studies. HRT was recognized as a parameter which does not significantly affect COD removal. These results are particularly important for economic viability of the process and, thus, its scale-up. It should be noted that in the full scale, the application of reduced HRT may be beneficial. Indeed, it may allow for a reduction of process costs due to: (i) smaller footprint, (ii) smaller sludge production, (iii) decreased energy demand and (iv) increased methane production [150,160]. For instance, in [150] it was reported that the energy demand of a submerged AnMBR applied for the treatment of mainstream municipal wastewater at 15 °C decreased from about 0.8 to 0.3 kWh/m³ of treated water with a decreasing of the HRT from 24 to 6 h.

Moreover, HRT is the parameter affecting the methane yield and membrane performance. For instance, in [171] it was determined that the reduction in HRT from 12 h to 6 h resulted in a reduction of the CH₄ fraction recovered from the synthetic municipal wastewater from 48 to 35%. As it was pointed out in the above-mentioned study, this observation was due to the increase of accumulated soluble COD in the reactor. In turn, it is well known that decreasing the HRT may enhance the accumulation of foulants in the feed [172], which is due to the fact that reduced HRT leads to a decrease in the fraction of degraded soluble microbial products which accelerates the membrane fouling phenomenon [175] and, consequently, leads to a decrease in the module performance. In [176], it was clearly indicated that the decrease of HRT may lead to an increase in specific fouling cake resistance and a decrease in particle size.
Figure 5. The impact of average temperature and HRT on the average COD removal during treatment of real and synthetic municipal wastewaters in AnMBRs. Results presented in [70,75,115,120,123,149–154,156,157,161,162,171,172,174,177–179].

3.2.3. Impact of Membrane Type

The studies devoted to investigating the effect of membrane pore size on the performance of AnMBRs used for the treatment of municipal wastewaters are limited. Ji et al. [151] used the AnMBRs with the 0.4 μm and 0.05 μm pore size membranes for treating real municipal wastewater (Table 3). The above-mentioned authors showed that COD and BOD removal efficiencies for both applied AnMBRs under ambient temperature (25 °C) and wide range of HRT (from 24 to 12 h) were very similar (Table 4). Therefore, it was concluded that in terms of suspended solids and organic matters removal, the AnMBRs performance was not affected by the pore size of the membranes. However, with regards to the energy consumption, a membrane with the pore size of 0.4 μm is more appropriate than with an 0.05 μm pore size. As indicated by the above-mentioned authors, the use of a membrane with the pore size of 0.05 μm requires higher suction pressure to overcome the higher filtration resistance during the treatment process.

In turn, Liu et al. [158] compared polymeric and ceramic membranes in a laboratory scale AnMBR applied for treating municipal wastewater (Table 3). For this purpose, a PVDF flat sheet membrane and Al₂O₃ ceramic flat-tubular membrane were used (Table 4). It was demonstrated that polymeric membrane was characterized by more severe irremovable fouling and a faster fouling rate. The authors indicated that this observation was due to the hydrophobic nature of the PVDF membrane, leading to high affinity of the material to organic foulants which accelerates pore clogging and formation of a gel layer.

Based on these findings, it can be deduced that the use of UF ceramic membranes is favored for AnMBRs applied for treating municipal wastewaters.
Table 4. Summary of the studies on the treatment of municipal wastewaters in AnMBRs.

| Wastewater Type | Origin | Scale | Configuration | AnMBR | Membrane | Process Conditions | Average Removal Performance |
|-----------------|--------|-------|---------------|-------|----------|-------------------|-----------------------------|
| municipal real   | industrial | external submerged | UF | hollow fiber | NI | T = 33–17 °C; HRT = 6–26 h | ~85 | [42] |
| municipal real   | industrial | external submerged | UF | hollow fiber | NI | T = 25–30 °C; HRT = 60 h; 24 h; 40 h; SRT = 190 d; 120 d; 70 d | 92 | [173] |
| municipal real   | semi-industrial | external | UF | hollow fiber | NI | T = 17–33 °C; SRT = 30–70 d | NI | NI | [47] |
| municipal real   | pilot | submerged | UF | hollow fiber | NI | T = 26–20 °C; HRT = 6–8 h; T = 10 °C; T = 25 °C | 85 ± 6.5%; 88 ± 9.7%; 94 ± 3% | NI | [67] |
| municipal real   | pilot | submerged | NI | flat sheet | PVDF | T = 23 ± 1 °C; 18 ± 4 °C; HRT = 2 d; 1 d; 0.5 d | from 69 ± 5 to 89 ± 2 | NI | [70] |
| municipal real   | pilot | submerged | UF | hollow fiber | PVDF | T = 18 ± 2 °C; HRT = 9.8–20.3 h | from 74.9 ± 9.9 to 90.0 ± 0.9 | NI | [123] |
| municipal real   | pilot | submerged | MF | hollow fiber | PVDF | T = 15 °C; HRT = 6–24 h; SRT = 20.7–515.7 d | 77.4; 90.5 | 81.9; 90.5 | [150] |
| municipal real   | pilot | submerged | MF | hollow fiber | PVDF | T = 15 °C; 20 °C; 25 °C; HRT = 8 h | 84.5 ± 8.0; 90.7 ± 2.6; 90.0 ± 2.6 | 92.5 ± 3.0; 90.9 ± 3.7; 92.4 ± 5.1 | [152] |
| municipal real   | large pilot | submerged | MF | hollow fiber | PVDF | T = 25 °C; HRT = 24 h; 12 h; 8 h; 6 h | 89.5 ± 2.4; 90.0 ± 1; 93.2 ± 1.5; 91.1 ± 0.8; 94.3 ± 1.9; 94.0 ± 0.9; 95.3 ± 1.4; 95.9 ± 0.6 | [156] |
| municipal real   | pilot | submerged | MF | tubular | PET | T = 15–20 °C; HRT = 2.6 h | 61.6 | NI | [160] |
| municipal real   | pilot | external submerged | UF | hollow fiber | NI | T = 33 °C; HRT = 20–6 h | 87 | NI | [44] |
| Wastewater   | AnMBR                | Membrane                  | Process Conditions                                                                 | Average Removal Performance | Ref. |
|-------------|----------------------|---------------------------|-------------------------------------------------------------------------------------|-------------------------------|------|
| municipal   | real pilot            | external submerged UF     | hollow fiber PVDF \(T = 18 \pm 2 \, ^\circ \text{C};\) \(\text{HRT} = 7-17.1 \, \text{h}\) | from 72.9 \pm 7.8 to 89.8 \pm 1.2 | [120]|
| municipal   | real pilot            | external submerged UF     | hollow fiber PVDF \(T = 23 \pm 1 \, ^\circ \text{C};\) \(\text{HRT} = 8.5 \, \text{h};\) \(\text{SRT} = 100 \, \text{d}; 70 \, \text{d}; 40 \, \text{d}\) | 88.0 \pm 3.9; 90.6 \pm 3.0; 91.8 \pm 1.9 | [172]|
| municipal   | real pilot            | external submerged UF     | hollow fiber PVDF \(T = 23 \pm 1 \, ^\circ \text{C};\) \(\text{HRT} = 8.5 \, \text{h};\) \(\text{SRT} = 70 \, \text{d}\) | 79.9 \pm 7.7; 93.7 \pm 2.0 | [177]|
| municipal   | real pilot            | external submerged MF     | hollow fiber NI \(T = 35 \, ^\circ \text{C};\) \(\text{HRT} = 2.2 \, \text{h}\) | 87 | [178]|
| municipal   | real pilot            | external submerged UF     | flat sheet PES \(T = 35 \, ^\circ \text{C}, 28 \, ^\circ \text{C}; 20 \, ^\circ \text{C}\) | \sim 90 | [180]|
| municipal   | real pilot            | external UF               | hollow fiber NI \(T = 30 \, ^\circ \text{C};\) \(\text{HRT} = 5 \, \text{h}\) | 83 \pm 1 | [75]|
| municipal   | real pilot            | external UF               | hollow fiber NI \(T = 27 \, ^\circ \text{C};\) \(\text{HRT} = 24.4 \pm 0.4 \, \text{h}\) | 88.0 \pm 4.3 | [153]|
| municipal   | real pilot            | external UF               | tubular PVDF \(\text{HRT} = 6 \, \text{h}\) | 92 | [155]|
| municipal   | real pilot            | external UF               | hollow fiber PVDF \(T = 12.5 \pm 1.8 \, ^\circ \text{C}; 13.1 \pm 1.8 \, ^\circ \text{C},\) \(\text{HRT} = 8 \, \text{h}\) | 76 \pm 4; 89 \pm 4 | 89 \pm 5; 91 \pm 2 | [161]|
| municipal   | real mini-pilot       | submerged MF               | hollow fiber PVDF \(T = 15 \, ^\circ \text{C}; 20 \, ^\circ \text{C}; 25 \, ^\circ \text{C};\) \(\text{HRT} = 6 \, \text{h}\) | 77.4 \pm 4.4; 90.0 \pm 1.8; 90.1 \pm 1.6 | 81.9 \pm 8.5; 92.2 \pm 2.4; 91.4 \pm 1.7 | [149]|
| municipal   | real laboratory       | submerged MF               | hollow fiber NI \(T = 25 \, ^\circ \text{C};\) \(\text{HRT} = 12-24 \, \text{h}\) | from 88.1 \pm 1.7 to 89.0 \pm 2.7 | from 91.4 \pm 4.3 to 93.9 \pm 1.3 | [151]|
| municipal   | real laboratory       | submerged UF               | hollow fiber NI \(T = 25 \, ^\circ \text{C};\) \(\text{HRT} = 12-24 \, \text{h}\) | from 88.9 \pm 2.1 to 89.5 \pm 3.0 | from 92.4 \pm 2.3 to 94.0 \pm 1.3 | [151]|
| municipal   | real laboratory       | submerged UF               | flat sheet PVDF \(T = 30 \pm 3 \, ^\circ \text{C};\) \(\text{HRT} = 10 \, \text{h}\) | 90 | NI | [179]|
| Wastewater | AnMBR | Membrane | Process Conditions | Average Removal Performance | Ref. |
|------------|-------|----------|--------------------|-----------------------------|-----|
| municipal  | real  | laboratory | external submerged | UF, hollow fiber, PVDF | T = 23 ± 1 °C; HRT = 12.5 h; SRT = 40 d; 70 d | 92.3 ± 1.5 | 94.6 ± 2.2 | [172] |
| municipal  | real  | laboratory | external | MF, tubular, PVDF | T = 25 °C; HRT = 0.5–2 d; SRT = 19–233 d | from 55 ± 18 to 68 ± 8 | NI | [40] |
| municipal  | real  | laboratory | external | MF, NI, NI | HRT = 0.5–2 d, SRT = 40–217 d | from 55 to 72 | NI | [41] |
| municipal  | real  | laboratory | external | UF, NI, NI | T = 37 °C; HRT = 15 h | 88 | 90 | [154] |
| municipal  | syntethic | pilot | submerged | MF, hollow fiber, PP | T = 22 ± 2 °C; SRT = 10–40 d | from 98.13 to 99.35 | NI | [181] |
| municipal  | syntethic | semi-pilot | submerged | MF, flat sheet, PVDF | T = 37 ± 1 °C; HRT = 47 d | 98.84; 98.75 | NI | [182] |
| municipal  | syntethic | laboratory | external submerged | MF, flat sheet, PVDF | T = 35 °C; HRT = 7 h | NI | NI | [158] |
| municipal  | syntethic | laboratory | external submerged | MF, flat tubular, ceramic | T = 35 °C; HRT = 7 h | NI | NI | [158] |
| municipal  | syntethic | laboratory | external | MF, tubular, PVDF | T = 35 ± 1 °C; HRT = 0.8 d | 85 ± 8.9 | NI | [115] |
| municipal  | syntethic | laboratory | external | UF, tubular, PES | T = 15 °C; 25 °C; HRT = 6 h | 90; 92% | NI | [157] |
| municipal  | syntethic | laboratory | external | MF, tubular, ceramic | T = 35 °C; HRT = 0.4–13.8 h | NI | NI | [159] |
| municipal  | syntethic | laboratory | external | MF, tubular, PTFE | T = 25 ± 1 °C; 15 ± 1 °C; HRT = 12 h | >95; >85 | NI | [162] |
| municipal  | syntethic | laboratory | external | MF, tubular, PTFE | T = 25 ± 1 °C; HRT = 6 h; 8 h; 12 h | 93.8 ± 1.9; 95.9 ± 1.0; 95.0 ± 0.8 | NI | [171] |
Table 4. Cont.

| Wastewater Type | Origin | Scale | Configuration | Membrane Type | Module | Material | Process Conditions | COD [%] | BOD [%] | Ref. |
|-----------------|--------|-------|---------------|---------------|--------|----------|--------------------|---------|---------|------|
| municipal        | syntethic | laboratory | external       | UF            | hollow fiber | PVDF     | T = 35 ± 1 °C; HRT = 6 h; 12 h | ~98     | NI      | [174] |
| municipal        | syntethic | laboratory | external       | MF            | flat sheet  | PVDF     | T = 35 °C; HRT = 26 h | from 90 to 96 | NI      | [175] |
| municipal        | syntethic | laboratory | external       | MF            | flat sheet  | PES      | T = 35 °C; HRT = 26 h | from 90 to 96 | NI      | [175] |
| municipal        | syntethic | laboratory | external       | UF            | tubular     | PVDF     | HRT = 4 h; 8 h; 12 h | ~80     | NI      | [176] |
| municipal        | syntethic | laboratory | external       | UF            | flat sheet  | PVDF     | HRT = 2.4 ± 0.6–3.6 ± 1.1 d | from 90.9 ± 6.0 to 95.9 ± 0.7 | NI      | [183] |

NI—no information; PE—polyethylene; PES—polyethersulfone; PET—polyethylene terephthalate; PVDF—polyvinylidene fluoride; PTFE—poly-tetrafluoroethylene; \(^1\) pore size < 0.1 \(\mu\)m; \(^2\) sCOD removal.
3.3. Energy Demand

There is a clear evidence that the economic feasibility of membrane-based processes is of great importance for the pilot- and full-scale plants [184]. In determining the cost of MBRs plants, energy consumption is one of the most important factors [185].

Some studies have demonstrated that biogas production could be observed in AnMBRs during treatment of municipal wastewaters. Indeed, biogas may be produced via organic matter degradation by methanogens through anaerobic digestion [26,150,186–188]. Generally, the biogas generated is a combination of methane, carbon dioxide, and trace gases. Therefore, CH$_4$ can be recovered (both as biogas and dissolved in effluent) and be used to cover a part of the energy demand of the treatment process, leading to a need to meet little or no energy requirements [172,177]. Seco et al. [153] demonstrated that considering the calorific power of methane (38,000 KJ/m$^3$) and the typical efficiency of a combined heat and power device (35%), the specific energy produced by the biogas in the AnMBR pilot plant would be 0.073 kWh/m$^3$ of treated water. In another study [156], it was found that the energy recovered from both biogas and dissolved CH$_4$ produced in the AnMBR used for the treatment of municipal wastewater can be up to 0.347–0.389 kWh/m$^3$.

With regards to the AnMBRs energy demands, Ferrer et al. [189] have proposed guidelines for designing submerged AnMBR used for treating sulphate-rich (100 mg SO$_4$–S/L) and low-sulphate (10 mg SO$_4$–S/L) municipal wastewaters. It was demonstrated that the energy requirements of AnMBRs technology depend on the methane capture and feed type. Indeed, it was presented that when no energy is recovered from methane, the energy demand of AnMBRs used for treating sulphate-rich municipal wastewater is equal to 0.22 kWh/m. In turn, when energy is recovered from methane, the energy of 0.14 kWh/m$^3$ is required. In addition, the authors showed that the energy requirements for the treatment of low-sulphate municipal wastewater are significantly lower and equal to 0.21 kWh/m$^3$ and 0.07 kWh/m$^3$ for the two aforementioned scenarios, respectively. However, it should be mentioned that in the discussed study, the energy needed for nutrient removal was not taken into account. Similar results have been obtained in [47], wherein it was demonstrated that the theoretical maximum energy production during the treatment of low-sulphate municipal wastewater (10 mg SO$_4$–S/L) in AnMBRs is equal to 0.11 kWh/m$^3$.

What becomes apparent from these studies is that the AnMBRs technology used for treating of low-sulphate municipal has the potential to be a net energy producer.

With regards to the treatment of sulphate-rich municipal wastewaters (105 mg SO$_4$–S/L) in AnMBRs, in the above-mentioned study [47] the energy requirements were calculated for the three following different conditions of methane productions: (i) $T = 33$ °C and SRT = 70 days, (ii) $T = 22$ °C and SRT = 38 days, (iii) $T = 17$ °C and SRT = 30 days and two different levels of energy recovery: (i) from biogas methane, (ii) from biogas methane and methane dissolved in the effluent. The obtained results clearly demonstrated that reduction in the energy requirements of AnMBRs used for treating sulphate-rich municipal wastewaters can be achieved when CH$_4$ is captured and used as the energy resource. For instance, it was shown that for the process performed under SRT of 30 days and a temperature equal to 17 °C, the energy demand of AnMBR is equal to 0.19 kWh/m$^3$ when the methane is not captured, while it decreased to 0.17 kWh/m$^3$ when using the methane as the energy source.

Figure 6 shows the reported in the literature [47,190,191] net energy demand of submerged AnMBRs used for treating sulphate-rich municipal wastewater under wide ranges of the process parameters: temperature (17–33 °C), SRT (30–70 days), TMP (0.09–0.35 bar) and the 20 °C-standardized critical fluxes ($J_{20}$) (9–20 L/m$^2$h). The data presented in the above-mentioned studies have been obtained by assuming that the energy recovery is conducted from biogas methane and methane dissolved in the effluent. For the comparison, the energy demand of WWTPs (from 0.45 to 1.25 kWh/m$^3$) reported by Guerra–Rodríguez et al. [35] have been used.
Figure 6. Net energy demand of submerged AnMBRs is used for the treatment of sulphate-rich municipal wastewaters. Assumption: energy recovery from biogas methane and methane dissolved in the effluent. Based on [47,190,191]. n/a—no information; T—temperature; SRT—solid retention time; TMP—transmembrane pressure; $J_{20}$—20 °C-standardized critical fluxes.
The presented results reveal that the energy demand in AnMBRs ranges from 0.06 to 0.47 kWh/m$^3$. It can be inferred from this investigation that for most of the analysed process parameters, the energy demand of AnMBRs are significantly lower than those reported for WWTPs. Moreover, it is important to highlight that the lowest (<0.1 kWh/m$^3$) and the highest (≥0.44 kWh/m$^3$) values of the AnMBRs energy requirement have been reported for AnMBRs operated under strictly defined, narrow ranges of the process parameters. Indeed, it was found that the lowest AnMBRs energy demand can be obtained under temperature of 25 °C and 33 °C, SRT of 40 days and 70 days, TMP equal to 0.09 bar to 0.2 bar and $J_{20}$ between 15 L/m$^2$h and 20 L/m$^2$h. Contrary, the highest value of the AnMBRs energy demand have been noted for temperature of 17 and 22 °C, SRT of 30 and 38 days, TMP of 0.26 bar and 20 °C-standardized critical fluxes ($J_{20}$) of 20 L/m$^2$h. Therefore, it can be indicated that performing the process under the above-mentioned conditions may have the positive impact on the economic viability of the AnMBRs technology. Based on these results, it can be concluded that conducting the treatment process under: (i) high ambient temperature and high SRT, (iii) low TMP and (iii) high $J_{20}$ is favored for AnMBRs. This observation can be explained by the fact that high temperature and high SRT lead to increase biogas production, hence, more energy can be recovered from CH$_4$ capture [189]. With regard to TMP, low values ensure a reduction in the energy consumption and intensity of the fouling phenomenon. Finally, operating the AnMBR at high $J_{20}$ allows to reduce the energy needed to scour the membrane with biogas [189]. Based on the above discussed results it should be indicated that AnMBRs can be considered as the reliable, energy-efficient potential technology for treating municipal wastewaters.

4. Treatment of Domestic Wastewaters

4.1. Characteristics of Domestic Wastewater

Domestic wastewater consists mainly of nutrients, biodegradable organic matter, pathogenic microorganisms and various organic micropollutants excreted by humans [13]. Generally, this type of wastewater is characterized by low biodegradability, high fractions of suspended solids and strong fluctuations in both hydraulic and organic loading rates [192,193]. For instance, the significant fluctuations in the composition of domestic wastewater during the treatment in AnMBRs was reported by Saddoud et al. [194].

Table 5 shows the presented in the literature characteristics of real domestic wastewater treated in AnMBRs. The analysis of the recorded data allowed to determine both the similarities and differences to the characteristics of urban wastewaters (Table 3). Firstly, likewise to municipal wastewaters, the wide ranges of COD, BOD and TSS concentration have been noted. Indeed, COD and BOD from 269 to 939.2 ± 53.2 mg/L and from 160 to 315 mg/L have been reported, respectively. In turn, TSS between 84 and 600 ± 210 mg/L was noted. Worthy of note, in [192] it was indicated that 50% of the COD in domestic sewage consists of suspended solids. It can be concluded that the literature provides studies on the AnMBRs performance in treating domestic wastewater classified as weak, medium and strong wastewater. Importantly, contrary to municipal wastewaters, no data on the VFA concentration were recorded. In turn, it was determined, that the domestic wastewaters used as a feed in AnMBRs technology were characterized by VSS concentration in the range from 130 to 286 mg/L. This range is much smaller than that reported for municipal wastewaters. In addition, it was reported that the concentration of N and P was in the range of 27.30 ± 1.30 to 166 mg/L and from 2.10 ± 0.27 to 52.5 mg/L, respectively. Therefore, it can be clearly indicated that the domestic wastewaters were characterized by a much wider range of phosphorus concentration values than municipal wastewaters. In turn, the concentration of ammonium nitrogen between 21.59 and 44.65 mg/L was reported. Likewise, to municipal wastewaters, the pH of domestic wastewaters was neutral to slightly alkaline.
Table 5. The characteristics of real domestic wastewaters treated in AnMBRs.

| City, Country       | tCOD [mg/L] | sCOD [mg/L] | tBOD [mg/L] | TSS [mg/L] | VSS [mg/L] | TN [mg/L] | Ammonium Nitrogen [mg/L] | TP [mg/L] | pH   | Ref. |
|---------------------|-------------|-------------|-------------|------------|------------|-----------|--------------------------|-----------|------|------|
| Beer-Sheva, Israel  | 939.2 ± 53.2| 706.3 ± 14.1| NI          | 600 ± 210  | NI         | 82.4 ± 3.74| NI          | 9.60 ± 0.70 | 7.00 ± 0.2 | [49] |
| Tagajo, Japan       | 422 ± 74    | NI          | 182 ± 67    | 197 ± 74   | NI         | 27.30 ± 1.30 | NI          | 2.10 ± 0.27 | 7.16 ± 0.17 | [195] |
| Beijing, China      | 269–712     | NI          | NI          | 197 ± 74   | NI         | 21.59–44.65 | 4.17–5.88  | 6.95–7.10   | [196] |
| Tunis, Tunisia      | 670         | NI          | 180         | 288        | 180        | 49.35     | NI          | 10.40      | 7.62   | [194] |
| Tunis, Tunisia      | 900         | NI          | 280         | 540        | 130        | 57.00     | NI          | 16.00      | 7.70   | [194] |
| Tunis, Tunisia      | 419         | NI          | 160         | 220        | 200        | 51.47     | NI          | 52.50      | 7.80   | [194] |
| Ksour Essef, Tunisia| 786         | NI          | 315         | 377        | 286        | 166.00    | NI          | 11.79      | 7.23   | [194] |
| Tunis, Tunisia      | 663 ± 240   | NI          | 206.66 ± 64 | 350 ± 160  | 170 ± 36   | 52.6 ± 3.95| NI          | 26.30 ± 22.86 | 7.70 ± 0.09 | [197] |
| Singapore           | 330.4 ± 89.8| 68.2 ± 47.6 | NI          | 341.1 ± 94.9| NI         | 68.2 ± 10.2| 32.7 ± 5.5  | NI         |      | [198] |

NI—no information; tCOD—total chemical oxygen demand; sCOD—soluble chemical oxygen demand; tBOD—total biochemical oxygen demand; TSS—total suspended solids; VSS—volatile suspended solids; TN—total nitrogen; TP—total phosphorus.
4.2. Performance of COD Removal

Information on the use of AnMBRs technology for treating domestic wastewaters available from the literature comes from experiments performed on the pilot and laboratory scale studies (Table 6). Worthy of note, published studies have been carried out for both synthetic and real domestic wastewaters.

The bibliographic investigation showed that both submerged and external AnMBRs are an excellent technology for the treatment of domestic wastewaters. For instance, in [111], the performance of pilot scale external side-stream AnMBR equipped with the MF hollow fiber module was investigated (Table 6). The treatment process was conducted under 25 °C and HRT of 6 h for three years. It was found that the used membrane ensures the retention of biomass and influent particulate matter in the bioreactor. Moreover, it was determined that the system provides an average COD removal equal to 88%. In turn, with regards to synthetic feed, Sanchez et al. [199] investigated the treatment process in the submerged AnMBR with an ultrafiltration PES flat sheet membrane (Table 6). The experiments have been carried out under temperature and HRT of 25 °C and 13 h, respectively. The above-mentioned authors clearly demonstrated that the membrane led to the retention of biomass, suspended solids and organic matter. The COD removal of 92.3% was achieved.

It should be pointed out that, as in the case of municipal wastewaters, the studies focused on the treatment of domestic wastewaters in AnMBRs have some limitations. Indeed, although the performance of AnMBRs equipped with polymer membranes was investigated by many researchers, the studies with the use of ceramic membranes are very limited [134,198,200–202] (Table 6).

It is necessary to mention that likewise to the treatment of municipal wastewaters, regarding the full-scale implementation of the AnMBRs technology, the impact of process parameters on system performance should be considered.

4.2.1. Impact of Temperature

Performing a comprehensive review allows for indicating that the studies devoted to examine the impact of temperature on the performance of AnMBRs used for the treatment of domestic wastewater were rarely reported in the literature for both real [203] and synthetic feeds [204,205]. The temperature of AnMBRs used for treating domestic wastewater was commonly reported to be in the range of 25–35 °C (Table 6).

In [203], the influence of temperature on the COD removal in a laboratory scale AnMBR, coupled with hollow fiber membranes made of PET, was studied (Table 6). It was found that the membrane module provided effective retention of biomass in the bioreactor. Moreover, it was determined that the COD removal was affected by feed temperature in the range from 12 °C to 27 °C. Indeed, at the temperature of 12 °C, the COD removal of 88% was noted. Under a temperature ≥17–18 °C, an efficiency of around 99% was achieved. In turn, Chu et al. [204] studied the performance of an expended granular sludge bed reactor coupled with a hollow fiber membrane submerged in the upper part of the reactor. As a feed, synthetic domestic wastewater was used. The treatment process was carried out at gradually reduced temperature from 25 °C to 20 °C, 15 °C and 11 °C and three values of HRT equal to 3.5 h, 4.6 h and 5.7 h. The above-mentioned authors have demonstrated that temperature has the significant impact on the COD removal under each of the HRT applied. For instance, it was reported that under HRT of 3.5 h, the decrease in the temperature from 25 °C to 11 °C led to a decrease in the COD removal from 92.8% to 76%. In turn, it was noted that under HRT of 5.7 h, the same reduction of the temperature led to decrease in the COD removal from 95.7% to 80.9%. It was indicated that the dependence effluent quality on the temperature is related to the fact that temperature affects the rates of the anaerobic conversion processes. In another paper, Smith et al. [205] studied the performance of AnMBR used for the treatment of synthetic domestic wastewater at psychrophilic temperatures. For this purpose, the process was conducted under the following values of process parameters (temperature, HRT): 15 °C and 17 ± 0.79 h, 12 °C and 17 ± 1.0 h, 9 °C and 19 ± 1.3 h, 6 °C and 26 ± 3.5 h, 3 °C and 29 ± 2.2 h, respectively.
It was noted that the average value of the COD removal was equal to 95 ± 1.6%, 95 ± 1.1 and 96 ± 1.8% at a temperature of 12 °C, 9 °C and 6 °C, respectively. However, at the lowest temperature (3 °C), a significant reduction in COD removal was observed (86 ± 4.0%). It was pointed out that the suspended biomass activity declined at a lower temperature and at 3 °C it was responsible only for hydrolysis of the particulate COD. These noteworthy results point to the possibility of obtaining the high efficiency of COD removal (>90%) at psychrophilic temperatures. However, it should be mentioned that the above discussed results have been achieved for the synthetic feed. In another study [206], it was shown that the feed type has the significant impact of the AnMBRs performance. Indeed, the COD removal during the treatment of synthetic domestic wastewaters in AnMBR at psychrophilic conditions (15 °C) was equal to 92 ± 5%, while for the real feed it was significantly lower and equal to 69 ± 10% (Table 6). In the above-mentioned study, it was indicated that the noted difference was due to the lower strength of the real wastewater (259 ± 82 mg/L) compared to the synthetic one (440 ± 68 mg/L). Therefore, it can be indicated that further research on the AnMBRs performance in the treatment of various strength real domestic wastewaters under psychrophilic temperatures is required.

To summarize, it should be pointed out that during the treatment of domestic wastewaters in AnMBRs, a decrease of the COD removal efficiency with decreasing the temperature is expected, which itself is related to the fact that, at lower temperatures, the activity of microorganisms decreases. This conclusion is in line with the results presented in Figure 7, which shows the impact of the average feed temperature on the average COD removal in AnMBRs used for the treatment of real and synthetic domestic wastewaters. All the data presented are based on results published in the reviewed studies.

Figure 7. The impact of average temperature and HRT on the average COD removal during treatment of real and synthetic domestic wastewaters in AnMBRs. Results presented in [111,134,193,195,199,202,204,206–209].
## Table 6. A summary of the studies on the treatment of domestic wastewaters in AnMBRs.

| Wastewater Type | Origin | Scale | Configuration | Membrane Type | Module | Material | Process Conditions | Average Removal Performance | Ref. |
|-----------------|--------|-------|---------------|---------------|--------|----------|---------------------|----------------------------|------|
| domestic        | real   | pilot | submerged     | UF            | hollow fiber | PVDF      | without external temperature control, HRT = 16 h | COD [%] | BOD [%] | |
|                 |        |       |               |               |         |          | 84                  | 93            | [48]  |
| domestic        | real   | pilot | external      | UF            | hollow fiber | PVDF      | without external temperature control, HRT = 16 h | 86     | 95     | [210] |
|                 |        |       |               |               |         |          | 88                  | NI            |      |
| domestic        | real   | pilot | external      | MF            | hollow fiber | NI        | T = 37 °C          | >76            | >84   |      |
|                 |        |       |               |               |         |          | 88                  | NI            |      |
| domestic        | real   | mini-pilot | submerged | MF          | hollow fiber | PVDF      | T = 25 °C; HRT = 6 h, SRT = 53 d, 65 d, 76 d, 88.4 ± 3.7; 87.9 ± 7.4; 86.3 ± 9.7 | 87.3; 92–94 |      | [195] |
| domestic        | real   | laboratory | submerged | MF; UF   | NI        | ceramic  | T = 25–30 °C; HRT = 7.5 h, SRT = 60 d | 88.6 ± 9.0; 87.9 ± 7.4; 86.3 ± 9.7 | NI    | [198] |
|                 |        |       |               |               |         |          | 60–95               | 69 ± 10       |      | [203] |
| domestic        | real   | laboratory | submerged | MF        | flat sheet | PES      | T = 15 °C, HRT = 16–24 h | 60–95         | NI    | [206] |
|                 |        |       |               |               |         |          | 80.23; 82.69; 78.19 | 78.19         |      | [207] |
| domestic        | real   | laboratory | submerged | MF        | NI       | NI       | T = 35 ± 1 °C; HRT = 6 h, 88.9 to 88.2 | 88.9 to 88.4 | NI    | [208] |
|                 |        |       |               |               |         |          | from 88.9 to 88.2 | 88.9 to 88.2 |      | [208] |
| domestic        | real   | laboratory | external submerged | MF    | flat sheet | ceramic  | T = 30 ± 3 °C; HRT = 17 h, SRT = 30 d | 88.97 ± 4.13; 91.33 ± 4.24 | 77.38 ± 8.19; 80.27 ± 7.79 | [134] |
|                 |        |       |               |               |         |          | 88.97 ± 4.13; 91.33 ± 4.24 | 77.38 ± 8.19; 80.27 ± 7.79 |      | [140] |
| Wastewater Type | Origin | Scale | Configuration | Membrane Type | Membrane Module | Membrane Material | Process Conditions | Average Removal Performance |
|-----------------|--------|-------|---------------|---------------|-----------------|-------------------|-------------------|----------------------------|
| domestic        | real   | laboratory | external, submerged | MF | frame | PES | T = 25–30 °C; HRT = 10 h; SRT = 30 d; 60 d; 90 d | 84 ± 6; 85 ± 3; 86 ± 3 | NI [193] |
| domestic        | real   | laboratory | submerged | UF | flat sheet | PES, PAN, PVDF | T = 25 ± 0.3 °C | 89.7 ± 3 | NI [49] |
| domestic        | real and synthetic | pilot | external | UF | NI | NI | T = 37 °C | NI | NI [197] |
| domestic        | synthetic | pilot | external | MF | tubular | ceramic | T = 35 ± 1 °C; HRT = 4 d | 84 | NI [200] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | alumina-based ceramic | HRT = 44 ± 3.1 h; 18 ± 1.3 h | 90.5 ± 6.8; 96.1 ± 5.1 | NI [201] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | pyrophyllite-based ceramic | HRT = 43 ± 1.6 h; 18 ± 1.6 h | 92.9 ± 5.5; 42.6 ± 19.2 | NI [201] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | alumina-based ceramic | T = 33 ± 2 °C; HRT = 28 h | 91.0 ± 13.8 | NI [202] |
| domestic        | synthetic | laboratory | submerged | UF | flat sheet | PVDF | T = 33 ± 2 °C; HRT = 22.5 h | 77.8 ± 20.5 | NI [202] |
| domestic        | synthetic | laboratory | submerged | MF | hollow fiber | PE | T = 25; 20; 15; 11 °C; HRT = 3.5; 4.6 and 5.7 h | from 76 to 96 | NI [204] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | PES | T = 15–3 °C; HRT = 17 ± 0.79–29 ± 2.2 h | from 95 ± 1.6 to 86 ± 4.0 | NI [205] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | PES | T = 15 °C; HRT = 16–24 h | 92 ± 5.0 | 92 [206] |
| domestic        | synthetic | laboratory | submerged | MF | plate and frame | PES | T = 25–30 °C; HRT = 12 h; 10 h; 8 h; SRT = 30; 60 d; infinity | >97 | NI [209] |
| domestic        | synthetic | laboratory | submerged | MF | flat sheet | polyolefin | HRT = 12 h | ~83% | NI [211] |
| domestic        | synthetic | laboratory | submerged | UF | flat sheet | PES | T = 25 °C; HRT = 13 h | 92.3 ± 4.1 | NI [199] |
| domestic        | synthetic | laboratory | submerged | MF | NI | NI | T = 15 °C | NI | NI [212] |
| domestic        | synthetic | laboratory | submerged | MF | hollow fiber | NI | NI | 97.1 ± 0.3 | NI [213] |

NI—no information; PAN—polyacrylonitrile; PES—polyethersulfone; PVDF—polyvinylidene fluoride.
4.2.2. Impact of Hydraulic Retention Time

As discussed above (Section 4.2.2), the AnMBR performance used for the treatment of municipal wastewater is not significantly affected by HRT. Surprisingly, with regards to treating domestic wastewaters, the results presented in the literature are different. Figure 7 shows the impact of the average HRT on the average COD removal in AnMBRs used for treating real and synthetic domestic wastewaters. All the data presented are based on the results published in the reviewed studies.

The impact of HRT on the AnMBRs technology performance in the treatment of low-concentration domestic wastewater was investigated by Liu et al. [207]. For this purpose, a submerged AnMBR with microfiltration PVDF hollow fiber membranes was used. The process was conducted under temperatures of 35 ± 1 °C. It was demonstrated that COD removal efficiency is strongly affected by HRT. Indeed, for the HRT equal to 15 h, 10 h and 6 h, the average COD removal efficiency was 80.23%, 82.69% and 78.19%, respectively (Table 6). Therefore, it can be indicated that reducing HRT below 10 h may lead to a significant decreasing in the COD removal efficiency of AnMBRs technology.

In turn, Ni et al. [208] studied the performance of submerged AnMBRs operated under lower temperature, equal to 25 °C and fed with real domestic wastewater (Table 6). The AnMBRs were equipped with different membranes pore size (0.4 or 0.05 µm). The AnMBR with the MF membrane (0.4 µm) was operated at HRT from 24 to 4 h. It was determined that the average COD removal rate was equal to 90% when the HRT exceeded 12 h and decreased to 84% in the case of an HRT equal to 4 h. In turn, the AnMBR with the UF membrane (0.05 µm) was examined at HRT in the range from 24 to 10 h. It was found that the average COD removal efficiency was over 88%; however, that stable performance can only be maintained under the HRT of 12 h or longer. It can be inferred from this study that shortened HRT does not ensure the insufficient contact time between the microorganism and the substrate, and as a consequence, the COD removal efficiency decreases. These findings correspond with the findings of Ji et al. [195], who used a submerged AnMBR for treating real domestic wastewater (Table 5) at 25 °C with HRT ranging from 4 to 12 h. It was found that for HRT between 6 and 12 h, the AnMBR demonstrated a good COD removal equal to 89%. In turn, at HRT of 4 h, the COD removal was lower and equal to 84% (Table 6). The results discussed above are in good agreement with other studies [202,204], wherein it was found that HRT has an impact on the performance of AnMBRs used for treating domestic wastewaters.

However, different observations were made by Huang et al. [209], who studied the effect of HRT on the performance of the submerged AnMBRs used for treating synthetic domestic wastewaters. For this purpose, the process was carried out at a temperature of 25 °C, SRT of 30 days, 60 days and infinite days and HRT equal to 12 h, 10 h and 8 h. It was found that COD removal efficiency higher than 97% was achieved at all operating conditions applied (Table 6). The authors indicated that this observation may be related to the fact that the used MF membrane (0.45 µm) ensured complete retention of biomass and soluble COD after biological treatment present in the mixed liquor which, in turn, allowed for producing a high-quality effluent with non-detectable solids. However, it should be pointed out that the experimental results reported above were obtained under a relatively low range of HRT values, and the synthetic low-strength wastewater was used as a feed, which obviously could have an impact on the results obtained. The investigation on the impact of the longer HRT on the AnMBRs performance used for treating synthetic domestic wastewaters was performed by Jeong et al. [201]. For this purpose, the AnMBRs equipped with the MF flat-sheet pyrophyllite based or alumina-based ceramic membranes were used. Reactors were operated under HRT for approximately 44 h and 18 h. It was noted that under longer HRT, both AnMBRs demonstrated successful organic removal (COD equal to 90.5 ± 6.8% and 95.9 ± 5.3%, respectively). In turn, at reduced HRT, for AnMBR with the alumina-based membrane, the removal of COD reached 96.1 ± 5.1%, while for the AnMBR with pyrophyllite based membrane, it was equal to 42.6 ± 19.2%. The authors have pointed out that the decrease in the AnMBR performance could be due to the washing-out...
of a part of the microorganisms. It can be inferred from this study that the impact of the process parameters on the AnMBRs, coupled with ceramic membranes, is still in need of further investigation.

### 4.2.3. Impact of Membrane Type

As mentioned above, the membrane type may have an impact on the AnMBRs performance in terms of organic matter removal. Yue [198] examined the performance of AnMBRs equipped with ceramic membranes used for treating real domestic wastewater (Table 6). It was noted that the total COD removal efficiency was equal to 88.6 ± 9.0, 87.9 ± 7.4 and 86.3 ± 9.7% for membranes characterized by pore size of 80 nm, 200 nm and 300 nm, respectively. Therefore, it was concluded that each of used AnMBRs ensured the high COD removal efficiencies, with no significant difference for the various membranes used. It also may be indicated that, as in the case of municipal wastewaters treatment (Section 4.2.3), the membrane pore size has no significant impact on the COD removal efficiency. However, the above-mentioned authors have indicated that higher fouling rates were observed for larger pore-sized membranes, which was attributed to higher occurrences of pore blockages. Therefore, it is important to point out that the membrane pore size has the significant impact on the process performance in terms of the permeate flux.

In turn, Jeong et al. [202] compared the performance of polymeric and ceramic membranes in AnMBR treatment of synthetic domestic wastewater. For this purpose, two membranes for identical AnMBR have been used: a flat-sheet Al-based ceramic membrane with a nominal pore size of 0.1 µm and a flat-sheet PVDF polymeric membrane with a nominal pore size equal to 0.08 µm. It was noted that AnMBR equipped with the ceramic membrane ensured much higher COD removal (91.0 ± 13.8%) than the AnMBR with the polymeric membrane (77.8 ± 20.5%). The above-mentioned authors attributed two explanations to this observation. First, it was indicated that the concentration of mixed liquor suspended solids and mixed liquor volatile suspended solids in the AnMBR with the Al-based membrane were higher than those noted in the AnMBR with PET membrane. Consequently, it allowed us to enhance the COD removal rate due to an increase of the biological activity. Secondly, the obtained result could be due to the difference in pore size distribution of the membranes used. Indeed, polymeric membranes are characterized by the wider pore size distribution than ceramic ones. Hence, it should be emphasized that the membrane material may have a significant impact on the AnMBRs performance in terms of COD removal.

### 4.3. Energy Demand

Information on the energy demand of the AnMBRs used for treating domestic wastewaters presented in scientific literature is scarce. Mei et al. [140] have analyzed the recovery of energy from domestic wastewater in the external submerged AnMBR equipped with three PVDF flat-sheet membrane modules (a mean pore size of 0.2 µm). For this purpose, it was assumed that the CH₄ energy potential is equal to 11 kWh/(m³ CH₄). It was shown that the energy recovery from methane production depends on temperature. Indeed, it was equal to 0.42 kWh/m³, 0.34 kWh/m³ and 0.17 kWh/m³ at temperatures equal to 35 °C, 25 °C and 15 °C, respectively. The above-mentioned authors have demonstrated that at temperatures of 25 °C and 35 °C, the overall energy demand of AnMBR is in the range from −0.31 kWh/m³ to 0.33 kWh/m³ and from −0.19 kWh/m³ to 0.58 kWh/m³, respectively. In turn, at 15 °C, the energy demand between 0.04 kWh/m³ and 0.97 kWh/m³ was reported (Figure 8). These noteworthy results indicate that performing the treatment process at 35 °C and 25 °C is more efficient at the same permeate flux than when conducted under 15 °C and also ensures net energy recovery. Moreover, it should be pointed out that under these conditions, the AnMBR energy demand is lower than that reported for a typical domestic WWTP (0.6 kWh/m³ [22]).
5. Conclusions

Nowadays, AnMBRs technology has gained great attention with a high interest in the application for the treatment of municipal and domestic wastewaters, due to the fact that it offers several advantages over WWTPs.

As discussed in this review, AnMBRs have the potential to reduce the net energy requirements in comparison with aerobic-based processes applied in WWTPs. Based on the findings presented in the literature, it was deduced that the net energy demand of submerged AnMBRs used for the treatment of sulphate-rich municipal wastewaters ranges from 0.06 kWh/m$^3$ to 0.47 kWh/m$^3$, while the energy demand of municipal WWTPs is from 0.45 kWh/m$^3$ to 1.25 kWh/m$^3$. Moreover, it was clearly indicated that the AnMBRs technology used for the treatment of low-sulphate municipal has the potential to be a net energy producer. With regards to domestic wastewaters, similar conclusions have been drawn. According to the literature, the energy demand of submerged AnMBRs is between $-0.31$ kWh/m$^3$ and 0.97 kWh/m$^3$, while for a typical domestic, WWTP is equal to 0.6 kWh/m$^3$.

To summarize, it should be pointed out that this review builds on a growing body of evidence linking wastewaters treatment with energy-efficient AnMBRs technology and creates a framework for future research involving AnMBRs regarding the treatment of municipal and domestic wastewaters.

6. Challenges and Perspectives

Although the application of AnMBRs for treating municipal and domestic wastewaters was investigated by many researchers, there remain notable challenges. Indeed, the conducted review of the published studies demonstrates that most of the information available from the literature comes from experiments performed on the laboratory scale. Therefore, it can be concluded that there is still a lack of enough feasibility studies on the
application of pilot and full-scale systems. Moreover, as shown in this review, only a few studies provide results on the application of ceramic membranes in AnMBRs, likely due to the fact that they are considered excellent material for wastewater treatment. Therefore, more attention should be paid to their use. They demonstrate high thermal, chemical and mechanical stability and thus, longer life-time, which is beneficial from an economic point of view on AnMBRs technology. In addition, findings presented in the literature have shown that ceramic membranes may provide a lower fouling rate than polymeric ones. Therefore, the use of ceramic membranes may lead to decreasing the energy requirement for control of flux decline in AnMBRs. Furthermore, the key challenge in AnMBRs technology is to optimize process parameters. However, to date, a full understanding on the influence of temperature and HRT on the AnMBRs performance in terms of organic matter removal is insufficient. For instance, further research on the AnMBRs performance in the treatment of various strength domestic wastewaters under psychrophilic temperatures is required. Therefore, from an engineering aspect, improvements recommended for AnMBRs should be focused on the optimization of operational conditions for maximizing high net energy production, which can contribute to the practical application of AnMBRs. It was demonstrated that the capture of CH$_4$ biogas and dissolved in the effluent can meet AnMBRs energy requirements. Therefore, additional findings on effective methods for recovering dissolved CH$_4$ are recommended. Obviously, these would lead to an enhancement of the economic viability of the process.

Author Contributions: Conceptualization, W.T.; methodology, W.T.; validation, M.G.; formal analysis, W.T.; investigation, W.T.; data curation, W.T.; writing—original draft preparation, W.T.; writing—review and editing, M.G.; visualization, W.T.; supervision, M.G.; project administration, M.G.; funding acquisition, M.G. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by National Science Centre, Poland, grant number 2018/29/B/ST8/00942.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: The data presented in this study are available on request from the corresponding author. The data are not publicly available due to the institutional repository being under construction.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, or in the decision to publish the results.

Abbreviations

AeMBR aerobic membrane bioreactors
AnMBR anaerobic membrane bioreactor
BOD biological oxygen demand
CAS classical activated sludge
COD chemical oxygen demand
EPS extracellular polymeric substance
HRT hydraulic retention time
$J_{20}$ 20 °C-standardized critical flux
MF microfiltration
MBR membrane bioreactor
MLSS mixed liquor suspended solid
N nitrogen
NF nanofiltration
NTU nephelometric turbidity unit
OLR organic loading rate
P phosphorus
References

1. Sonune, A.; Ghate, R. Developments in Wastewater Treatment Methods. Desalination 2004, 167, 55–63. [CrossRef]
2. Qadir, M.; Drechsel, P.; Jiménez Cisneros, B.; Kim, Y.; Pramanik, A.; Mehta, P.; Olaniyan, O. Global and Regional Potential of Wastewater as a Water, Nutrient and Energy Source. Nat. Resour. Forum 2020, 44, 40–51. [CrossRef]
3. Available online: https://www.unwater.org/water-facts/quality-and-wastewater/ (accessed on 26 May 2022).
4. Boretti, A.; Rosa, L. Reassessing the Projections of the World Water Development Report. Npj Clean Water 2019, 2, 15. [CrossRef]
5. Chahal, C.; van den Akker, B.; Young, F.; Franco, C.; Blackbeard, J.; Monis, P. Pathogen and Particle Associations in Wastewater. In Advances in Applied Microbiology; Elsevier: Amsterdam, The Netherlands, 2016; Volume 97, pp. 63–119. ISBN 978-0-12-804816-0.
6. Jasim, N.A. The Design for Wastewater Treatment Plant (WWTP) with GPS X Modelling. Cogent Eng. 2020, 7, 1723782. [CrossRef]
7. Liu, Y.-J.; Gu, J.; Liu, Y. Energy Self-Sufficient Biological Municipal Wastewater Reclamation: Present Status, Challenges and Solutions Forward. Bioresour. Technol. 2018, 269, 513–519. [CrossRef]
8. Bohra, V.; Ahamad, K.U.; Kela, A.; Vaghela, G.; Sharma, A.; Deka, B.J. Energy and Resources Recovery from Wastewater Treatment Systems. In Clean Energy and Resource Recovery; Elsevier: Amsterdam, The Netherlands, 2022; pp. 17–36, ISBN 978-0-323-90178-9.
9. Lehtoranta, S.; Malila, R.; Särkilahti, M.; Viskari, E.-L. To Separate or Not? A Comparison of Wastewater Management Systems for the New City District of Hiedanranta, Finland. Environ. Res. 2022, 208, 112764. [CrossRef]
10. Zickler, D.; Bader, G.; Bullock, T.S.; Gehri, C.; Gessner, M.O.; Schiegg, T.; Schnitzler, E.; Zöller, M.; Ageykin, V.; Albert, C.; et al. The Circular Economy of Municipal Wastewater Treatment Plants: An Overview. Water Air Soil Pollut. 2021, 232, 318. [CrossRef]
11. Herrera-Navarrete, R.; Arellano-Wences, H.J.; Colín-Cruz, A.; Sampedro-Rosas, M.L.; Rosas-Acevedo, J.L.; Rodríguez-Herrera, A.L. Thematic and Geographical Trend in Scientific Research Applied in Municipal Wastewater Treatment Plants: An Overview. Water Air Soil Pollut. 2021, 232, 318. [CrossRef]
18. Parravicini, V.; Svardal, K.; Krampe, J. Greenhouse Gas Emissions from Wastewater Treatment Plants. *Energy Procedia* 2016, 97, 246–253. [CrossRef]

19. Lv, Z.; Shan, X.; Xiao, X.; Cai, R.; Zhang, Y.; Jiao, N. Excessive Greenhouse Gas Emissions from Wastewater Treatment Plants by Using the Chemical Oxygen Demand Standard. *Sci. China Earth Sci.* 2022, 65, 87–95. [CrossRef]

20. Zhang, X.; Liu, Y. Circular Economy Is Game-Changing Municipal Wastewater Treatment Technology towards Energy and Carbon Neutrality. *Chem. Eng. J.* 2022, 429, 132114. [CrossRef]

21. Goliopoulos, N.; Mamais, D.; Noutsopoulos, C.; Dimopoulou, A.; Kounadis, C. Energy Consumption and Carbon Footprint of Greek Wastewater Treatment Plants. *Water* 2022, 14, 320. [CrossRef]

22. Gude, V.G. Wastewater Treatment in Microbial Fuel Cells—An Overview. *J. Clean. Prod.* 2016, 122, 287–307. [CrossRef]

23. Bao, Z.; Sun, S.; Sun, D. Assessment of Greenhouse Gas Emission from A/O and SBR Wastewater Treatment Plants in Beijing, China. *Int. Biodeterior. Biodegrad.* 2016, 108, 108–114. [CrossRef]

24. Fighir (Arsene), D.; Teodosiu, C.; Fiore, S. Environmental and Energy Assessment of Municipal Wastewater Treatment Plants in Italy and Romania: A Comparative Study. *Water* 2019, 11, 1611. [CrossRef]

25. Tumenelger, A.; Alshboul, Z.; Lorke, A. Methane and Nitrous Oxide Emission from Different Treatment Units of Municipal Wastewater Treatment Plants in Southwest Germany. *PLoS ONE* 2019, 14, e0209763. [CrossRef] [PubMed]

26. Crone, B.C.; Garland, J.L.; Sorial, G.A.; Vane, L.M. Significance of Dissolved Methane in Effluents of Anaerobically Treated Low Strength Wastewater and Potential for Recovery as an Energy Product: A Review. *Water Res.* 2016, 104, 520–531. [CrossRef] [PubMed]

27. Gu, Y.; Li, Y.; Li, X.; Luo, P.; Wang, H.; Wang, X.; Wu, J.; Li, F. Energy Self-Sufficient Wastewater Treatment Plants: Feasibilities and Challenges. *Energy Procedia* 2017, 105, 3741–3751. [CrossRef]

28. Neczaj, E.; Grosser, A. Circular Economy in Wastewater Treatment Plant–Challenges and Barriers. In *Proceedings of the EWaS3, Lefkada Island, Greece, 31 July 2018*; MDPI: Basel, Switzerland, 2018; p. 614.

29. Gandiglio, M.; Lanzini, A.; Soto, A.; Leone, P.; Santarelli, M. Enhancing the Energy Efficiency of Wastewater Treatment Plants through Co-Digestion and Fuel Cell Systems. *Front. Environ. Sci.* 2017, 5, 70. [CrossRef]

30. Vaccari, M.; Foladori, P.; Nembrini, S.; Vitali, F. Benchmarking of Energy Consumption in Municipal Wastewater Treatment Plants—A Survey of over 200 Plants in Italy. *Water Sci. Technol.* 2018, 77, 2242–2252. [CrossRef] [PubMed]

31. Available online: [https://www.iea.org/reports/water-energy- nexus/](https://www.iea.org/reports/water-energy-nexus/) (accessed on 26 May 2022).

32. Gude, V.G. Energy and Water Autarky of Wastewater Treatment and Power Generation Systems. *Renew. Sustain. Energy Rev.* 2015, 45, 52–68. [CrossRef]

33. Roccaro, P.; Vaglasiindi, F.G.A. Membrane Bioreactors for Wastewater Reclamation: Cost Analysis. In *Current Developments in Biotechnology and Bioengineering*; Elsevier: Amsterdam, The Netherlands, 2020; pp. 311–322. ISBN 978-0-12-819854-4.

34. Pabi, S.; Amarnath, A.; Goldstein, R.; Reekie, L. *Electricity Use and Management in the Municipal Water Supply and Wastewater Industries*; Electric Power Research Institute: Washington, DC, USA, 2013.

35. Guerra-Rodriguez, S.; Oulego, P.; Rodriguez, E.; Singh, D.N.; Rodriguez-Chueca, J. Towards the Implementation of Circular Economy in the Wastewater Sector: Challenges and Opportunities. *Water* 2020, 12, 1431. [CrossRef]

36. He, Y.; Zhu, Y.; Chen, J.; Huang, M.; Wang, P.; Wang, G.; Zhou, G. Assessment of Energy Consumption of Municipal Wastewater Treatment Plants in China. *J. Clean. Prod.* 2019, 228, 399–404. [CrossRef]

37. Trapote, A.; Albaladejo, A.; Simón, P. Energy Consumption in an Urban Wastewater Treatment Plant: The Case of Murcia Region (Spain). *Civ. Eng. Environ. Syst.* 2014, 31, 304–310. [CrossRef]

38. Available online: [https://www.bcresearch.com/market-research/membrane-and-separation-technology/membrane-bioreactors.html/](https://www.bcresearch.com/market-research/membrane-and-separation-technology/membrane-bioreactors.html/) (accessed on 26 May 2022).

39. Akkooyunlu, B.; Daly, S.; Casey, E. Membrane Bioreactors for the Production of Value-Added Products: Recent Developments, Challenges and Perspectives. *Biotechnol. Biofuels* 2014, 7, 125793. [CrossRef]

40. Baek, S.H.; Papilla, K.R. Aerobic and Anaerobic Membrane Bioreactors for Municipal Wastewater Treatment. *Water Environ. Res.* 2006, 78, 133–140. [CrossRef] [PubMed]

41. Baek, S.H.; Papilla, K.R.; Kim, H.-J. Lab-Scale Study of an Anaerobic Membrane Bioreactor (AnMBR) for Dilute Municipal Wastewater Treatment. *Biotechnol. Bioproc. E* 2010, 15, 704–708. [CrossRef]

42. Robles, A.; Ruano, M.V.; García-Usach, F.; Ferrer, J. Sub-Critical Filtration Conditions of Commercial Hollow-Fibre Membranes in a Submerged Anaerobic MBR (HF-SAnMBR) System: The Effect of Gas Sparging Intensity. *Biotechnology. Technol.* 2012, 114, 247–254. [CrossRef] [PubMed]

43. Judd, S. The Status of Membrane Bioreactor Technology. *Trends Biotechnol.* 2008, 26, 109–116. [CrossRef] [PubMed]

44. Giménez, J.B.; Robles, A.; Carretero, L.; Durán, F.; Ruano, M.V.; Gatti, M.N.; Ribes, J.; Ferrer, J.; Seco, A. Experimental Study of the Anaerobic Urban Wastewater Treatment in a Submerged Hollow-Fibre Membrane Bioreactor at Pilot Scale. *Biotechnology. Technol.* 2011, 102, 8799–8806. [CrossRef]

45. Aslam, A.; Khan, S.J.; Shahzad, H.M.A. Anaerobic Membrane Bioreactors (AnMBRs) for Municipal Wastewater Treatment-Potential Benefits, Constraints, and Future Perspectives: An Updated Review. *Sci. Total Environ.* 2022, 802, 149612. [CrossRef]

46. Jegatheesan, V.; Pramanik, B.K.; Chen, J.; Navaratna, D.; Chang, C.-Y.; Shu, L. Treatment of Textile Wastewater with Membrane Bioreactor: A Critical Review. *Biotechnology. Technol.* 2016, 204, 202–212. [CrossRef]
47. Pretel, R.; Robles, A.; Ruano, M.V.; Seco, A.; Ferrer, J. The Operating Cost of an Anaerobic Membrane Bioreactor (AnMBR) Treating Sulphate-Rich Urban Wastewater. Sep. Purif. Technol. 2014, 126, 30–38. [CrossRef]
48. Martin-Garcia, I.; Monsalvo, V.; Pidou, M.; Le-Clech, P.; Judd, S.J.; McAdam, E.J.; Jefferson, B. Impact of Membrane Configuration on Fouling in Anaerobic Membrane Bioreactors. J. Membr. Sci. 2011, 382, 41–49. [CrossRef]
49. Grossman, A.D.; Yang, Y.; Yoge, U.; Camarena, D.C.; Oron, G.; Bernstein, R. Effect of Ultrafiltration Membrane Material on Fouling Dynamics in a Submerged Anaerobic Membrane Bioreactor Treating Domestic Wastewater. Environ. Sci. Water Res. Technol. 2019, 5, 1145–1156. [CrossRef]
50. Gharibian, S.; Hazzati, H. Towards Practical Integration of MBR with Electrochemical AOP: Improved Biodegradability of Real Pharmaceutical Wastewater and Fouling Mitigation. Water Res. 2022, 218, 118478. [CrossRef]
51. Xiao, Y.; Yaohari, H.; De Araujo, C.; Sze, C.C.; Stuckey, D.C. Removal of Selected Pharmaceuticals in an Anaerobic Membrane Bioreactor (AnMBR) with/without Powdered Activated Carbon (PAC). Chem. Eng. J. 2017, 321, 335–345. [CrossRef]
52. Kanafin, Y.N.; Kanafina, D.; Malamis, S.; Katsou, E.; Inglezakis, V.J.; Poulopoulos, S.G.; Arkhangelsky, E. Anaerobic Membrane Bioreactor: Efficiencies, Fates and Impact Factors. Case Stud. Chem. Environ. Eng. 2019, 102061. [CrossRef]
53. Akca, M.S.; Bostancı, O.; Aydin, A.K.; Koyuncu, I.; Altinbas, M. BioH2 Production from Food Waste by Anaerobic Membrane Bioreactor. Int. J. Hydrog. Energy 2021, 46, 27941–27955. [CrossRef]
54. Yurtsever, A.; Sahinkaya, E.; Çınar, O. Performance and Fouulant Characteristics of an Anaerobic Membrane Bioreactor Treating Real Textile Wastewater. J. Water Process Eng. 2020, 33, 101088. [CrossRef]
55. Deschamps, L.; Merlet, D.; Lemaire, J.; Imatoukene, N.; Filali, R.; Clémont, T.; Lopez, M.; Theoleyre, M.-A. Excellent Performance of Anaerobic Membrane Bioreactor in Treatment of Distillery Wastewater at Pilot Scale. J. Water Process Eng. 2021, 41, 102061. [CrossRef]
56. Yurtsever, A.; Sahinkaya, E.; Akta¸s, Ö.; Uçar, D.; Çınar, Ö.; Wang, Z. Performances of Anaerobic and Aerobic Membrane Bioreactors for the Treatment of Synthetic Textile Wastewater. Bioresour. Technol. 2015, 192, 564–573. [CrossRef]
57. Vinardell, S.; Dosta, J.; Lopez, M.; Theoleyre, M.-A. Excellent Performance of Anaerobic Membrane Bioreactor Treating Municipal Wastewater at Ambient Temperature: Operation and Potential Use for Municipal Wastewater Treatment. Sep. Purif. Technol. 2014, 126, 30–38. [CrossRef]
58. Al-Asheh, S.; Bagheri, M.; Aidan, A. Membrane Bioreactor for Wastewater Treatment: A Review. Case Stud. Chem. Environ. Eng. 2021, 4, 100109. [CrossRef]
59. Asante-Sackey, D.; Rathilal, S.; Tetteh, E.K.; Armah, E.K. Membrane Bioreactors for Produced Water Treatment: A Mini-Review. Membranes 2022, 12, 275. [CrossRef]
60. De Vela, R.J. A Review of the Factors Affecting the Performance of Anaerobic Membrane Bioreactor and Strategies to Control Membrane Fouling. Rev. Environ. Sci. Biotechnol. 2021, 20, 607–644. [CrossRef]
61. Kanafin, Y.N.; Kanafina, D.; Malamis, S.; Katsou, E.; Inglezakis, V.J.; Poulopoulos, S.G.; Arkhangelsky, E. Anaerobic Membrane Bioreactors for Municipal Wastewater Treatment: A Literature Review. Membranes 2021, 11, 967. [CrossRef] [PubMed]
62. Zhang, S.; Lei, Z.; Dazkapasu, M.; Li, Q.; Li, Y.-Y.; Chen, R. Removal of Trace Organic Contaminants in Municipal Wastewater by Anaerobic Membrane Bioreactor: Efficiencies, Fates and Impact Factors. J. Water Process Eng. 2021, 40, 101953. [CrossRef]
63. Kehrein, P.; van Loosdrecht, M.; Osseweijer, P.; Garf

[CrossRef]
64. Lin, H.; Gao, W.; Meng, F.; Liao, B.-Q.; Leung, K.-T.; Zhao, L.; Chen, J.; Hong, H. Membrane Bioreactors for Industrial Wastewater Treatment: A Critical Review. Crit. Rev. Environ. Sci. Technol. 2012, 42, 677–740. [CrossRef]
65. Kehrein, P.; van Loosdrecht, M.; Osseweijer, P.; Garf

[CrossRef]
66. Peña, M.; de Nascimento, T.; Gouveia, J.; Escudero, J.; Gómez, A.; Letona, A.; Arrieta, J.; Fd-Polanco, F. Anaerobic Submerged Membrane Bioreactor (AnSMBR) Treating Municipal Wastewater at Ambient Temperature: Operation and Potential Use for Agricultural Irrigation. Bioresour. Technol. 2019, 282, 285–293. [CrossRef]
67. Habr, M.; Hong, P.-Y. Anaerobic Membrane Bioreactor Effluent Reuse: A Review of Microbial Safety Concerns. Fermentation 2017, 3, 39. [CrossRef]
68. Lee, E.; Rout, P.R.; Bae, J. The Applicability of Anaerobically Treated Domestic Wastewater as a Nutrient Medium in Hydroponic Lettuce Cultivation: Nitrogen Toxicity and Health Risk Assessment. Sci. Total Environ. 2021, 780, 146482. [CrossRef]
69. Plevri, A.; Maimaris, D.; Noutsopoulos, C. Anaerobic MBR Technology for Treating Municipal Wastewater at Ambient Temperatures. Chemosphere 2021, 275, 129961. [CrossRef]
70. Dereli, R.K.; Ersahn, M.E.; Ozgur, H.; Ozturk, I.; Jeison, D.; van der Zee, F.; van Lier, J.B. Potentials of Anaerobic Membrane Bioreactors to Overcome Treatment Limitations Induced by Industrial Wastewaters. Bioresour. Technol. 2012, 122, 160–170. [CrossRef] [PubMed]
71. Maaz, M.; Yasin, M.; Aslam, M.; Kumar, G.; Aatabi, A.E.; Idrees, M.; Anjum, F.; Jamil, F.; Ahmad, R.; Khan, A.L.; et al. Anaerobic Membrane Bioreactors for Wastewater Treatment: Novel Configurations, Fouling Control and Energy Considerations. Bioresour. Technol. 2019, 283, 358–372. [CrossRef] [PubMed]
201. Jeong, Y.; Cho, K.; Kwon, E.E.; Tsang, Y.F.; Rinklebe, J.; Park, C. Evaluating the Feasibility of Pyrophyllite-Based Ceramic Membranes for Treating Domestic Wastewater in Anaerobic Ceramic Membrane Bioreactors. *Chem. Eng. J.* 2017, 328, 567–573. [CrossRef]

202. Jeong, Y.; Kim, Y.; Jin, Y.; Hong, S.; Park, C. Comparison of Filtration and Treatment Performance between Polymeric and Ceramic Membranes in Anaerobic Membrane Bioreactor Treatment of Domestic Wastewater. *Sep. Purif. Technol.* 2018, 199, 182–188. [CrossRef]

203. Wen, C.; Huang, X.; Qian, Y. Domestic Wastewater Treatment Using an Anaerobic Bioreactor Coupled with Membrane Filtration. *Process Biochem.* 1999, 35, 335–340. [CrossRef]

204. Chu, L.-B.; Yang, F.-L.; Zhang, X.-W. Anaerobic Treatment of Domestic Wastewater in a Membrane-Coupled Expended Granular Sludge Bed (EGSB) Reactor under Moderate to Low Temperature. *Process Biochem.* 2005, 40, 1063–1070. [CrossRef]

205. Smith, A.L.; Skerlos, S.J.; Raskin, L. Anaerobic Membrane Bioreactor Treatment of Domestic Wastewater at Psychrophilic Temperatures Ranging from 15 °C to 3 °C. *Environ. Sci. Water Res. Technol.* 2015, 1, 56–64. [CrossRef]

206. Smith, A.L.; Skerlos, S.J.; Raskin, L. Psychrophilic Anaerobic Membrane Bioreactor Treatment of Domestic Wastewater. *Water Res.* 2013, 47, 1655–1665. [CrossRef]

207. Liu, J.; Tian, H.; Luan, X.; Zhou, X.; Chen, X.; Xu, S.; Kang, X. Submerged Anaerobic Membrane Bioreactor for Low-Concentration Domestic Sewage Treatment: Performance and Membrane Fouling. *Environ. Sci. Pollut. Res.* 2020, 27, 6785–6795. [CrossRef]

208. Ni, J.; Ji, J.; Li, Y.-Y.; Kubota, K. Microbial Characteristics in Anaerobic Membrane Bioreactor Treating Domestic Sewage: Effects of HRT and Process Performance. *J. Environ. Sci.* 2022, 111, 392–399. [CrossRef]

209. Huang, Z.; Ong, S.L.; Ng, H.Y. Submerged Anaerobic Membrane Bioreactor for Low-Strength Wastewater Treatment: Effect of HRT and SRT on Treatment Performance and Membrane Fouling. *Water Res.* 2011, 45, 705–713. [CrossRef]

210. Martin, I.; Pidou, M.; Soares, A.; Judd, S.; Jefferson, B. Modelling the Energy Demands of Aerobic and Anaerobic Membrane Bioreactors for Wastewater Treatment. *Environ. Technol.* 2011, 32, 921–932. [CrossRef]

211. Achilli, A.; Marchand, E.A.; Childress, A.E. A Performance Evaluation of Three Membrane Bioreactor Systems: Aerobic, Anaerobic, and Attached-Growth. *Water Sci. Technol.* 2011, 63, 2999–3005. [CrossRef]

212. Smith, A.L.; Skerlos, S.J.; Raskin, L. Membrane Biofilm Development Improves COD Removal in Anaerobic Membrane Bioreactor Wastewater Treatment. *Microb. Biotechnol.* 2015, 8, 883–894. [CrossRef] [PubMed]

213. Liu, J.; Zhang, L.; Zhang, P.; Zhou, Y. Quorum Quenching Altered Microbial Diversity and Activity of Anaerobic Membrane Bioreactor (AnMBR) and Enhanced Methane Generation. *Bioresour. Technol.* 2020, 315, 123862. [CrossRef] [PubMed]