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A comprehensive review of the direct membrane filtration of municipal wastewater

Pau Sanchis-Perucho [a,*], Daniel Aguado [b], José Ferrer [b], Aurora Seco [a], Ángel Robles [a]

[a] CALAGUA – Unidad Mixta UV-UPV, Departament d'Enginyeria Química, Universitat de València, Spain
[b] CALAGUA – Unidad Mixta UV-UPV, Institut Universitari d'Investigació d'Enginyeria de l'Aigua i Medi Ambient – IIAMA, Universitat Politècnica de Valencia, Spain

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ABSTRACT

Direct membrane filtration is attracting increasing interest as a potential alternative to the current municipal wastewater treatment schemes. Extremely high and stable water quality can be obtained by membrane filtration, while the organic material recovered in the membrane tank can be directly fed to anaerobic digesters, boosting the recovery of resources and reducing the cost of the treatment. Unfortunately, the severe fouling involved in the process is still a serious drawback to achieving viable full-scale schemes. Although the literature provides numerous strategies designed to minimize fouling and achieve feasible long-term filtrations, due to the diverse membranes and operating conditions used, practical solutions are difficult to reach. This work therefore aimed to cover a comprehensive review on the direct filtration of municipal wastewater, describing most suitable membrane types (particularly microfiltration, ultrafiltration, osmosis and dynamic membranes), influent pre-treatment, operating conditions and fouling control strategies. Resource recovery, including reclaimed water, essential nutrients (nitrogen and phosphorus), and energy, as well as the quality of the recycled water produced, are summarized and discussed. Finally, the potential economic and environmental benefits of this alternative are included, along with an evaluation of its practical application.

1. Introduction

Fresh water stress and climate change are worldwide problems that need to be addressed in the coming years (World Economic Forum, 2022; 2019). Global water scarcity is a growing problem (Wang et al., 2021) even in regions with consistent rainfall (Gassert et al., 2013), making the search for good quality water sources a prominent policy in different countries (Jiménez-Benítez et al., 2020). On the other hand, the forecast greenhouse gas (GHG) emissions predict significantly higher trends than those necessary for the 2 ºC-target for global temperature stabilization (UNEP, 2019) and are not on track for meeting the Paris Agreement objectives. The energy crisis is also an important issue to be faced (Khalil te al., 2014), while other essential resources, such as phosphorous for use in agriculture, are also close to exhaustion due to the increasing population growth and food demands, making scarcities possible in the near future (World Economic Forum, 2022). All these interrelated problems are exposing the limitations of the current economic models and worldwide policies and demand a new approach to human interaction with the environment. Indeed, the exploitation of

* Corresponding author.
E-mail address: pau.sanchis-perucho@uv.es (P. Sanchis-Perucho).

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non-renewable resources and the linear economy approaches currently used have proven unsuitable to sustain the developed countries in the long-term. Numerous experts are thus claiming than there is an urgent need to adapt our consumption systems to circular economy (CE) to achieve a sustainable future human activity (Ferronato et al., 2019), as well as developing greener and more energy-efficient technologies to minimize our impact on the environment.

In the context of municipal wastewater (MWW) treatment, the CE approach has gained special interest due to the potential resources that MWW contains. MWW is now regarded as a source of reclaimed water, carbon, energy and essential nutrients (mainly nitrogen and phosphorus) (Donoso-Bravo et al., 2020). Unfortunately, most MWW treatment processes fail to recover a large fraction of these resources. MWW management generally depends on aerobic biological treatments, commonly known as activated sludge (AS) process, to remove organic matter (OM) through biological oxidation. This process is undesirable from a CE point of view since it fails to recover carbon and energy from OM while requiring significant energy inputs associated to aeration (see Fig. 1a). Indeed, AS is recognized as an energy-intensive process with energy demands that can reach up to 50 % of the total energy requirements of a full-scale WWTP (Sid et al., 2017), while the associated nitrification-denitrification and chemical phosphorous precipitation fail to recover valuable nutrients from the MWW, representing even a negative impact due to possible N_2O gas emissions to the atmosphere (Tallec et al., 2006) plus chemicals expenses.

Anaerobic technology is thus more appropriate for MWW management since it does not require aeration, while energy can be obtained revalorizing OM via methane production. The influent nutrients can also be mineralized during the treatment, allowing their

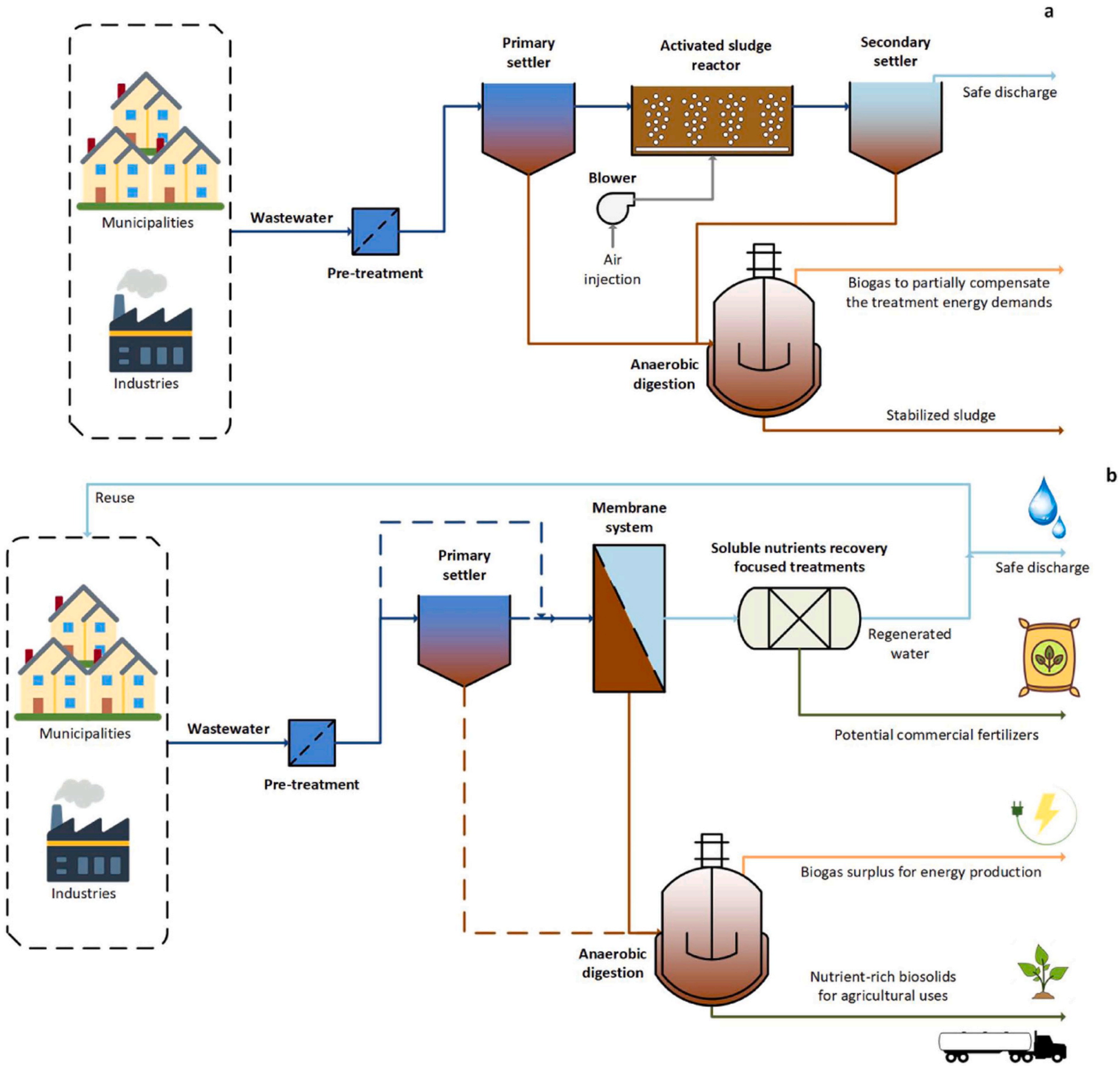

Fig. 1. MWW treatment scheme: (a) classic management and (b) direct filtration concept.

later recovery, while lower sludge production rates can be expected than in aerobic systems, reducing the associated management expenses (McCarty et al., 2011). Unfortunately, the slow growth rate of anaerobic biomass has historically forced the application of aerobic technologies for MWW treatment, anaerobic systems only feasible to manage the primary and biological sludge generated in the treatment mainline. Nevertheless, different solutions have been proposed to boost the potential of anaerobic technology for resource recovery, transforming the former WWTPs into new water resource recovery facilities (WRRF), in which not only reclaimed water but also other valuable resources can be obtained such as bio-products and bio-energy (Batstone et al., 2015; Donoso-Bravo et al., 2020).

The direct filtration of MWW by membranes – commonly known as direct membrane filtration (DMF) – has recently emerged as an interesting alternative to enhance the current MWW treatment (Zhao et al., 2019). DMF consists of installing a membrane filtration system as previous step to the secondary treatment, allowing the capture of all the MWW particulate fraction before the conventional AS process. Thanks to this separation process, only soluble compounds are fed to the AS treatment, thus dramatically reducing energy demands. Indeed, the biological treatment of the generated effluent could even be unnecessary when using DMF (Hube et al., 2020), allowing the implementation of other nutrients recovery systems as potential post-treatments (see Fig. 1b). The sludge retained in the membrane module during DMF could also be valorized via anaerobic digestion (AD) jointly with other WWTP sludge, increasing the energy recovery of MWW treatment. Additionally, the application of this alternative to existing WWTPs would only require installing compact membrane modules, minimizing the space requirements for its implementation. DMF technology thus represents an interesting alternative for up-grading the currently-in-operation WWTPs without involving important structural or operational modifications.

Despite all the potential benefits that the DMF of MWW entails, fouling is one of most challenging issues in any membrane-based system (Li et al., 2020), seriously hindering DMF due the large variety of pollutants contained in untreated MWW. In fact, numerous DMF studies have focused on developing efficient fouling control strategies to boost its feasibility (Hube et al., 2020). Microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and forward/reverse osmosis (FO/RO) membranes have been applied in several schemes to enhance both energy and nutrients recovery. MF and UF membranes are generally used for primary and secondary sewage treatment, including DMF, while NF and RO membranes are generally used for tertiary treatment due to the higher membrane resistance and driving forces required. Dynamic membranes (DM) have also attracted interest due to their lower prices and operating costs, together with easier physical membrane fouling cleaning than classic membrane systems (Hu et al., 2017a; Xiong et al., 2019), representing an interesting option to DMF. Regrettably, although numerous studies covering this matters can be found in literature, a dramatic variety of results can be found when directly filtering MWW, especially when considering membrane types employed, extremely diverse operating conditions and fouling control strategies. This makes it difficult to achieve general conclusions and to unveil the most suitable conditions for applying this technology.

In this context, this paper aims to thoroughly review the reported DMF studies for MWW treatment and resources recovery. The influence of membrane technology, sewage fed and operating conditions on filtration process feasibility, reclaimed water quality and resource recovery are summarized and discussed, while results from different membrane fouling control strategies are extensively examined, focusing on improving DMF performance. The potential economic and environmental benefits of this alternative are also compared with conventional treatments.

2. Application of DMF technology to MWW treatment

2.1. Pre-treatment

Although a classic pre-treatment is included in DMF when filtering raw MWW (i.e. screening, sieving, de-sanding and de-greasing), several authors have recommended the use of a low size solids screening before the membrane tanks to achieve better filtration performance (Hey, 2016; Nascimento et al., 2018). Thanks to solids screening, higher foulants can be removed from the treated wastewater, reducing the fouling formation propensity (Nascimento et al., 2018) and the possibility of membrane damage. Also, higher flux stabilities have been reported when low size solids screening is included before the membrane filtration due to the reduction in the matter size range (Hey et al., 2017). Strainer or micro-sieving ranging from 500 to 100 μm are usually proposed (Ahn et al., 2001; Hey et al., 2017; Juby et al., 2013; Ravazzini et al., 2005; Sanchis-Perucho et al., 2022a). Alternatively, a two-stage high size screening (1190 and 600 mm) (Sethi and Juby, 2002) or sand-filters (Abdessemed et al., 1999) have also been recommended.

2.2. System configuration

As in membrane bioreactors (MBRs), two main configurations have been defined for DMF technology based on the integration of the membrane and treated influent: side-stream and submerged systems. In the side-stream configuration, the membrane and concentration tanks are placed separately, continuously recycling the treated sludge into the membrane tank at relatively high flow rates and pressures. The main advantage of this configuration is the possibility of applying high cross-flow velocities onto the membrane, thereby providing membrane scouring by the tangential liquid forces during filtration (Shin and Bae, 2018). However, due to the high pumping demands, large energy requirements can be expected when using this configuration (Robles et al., 2018). The side-stream configuration is generally used when treating high-strength sewage (Robles et al., 2018) and has been evaluated for DMF in different studies (Abdessemed and Nezzal, 2002; Bendick et al., 2005; Ramon et al., 2004). In the submerged configuration, the membrane is directly immersed in the concentration tank, avoiding continuous pumping recycling requirements and thereby reducing the energy demands (Huang et al., 2010). However, additional membrane cleaning strategies are required for fouling control, which

Table 1
Application of DMF technology for the treatment of MWW in the middle-/long-term.

Membrane type	Treated influent	Fouling control strategy applied	Operation	Operating time	Ref.
Microfiltration	Raw MWW	Coagulant dosing combined with permeate backwashing every filtration-relaxation cycle. Air scouring applied during backwashing and relaxation periods.	Average operating flux of 41.7 LMH. Off-line chemical cleaning every 3 – 5 days once reached a TMP of 35 kPa.	700 hours	(Zhao et al., 2019)
		Coagulant dosing combined with micro-sieving and membrane air scouring.	Constant TMP of 3 kPa achieving a permeate flux of 6.1 LMH.	159 hours	(Hey et al., 2017)
		Mechanical agitation.	Filtration of low-strength sewage. The filtration resistance raised from $8{\cdot}10^9$ to $4{\cdot}10^{10}$ m^{-1} for an operating flux of 20 LMH.	120 days	(Ahn et al., 2001)
		Air-assisted membrane scouring and permeate backwashing.	Operating flux of 10 LMH. Average total suspended solids concentration in the membrane tank of 1.0 and 2.7 gL^{-1}.	2 – 4 hours	(Sanchis-Perucho et al., 2022a)
		Continuous coagulant dosing combined with air backwashing.	Average operating flux of 10 – 15 LMH achieving a pseudo-steady operation with average TMPs of 20 – 40 kPa.	1600 hours	(Jin et al., 2017)
		Air-assisted membrane scouring combined with intermittent CEBs every 12 hours.	Concentration of the influent COD in two different tanks: Tank 1: concentration factor of 21 at an operating flux of 20.8 LMH. Tank 2: concentration factor from 21 to 50 at an operating flux of 16.7 LMH.	200 hours	(Lateef et al., 2013)
		Coagulant dosing combined with air backflushing.	Operating flux of 20 LMH.	93 hours	(Jin et al., 2015)
		Coagulant + adsorbent (activated carbon) dosing.	Operating flux of 5 LMH.	144 hours	(Gong et al., 2015)
		Continuous coagulant dosing combined with intermittent air scouring.	Operating flux of 13.3 LMH.	295 hours	(Jin et al., 2016)
		Backwashing with ozonized water.	Operating flux about 8 LMH.	180 days	(Kim et al., 2007)
		Membrane vibrations combined with air scouring.	Operating flux of 12.8 LMH.	20 hours	(Mezohegyi et al., 2012)
	Primary settler effluent	Air-assisted membrane scouring and permeate backwashing.	Operating flux of 10 LMH. Average total suspended solids concentration in the membrane tank of 1.1 and 2.8 gL^{-1}.	6 – 8 hours	(Sanchis-Perucho et al., 2022a)
		Intermittent membrane vibrations combined with mechanical agitation and periodical CEBs.	TMPs under 10 kPa for an operating flux of 6.5 – 4.2 LMH.	600 hours	(Kimura et al., 2017)
Membrane type	**Treated influent**	**Fouling control strategy applied**	**Operation**	**Operating time**	**Ref.**
Ultrafiltration	Raw MWW	Cross-flow operation and demineralized water backwashing.	Operating at a constant TMP of 0.3 bar achieved an average flux of 120 LMH.	120 min	(Ravazzini et al., 2005)
		Cross-flow operation.	For an operating TMP of 1.8 bar the flux decreased from 262 to 143 during continuous filtration.	7 hours	(Abdessemed et al., 1999)
		Air-assisted membrane scouring and permeate backwashing.	Operating flux of 10 LMH. Average total suspended solids concentration in the membrane tank from 1.1 to 11.1 gL^{-1}.	35 – 90 days	(Sanchis-perucho et al., 2023)
		Cross-flow operation.	TMPs about 0.2 – 0.4 bar achieved for an average operating flux of 300 LMH.	7 hours	(van Nieuwenhuijzen et al., 2002)
		Intermittent air scouring and permeate backwashing	TMPs around 0.3 – 0.5 bar for an operating flux of 12.7 LMH.	32 days	(Nascimento et al., 2021)
		Coagulant + adsorbent (activated carbon) dosage.	TMPs around 50 – 70 kPa for a changing operating flux of between 5 – 11 LMH.	2 months	(Gong et al., 2019)
		Rotating hollow fibers	Fouling growth rates below 0.1 mbar s^{-1} at an average operating flux between 24 – 30 LMH.	17 days	(Ferrera et al., 2024)
	Primary settler effluent	Cross-flow operation and demineralized water backwashing.	Operating at a constant TMP of 0.3 bar achieved an average flux of 160 LMH.	100 min	(Ravazzini et al., 2005)
		Coagulant + adsorbent (activated carbon) dosing combined with cross-flow operation.	Operating flux of 59.8 LMH.	180 min	(Abdessemed and Nezzal, 2002)
		Air-assisted membrane scouring.	Operating flux of 20 LMH.	72 hours	(Tuyet et al., 2016)
		Coagulant + flocculent dosing combined with permeate backwashing.	Operating flux of 23 LMH.	Few hours	(Delgado Diaz et al., 2012)

(*continued on next page*)

Table 1 (*continued*)

Membrane type	Treated influent	Fouling control strategy applied	Operation	Operating time	Ref.
		Membrane air scouring and permeate backwashing.	Operating flux of 10 LMH.	54 days	(Nascimento et al., 2017)
		Air-assisted membrane scouring and permeate backwashing.	Operating flux of 10 LMH. Average total suspended solids concentration in the membrane tank from 1.2 to 10.6 gL^{-1}.	35 – 122 days	(Sanchis-perucho et al., 2023)
Membrane type	**Treated influent**	**Fouling control strategy applied**	**Operation**	**Operating time**	**Ref.**
Forward osmosis	Raw MWW	Cross-flow operation.	Asymmetric CTA membrane. DS: seawater (osmotic pressure of 26.45 bar).	70 days	(Sun et al., 2016)
		Cross-flow operation (0.28 m s^{-1}).	CTA membrane. DS: NaCl solution (35 g L^{-1}). Continuous DS regeneration by membrane distillation. Stable flux of 17.6 LMH during filtration.	120 hours	(Li et al., 2018)
		Cross-flow operation (from 0.29 to 0.48 cm s^{-1}).	TFC membrane. DS: $MgCl_2 \cdot 6\,H_2O$ (osmotic pressure of 150 bar). Average flux of 5.3 LMH (53 % water recovery). Membrane cleaning required every 24 hours.	24 hours	(Singh et al., 2019)
		-	Pilot-scale spiral wound membrane. DS: NaCl (0.5 M). Average flux of 6 LMH.	450 hours	(Wang et al., 2016)
		Cross-flow operation (from 0.5 to 3 L min^{-1}).	CTA membrane. DS: NaCl (from 0.2 to 4 M).	1500 min	(Gao et al., 2018)
		Cross-flow operation (10 cm s^{-1}).	CTA membrane. DS: synthetic seawater (osmotic pressure of 26.45 bar).	30 days	(Sun et al., 2018)
		Cross-flow operation.	TFC membrane. DS: synthetic seawater.	24 – 46 hours	(Yang et al., 2019)
Dynamic membrane	Raw MWW	External physical cleaning with air backwashing and surface brushing each 48 h.	Three-layer stainless steel mesh of 25 μm as supportive material. Continuous filtration without relaxation at a flux between 33 – 54 LMH for 48 h.	200 hours	(Xiong et al., 2019)
		External physical cleaning with tap water after 50 days of filtration.	One/two layers of a flat open monofilament woven polyamide meshes with 1–5 μm of average pore size as supportive material. Operating flux of 15 LMH.	20 – 102 days	(Sanchis-Perucho et al., 2022c)
		Continuous coagulant dosing. External physical cleaning with tap water after reaching a TMP of 35 kPa.	Dacron mesh of 61 μm as supportive material. Operation flux of 60 LMH. External physical cleaning required every 25 – 50 days.	300 days	(Ma et al., 2013)
	Primary settler effluent	-	Two layers of a flat open monofilament woven polyamide meshes with 1 μm of average pore size as supportive material. Operating flux of 15–45 LMH.	150 days	(Sanchis-Perucho et al., 2022b)

TMP: Transmembrane pressure.
CTA: cellulose triacetate.
TFC: thin film composite.
DS: draw solution.

usually represents the system's main energy input (*e.g.* air-assisted membrane scouring). In conventional MBRs, the membrane can be immersed directly in the bioreactor or placed in an auxiliary membrane tank for sludge filtration (Robles et al., 2018). However, since negligible biological activity is expected in DMF, the reactor and membrane tank can be integrated, thus simplifying the configuration scheme. The submerged DMF configuration is generally used to treat low-strength wastewater (such as MWW) and is the predominant configuration when applying DMF technology to wastewater treatment (see for instance (Delgado Diaz et al., 2012; Nascimento et al., 2017; Sanchis-Perucho et al., 2022a; Tuyet et al., 2016)).

Also as in MBR, hollow fiber (HF) is the most extended membrane module configuration in DMF for treating MWW (see for instance (Jin et al., 2017; Kimura et al., 2017; Nascimento et al., 2017; Sanchis-Perucho et al., 2022a)), although the plate-and-frame configuration is typically used when employing DMs (Sanchis-Perucho et al., 2022b, 2022c; Zhao et al., 2019), which are gaining increasing interest for developing this alternative.

2.3. Membrane technology

Different membrane technologies have been tested for the DMF of MWW, including MF, UF, NF and FO/RO membranes. The lower the membrane pore size, the more the substances retained, increasing both filtration resistance and the required driving force. Due to this increasing filtration demand, MF and UF membranes have been used for primary and secondary sewage treatment, while NF and RO membranes have been more common in tertiary treatments. Nonetheless, FO membranes have been shown to be a technically feasible alternative for treating primary MWW effluents (Lutchmiah et al., 2011) and can be used for producing high quality clean water, whilst secondary and tertiary effluents can be filtered by these membranes for water production (Cath et al., 2010). Alternatively to classic membrane technology, DMs have attracted great interest in the DMF of MWW, which has so far accumulated promising results thanks to the low prices of the supporting materials and relatively easy fouling control. Table 1 summarizes the different operating conditions (membrane technology, influent, and fouling control methodology) in which several authors have studied the DMF of MWW.

2.3.1. Micro- and ultra-filtration membranes

The DMF of MWW is mostly performed by MF and UF membranes (Nascimento et al., 2018). Unfortunately, a severe fouling has been found when filtering untreated MWW (Fujioka and Nghiem, 2015; Gong et al., 2015; Jin et al., 2016, 2015; Kimura et al., 2017), considerably higher than in other MBR systems (Jin et al., 2016; Nascimento et al., 2017). For example, Jin et al. (2016) noted sharply reduced membrane permeability (from 1000 to 20 LMH bar^{-1}) in the first 5 h of operation when treating raw MWW by MF membranes, further declining to about 10 LMH bar^{-1} after 20 h of operation. Jin et al. (2015) also reported a rapid decline in membrane permeability (from 2000 to 20 LMH bar^{-1}) in the first 4 hours of operation, reaching values lower than 10 LMH bar^{-1} in less than 7 hours, while Kimura et al. (2017) found a drastic increase in the operating transmembrane pressure (TMP) (about 45 kPa) in only 5 min of operation. Gong et al. (2015) studied the evolution of the membrane resistance of an MF membrane when treating raw MWW, noting a similar multi-stage performance to those reported when operating other MBRs, but with faster and higher resistance increasing stages. The study of the sources and mechanisms of fouling, as well as the development of efficient fouling control strategies, is thus imperative to achieve feasible DMF applications.

In the case of untreated MWW, a low bio-fouling influence could be expected since any biological activity is involved or desired. However, it contains a large PSD, colloids and dissolved organic and inorganic compounds which can interact with each other and/or cause different problems on the membrane during filtration (Guo et al., 2012; Ravazzini, 2008). Given this diversity of contaminants, pretty complex, diverse, and changing fouling mechanisms can be anticipated when filtering MWW. Membrane pore blocking is characterized by rapid development in the first few minutes of each filtration stage (Iritani, 2013) dramatically reducing membrane permeability. During raw MWW filtration, a large part of the fouling could be related to this mechanism. Sanchis-Perucho et al. (2022a) concluded that irreversible fouling when directly filtering raw MWW and PSE by a MF membrane could be related to intermediate/complete pore blocking, while standard/complete pore blocking controls fouling when using UF membranes. Hube et al. (2021) also determined by means of optical coherence tomography that intermediate pore blocking is dominant during the earlier fouling stages (about 0.5 h of continuous filtration in the cited study), followed by cake layer filtration. Nevertheless, as several authors conclude, other fouling mechanisms could also produce similar outputs. For instance, the wide PSD in MWW may promote the rapid formation of a consistent thick cake layer on the membrane surface. In this respect, Fujioka and Nghiem (2015) and Ravazzini (2008); (2005) studied the fouling mechanisms affecting MF/UF membranes by mathematical approaches during each filtration cycle and concluded that although pore blocking could occur at the very beginning of the filtration stage, the filtration process was mostly governed by cake layer formation. In this case, the faster fouling growth rate in the following filtrations cycles would be related to the compression of the cake layer, reducing the average porosity (Fujioka and Nghiem, 2015). Other authors have also indicated that fouling could be significantly affected by extracellular polymeric substances (EPS) and soluble microbial products (SMP) (Jin et al., 2016; Ravazzini, 2008; Sanchis-perucho et al., 2023). In this respect, Sanchis-perucho et al. (2023)) and Ravazzini (2008) showed that when using UF membranes for both raw and PSE MWW filtration, a significant fraction of the total influent EPS and SMP was captured by the membrane (around 10 – 40 % and 5 – 30 % for EPS and SMP, respectively). These substances could form aggregates that would be absorbed and accumulated on the membrane surface or penetrate into the pores (Ravazzini, 2008). Moreover, Jin et al. (2016) suggested that these substances could induce the formation of a sticky gel layer on the membrane, which strongly retaining deposited particles, especially when using air scouring. Therefore, other fouling mechanisms more related to EPS and SMP concentrations, such as gel formation on the membrane surface, could also significantly contribute to the severe fouling noticed during the DMF.

Regarding fouling nature, different studies using oxidant chemicals (Kimura et al., 2017; Kramer et al., 2015; Lateef et al., 2013; Nascimento et al., 2017; Sanchis-Perucho et al., 2023; Sanchis-Perucho et al., 2022a) and ozone (Fujioka and Nghiem, 2015; Kim et al., 2007) as cleaning agents have identified OM as the main fouling precursor, representing between 68 – 89 % of total membrane resistance. Furthermore, Kramer et al. (2015) evaluated the composition of the cake layer on a ceramic UF membrane and attributed about 98 % of the mass to organic materials. Since only 2 % of the cake layer was attributed to inorganic elements (Na, Al, Si, S and Cl), the authors concluded that salt precipitation was insignificant. However, Nascimento et al. (2017), Lateef et al. (2013) and Sanchis-Perucho et al. (2023); (2022a) maintained that significant inorganic fouling can happen during DMF of MWW, showing the effectiveness of acid chemicals (such as citric acid) for membrane fouling control. Inorganic compound deposition should thus also be considered as a fouling precursor in DMF processes, although to a lesser extent.

Since fouling mitigation is one of the major keys to boosting the feasibility of this alternative at full-scale, the study of the most suitable designs and operating conditions has been the focus of multiple works. The most relevant findings covering this aspect are summarized below, including the membrane materials and pore sizes used, the influent treated, and the most relevant operating conditions evaluated.

2.3.1.1. Membrane materials. Different membrane materials have been used in DMF for MWW treatment, polyvinylidene fluoride (PVDF) being the most common (see for instance (Jin et al., 2016, 2015; Kimura et al., 2017; Nascimento et al., 2018; Sanchis-Perucho et al., 2022a)). However, other membrane materials such as polypropylene (PP) (Hey et al., 2017), polyethylene (PE) (Ahn et al., 2001; Mezohegyi et al., 2012), polyamide (PA) (Ramon et al., 2004), polyethersulfone (PS) (Tuyet et al., 2016) and ceramics (Abdessemed and Nezzal, 2002; Bendick et al., 2005; Fujioka and Nghiem, 2015) have also been used in this field. Despite the numerous membrane materials used, few studies are available on its effects on filtration performance, mainly focusing on MF technology. For instance, Mezohegyi et al. (2012) reported similar short-term fouling propensities (20 h of operation) when using PE and PVDF membranes, while Fujioka and Nghiem (2015) observed similar sharp TMP increases (from 49 to 143 kPa in the first 40 min of filtration) using ceramic membranes. Despite the little research performed so far, it can be concluded that the membrane material used is not a key variable in the DMF of MWW, as similar filtration performance can be expected whatever the material used.

2.3.1.2. Membrane pore size. The selection of an adequate membrane pore size is a crucial issue in any membrane-based process. Different fouling mechanisms and propensities are triggered by the interaction between the applied membrane pore size and the treated particle suspension (Ravazzini, 2008). The effective pore size is thus crucial in controlling membrane fouling and is a key aspect in achieving feasible processes. In this respect, few studies have been conducted to compare MF and UF membrane performance when filtering untreated MWW, results from different studies difficult to compare due to the extremely different flux, fouling control strategy, and influent source used in each case (see Table 1).

MF membranes could present better performance due to their large pore size, which reduces membrane filtration resistance. Nevertheless, due to the large particle size distribution (PSD) range and the sticky substances present in untreated MWW, large pores may favor the settling of different types of particles in the pores, favoring pore narrowing, which would finally result in pore blocking. On the other hand, UF membranes require higher energy inputs to perform filtration due to their inherent higher filtration resistance. However, reducing the effective pore size may give better filtration performance by reducing the number of particles that can interact with the pores. In this regard, Sanchis-Perucho et al. (2022a) operated MF (0.4 μm pore size) and UF (0.03 μm pore size) membranes for filtering untreated MWWs, reporting higher membrane fouling increases in the former than in the later, despite filtering the same influent in parallel. Filtration was only possible for 2 – 8 hours when using the MF membrane, rapidly achieving TMPs of over 0.4 bars, whilst the UF module was operated for 30 – 70 days before reaching similar TMPs. Additionally, Kramer et al. (2015) operated a UF ceramic membrane (3 kDa pore size) for the treatment of raw MWW for 22 h and found a reduced membrane permeability of about 16 %, which is clearly lower than that reported in many other studies on MF membranes. Nonetheless, it is important to consider that reducing membrane pore size increases not only the inherent membrane resistance but also favors thicker cake layers due to the accumulation of more types of particles and substances on the membrane surface. In fact, when the last authors (Kramer et al., 2015) compared the filtration performance of raw MWW by UF and NF membranes (3000 and 450 Da of pore sizes, respectively), permeability decreased more when using NF membranes (from 5.9 to 2.5 LMH bar^{-1} and from 5.8 to 4.8 LMH bar^{-1} for the NF and UF membranes in the first 22 – 24 h, respectively), showing that reducing the membrane pore size does not always reduce the membrane fouling propensity. A similar experience was reported by Ahn et al. (1998), who observed a significant resistance increase when a lower pore size UF membrane was used (from 300 to 15 kDa) for treating hotel wastewater in the short-term. The above-mentioned studies seem to indicate that the most convenient membrane pore size is closely related to the influent's characteristics, further studies thus required on this field for helping with DMF proper membrane selection.

2.3.1.3. Treated influent. As above stated, the influent MWW can also significantly impact filtration performance in the DMF of MWW. Indeed, the treated influent's PSD has been shown to have a strong influence on membrane fouling in different membrane systems (Abdessemed and Nezzal, 2002; Ahn and Song, 2000). Some authors have suggested that the large PSD in raw MWW can rapidly produce the formation of thick cake layers on the membrane surface (Ravazzini, 2008). In addition, the irregular shape and diverse material in the particles present in MWW could favor the development of simultaneous fouling mechanisms during filtration, thus increasing filtration resistance and reducing the effectiveness of on-line cleaning strategies.

The treatment of more depurated influents (*i.e.* after a pre-treatment) has been recommended to reduce fouling during DMF (Kimura et al., 2017; Ravazzini et al., 2005; Sanchis-Perucho et al., 2022a), such as coagulants and micro-sieving, which can be used to reduce the particulate material entering the filtration membrane modules (Hey et al., 2018, 2017). Nevertheless, a conventional primary settler is the option most often used (Kimura et al., 2017; Ravazzini et al., 2005; Sanchis-Perucho et al., 2022a). Primary settling can recover and concentrate a great fraction of the influent's total suspended solids (TSS) by sedimentation, so that only the smaller particles are fed to the DMF modules. Also, since primary settling is a fundamental part of almost all WWTPs, using the primary settler effluent (PSE, *i.e.* supernatant from primary settling) to feed the DMF unit would not require additional WWTP up-grading, making it an attractive alternative. However, using PSE instead of raw MWW during DMF does not seem to provide filtration benefits. Kimura et al. (2017) found a rapid TMP increase (up to 50 kPa in less than 5 h) when treating PSE MWW by an MF HF membrane. Since similar results have been reported when treating raw MWW by this type of membrane, similar filtration performance can be expected, regardless of the influent used. In fact, Ravazzini et al. (2008); (2005) concluded that similar fouling levels and mechanisms can be expected during continuous filtration, regardless of the influent treated (*i.e.* raw or PSE), while Sanchis-Perucho et al. (2022a) found that fouling propensities are even higher with PSE instead of raw MWW due to the reduction of the average particle size in the treated sludge. On the other hand, these authors (Ravazzini, 2008; Ravazzini et al., 2005) also reported that higher permeate production rates can be obtained during short-term filtration when treating PSE (around 20 – 70 % higher compared with raw MWW), depending on the operating conditions, while further improvements could be achieved in membrane permeability by physical cleaning when treating PSE, especially with high TMPs and low cross-flow velocities. These results were attributed to both the reduced particulate material fed to the filtration modules and the smaller average particle size when treating PSE than in raw MWW, promoting the formation of thinner but denser cake layers in the former than in the latter (Ravazzini, 2008). Therefore, although the severe fouling during the DMF of MWW could not be directly mitigated by using a primary settler as influent pre-treatment, this alternative could be an interesting option to improve fouling control strategies effectivity during filtration and improve DMF feasibility.

2.3.1.4. Operating conditions. Operating conditions when treating MWW seem to play a more important role in filtration performance than changes to the feed characteristics (Ravazzini, 2008), selecting a suitable permeate flux, TMP or filtration/relaxation ratio essential to achieve energy-efficiency and cost-effectiveness. In this respect, Ravazzini (2008) and Ahn et al. (1998) studied the effect of different TMPs during raw MWW filtration with UF membranes and reported that raising the TMP may only increase permeate productivity in the early stage of operation, reaching later similar low fluxes to those obtained at lower TMPs. However, high TMPs may favor fouling in the membrane by faster material transfer onto the membrane surface (Ravazzini, 2008). High TMPs could also favor the formation of denser and more compact cake layers during DMF of MWW and raise filtration resistance, since cake layers formed in different MF and UF MBRs are often compressible (Ravazzini, 2008). In fact, Ravazzini et al. (2005) and Fujioka and Nghiem (2015) suggested the development of compressible cake layers during DMF, while Jin et al. (2016) reported that relatively dense cake layers could be formed in these systems. Ravazzini (2008) also used a mathematical model to show that compressible cake layers can be expected during the DMF of MWW, concluding that TMP is the main factor affecting filtration resistance. The use of medium-to-low TMPs during filtration (about 0.3 – 0.5 bar) has thus been recommended to minimize fouling in DMF systems (Ravazzini, 2008; Ravazzini et al., 2005). The same authors (Ravazzini, 2008; Ravazzini et al., 2005) also recommended operating at short filtrations times of only a few minutes to reduce backwashing frequency and achieve higher net permeate production.

Regarding permeate flux, the most suitable values have not yet been clarified. Sanchis-Perucho et al. (2023) reported severe fouling when the operating transmembrane flux was raised from 10 to 12.5 and 15 LMH when directly filtering raw MWW and PSE with a UF membrane (fouling rose from 0.5 – 0.54 to 4.43 – 5.30 and 11.91 – 12.25 mbar day^{-1}, respectively). These authors thus concluded that moderate transmembrane fluxes (about 10 LMH in the cited study) are recommended in this process to reduce filtration operating costs. Other authors have furthermore reported a negative effect on filtration performance when using high initial permeate fluxes in various MBRs. For instance, Madaeni (1998) found these hindering effects when treating inorganic colloidal solutions, Ho and Zydney (2000) with protein solutions and Ahn et al. (1998) when filtering MWW. However, high permeate fluxes may overcome the electrostatic rejection between the treated mixed liquor and surface membrane (Ravazzini, 2008) and accelerate the transition from initial pore blocking fouling to cake layer filtration (Velasco et al., 2003), thereby preventing irreversible fouling. In any case, when Nascimento et al. (2017) studied the effects of starting with high fluxes or applying a step-flux-increase method for filtering PSE with UF membranes, similar fouling growth rates were obtained whatever the operating flux and so concluded that both strategies are valid approaches for filtering MWW.

On the other hand, increasing the operating solids concentration in the treated sludge appears to offer benefits in terms of fouling mitigation. Lateef et al. (2013), Jin et al. (2016) and Mezohegyi et al. (2012) reported that despite raising the concentration of the bulk to be filtered (sludge concentration factors referring to influent concentration of from 5 to 25, 21 to 50 and around 11 to 46, respectively), similar fouling propensities were achieved, in some cases even less fouling detected than in fresh raw MWW filtration (all results obtained from filtering raw MWW by MF membranes). In fact, Sanchis-Perucho et al. (2023); (2022a) showed that significantly less fouling growth can be obtained for both raw MWW and PSE UF by increasing the total suspended solids concentration in the membrane tank (from about 1 to 11 g L^{-1}). There was therefore stated that the particulate material was not the main fouling promotor but the colloidal and/or soluble fraction. Similarly to other MBR studies (see for instance (Drews, 2010; Meng et al., 2009)), the concentration of SMPs or colloidal EPSs could thus be the key factor in fouling during DMF of MWW (Mezohegyi et al., 2012).

2.3.2. Osmosis membranes

Osmosis membranes are built on semi-permeable materials able to retain from colloidal material to even different monovalent ions depending on their molecular size (Kumar et al., 2013). Polymeric materials dominate the market due to their excellent performance and low cost (Yang et al., 2019), mainly in the form of thin-film composite (TFC) and cellulose base (CA) membranes. TFC membranes are clearly predominant due to their higher permeability, lifespan, pH stability and resistance to hydrolysis and biological degradation (Ferrari, 2020). However, CA membranes, especially those of cellulose triacetate (CTA), are still available due to their exceptional resistance to chlorine (Yang et al., 2019). This chlorine resistance is particularly interesting since membrane cleaning and disinfection are usually conducted using NaOCl, allowing these membranes to achieve longer lifespans when this chemical is used.

Osmosis membranes can be operated either by pressure or concentration gradients, classically differentiating between RO and FO processes. Concerning their applicability to DMF, FO membranes have been proposed for directly treating MWW (Ansari et al., 2017) since these membranes generally have lower fouling propensities than other pressure-driven membranes (Wang et al., 2016). Instead, RO membranes require an additional pre-treatment for MWW filtering, since their higher sensitivity to membrane fouling (Wang et al., 2016) involves higher energy costs due to the high operating pressures required. RO membranes are thus not normally applied to DMF of MWW, the application of these membranes however especially interesting for treating the effluent produced from DMF, which can be used to boost nutrient recovery and enhance the quality of the permeate produced.

2.3.2.1. Viability of FO for directly filtering MWW. Despite their higher filtration resistance, the viability of FO membranes for DMF of MWW has been proven by several authors. Zhang et al. (2014) were able to multiply the influent MWW concentration by 6 in a lab-scale CTA cross-flow membrane module, while, Ansari et al. (2016) demonstrated the feasibility of using a CTA membrane, also on a lab-scale, for concentrating raw MWW (concentration factor of 10), reaching COD concentrations in the membrane retentate high enough for directly feeding an AD. Moving to pilot-scale studies, Wang et al. (2016) reported stable long-term filtration (average flux 6 LMH for 51 days) when treating MWW with a CTA spiral-wound membrane, achieving a concentration factor of 5 (Wang et al., 2016). Unfortunately, membrane fouling is also a serious issue when filtering MWW with FO membranes, especially when operating at high feed concentration factors (Ansari et al., 2016; Gao et al., 2018). In fact, more severe organic, inorganic and even biological fouling has been reported in DMF than in activated sludge filtration in similar operating conditions (Sun et al., 2018). Gao et al. (2018) reported flux decreases of from 25 – 20 to 10 – 5 LMH in a TCA FO membrane when filtering MWW for 1500 min, due to both DS dilution and fouling. In this case, the authors determined that a significant fouling cake layer is formed after 500 – 700 min of operation, which rose as the initial flux was increased. Moderate osmotic pressures can thus be recommended for fouling mitigation (Gao et al., 2018). On the other hand, Ansari et al. (2016) reported a 50 % flux decline after 70 h of filtration, starting after the first 40 h and evolving consistently (linear trend) as the filtration continued, whilst Zhang et al. (2014) also reported flux reductions (from 8 – 6 to 4 – 3 LMH after 17 h filtration) when operating with a CTA FO membrane. In this last study, however, two different membrane alignments were tested (active layer in contact with the DS or feed), with a clear benefit when the active layer was in contact with the feed. As the authors concluded, a less thick fouling cake layer was formed in the latter orientation, related to the smoothness of the active membrane layer (Zhang et al., 2014). Ferrari (2020), on the other hand, reported that the drop in flux during the filtration of MWW (39.9 hours) was associated with both fouling and concentration polarization.

Concerning the nature of the fouling formed, some authors reported a loose fouling layer when filtering MWW by FO membranes that could easily be removed (Ansari et al., 2016; Gao et al., 2018; Lutchmiah et al., 2011) making it mostly reversible. In this respect, Gao et al. (2018) reported that although about 20 – 40 % of the original membrane permeability was lost after 1500 min of MWW filtration, 90 % was recovered after physical cleaning. Similarly, Ansari et al. (2016) reported complete membrane permeability recovery after a simple membrane flushing, despite having lost 50 % of the permeate flux in 70 h of continuous filtration. Nevertheless, significant irreversible fouling has also been reported when treating MWW. Gao et al. (2018) found that chemical cleaning (1 % NaOCl for 30 min) was necessary to raise membrane permeability from 90 % to 96 % of its original value after 1500 min of operation. These authors consequently assumed that 90 % of the fouling was reversible, 6 % could be considered as easy to remove (practically reversible) and the rest was 4 % irreversible. Less satisfactory results were reported by Ferrari (2020), who only recovered 80 % of the original operating flux after an osmotic backwash (OBW). Fouling besides later intensified, reporting average flux declines after every filtration cycle (4 in total) of 14.6, 24.4 and 26.8 %, respectively, regarding the first filtration cycle (cycle filtration times between 24.5 – 37.2 hours). Regarding the fouling source, Gao et al. (2018) determined that the developed cake layer was mainly formed by carbon, calcium, magnesium, silicon, aluminum and phosphorus, mostly composed of polysaccharides, humic acids and proteins present in the MWW. Light chemical cleaning (1 % NaOCl for 30 min) was easily able to remove the aluminum and phosphorus, which were considered as reversible foulants. Instead, calcium and silicon were more adhesive and not easy to remove, considered as irreversible fouling promotors (Gao et al., 2018).

2.3.2.2. Other aspects to consider. In addition to membrane fouling, when operating osmosis membranes other phenomena are important, such as concentration polarization and reverse solute flux (also known as negative rejection) (Tang et al., 2010). Concentration polarization can take place in the proximity of the membrane, concerning the feed and permeate solutions themselves, which is then defined as external concentration polarization, or in the pores or dense material of the membrane, being known as internal concentration polarization (Qin et al., 2010). External concentration polarization can be mitigated by any fouling control strategy that affects the hydrodynamics, such as gas sparging or crossflow operation, while internal concentration polarization is difficult to treat. Concentration polarization has a significant impact when filtering MWW and was found to be the most important filtration resistance in several studies. Wang et al. (2016) identified an enhanced concentration polarization cake as the major contributor to flux decline during MWW filtration, while Ferrari (2020) recognized external concentration polarization as an important flux reducer, whose effects were mitigated by gas scouring. Zhang et al. (2014) informed that facing the active membrane layer to the feed solution could be an interesting option to reduce the negative effects of internal concentration polarization, since it could reduce the fouling thickness and pore plugging, thereby improving contact between the membrane surface and operating solutions. On the other hand, reverse solute flux is exclusive to FO filtration and is due to the flux of salts from the DS to the feed during filtration. This salt flux is contrary to permeate flux and reduces the system's effective osmotic pressure and raises filtration costs (Vinardell et al., 2021). The salts accumulated in the feed solution can also inhibit the subsequent AD and compromise the main objective of this alternative treatment (Ansari et al., 2016; Gao et al., 2018). Several authors have reported a significant reverse solute salt flux when filtering MWW with FO membranes (Gao et al., 2018; Vinardell et al., 2021), which could force the use of moderate DS salt concentrations or limit the MWW concentration to moderate values to prevent AD inhibiting effects.

The accumulation of toxic substances is another limitation that needs to be taken into account when concentrating MWW with FO membranes. NH_3 and different salts can easily be retained and concentrated in FO membranes, which are known to be strong AD inhibitors when achieving high enough concentrations in the feed (Ferrari, 2020). Other classical or emerging contaminants (SO_4^{2-}, heavy metals, pharmaceutical products, hormones, microplastics, etc.) are also susceptible to being retained in high extents by FO membranes (Al-Obaidi et al., 2020), which may hinder AD and cause negative effects on methane production. Depending on the concentration of these substances in the influent MWW, moderate concentration factors of around 1 – 10 can be recommended to avoid possible problems with the subsequent AD system (Ferrari, 2020).

Apart from limiting the operating FO retentate concentration, the draw solution (DS) used is also a relevant matter to consider. Different artificial saline solutions can be efficiently used for filtration, such as NaCl, KCl, or $MgSO_4$, among others (Chekli et al., 2012). However, seawater is emerging as an attractive alternative in the DMF concept as it would not only eliminate the chemical demand and DS regeneration requirements, but the final salt-diluted solution obtained after the process could be used to produce drinking water by RO (Teusner et al., 2017). This combination has already demonstrated several advantages, like lower RO fouling tendencies, and has achieved higher energy and economic savings than the classic RO desalination (Teusner et al., 2017). Unfortunately, however, a high salt content may reach the AD due to the high flux of salts that may travel from the DS towards the concentrated MWW during filtration and inhibit the process (Ansari et al., 2016; Gao et al., 2018). Consequently, the use of organic DS, such as sodium acetate or EDTA-2Na, has been recommended when treating MWW (Ansari et al., 2016), as have moderate DS concentrations (Gao et al., 2018) due to the higher and quicker fouling formations noticed when operating at relatively elevated permeate fluxes (20 – 25 LMH).

2.3.3. *Dynamic membranes*

DMs have also attracted a great interest for the DMF of MWW. DMs consist of two different layers: (1) a low filtration-resistance supporting material, and (2) a developed cake layer. The supporting material acts as a base on which the particles in the influent can settle until a stable cake layer is formed that acts as the main filter (Usman et al., 2021). Thanks to removing the intrinsically filtration resistance of conventional membranes, better permeability can be achieved during filtration, while the filtration performance can be controlled by acting on the thickness and density of the formed cake layer (Xiong et al., 2019). This changes the membrane fouling paradigm, in DMs playing a partially beneficial role often easier to control by low energy-demanding physical cleaning (Xiong et al., 2019). In addition, the supporting structures used are generally low-cost materials such as woven meshes or filter-cloths, representing a significant lower acquisition and/or replacement cost than conventional membrane modules (Hu et al., 2018). Despite all DMs' potential benefits, the influent characteristics (e.g. particles size, pH, temperature, etc.) and operating conditions (*e.g.* supporting material, flux, filtration time, cleaning, etc.) have a strong influence on filtration resistance and permeate quality (Gong et al., 2014; L. Li et al., 2018), in some cases hindering their applicability. Indeed, significantly worse permeate qualities can be expected when using DMs instead of other membrane technologies, since the formed cake layer presents a less homogeneous structure with higher porosity than commercial membranes. Also, since the formed cake layer mainly controls the filtration performance, the generated permeate in DM filtration may be unstable to some degree, with changes as the DM structure evolves during filtration. On the other hand, the formation of a stable DM can also be a difficulty in some cases, since its formation strongly depends on the characteristics and concentration of the influent's suspended material and its interaction with the supporting structure, so that selecting a suitable supporting material for the treated influent is a key factor in achieving good DM performance.

2.3.3.1. Dynamic cake layer formation. There are two different DM types, depending on how the filtering cake layer is formed on the supporting structure: self-forming and pre-coated (Mohan and Nagalakshmi, 2020; Usman et al., 2021). The former is formed when the filtering cake is developed by direct deposition of the particulate material onto the supporting structure during filtration, while the later consist of previously forming a stable structure on which the influent particulate material can be deposited to form the filtering cake layer. Pre-treatment with external solutions containing additional particulate material (*e.g.* powdered carbon or kaolite) can be used to pre-coat the supporting structure before starting real influent filtration. Pre-coating DMs are in theory less advantageous, since they require auxiliary chemical dosing, which increases the operating cost (Mohan and Nagalakshmi, 2020; Sanchis-Perucho et al., 2022b). However, due to the high PSD and fewer sticky substances in MWW than in other membrane-based bio-reactors, DM formation-improving strategies may be required to properly apply this technology in the DMF alternative. Instead of previously pre-coating the supporting material, adding chemicals during filtration has also been a common solution (Ersahin et al., 2012). Thanks to increasing the average particles size by flocculation, quicker and more stable DM developments can be achieved, allowing consequently the use of higher supporting material pore sizes. Coagulant dosing could be used during the first DM development step (as a kind of pre-coating), favoring DM formation only, or during the entire operating period, also controlling membrane fouling and permeate quality. In this regard, Ma et al. (2013) reported stable permeate quality in 1 – 2 days of operation when dosing polyferric sulfate in a 61 μm Dacron mesh, while Gong et al. (2014) achieved a stable DM in just 2 hours when dosing diatomite in a 100 μm propane polymer mesh, both treating raw MWW. Similarly, Sanchis-Perucho et al. (2022b) added polyaluminum chloride (PACl) during PSE filtration with a 1 μm polyamide monofilament mesh, reducing DM formation from about 17–7 days. Another alternative to enhance DM development and permeate quality is to modify the supporting material's sewing or thickness (Ersahin et al., 2013). Xiong et al. (2019) studied the effect of using multi-layer supporting materials for raw MWW treatment, reporting significant increments on the OM recovery and operating TMP when increasing the added layers. Despite the membrane resistance raise, the authors selected the three-layer structure due to the significant OM recovery improvement and rapid DM layer formation. This conclusion is in disagreement with the results reported from AnMBRs operated with DMs (AnDMBR), which usually recommend the use of simple mono-filament mesh supporting materials due to their lower filtration resistance (Ersahin et al., 2013; Li et al., 2018). This disagreement is probably due to the significant difference in the PSD of raw MWW compared to biological sludge, the later easier to retentate by DMs.

2.3.3.2. Supporting materials and pore size. Metallic, ceramic and polymeric materials are commonly used as supporting layers when working with dynamic membranes (see for instance Ersahin et al., 2012; Li et al., 2018). When treating raw MWW, Xiong et al. (2019) found that nylon and stainless-steel meshes can be used to quickly develop a dynamic membrane, with better performance at the same pore size by the stainless-steel mesh when considering OM recovery and filtration resistance. Dacron and propane polymer meshes have also been shown as suitable supporting materials when treating raw MWW (Gong et al., 2014; Ma et al., 2013), while polyamide has been used for both raw MWW and PSE filtration (Sanchis-Perucho et al., 2022b, 2022c). Regarding the supporting material pore size, a range of between 1 and 25 μm has been used for both raw MWW and PSE treatment (Gong et al., 2014; Ma et al., 2013; Sanchis-Perucho et al., 2022b, 2022c), achieving stable DM formations in between 10 – 24 hours and 4 – 17 days. In this respect, focusing on raw MWW treatment, Xiong et al. (2019) and Sanchis-Perucho et al. (2022c) identified 25 – 5 μm, respectively, as the most suitable pore size when considering permeate quality and the TMP operating rise, while Gong et al. (2014) suggested 1 μm due to its higher OM recovery efficiency and lower fouling growth rate. Considering PSE treatment, Sanchis-Perucho et al. (2022b) found that a 1 μm pore size may not be enough for the self-formation of a stable DM in the short-term, requiring the use of additional supporting material layers in this case. This was due to the lack of particulate materials when filtering this type of influent, as well as a reduction in the average size of the influent particles. On the other hand, Gong et al. (2014) reported that with 1 and 10 μm pore sizes, a cake layer filtration mechanism was developed when treating raw MWW, identifying a complete blocking for 50 μm and a standard blocking for 100 μm. As these authors suggest, the DM filtration mechanism change was probably due to the PSD in the influent (particles size concentrations between 20 and 50 μm). Because of the dramatic rise in membrane resistance observed for 50 and 100 μm pore sizes, small pore sizes can be recommended to avoid complete membrane pore blocking (Gong et al., 2014). The proposed supporting material pore sizes are similar to those reported by AnDMBR (Ersahin et al., 2016, 2013), which could indicate that similar DM formation mechanisms take place, despite the different characteristics of the wastewater treated.

2.3.3.3. Operating aspects. Good filtration performance was obtained during DM operation for raw MWW and PSE treatment when fluxes of around 45 – 60 LHM were applied (Ma et al., 2013; Sanchis-Perucho et al., 2022b; Xiong et al., 2019), which clearly overcomes regular MF and UF fluxes. Stable operating TMPs of between 20 – 40 kPa have also been reported in the middle-term (200 h of operation) (Xiong et al., 2019) and long-term (100 – 300 days of operation) (Ma et al., 2013; Sanchis-Perucho et al., 2022b, 2022c). Regarding fouling development, Gong et al. (2014) found that all fouling after more than 60 h of operation could be considered reversible, completely recovering the initial flux by mechanical cleaning. However, Xiong et al. (2019) observed a slight descend in the operating flux around the first 100 h of operation, despite the physical cleanings performed, which was attributed to significant irreversible fouling, as has been noted in other DM systems (Hu et al., 2017b, 2016). Similar results were also reported by Sanchis-Perucho et al. (2022c), who informed of a significant reduction of the initial membrane permeability (from about 370 to 90 LMH bar^{-1}) despite physical cleaning after 50 days of operation. Additionally, Ma et al. (2013) determined that seasonal temperature changes did not affect the fouling propensity. Air backwashing and surface brushing are commonly used as cleaning methods when operating DMs (Hu et al., 2017a; Xiong et al., 2019). However, other strategies can also be used to mitigate fouling, such as any method employed in MF and UF systems.

3. Membrane fouling control strategies

Membrane fouling is an important issue in any membrane-based system. Fouling during filtration entails significant increases in the process energy demands that may affect its technical/economic viability. Once membrane permeability is hindered by irreversible fouling, chemical cleaning by basic and acid solutions is the only effective method to recover it. However, proper membrane chemical cleaning is done off-line and requires stopping the process for several hours (membrane suppliers may recommend direct contact between membrane and the cleaning solution of 14 – 20 hours) while their use sharply reduces the membrane lifespan. To prevent premature membrane chemical cleaning and enhance filtration feasibility when filtering with solutions with high propensities to form fouling (such as MWW), multiple on-line cleaning strategies have been developed.

3.1. Physical methods

Physical cleaning strategies for fouling control have been extensively used in different MBRs (Weerasekara et al., 2014) and are mainly based on mechanical forces to continuously detach fouling formations from the membrane surface. Due to the nature of the applied cleaning, physical methods mainly act on reversible fouling mitigation, losing effectivity as irreversible fouling appears (Weerasekara et al., 2014). Since significant energy inputs can be required depending on the employed fouling control strategy, their proper selection, application, and optimization is an essential matter in any energy-efficient and cost-effective filtration process.

3.1.1. Air scouring

Air-assisted membrane-scouring is widely used for membrane fouling control and has been successfully applied in numerous membrane systems (Weerasekara et al., 2014). Continuous air sparging is generally able to prevent fast particle deposition on the membrane surface by the induction of local shear stress and liquid flow fluctuations near the membrane surface, thus improving membrane permeability, especially in the early stages. In HF membranes, air sparging also favor cake layer detachment by the swaying of the membrane fibers in the module package, reducing clogging problems, especially important in HF high packing density modules (Robles, 2013).

In the case of DMF of MWW, different studies have shown the effectiveness of air scouring for reversible fouling control in MF and UF membranes (Jin et al., 2016; Mezohegyi et al., 2012; Nascimento et al., 2017). Jin et al. (2016) showed that after 5 h of operation, membrane permeability could be held around 10 times higher when high specific air demands (SAD) (about 0.6 $Nm^3\ m^{-2}\ h^{-1}$) are applied, whilst Nascimento et al. (2017) reported that the application of intensive aeration (56 m^3 of air per m^3 of permeate) enabled filtration at low TMPs (around 150 mbar) and moderate fouling growth rates for a period of 40 days. Air/gas sparging have also been shown to be effective in improving FO membrane filtration. For instance, Ferrari (2020) reported that air/gas sparging significantly reduced FO filtration flux decline by reducing membrane fouling and concentration polarization. 70 % water recovery was achieved in 40.1 % less time with continuous air sparging, while 17.4 % less time was required when intermittent N_2 sparging was used (both compared with an FO test without air-assisted fouling control). This author also performed a flux test after around 24 – 40 hours of filtration, noting that the flux decline compared to the original clean membrane was about 5.7, 12.4 and 31.3 % for continuous air sparging, intermittent N_2 sparging and no gas injection, respectively (Ferrari, 2020). The author concluded that gas sparging (even intermittent) has a strong anti-fouling effect and can control both, reversible and irreversible fouling.

Despite the effectiveness of air sparging in controlling membrane fouling, several drawbacks are involved when applying this strategy to DMF. First of all, intensive SADs are usually required to suitably mitigate fouling, and these are significantly higher than those reported in MBRs (Nascimento et al., 2017). As some authors point out (see for instance (Nascimento et al., 2017)), these high SADs requirements could be related to a specific rise in resistance of the solids deposited on the membrane surface during filtration of untreated MWW compared to that in MBR-activated sludge. Since air scouring commonly represents the most energy demanding element in MBRs (Weerasekara et al., 2014), other cleaning strategies should be considered to improve the DMF energy balance (Kimura et al., 2017; Nascimento et al., 2017). On the other hand, the combination of small particles in untreated MWW (*e.g.* sand or crystalline precipitates) and air sparging may produce abrasions on the membrane surface and reduce the membrane lifespan (HEY, 2016). Moreover, continuous membrane cleaning by air scouring could result in a more vulnerable membrane surface against EPS and SMP (Jin et al., 2016), these compounds being identified as the main membrane fouling promotors in MBRs (Drews, 2010). In this respect, Jin et al. (2016) reported that air sparging was ineffective in the later stages of filtration, which the authors justified as due to the possibly faster gel layer formation in these systems than in other MBRs. This gel layer could strongly attach particles onto the membrane surface, drastically reducing the effectiveness of air sparging to control membrane fouling (Jin et al., 2016; Yu et al., 2006). On the other hand, the continuous injection of air into the system can reduce the DMF energy recovery potential by promoting the development of heterotrophic organisms, which consume the OM retained in the system (Gong et al., 2015; Jin et al., 2016, 2015). OM degradation by heterotrophs could besides raise the EPSs and SMPs in the membrane tank, contributing to membrane fouling. Then, since significant heterotrophic activity has been reported, even with low air flow rates (Ferrari, 2020; Jin et al., 2015; Lateef et al., 2013), mechanical systems have been recommended instead of aeration, even for mixing (Jin et al., 2016; Kimura et al., 2017; Lateef et al., 2013).

3.1.2. Cross-flow operation

Based on the side-stream configuration, cross-flow operation has also been applied in numerous membrane systems for fouling mitigation. Thanks to introducing a flux perpendicular to the permeate flow with significant tangential velocity, rapid particle accumulation can be mitigated on the membrane, reducing the thickness of the formed cake layer (Ravazzini, 2008). Also, due to the turbulence and scouring effects promoted on the membrane, the back-transport of the deposited solids is favored (Ravazzini, 2008), especially during relaxation periods, when the cake layer can be detached from the membrane surface.

The effectiveness of cross-flow for fouling mitigation in DMF has been thoroughly proven (Ahn et al., 1998; Ansari et al., 2016; Hao et al., 2006; Ravazzini, 2008; Ravazzini et al., 2005). Ravazzini (2008) showed the usefulness of this strategy for fouling control when treating both raw and PSE MWW using UF membranes. In this study, filtration resistance was considerably reduced by increasing the cross-flow velocity from 1 to 2 m s^{-1}, allowing to operate for 7 h without serious fouling. Ahn et al. (1998) reported similar results when operating MF and UF ceramic membranes for hotel wastewater filtration, achieving stable TMPs for more than 12 h of continuous filtration using cross-flow (2.5 – 4 m s^{-1}). Ansari et al. (2016) also showed that cross-flow can be used when employing a CTA FO membrane to filter MWW. Water recoveries of only 70 % were achieved in this last study when the cross-flow velocity was set at 9 cm s^{-1}, reporting a considerable flux decline during filtration due to severe fouling. However, this fouling was significantly reduced when the cross-flow velocity was almost doubled (17 cm s^{-1}), in this case achieving 90 % water recovery efficiency. This strategy thus showed its effectiveness, regardless of the membrane pore size used (MF, UF and FO membranes) and treated influent (raw or PSE MWW). Nevertheless, since high cross-flow velocities during filtration represent considerable energy costs due to intensive pumping, the operation must be optimized. According to Hao et al. (2006), who used MF membranes for raw MWW filtration, increasing cross-flow velocity (ranging from 2 to 3.7 m s^{-1}) always significantly improves the flux. Similar improvements of the permeate flux regarding the applied cross-flow velocity (ranging from 1 to 4 m s^{-1}) were reported by Ahn et al. (1998) operating ceramic MF and UF membranes for hotel wastewater treatment. with a UF membrane for raw and PSE MWW treatment, Ravazzini (2008) showed that cross-flow velocities above 2 m s^{-1} did not significantly improve the process stability or permeate production, so that they recommended operating around 1.5 m s^{-1}.

Besides controlling the fouling growth rate during filtration, enhanced shear conditions provided by cross-flow operations could affect the developed fouling by changes in the membrane's material transportation. Shear conditions can affect the protein aggregates in the treated solution or induce the collection of inorganic colloids around the membrane pores (Ravazzini, 2008). High cross-flow velocities can also boost irreversible fouling due to the higher permeation drag under these conditions (Choi et al., 2005). In this respect, Ravazzini (2008) found that, although fouling was mainly reversible when filtering raw and PSE MWW with a UF membrane in cross-flow mode, significant irreversible fouling always appeared at the beginning of the operation, regardless of the cross-flow velocity applied. On the other hand, the main interacting forces among the particles are related to particle size in cross-flow UF (Ravazzini et al., 2005). The use of more depurated influents (*e.g.* PSE) could thus affect cross-flow cleaning effectiveness. In this regard, when treating PSE by UF membranes, Ravazzini et al. (2005) detected that a denser cake layer could be formed on the membrane surface due to a reduction in the average pore size of the deposited particles. This phenomenon was highlighted by the authors, since the cake layer formed in these conditions was more effectively removed by the applied cross-flow, thereby reducing membrane resistance. Concerning the operating permeate flux, the use of middle/low fluxes can be recommended due to the reduction of the transporting material forward of the membrane surface during filtration, which has been reported as benefitting the cross-flow scouring effects (Ravazzini et al., 2005). However, the permeate flux needs to be optimized in each case to maximize wastewater treatment without compromising membrane fouling.

3.1.3. Backwashing

Together with air scouring, the use of backwashing with tap water or generated permeate is the usual practice for cake layer removal in MF and UF membranes (Kramer et al., 2015). Thanks to periodic backwashing, the formed cake layer can be detached from the membrane surface, recovering membrane permeability and relieving the cake layer compression. Unfortunately, backwash has been shown to be an inefficient strategy for fouling mitigation in DMF. Kramer et al. (2015) found no differences in the filtration performance when intensive backwashing was applied hourly with a ceramic UF membrane. Similarly, Fujioka and Nghiem (Fujioka and Nghiem, 2015) showed that backwashing alone could not control membrane fouling, even when it was performed every filtration cycle. Sanchis-Perucho *et al.* (Sanchis-Perucho et al., 2022a) also confirmed the ineffectiveness of backwashing in reducing filtration resistance, regardless of whether raw or PSE MWW was filtered. The inability of backwashing to control fouling growth could confirm the different hypotheses of different authors regarding the main fouling mechanism during DMF of MWW, *i.e.* pore blocking (by colloidal particles or by gel layer formations) or consolidated cake layer formation that could not be removed by inverting the flux (Jin et al., 2016; Ravazzini, 2008). However, despite these results, backwashing could be useful as complementary cleaning to alleviate possible OM concentration polarization on the membrane surface and the compression of the cake layer (Jin et al., 2015). Indeed, some authors have reported that when using cross-flow operation to control fouling, backwashing is strictly necessary and raising its frequency can help to mitigate fouling (Ravazzini, 2008).

When operating FO membranes, backwashing is usually carried out by osmotic pressure as the driving force. In this case, after some conventional filtration, the membrane permeate side is replaced by tap water or any other cleaning solution, feeding the membrane waste side with the same DS as that used on the permeate side during filtration. Since the driving force changes directions towards the membrane waste side, a permeate flux appears which is used as in other pressure-driven membrane backwashing for detaching any foulant remaining in the membrane surface. This method has been shown to be efficient for mitigating fouling, even when its source is biological (Daly et al., 2021). Unfortunately, as in porous membranes, this strategy alone does not seem to be enough to control fouling in direct MWW filtration. In this respect, Ferrari (2020) reported that after around 23.9 – 39.9 hours of filtration, an OBW was only able to recover 80 % of the original membrane flux, indicating that considerable fouling still remained, despite the cleaning. The membrane permeability recovery was enhanced only when other cleaning methods (continuous or intermittent gas sparging) were used, together with OBW, in this case obtaining nearly full recoveries (95.1 % of the original membrane flux).

3.1.4. Use of granular materials

Placing granular materials in the membrane tanks during filtration has been proposed to provide direct mechanical membrane surface cleaning (Kimura et al., 2017). Sparging this material can detach fouling depositions by scouring the membrane surface having displayed good results in different flat-sheet MBRs (Kurita et al., 2015, 2014). This alternative has captured some interest for application during DMF of MWW, since the required sparging can easily be provided by mixing, such as vibration or paddle stirring, avoiding injecting air into the system and significantly reducing the cleaning energy input (Kimura et al., 2017). Nevertheless, using this strategy with HF membranes seems to be inadequate. For instance, Kimura et al. (2017) studied the efficiency of granular materials in operating an HF MF membrane treating PSE MWW and reported similar results, despite the granular material, due to the capture of the granules by the membrane fibers, which inhibited physical cleaning. To avoid the granules being captured by the membrane, Kimura et al. (2017) combined this strategy with a vibrating mechanism, without a significant improvement in membrane cleaning, even reporting negative effects when the sludge was concentrated to around 8 g L^{-1} of COD. In this case, the increase in medium viscosity could have negatively affected the granule dynamics and reduced filterability. This phenomenon has also been reported by other studies (Kurita et al., 2014), resulting in alterations to the foulants' characteristics because of the granules performance that revealed increments in the irreversible fouling. Since one of the main objectives of DMF is to increase as much as possible the sludge concentration in order to reduce pumping energy and AD volume requirements, this alternative does not appear to be a proper approach for filtering untreated MWW, especially with high density packing membranes.

3.1.5. Membrane movement

Strategies based on membrane movement have also been proposed to control MBR fouling. This method focuses on providing an intensive movement to the membranes to raise their liquid-membrane interface shear and favor membrane fouling detachment (Jaffrin, 2012). This movement is generally produced by vibrating fibers or rotating disks and has been effective in diverse submerged MBRs (Bilad et al., 2012; Jaffrin, 2012; Li et al., 2013). Concerning DMF, Kimura et al. (2017) showed that when operating an HF MF membrane for PSE MWW treatment, membrane permeability could be significantly enhanced by applying intermittent vibration, reporting that filtration was extended by 100 h at low TMPs (under 30 kPa) compared to the 45 kPa in less than 5 h obtained when this mechanism was not used. Moreover, Mezohegyi et al. (2012) reported that vibrating membrane cleaning could control fouling better than aeration, achieving critical fluxes of 10, 15, and 25 LMH with no control strategy, air-assisted membrane scouring, and vibrations, respectively. Due to the significantly lower energy demand of vibrations than aeration, this cleaning strategy is considered as an attractive alternative for DMF fouling control (Mezohegyi et al., 2012). Despite the promising results reported of membrane vibration for fouling control during DMF of MWW, some undesirable effects have also been found when applying this alternative. Although fouling removal may be improved by increasing the vibration intensity and/or frequency, intensive vibrations could negatively affect OM recovery (Mezohegyi et al., 2012). Raising vibration intensity can generate strong perturbations in the medium, enhancing air capture and gas transfer in the liquid phase (Grinis and Kholmer, 2007) and ultimately increasing the development of aerobic bacteria. Intensive vibrations can also perturb the particulate material, inducing partial solubilization of the low-size suspended solids fraction (Mezohegyi et al., 2012), thereby negatively affecting the resource recovery efficiency and permeate quality. A further study and optimization of this alternative is thus required to determine its suitability for DMF.

As alternative to vibrations, other authors have evaluated membrane bundle rotation for fouling control (Ruigómez et al., 2022, 2020, 2019). As the cited works show, this strategy minimized fouling when directly filtering PSE with a UF membrane, allowing long-term filtration of about 54 days when combined with a coagulation/flocculation pre-treatment of the influent. Ferrera et al. (2024) also proved the efficacy of rotating hollow fibers for fouling control, achieving even better fouling mitigations with this alternative than with air sparging. These results were corroborated in both lab and pilot scales and demonstrated the benefits of this kind of alternatives when directly filtering MWW, achieving good fouling mitigations at extremely low energy costs.

3.2. *Chemical methods*

In contrast to physical methods, chemical cleaning strategies mainly focus on mitigating/removing irreversible membrane fouling (Weerasekara et al., 2014). Although this fouling is commonly associated with long-term operations, several authors found short-term physical methods to be ineffective when operating with untreated MWW (see for instance (Nascimento et al., 2017; Sanchis-Perucho et al., 2022a)), indicating a need to implement more effective methods to achieve viable systems. Since these strategies involve considerable reagent dosing (*e.g.* chemical membrane cleaner reagents) or energy-intensive associated processes (*e.g.* ozone generation) the correct selection, use and optimization regarding the treated influent is essential in order to achieve feasible processes.

3.2.1. *Coagulation/flocculation*

Coagulation/flocculation is generally consolidated as an efficient method to reduce membrane fouling. Coagulant addition promotes the aggregation of small particles to produce more voluminous flocs that can prevent the membrane pores being blocked by colloidal materials (Jin et al., 2015). Soluble compounds can also be captured by other mechanisms, such as adsorption or precipitation to reduce these components hindering effects on the membrane surface or pores. Additionally, the voluminous flocs formed during this treatment can quickly cover the membrane surface, protecting the membrane from other fouling promotors (Ravazzini, 2008) and reducing the interaction between the new influent particulate material and the membrane surface (Jin et al., 2016). Increasing the bulk average particle size can have other additional beneficial effects, such as the generation of thinner and more porous cake layers (Ravazzini, 2008) more susceptible to being removed by back-transport mechanisms such as gravitational and/or shear forces (Jin et al., 2016, 2015; Ravazzini, 2008). These easier to remove cake layers would allow operating at higher permeate fluxes (Jin et al., 2015), representing an important benefit for the process viability by reducing the membrane area requirements for MWW filtration. Nonetheless, the cost of coagulation can be a key issue when requiring a continuous dosage while increases sludge productions and may affect its biodegradability due to chemical contamination (Ravazzini, 2008), compromising the main aim of DMF technology. Therefore, this strategy needs to be carefully studied in each case in order to select a suitable reagent and dosing concentration.

Concerning DMF of MWW, numerous studies have reported the beneficial effects of coagulation on wastewater filterability (Hey, 2016; Jin et al., 2016, 2015; Ravazzini, 2008). Jin et al. (2016) reported high filtration improvements when operating a MF membrane for raw MWW treatment, enhancing the membrane permeability after 5 hours of operation from 20 to 300 LMH bar^{-1} when coagulation was used. Similarly, Jin et al. (2015) reported a permeability more than 5 times higher after 1 hour of operation under similar conditions, whilst Ravazzini (2008) showed filtration resistance reductions of up to 35 % when operating a UF membrane for raw MWW treatment. Moreover, average fouling rates of around 5.5 – 6.7 mbar h^{-1} have been found when employing coagulation in DMF (Jin et al., 2016), close to the values achieved in MBR systems (Jin et al., 2016), while thanks to the promoted capture of colloidal and soluble compounds, coagulation seems to significantly reduce irreversible fouling in the middle-term (about 80 – 300 h) (Jin et al., 2016; Ravazzini, 2008). Indeed, thinner and looser cake layers have been obtained when using coagulation (Jin et al., 2015), so that frequent chemical cleaning could be prevented by this method (Jin et al., 2016), resulting in higher membrane lifespans. However, despite all these reported beneficial aspects, the effect of coagulation itself may not be enough for long-term fouling mitigation. In fact, whatever the coagulation dose used, a consistent permeability reduction has been reported during filtration in middle-term operations (between 7 and 70 h), reaching values similar to those obtained when no chemicals were used (Gong et al., 2015; Jin et al., 2016, 2015) due to an inability to remove deposited material on the membrane by chemicals only. This strategy should thus always be implemented together with other assistant methods for effectively controlling membrane fouling in the long-term, such as air sparging, cross-flow, or backwashing.

3.2.1.1. Coagulant reagents. Numerous reagents have been proposed to improve the feasibility of MMW DMF. In fact, different inorganic/metallic coagulants (*e.g.* iron chloride ($FeCl_3$) (Hey, 2016; Ravazzini, 2008; van Nieuwenhuijzen, 2011; Zhao et al., 2020), alumina salts (*e.g.* $Al_2(SO_4)_3$) (Ravazzini, 2008) or PACl (Delgado Diaz et al., 2012; Hey, 2016; Jin et al., 2015; Ravazzini et al., 2008; van Nieuwenhuijzen, 2011; Zhao et al., 2020)) and organic/polymeric coagulants (*e.g.* C-492 (Ravazzini, 2008; Remy et al., 2014), C-581 (Ravazzini, 2008) or C-592 (Ravazzini, 2008)) have been employed for both fouling mitigation and OM recovery. Of the proposed coagulants, PACl is often selected for its ability to form larger aggregates and its higher OM capture than other coagulants, generally achieving better results in jar tests (Gong et al., 2015; Hey et al., 2018; Jin et al., 2016; Te Poele, 2006). Stronger flocs and better performance regarding membrane filtration have also been reported when using PACl (Qin et al., 2004; Ravazzini, 2008; Zhao et al., 2020). However, $FeCl_3$ is the cheapest and most often used chemical reactive in MWW treatment, while other aluminum-based salts also display high coagulation performances at low prices (Rulkens et al., 2005). These alternative reagents can thus also be considered an attractive option for reducing the cost of coagulant-assisted DMF. On the other hand, the benefits of using polymeric coagulants for DMF fouling mitigation are unclear. Rulkens et al. (2005) reported negative effects on the membrane fouling rate when using C-492, and Hey (2016) reported low permeate fluxes from using cationic polymeric coagulants and concluded that filtration performance is negatively affected by these reagents. Nevertheless, in the short-term (around 40 min), Ravazzini (2008) reported reductions in membrane resistance of up to 18, 32 and 61 % when using $FeCl_3$, PACl and C-592, respectively, showing the better performance of organic coagulants when treating MWW. Based on these results, it can be assumed that organic coagulants can be a competitive alternative under certain circumstances, although their suitability should be studied for specific scenarios. Regarding polymeric coagulants, C-592 seems to perform better than C-581 at low doses (Ravazzini, 2008) and, although C-492 shows better particle capture at low dosages, its use is not recommended due to the sticky flocs generated, which can negatively affect filtration (Ravazzini, 2008).

3.2.1.2. Dosing protocol. The coagulant dosing protocol is one of most important considerations in achieving proper DMF performance. Indeed, as several studies shown (Gong et al., 2015; Ravazzini, 2008), coagulant performance is more closely related to dosing methodology, concentration, and medium interaction than with the reagent itself. Since coagulants represent an extra operating cost, this reagent should be limited to small doses (Ravazzini, 2008). However, both underdosing and overdosing coagulants have been shown to have negative effects on DMF (Gong et al., 2015; Ravazzini, 2008). The former is ineffective and even prejudicial to filtration performance since it raises the TSS concentration in the system without forming flocs (Ravazzini, 2008). On the other hand, the latter produces an excess of available cations, positively charging the suspended particles and forming stable suspensions instead of flocculation (Yu et al., 2014). The best dosage, considering the selected coagulant and its interaction with the influent wastewater characteristics, is thus imperative. In the case of PACl, concentrations of about 15 – 30 mg L^{-1} have generally been reported as the optimum dosage when filtering untreated MWW (Hey, 2016; Jin et al., 2016, 2015; Te Poele, 2006; Zhao et al., 2020).

The method of adding the coagulant also plays an important role in fouling control. As Gong et al. (2015) reported, the punctual addition of PACl produced more severe fouling problems than direct filtration itself, while a continuous dosage achieved the opposite results. Rulkens et al. (2005) reported similar results when applying cationic electrolyte (C-492) punctually, noting a negative effect on the membrane fouling rate. This phenomenon could be attributed to the sudden increase of the TSS concentration due to particle flocculation in the single addition, together with greater colloidal deposition on the membrane surface due to the more porous initial cake layer formed (Gong et al., 2015). A single coagulant addition can also promote the formation of thicker and denser cake layers, increasing the filtration resistance (Zhang et al., 2011). To avoid these unfavorable effects, Gong et al. (2015) recommended using continuous low dosing instead of intermittent additions to continuously control the amount of soluble and colloidal particles that reach the membrane surface.

3.2.1.3. Strategies to improve coagulant effectiveness. As Jin et al. (2015) reported, the use of a coagulant by itself may not be a suitable solution to control fouling due to the lack of a direct back-transport force during filtration. This strategy should thus always be implemented together with an additional cleaning strategy, such as permeate and/or air backwashing. The cake layer formed can become thinner by an appropriate back-transport force and make it more likely to be flushed back into the mixed liquor (Jin et al., 2015). Cross-flow operation has also been proposed as an extra cleaning strategy when dosing coagulants. However, temporary effectiveness (reducing membrane resistance for just several hours) have been reported when applying this option (Ravazzini, 2008). This phenomenon could be related to changes in the cake layer composition and structure during filtration (Gong et al., 2015) or to the flocs being broken by shear forces (Ravazzini, 2008). Consequently, Ravazzini (2008) recommended the use of PACl due to their stronger flocs when using cross-flow. Moving to coagulant concentration, Gong et al. (2015) studied the effect of increasing the reagent dose to improve this method fouling control. Unfortunately, even after raising the PACl dosing concentration by a factor of 10 (from 60 to 600 mg L^{-1}), the membrane resistance was only raised slightly (from $4.5 \cdot 10^{13}$ to $4.0 \cdot 10^{13}$ m^{-1}) in 30 h of operating time. Given the few benefits that the ever-increasing chemicals costs provide in improving filtration resistance, this option is not recommended. As an alternative, Hey (2016) studied using both coagulant and flocculants to control membrane fouling. However, as these authors reported, the flocculants only had a favorable effect on resource recovery and did not influence filtration performance in any of the tested coagulant/flocculant concentrations. Finally, Gong et al. (2015) determined that the combination of a continuous PACl dosage with the occasional addition of powdered activated carbon (PAC) could be a suitable alternative to mitigate fouling. Indeed, filtration resistance was dramatically reduced from $5.0 \cdot 10^{13}$ to $1.5 \cdot 10^{13}$ m^{-1} in 120 h of operation by PAC under the conditions described in this study. This improvement was attributed to its adsorbent capacity, reducing the presence of dissolved OM, such as proteinaceous matter and some humic-type substances (Hu et al., 2014), preventing their deposition on the membrane surface. The results reported could imply that membrane fouling was significantly controlled by SMP and EPS concentration and that PAC can be used to effectively control this fouling focus.

3.2.2. Chemical enhanced backwash

Irreversible membrane fouling is commonly removed by periodical chemical cleaning with basic and/or acid solutions for several hours to favor external and internal membrane cleaning. Given the effectiveness of this method in recovering membrane permeability, the same reagents can thus be used to minimize irreversible fouling development during continuous filtration (Huyskens et al., 2008). Chemical enhanced backwashing (CEB) consists of applying low basic and/or acid reagent concentrations during backwashing to make it more effective. In DMF, Lateef et al. (2013) studied the effect of CEB on the fouling control of an MF membrane in treating raw MWW, reporting significant permeability improvement when using low frequency CEBs. A 0.1 % NaOCl CEB for 30 s every 12 h at a flux of 210 LMH allowed continuous membrane operation for more than 200 h at a TMP below 30 kPa in comparison to the 45 kPa reached at 96 h of operation when only pure water was used. Similarly, Kramer et al. (2015) reported permeability recoveries of around 93 % by backwashing every 22 h with 0.1 % NaOCl and 0.1 mol L^{-1} HCl solutions (15 min each) on a ceramic UF membrane. Unfortunately, as Fujioka and Nghiem (2015) and Gong et al. (2015) suggested, the chemical cleaning effects may only be temporary, thus requiring high CEB frequencies to improve long-term filtration. The type of fouling as well as the CEB reagent, concentration, intensity and frequency thus need to be thoroughly studied and optimized in each case. Additionally, due to the structural damage chemicals cause on polymeric membranes (Kimura et al., 2017), more robust membrane materials (*e.g.* ceramic membranes) could be recommended to extend the membrane lifespan (Kramer et al., 2015).

Since OM is identified as the main fouling precursor in the DMF of MWW (Kimura et al., 2017; Kramer et al., 2015; Lateef et al., 2013; Nascimento et al., 2017), oxidizing chemicals can be suggested to efficiently mitigate fouling, NaOCl being the most extended (Fujioka and Nghiem, 2015; Gong et al., 2015; Kramer et al., 2015; Lateef et al., 2013). In fact, compared to NaOH, NaOCl has reported higher reversible and irreversible fouling cleaning efficiencies (Lateef et al., 2013), achieving effective fouling mitigation even at high sludge concentrations (Lateef et al., 2013). However, other specific cleaning reagents (*e.g.* Divos109) can be more effective in DMF systems for MWW treatment (Ravazzini et al., 2005). On the other hand, inorganic matter can also represent an important fouling precursor in DMF (Lateef et al., 2013; Nascimento et al., 2017). As Lateef et al. (2013) reported, although NaOCl had the best cleaning performance, citric acid was more effective than NaOH. Nevertheless, the use of citric acid was shown to be not recommendable for irreversible fouling control by itself (Lateef et al., 2013). Moreover, Nascimento et al. (2017) determined that both citric acid and EDTA can be used for inorganic fouling removal, achieving similar performances. Due to the significant organic and inorganic fouling during DMF of MWW, the alternative use of NaOCl and citric acid CEBs can be proposed as a suitable option for fouling mitigation (Kramer et al., 2015; Lateef et al., 2013).

3.2.3. Ozone

As alternative to chemicals, ozone can also be used to remove OM-based fouling from the membrane surface and/or membrane pores (Hube et al., 2020; Zhu et al., 2011). Fujioka and Nghiem (2015) studied the effect of adding ozone during backwashing to oxidize organic foulants in a ceramic MF membrane treating raw MWW and found that with ozone concentrations of 2 mg O_3 L^{-1} after each filtration cycle (10 min continuous filtration), the membrane recovered its initial permeability with only 2.5 min of backwashing, which was used as an effective short-term fouling control strategy (60 min). They also found that this complete membrane cleaning was possible even when reducing the period of ozonized water backwashing to 1:4 (backwash:filtration cycle ratio), although applying longer backwashing periods (3.5 min). The effectiveness of this method could thus be controlled by adjusting the backwash time/-periodicity according to the degree of fouling. Similarly, Kim et al. (2007) showed that ozone-enhanced backwashing (ozone concentrations between 0.2 – 0.8 mg O_3 L^{-1}) significantly improved fouling removal when treating raw MWW with a metallic MF membrane, achieving permeability recoveries of around 47 % by simple backwashing, compared to 92 % recovery with ozone, with no damage to the metallic membranes. However, despite the potential benefits of this alternative, the in-situ generation of ozone is cost-intensive (Hube et al., 2020) and requires advanced control systems to avoid dangerous issues. Also, robust ceramic or metallic membranes need to be used when employing this method to avoid damaging the membranes during filtration, increasing the capital costs of DMF.

3.3. Combined strategies

As Kimura et al. (2017) suggested, as a single cleaning method may not be enough for long-term DMF operations, a combination of different physicochemical strategies could significantly improve filtration performance. Indeed, when intermittent membrane vibration, mechanical stirring and periodical CEB cleaning strategies were combined during the operation of an HF MF membrane treating PSE MWW, the TMP was perfectly controlled (Kimura et al., 2017). TMPs lower than 10 kPa were achieved for more than 600 h of operations, even when the concentration in the treated mixed liquor reached about 8 g L^{-1} of COD. On the other hand, Jin et al. (2015) used coagulation combined with intermittent air backwashing for raw MWW treatment by MF membranes, reporting increased membrane permeability after 7 h of operations (300 LMH bar^{-1} in comparison with the 20 LMH bar^{-1} observed with coagulant only). Jin et al. (2016) also compared this same strategy with intensive air sparging for fouling control in an MF membrane treating raw MWW, reporting a membrane permeability of about 100 and 45 LMH bar^{-1}, respectively, after 20 h of operation. These authors described this combination as an effective alternative for both membrane fouling mitigation and minimizing the energy demand. Furthermore, fouling growth rate was not only significantly reduced (from 80 kPa h^{-1} with direct filtration to 5 kPa h^{-1} with this strategy), but the maximum fouling increase was reached at high-solids concentrations (from a factor of 5 with direct filtration to a factor of 12 with this strategy) (Jin et al., 2015). This strategy thus proved to be effective in mitigating membrane fouling while allowing to filtrate at high solids concentrations in the membrane module. Finally, in comparison to CEB application, the authors concluded that better filtration performance can be obtained with the alternative, which also improves membrane productivity and reduces OM mineralization.

Fig. 2 summarizes all the strategies addressed in this review together with their main conclusions.

4. DMF resources recovery potential

As DMF technology's main aim is to enhance resource recovery from MWW, the most suitable membrane technology is decided not only by filtration performance but also on the membrane's capacity to capture influent resources.

4.1. Micro- and ultra-filtration membranes

Many MBR studies have reported that high permeate qualities can be achieved by MF or UF membranes. The pores of these membranes can capture all the particulate material and only allow soluble compounds and a fraction of the colloids to pass through. In their application to direct MWW filtration, turbidity can be reduced by around 97 – 99 % (Fujioka and Nghiem, 2015; Sethi and Juby, 2002), achieving NTU values in the permeate below 1.0 (Ahn et al., 2001; Ravazzini et al., 2008; 2005; Sethi and Juby, 2002; van Nieuwenhuijzen et al., 2000). However, due to the significant amount of colloidal material present in untreated MWW, large amounts of total solids can be reached in the permeate, in some cases with poor influent capture efficiencies, ranging between 20 – 28 % and 35 – 41 % of total solids (TS) and volatile solid (VS) concentrations, respectively (Nascimento et al., 2017).

Concerning OM, high fractions of influent COD can be recovered during direct MWW filtration, generally achieving COD recovery efficiencies of up to 70 – 80 % (Ahn and Song, 2000; Hey et al., 2017; Kimura et al., 2017; Kramer et al., 2015; Lateef et al., 2013; Sanchis-Perucho et al., 2023) and BOD recovery efficiencies about 47 – 95 % (Hey et al., 2016; 2017; Juby et al., 2013; Sethi and Juby, 2002). However, changes in the membrane pore size and influent wastewater characteristics can greatly affect both OM recovery efficiency and permeate quality. Indeed, the results of numerous studies are inconsistent, with a wide range of COD recovery values (from 35 % to 80 %), regardless of the membrane pore size (Gong et al., 2017; Hey, 2016; Kimura et al., 2017; Lateef et al., 2013; Nascimento et al., 2017; Ravazzini et al., 2005). In a study of the effect of membrane pore size on OM recovery, Ahn and Song (2000) tested the effect of using different UF membrane pore sizes for filtering MWW, observing limited improvements in COD recovery (around 80, 83 and 84 %) as the pore size was reduced (300,000, 150,000 and 30,000 Da, respectively). Kramer et al. (2015) also found insignificant differences in COD recovery when comparing UF and NF membranes to treat raw MWW, while other authors reported that no significant OM recovery efficiency differences can be expected by filtering raw or PSE MWW (Hey, 2016; Ravazzini et al., 2005), in some cases finding slight reductions in OM recovery when treating PSE (below 5 %), probably related to the prior reduction of particulate material content in the influent (Hey, 2016). In addition, Lateef et al. (2013) and Sanchis-Perucho et al. (2023) concluded that the increase in the membrane operating sludge concentration did not affect the permeate quality. Thus, as Ravazzini et al. (2005) concluded, OM recovery is more closely related to the characteristics of the influent (*e.g.* OM content, particulate fraction, PSD, etc.) than to the membrane or the influent pre-treatment.

On the other hand, different authors have proven the feasibility of OM concentration by DMF, achieving solids concentrations of between 5 – 19 gCOD L^{-1} (Gong et al., 2015; Jin et al., 2015; Lateef et al., 2013; Nascimento et al., 2017; 2021; Sanchis-Perucho et al., 2023) and even higher (up to 54 gCOD L^{-1} (Nascimento and Miranda, 2021)) which could be directly valorized in AD (Nascimento et al., 2017; Nascimento and Miranda, 2021). Additionally, the recovered OM had high biodegradability rates (Jin et al., 2016; Nascimento et al., 2017), similar to those reported from primary and activated sludge digestion (Nascimento et al., 2017), the smaller size particles being the most easily biodegradable (Jin et al., 2016). Unfortunately, operating at high sludge concentrations in DMF can

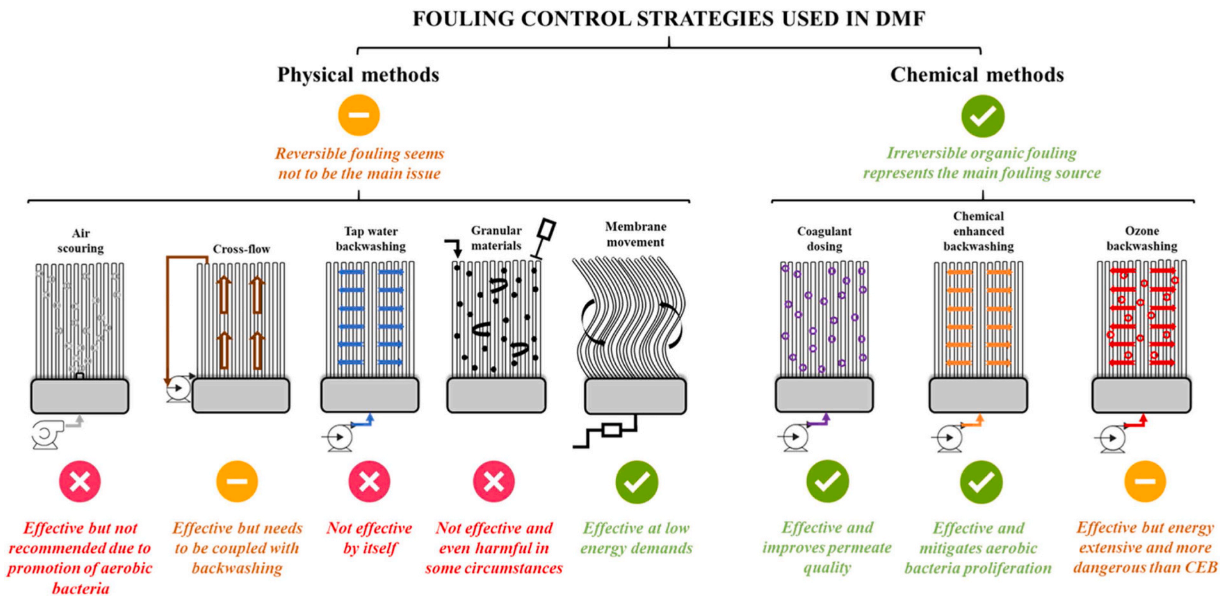

Fig. 2. Fouling control strategies used in DMF and main aspects to consider.

not only harm filtration performance but also reduce the process feasibility by the mineralization of retained OM. In this respect, Rulkens et al. (2005) reported about 50 % OM mineralization when operating in batch mode for 8 days and also detected the partial conversion of soluble COD into colloids, probably due to the generation of bio-flocs, which increased the average particle size. In agreement with this, Faust et al. (2014a), (2014b) reported considerable OM mineralization (up to 32 %) when operating a membrane module at solids retention times (SRT) of 5 days and concluded that this negative effect can be mitigated by operating at short SRTs. Indeed, small COD losses (around 1 – 2 %) were reported when operating at SRTs of 0.125 – 0.250 days (Faust et al., 2014a; Gong et al., 2017), Sanchis-Perucho et al. (2023) determining a maximum SRT of around 3 days to avoid significant OM loses in the membrane tank.

Regarding nutrient capture, total nitrogen (TN) and total phosphorous (TP) influent recoveries of between 9 – 36 % and 15 – 75 %, respectively, have been reported when operating MF/UF membranes (Hey et al., 2016; 2017; Ravazzini et al., 2005; Remy et al., 2014; Sanchis-Perucho et al., 2023; van Nieuwenhuijzen et al., 2000). Unfortunately, since high nutrient concentrations can be found in the MWW soluble fraction, low nutrient recovery rates could be reached in DMF (Lateef et al., 2013; Mezohegyi et al., 2012; Ravazzini, 2008; Sanchis-Perucho et al., 2023), the recovery efficiency being closely related to the nutrient content of the particulate material in the influent wastewater. Indeed, the influence of UF and even NF membranes on untreated MWW ions concentration is extremely limited, reporting recoveries below 10 % for NH_4^+, SO_4^{2-}, Mg^{2+}, and Ca^{2+} (Kramer et al., 2015; Sanchis-Perucho et al., 2023). In the case of PO_4^{3-}, recovery rates of up to 97 % have been reported when using NF membranes (Kramer et al., 2015) but dropping to comparable values to those reached in UF (14 – 9 %) as membrane fouling accumulates (Kramer et al., 2015). These results agree with those reported by Fujioka and Nghiem (2015), who found that small conductivity reductions of around 9 % can be expected during raw MWW filtration by MF membranes.

In addition to the capture of particulate resources, membrane-based systems usually act as a physical barrier to microorganisms, including pathogens and viruses, which can later be eliminated during AD. Membrane effluent can thus be partially disinfected, promoting its reclamation (Bali et al., 2020). For DMF, Sethi and Juby (2002) reported excellent coliform removal efficiencies when operating an MF membrane for PSE treatment. The average total coliforms and fecal coliform bacteria levels were reduced from $9.4 \cdot 10^6$ and $2.7 \cdot 10^6$ MPN to 204 and 25 MPN in 100 mL, respectively. Significant removal of coliphage viruses were also reported, achieving an average influent-to-permeate removal from $2.6 \cdot 10^5$ to 838 PFU in 100 mL. Other important pollutants (*e.g.* heavy metals and micro-plastics) can also be removed by MF and UF membranes (Hey, 2016). DMF technology thus has a high potential to solve some relevant issues in MWW treatment besides providing high quality permeates.

Despite all the above benefits, the high soluble compounds content, especially nutrients, may not allow direct discharge of the permeate produced into natural water bodies (Hey et al., 2017; Lateef et al., 2013; Nascimento et al., 2017). Effluent COD, BOD, TN and TP concentrations ranging between 7 – 550, 1 – 65, 11 – 48, and 6.4 – 0.5 mg L^{-1}, respectively, have been reported in DMF (Ahn et al., 2001; Hey et al., 2017; Jin et al., 2016; Mezohegyi et al., 2012; Nascimento et al., 2017; Sanchis-Perucho et al., 2023; Sethi and Juby, 2002), which may not meet the standards laid down in the EU regulations on discharged water quality (discharge in sensitive environments). Reverse osmosis (Kramer et al., 2015; Remy et al., 2014), packed biological filters (Kimura et al., 2017), electrodialysis (Kim et al., 2007) or anaerobic ammonium oxidation (Anammox) processes (Jin et al., 2016) have been proposed as potential post-treatments to recover and valorize the valuable soluble resources present in DMF permeate. On the one hand, as some authors have suggested (Hey, 2016; Jin et al., 2016), the permeate produced could be used for irrigation when possible to supply crops with viable nutrients.

4.2. Osmosis membranes

Given the nature of osmosis membranes, high resource recovery can be expected when filtering MWW. Complete retention of influent particles can be assumed, while microorganisms (viruses and bacteria) and other contaminants (heavy metals, pharmaceutical products, hormones, microplastics, etc.) can also be removed (Al-Obaidi et al., 2020). Almost complete COD removals (between 95 – 99.8 %) have been reported when using FO membranes for DMF of MWW (Li et al., 2018; Singh et al., 2019; Song et al., 2018). However, as reported in some studies (Wang et al., 2016), around 19 % of the influent COD can be lost due to degradation or attachment to the membrane surface. These COD losses should be properly identified to prevent fouling and enhance the process energy efficiency. Regarding the retentate characteristics, different authors have proven the viability of concentrating the MWW influent material by 5 – 10 fold using FO membranes (Ansari et al., 2016; Wang et al., 2016; Zhang et al., 2014). The FO retentate could thus be directly valorized via AD (Ansari et al., 2016). In addition to enhance energy recovery via AD, the DMF of raw MWW by FO could also benefit other treatments, such as AnMBR. For instance, Vinardell et al. (2021) reported increasing methane yield from 214 to 322 mL CH_4 g^{-1}COD when concentrating the AnMBR influent MWW from 1 to 10 using FO membranes. This phenomenon was mainly attributed to lower methane losses in the permeate when operating at higher organic load rates. In view of these results, any membrane technology designed to develop the DMF could also produce a similar beneficial effect on a subsequent AnMBR system. However, since FO membranes have the highest COD recovery rates, coupling these membranes with AnMBR could be an attractive option for increasing energy recovery and minimize MWW's environmental impact. In this context, over 10-fold MWW concentrations in the FO system are recommended to achieve economically-self-sufficient processes (Vinardell et al., 2021).

Regarding nutrient recovery, high TP captures (about 93 – 99 %) are easily achieved by FO membranes in the DMF of MWW, especially the CTA type (Li et al., 2018; Song et al., 2018; Sun et al., 2016). Unfortunately, relatively low TN recovery rates (up to 50 – 60 %) have generally been reported (Bao et al., 2019; Ferrari, 2020; Wang et al., 2016; Zhang et al., 2014). This is due to the extensive use of TFC membranes, which have a poor capacity to capture cations. While CTA membranes have a negligible charge, the TFC membrane surface has carboxyl groups, which serve as fixed ionic groups (Qiu et al., 2020). These membranes thus confer a cation exchange feature that dramatically reduces their capacity to retain cations (Lu et al., 2014). In this regard, Ferrari (2020) showed how high anion retention (captures of SO_4^{2-} and PO_4^{3-} over 90 %) can be achieved when operating a TFC membrane, while less than 20 % of the cations (NH_4^+, K^+, Mg^{2+} and Ca^{2+}) were retained. On the other hand, Gao *et al.* (Gao et al., 2018) reported the almost complete capture of anions (SO_4^{2-} and PO_4^{3-}) when operating a CTA membrane, while K^+, Mg^{2+}, and Ca^{2+} rejections were about 60, 90 and 80 %, respectively. Therefore, the low TN captures reported when filtering MWW by FO membranes can be solved by using CTA membranes, with which NT retentions of up to 96 % have been reported (Gao et al., 2018; Song et al., 2018), although at the cost of a lower membrane permeability. Another alternative is by modifying/functionalizing TFC membranes to improve the cation capture rate. Several studies have shown that modifying the membrane's active layer with polyethylenimine can improve NH_4^+ recovery by 15 – 25 % (Bao et al., 2019), while aquaporin modules incorporate a selective barrier that provides NH_4^+ recoveries comparable to those of CTA membranes without compromising the high water fluxes typical of TFC membranes (Song et al., 2018).

Concerning endocrine-disrupting chemicals, Gao et al. (2018) reported between 60 – 100 % rejections for Bisphenol A, Estrone, 17β-estradiol and Estriol when filtering MWW with a CTA FO membrane, in agreement with the findings by Cartinella et al. (2006), who obtained similar endocrine-disrupting rejection values (from 77 % to 99.9 %) using a similar FO membrane to treat diverse influents. Nevertheless, despite FO filtration, significant concentrations of these substances can still be found in the effluent, reporting Bisphenol A and 17β-estradiol contents in the permeate of 1388 and 30.3 ng L^{-1}, respectively (Gao et al., 2018). Consequently, as some authors recognize (Gao et al., 2018), FO filtration is not enough to remove all the substances of this type, requiring of tertiary treatments for its complete elimination.

4.3. Dynamic membranes

DM resource recovery potential depends on the characteristics of the cake layer on the supporting material. Since it can be heterogeneous and dynamic, smaller and more variable particulate captures can be expected than those reported in MF or UF membranes (Sanchis-Perucho et al., 2022b, 2022c; Xiong et al., 2019). Xiong et al. (2019) found 20 – 50 NTU effluent turbidity when operating a DM to treat raw MWW, in agreement with other DM studies treating wastewater (Li et al., 2017). Similarly, Sanchis-Perucho et al. (2022b), (2022c) reported effluent turbidity around 90 NTU with TSS concentrations of about 60 mg L^{-1} when filtering PSE, these values increasing to about 160 NTU and 70 mg L^{-1} on treating raw MWW. Regarding OM and nutrients recovery, COD, TN, and TP recoveries of around 20 – 50, 10 – 25, and 15 – 30 %, respectively, were obtained in the above cited studies (Sanchis-Perucho et al., 2022b, 2022c; Xiong et al., 2019), with a limited but significant capture of soluble compounds. SCOD, SN, and SP captures of 18.7, 8.7, and 5.8 %, respectively, were reported by Xiong et al. (2019), which were related to the mineralization or adsorption of compounds by the cake layer. Other authors have reported higher recoveries when dosing coagulants during raw and PSE MWW treatment, *e.g.* COD, TN, and TP captures around 65 – 80, 25 – 30, and 60 – 80 %, respectively (Gong et al., 2014; Ma et al., 2013; Sanchis-Perucho et al., 2022b). Like MF and UF membranes, DM permeate quality is unable to meet the standards of EU regulation on discharged water quality, reaching COD, SCOD, TN, NH_3-N, TP, and PO_4^{3-}-P concentrations in the permeate of about 75 – 180, 74, 35 – 50, 20 – 35, 2 – 5, and 5 mg L^{-1}, respectively (Gong et al., 2014; Ma et al., 2013; Sanchis-Perucho et al., 2022b). Therefore, similar alternatives than those cited for MF and UF membranes effluent treatment can also be proposed in this case to valorize the generated permeate.

For OM capture, Xiong et al. (2019) and Sanchis-Perucho et al. (2022b), (2022c) reported low COD and TSS concentrations in the membrane tank (up to 1.2 gCOD L^{-1} and 2.4 g L^{-1}, respectively) when directly self-forming a DM by raw or PSE MWW due to the significant loss of suspended material in the permeate. This represents a limitation to valorize the recovered OM via AD since the low concentrations reached by DM. However, these authors noted a significant trend towards higher particle sizes in the concentrated sludge, due not only to the capture of larger particles, but also to the aggregate formation during OM concentration (Sanchis-Perucho et al., 2022b; Xiong et al., 2019). Sludge sedimentation could thus be improved to allow AD treatment after conventional sludge thickening. Alternatively, high COD retentate concentrations (from 4.5 up to 27 g L^{-1}) can be reached by coagulant dosing (Gong et al., 2014; Ma et al., 2013), allowing its direct treatment by AD. Additionally, the reported bio-methane potential of the concentrate is similar or even higher than that reported for activated sludge (Ma et al., 2013; Xiong et al., 2019), which was related to the higher C:N ratio during DM filtration, which improves organic fermentation (Feng et al., 2009).

4.4. Side-effects of cleaning method on resource recovery and permeate quality

The cleaning strategies used during filtration have been shown to strongly influence the process, with either favorable or unfavorable side-effects on resource recovery and permeate quality. Wastewater characteristics should therefore be considered when selecting a fouling control strategy to not only achieve energy-efficient filtration processes but also to maximize resource recovery and achieve suitable permeate qualities.

4.4.1. Air-assisted membrane cleaning

Air injection during DMF is considered as unsuitable as it promotes the development of heterotrophic microorganisms. Lateef et al. (2013) observed losses in OM of about 10 – 20 % of the total influent COD due to aerobic biodegradation when air sparging was used for membrane scouring and/or tank mixing, even when operating at short SRTs of around 9 – 75 h, while Jin et al. (2016) reported about 19 % OM aerobic mineralization even when intermittent aeration was applied. Alternative cleaning strategies such as membrane vibration could also be unsuitable as they promote the re-aeration of the mixed liquor, reporting unintentional biodegradations of up to 50 % of the concentrated COD in only 22.5 h of operation (Mezohegyi et al., 2012), compared with the 8 days required in batch mode in a not-aerated system (Remy et al., 2014). Due to the unfavorable results, air sparging is normally avoided in DMF (Jin et al., 2016;

Table 2
Energy feasibility of DMF for MWW.

Membrane Type	Treated influent	Operation	Energy input (kWh per m^3 of treated water)	Energy output (kWh per m^3 of treated water)	Total energy demand (kWh per m^3 of treated water)	Ref.
Microfiltration	Raw MWW	Coagulant dosing combined with micro-sieving and membrane air scouring.	0.40			(Hey et al., 2017)
		Coagulant dosing and intermittent air-backwashing.	0.08	0.11	-0.03	(Jin et al., 2017)
		Air mixing combined with CEBs.	< 0.40[a]	0.14 – 0.50		(Lateef et al., 2013)
		Coagulant and powdered activated carbon dosing.		0.19		(Gong et al., 2015)
		Coagulant dosing and intermittent aeration.	0.09	0.10	-0.01	(Jin et al., 2016)
		Coagulant dosing combined with micro-sieving and membrane air scouring.	0.41	1.3		(Hey et al., 2018)
		Coagulant and powdered activated carbon dosing.		0.26 – 0.39		(Gong et al., 2017)
	Primary settler effluent	Vibrations combined with mechanical mixing and periodical CEBs.	< 0.40[a]			(Kimura et al., 2017)
Ultrafiltration	Raw MWW	Air sparging and permeate backwashing.	0.10	0.53	-0.43	(Sanchis-Perucho et al., 2023)
		Intermittent air scouring and permeate backwashing	0.15	0.19		(Nascimento and Miranda, 2021)
		Coagulant and powdered activated carbon dosing.	0.03 – 0.05	0.08	-[0.03 – 0.05]	(Gong et al., 2019)
	Primary settler effluent	Air sparging and permeate backwashing.	0.34	0.50		(Nascimento et al., 2017)
		Air sparging and permeate backwashing.	0.10	0.24	-0.14	(Sanchis-Perucho et al., 2023)
Forward osmosis	Raw MWW	Coagulant dosing combined with micro-sieving. Seawater as draw solution.	0.41	1.5		(Hey et al., 2018)
		Synthetic MWW. Water recovery in the FO dispositive from 50 % to 90 %.		1.0 – 6.9		(Vinardell et al., 2021)
Membrane Type	**Treated influent**	**Operation**	**Energy input (kWh per m^3 of treated water)**	**Energy output (kWh per m^3 of treated water)**	**Total energy demand (kWh per m^3 of treated water)**	**Ref.**
Dynamic membrane	Raw MWW	Three-layer stainless steel mesh of 25 µm as supportive material. External physical cleaning with air backwashing and surface brushing each 48 h.	0.01	0.10	-0.09	(Xiong et al., 2019)
		One/two layers of a flat open monofilament woven polyamide meshes with 1–5 µm of average pore size as supportive material. Operating flux of 15 LMH.	0.03	0.37	-0.33	(Sanchis-Perucho et al., 2022c)
		Dacron mesh of 61 µm as supportive material. Continuous coagulant dosing to perform the DM auto-formation and control fouling development. External physical cleaning with tap water after reach a TMP of 35 kPa.	0.24 – 4.57[b]	0.13 – 4.33[b]	0.11 – 0.24[b]	(Ma et al., 2013)
	Primary settler effluent	Two layers of a flat open monofilament woven polyamide meshes with 1 µm of average pore size as supportive material. Operating flux of 15–45 LMH. Coagulant dosed during the last period of filtration.		0.03 – 0.12		(Sanchis-Perucho et al., 2022b)

MWW: Municipal wastewater.
[a] Estimated from MBRs.
[b] Including AD and nutrients recovery equipment energy impacts.

Kimura et al., 2017; Lateef et al., 2013; Xiong et al., 2019).

4.4.2. Coagulant dosing

Besides improving filtration performance, coagulant dosing has also benefits for resource recovery and permeate quality in DMF. COD recovery improvements of up to 89 – 98 % in MF/UF membranes have been reported when coagulant chemicals were used (Delgado Diaz et al., 2012; Gong et al., 2017; Hey, 2016; Jin et al., 2016). This improved OM recovery has been attributed to the capture of small particulate materials in the formed flocs (Jin et al., 2015), including colloids and supra-dissolved solids (Ravazzini, 2008), while a portion of the soluble OM fraction can also be adsorbed onto the cake layer by adding chemicals (Jin et al., 2016). Coagulants have also shown favorable operating effects during OM retention, achieving higher OM concentrations in a shorter time (Gong et al., 2017, 2015; Jin et al., 2016). They use would therefore allow the membrane tanks to be operated at lower SRTs, preventing OM mineralization. Furthermore, Hafuka et al. (2020) studied the potential energy recovery of the captured OM when dosing PACl during DMF and reported that this coagulant does not seem to have unfavorable effects on sludge biodegradability by AD whenever the Al^{3+} concentration in the sludge is kept below 4.3 mg L^{-1}. Considerable phosphorous recovery can also be achieved by using chemicals. Indeed, coagulant dosing is conventionally used in WWTP for phosphate precipitation (Hey, 2016). Since membrane filtration presents higher particulate material retention efficiency than sedimentation, high phosphorous captures can be expected by combining coagulation and DMF, with TP removals of up to 95 – 99 % when using membranes or micro-sieving (Hey, 2016; Te Poele, 2006; Väänänen et al., 2016). Unfortunately, TN capture is insensitive to coagulants, reaching similar recovery efficiencies regardless using these chemicals (Hey, 2016; Ravazzini, 2008). Organic coagulants have also shown high OM recoveries with TP capture efficiency of around 75 % (Hey, 2016), without affecting other wastewater chemical features (*e.g.* pH or conductivity) (Ravazzini, 2008). However, although modest pH reductions and conductivity increases have been reported from inorganic coagulants (Ravazzini, 2008), they obtain similar or slightly higher OM recoveries (Hey, 2016; Ravazzini, 2008), greatly improving TP capture efficiency until achieving almost complete recovery (Hey, 2016). The difference in phosphorous capture is due to the lack of metallic ions in organic coagulants (Ravazzini, 2008; Väänänen et al., 2016). The application of inorganic coagulants can thus be recommended from the nutrient removal perspective. Of the inorganic coagulants, $FeCl_3$ and $AlCl_3$ salts seem to have similar performance (Abdessemed and Nezzal, 2002; Delgado Diaz et al., 2012; Hey, 2016), while slightly higher OM recovery has been reported when using PACl (Delgado Diaz et al., 2012; Hey, 2016). Hey (2016) proposed the use of flocculants together with coagulant dosing to further improve resource recovery. In this study, anionic and cationic flocculants showed similar behavior on OM capture, but improved TP capture by up to 99 %. Since PAC1 plus anionic flocculants provide better filtration performance, this combination was selected as the most suitable for DMF. Thanks to coagulant or coagulant + flocculent dosing, COD, BOD, TN, and TP concentrations of 25 – 73, 11 – 17, 16 – 42, and 0.02 – 2.5 mg L^{-1}, respectively, can be achieved in the permeate (Hey, 2016; Jin et al., 2016; Zhao et al., 2019). The addition of these chemicals can thus be considered an alternative to additional permeate post-treatment due to their improved permeate quality. However, traces of metals or cations can be found in the permeate according to the dose used, which needs to be taken into account according to the water's expected fate (Ravazzini, 2008).

4.4.3. Membrane cleaning chemicals

Besides adding coagulants, other chemicals can significantly affect sludge and permeate quality such as cleaning reagents for membranes. Using CEBs during DMF can greatly restrict biological activity during filtration, avoiding OM losses and preventing biological fouling problems (Kimura et al., 2017; Lateef et al., 2013), though, not all cleaning chemicals have the same effect. Lateef et al. (2013) showed that NaOH and citric acid slightly influence heterotroph activity, while NaOCl almost completely inhibits it, thus the latter being a most suitable reagent. However, NaOCl in DMF can favor solubilization of part of the particulate/colloidal material in MWW, resulting in significant losses of soluble COD with the permeate (Lateef et al., 2013). Since this issue is not observed when using NaOH and citric acid, reporting similar permeate COD concentrations after using these reagents to those obtained with pure water for backwashing (Lateef et al., 2013), they use can be preferable depending of the situation. In addition to these results, the effect of CEBs on biological permeate quality (*i.e.* pathogenic bacteria and virus removal) should also be considered, since chemicals can enhance the system's disinfecting capacity and improve the potential reuse of the water recovered. The suitable selection of chemicals and the optimum CEBs concentration and frequency is thus imperative to enhance both OM recovery and permeate quality (Lateef et al., 2013).

5. Feasibility of DMF technology for MWW treatment

The feasibility of an emerging technology can be determined by its energy balance, investment/implementation and operating costs, the area required for its application and its environmental impact, among others. Table 2 summarizes the results obtained by different authors when studying the energy balance of DMF of MWW by different membrane technologies.

The DMF energy demands of MF and UF membranes are usually calculated by considering other similar MBRs, finding energy inputs between 0.30 – 0.60 kWh per m^3 of treated MWW (Kimura et al., 2017; Lateef et al., 2013). However, aeration represents 60 – 70 % of the total energy demand in MBR operation (Hey et al., 2017; Kimura et al., 2017; Mezohegyi et al., 2012). Since air-assisted membrane scouring can be replaced by less energy-intensive cleaning strategies in DMF, different authors suggest that its energy cost can be significantly reduced (Jin et al., 2016; Kimura et al., 2017; Lateef et al., 2013). In this regard, some authors have also estimated the DMF energy input in lab/pilot-scale systems, with values around 0.08 – 0.09 kWh per m^3 of treated MWW when using coagulant and intermittent aeration (Jin et al., 2017, 2016), values of about 0.10 – 0.34 kWh per m^3 of treated MWW with continuous/intermittent air sparging and permeate backwashing (Nascimento et al., 2017; Nascimento and Miranda, 2021;

Sanchis-Perucho et al., 2023), and values around 0.40 – 0.41 kWh per m^3 of treated MWW using coagulant, micro-sieving and air sparging (Hey et al., 2018, 2017). On the other hand, thanks to OM capture, energy recoveries between 0.19 – 1.40 kWh per m^3 of treated MWW have been estimated in different studies (Gong et al., 2017, 2015; Hey et al., 2018, 2017; Lateef et al., 2013; Nascimento et al., 2017; Sanchis-Perucho et al., 2023). Besides, this energy output can be significantly improved by considering nutrient recovery and the equivalent cost of inorganic fertilizer production (Gong et al., 2017). AD heat requirements can also be completely covered according to some studies and in some cases even achieving a heat energy surplus (Hey et al., 2018, 2017). These promising results make numerous authors agree that neutral energy treatment or even net energy production can be achieved by the DMF of MWW when considering the overall process energy balance (Gong et al., 2017; Hey et al., 2018, 2017; Jin et al., 2017, 2016; Sanchis-Perucho et al., 2023). These results clearly exceed the energy input of current aerobic treatments, which can be from around 0.3–1.9 kWh per m^3 of treated MWW (Gong et al., 2015; Hey et al., 2017), so that from an energy point of view, DMF can be considered an attractive alternative for MWW treatment.

Other important aspects that also needs to be considered to assess DMF feasibility is the land area requirements and economic cost. Hey et al. (2016); (2018) estimated that the implementation of an MF system for MWW treatment would reach about 0.0075 m^2 per population equivalent (PE), rising to 0.046 m^2 per PE when considering all additional equipment (*i.e.* screening, sand-trap, micro--sieving, coagulant and equalization tanks, AD, etc.). Since smaller area requirements than those reported by conventional WWTP facilities (about 0.107 and 0.228 m^2 per PE) have been reported (Hey, 2016), up-grading existing WWTPs without increasing land demands could be achieved. On the other hand, according to the calculations performed by Mezohegyi et al. (2012), economic improvement can be expected from a realistic semi-continuous OM concentration by membrane filtration. As regards capital expenses, Ravazzini (2008) estimated a DMF investment cost of €0.05 – €0.06 per m^3 of influent water, which agrees with the range reported by Baker (2004), considering different UF membrane plants. Sanchis-Perucho et al. (2023) also estimated the DMF capital requirements applied to an MWWTP (treatment flow rate 30,000 m^3 d^{-1}) at different transmembrane fluxes when filtering both raw and PSE MWW by a UF membrane, reporting initial investments for membrane acquisition of M€3.1, M€3.8 and M€4.7 for operating permeate fluxes of 10, 12.5 and 15 LMH, respectively. Regarding operational expenses, Rulkens et al. (2005) and Ravazzini (2008) estimated an MWW treatment cost of €0.20 – €0.40 per m^3 of treated water, while Sanchis-Perucho et al. (2023) reported values between €0.11 and -€0.04 per m^3 of treated water, all the studies with UF membranes. Since the current overall MWW treatment cost can be estimated as between €0.57 – €1.02 per m^3 of treated water (Ravazzini, 2008), the DMF concept can be considered an attractive alternative that would reduce MWW treatment costs. It is also important to highlight that all the equipment required for DMF is commercially available (Hey et al., 2018). Furthermore, thanks to the improved energy recovery during MWW treatment, significant carbon footprint reductions can also be achieved by MWWTP operations. In this regard, Sanchis-Perucho et al. (2023) estimated that the carbon footprint of MWW treatment can be reduced by about 0.06 – 0.26 $kgCO_2$-eq per m^3 of treated water, when DMF energy demands are directly compared to conventional aerobic treatment, achieving even higher values when also considering the extra energy output obtained by the DMF of MWW due to the improved OM recovery. Nonetheless, further studies on the environmental and economic impact of the equipment and materials used (*i.e.* LCA and LCC analysis) are required to realistically assess DMF feasibility from the environmental and economic perspectives.

As far as the authors know, few studies have been carried out on the energy and economic viability of FO and DMs for MWW treatment. FO has low energy demands since the DS is the main driving force (Ansari et al., 2016; Cath et al., 2006). However, regenerating this DS can be an energy-intensive process generally conducted by RO or membrane distillation (Ansari et al., 2016; Shaffer et al., 2015). In this case, an energy requirement and environmental impact similar (or even higher) than those reported for water desalination have been estimated for MWW concentration by FO-RO membranes (Ansari et al., 2016), although recognizing the limitations of this comparison. Regarding FO-RO treatment costs, Vinardell et al. (2020) reported values around €0.81 per m^3 of treated water when water recovery was restricted to 50 %, rising to €1.01 and €1.27 per m^3 of treated water when increasing water recovery to 80 and 90 %, respectively. Using seawater or concentrated fertilizers as DS has been considered a potential alternative to avoid regeneration (Ansari et al., 2016). For instance, Hey et al. (2018) reported energy demands around 0.41 kWh per m^3 of treated MWW when combining FO filtration with a micro-sieving pre-treatment, obtaining competitive results regarding MF/UF. Moreover, slightly higher energy recoveries were reported when compared with MF, thanks to a higher COD capture rate (Hey et al., 2018), achieving an energy output potential of 1.5 kWh per m^3 of treated MWW. The area requirements for implementing DMF with FO membranes are similar to those reported for MF or UF, since similar membrane areas are required, estimating FO-DMF land demands of between 0.039 – 0.050 m^2 per PE (Hey et al., 2018).

Finally, promising results have been reported for DMs in direct MWW filtration. Concerning the energy balance, total energy demands of about 0.24 kWh per m^3 of treated MWW have been reported for raw MWW filtration (Ma et al., 2013), achieving neutral energy processes thanks to OM recovery (Sanchis-Perucho et al., 2022b, 2022c) or even energy surpluses around 0.09 kWh m^{-3} when aeration is avoided and chemicals dosed (Xiong et al., 2019). A thorough economic analysis has not yet been performed, although promising results can be expected due to DM's lower costs and maintenance requirements than conventional membrane systems (Hu et al., 2017a, 2017b; Xiong et al., 2019). Indeed, Sanchis-Perucho et al. (2022c) estimated an operating cost of less than €0.002 per m^3 of treated MWW, which entailed a net profit of around €0.065 per m^3 of treated MWW from the OM recovered by the system. These results mean that implementing DM involves an extremely low payback period, this study estimating it to be around 0.08 years (Sanchis-Perucho et al., 2022c). However, the authors state that DMs do not offer significant improvements over conventional systems (*i.e.* primary settling) due to their reduced capacity to capture small particles from the influent MWW. These systems are thus considered potential competitors to classic membrane systems only when loaded MWWs are treated (Sanchis-Perucho et al., 2022c)

6. Guidelines and future perspectives

As many studies have shown in the last decades, the DMF of MWW seems to be an attractive alternative to boost resource recovery in MWWT facilities. However, some issues need to be properly addressed before considering this alternative as a viable option for full-scale applications. In particular, the severe fouling reported in numerous works is one of the major problems to solve. Considering all the results obtained so far, UF and DO membranes seem to be the best options to carry out the DMF, with MF membranes generally displaying worse fouling issues. On the other hand, the membrane material or treatment influent does not seem to meaningfully contribute to the fouling evolution. Concerning operating conditions, low/moderate fluxes seem to be recommended to reduce fouling promotion, while operating at relatively high TSS concentration in the membrane tank (about 6 – 10 g L^{-1}) seems to contribute to reducing fouling appearance. Examples of successful UF long-term filtrations, all of them at pilot–scale, can be found in Ferrera et al. (2024), who filtered for about 17 days with a rotating hollow fiber membrane (0.9 m^2 of membrane area); Nascimento et al. (2021), who operated for about 32 days with an air scouring assisted hollow fiber membrane (0.93 m^2 of membrane area); Gong et al. (2019), who filtered for more than 66 days with a submerged hollow fiber membrane (24 m^2 of membrane area) assisted with coagulant and activated carbon dosing; and Sanchis-Perucho et al. (2023), who operated for more than 90 days with an air scouring assisted hollow fiber membrane (43.5 m^2 of membrane area). For successful FO long-term filtrations, there can be consulted Sun et al. (2018), (2016), who operated for 30 – 70 days a CTA membrane at cross-flow (20 cm^2 of membrane area) with synthetic sea water as DS.

The most suitable fouling control strategy to use is also a difficult matter to consider. Overall, physical methods do not seem to be the best option since reversible fouling is not the main issue in these systems. Particularly, air-related methods (air scouring specifically), although effective, are not recommended due to their significant energy demands and the potential hindering effects of promoting aerobic bacteria in the membrane tank. Instead, membrane vibrations or rotations could be considered as a more suitable alternative in this aspect, showing good results with low energy requirements. On the other hand, irreversible fouling seems to be the major issue in the DMF. To mitigate it, CEBs with basic solutions (particularly with NaOCl) seem to be the most recommended option. The dosage of coagulants is also a good alternative in this field, achieving great results in mitigating fouling and furthermore enhancing the quality of the generated permeate. In this last case, the use of PACl seems the most recommended option.

Concerning the quality of the produced permeate, numerous studies have proven than the UF regenerated water meets European standards for not sensible environments, thus being an excellent option for farmlands fertigation. However, when fertigation is not possible, other nutrients recovery focused post-treatments can be applied aiming to recover the soluble nutrients (nitrogen and phosphorus particularly) of MWW. Membrane contactors, ion exchangers, microalgae culture and harvesting or reverse osmosis filtration are some of the alternatives that could be used as post-treatment for this permeate, improving its quality while valorizing its nutrients content. The direct filtration with DO membranes or the dosing of coagulants during filtration could also be contemplated to avoid these kind of post treatments, since the generated permeate in this case would already meet the quality standards for sensible environments. On the other hand, the concentrated sludge would be generally employed to improve the energy savings of MWW by generating methane via AD. The mineralized sludge could be used for agriculture purposes while the nutrients concentrated

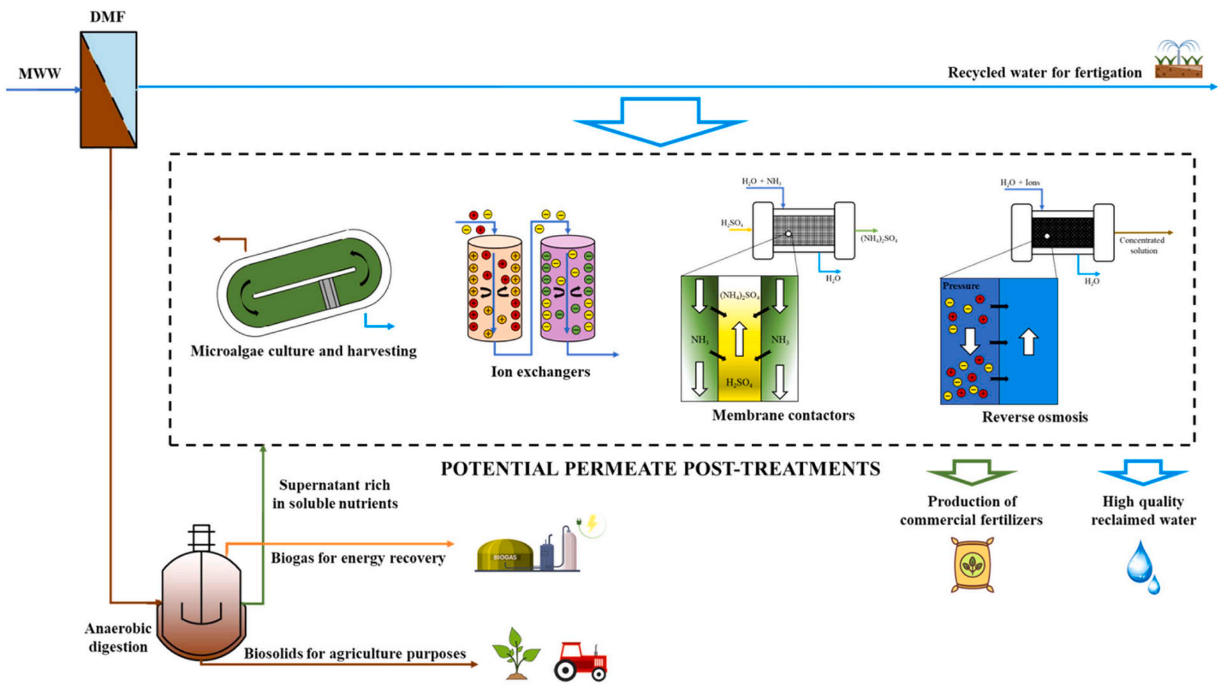

Fig. 3. DMF effluents potential uses and post-treatments.

supernatant could be valorized as a fertilizers source via any before mentioned alternative. A schematic view of the DMF effluents potential uses and post-treatments can be seen in Fig. 3.

Finally, although DM shows great potential to reduce fouling and capital expenses in DMF, the permeate produced does not have high enough quality to implement some of the previously introduced alternatives, requiring an associated AS or similar process. This considerably reduces its utility since the energy savings are not as significant as those obtained with other membranes. However, when treating influents with a high load of solids and COD, DMs could be considered an attractive alternative to reduce filtration operating and capital expenses, thus displaying an operative niche in this field. In the case of applying DMs, supporting materials with pore sizes of about 1 – 5 μm have proven feasible to self-form a DM when filtering raw MWW, whilst filtering PSE is not recommended due to the lack of particulate material. However, pore sizes can be increased to 25 – 60 μm when dosing coagulants, which can also be used to improve the quality of the generated permeate. For fouling control, occasional superficial brushing with tap water has proven to be sufficient to operate these membranes. Examples of successful DM long-term filtrations can be found in Ma et al. (2013), who achieved a 300 days continuous filtration with a 0.081 m^2 dracon mesh (61 μm of pore size), and Sanchis-Perucho et al. (2022c), who achieved a 100 days continuous filtration with a 2 m^2 polyamide mesh (5 μm of pore size).

Overall, as this review highlights, the DMF shows a great potential for improving current MWW treatments. However, besides needing to improve fouling mitigation, further research is still required to demonstrate the suitability of this alternative for full-scale applications. Fouling mechanisms need to be further investigated in order to propose the most effective fouling control strategies at the lowest possible operating costs. Also, transmembrane fluxes need to the increased aiming to reduce the important capital expenses for membranes acquisition, which is one of most important limitation in DMF. Finally, more studies focused on determining the cost and environmental benefits of this alternative (i.e. Life Cycle Costing (LCC) and Life Cycle Assessment (LCA)) are required to properly determine the best scenarios where this alternative can be proposed.

7. Conclusions

This review highlights the promising results of DMF technology for upgrading MWW treatment. Thanks to including a membrane filtration process prior to conventional aerobic treatment, high quality reclaimed water can be produced, while concentrated sludge can be used to produce energy. Essential nutrients can also be recovered from both the streams generated, transforming MWW into a source of multiple resources. Unfortunately, membrane fouling is still the most important issue to be addressed to improve DMF feasibility. UF and DO membranes seem to be the most promising types to use in this strategy, while DMs occupy an interesting operational niche when treating loaded MWWs. Of the fouling control strategies applied, physical methods do not seem to involve significant enhancements in filtration, suggesting the minimum use thanks to the small need to reduce the negligible amount of reversible fouling in these systems. Instead, chemical methods, such as coagulant dosing or CEBs, are a more suitable solution to minimize fouling during direct MWW filtration, achieving to date good results that justify the implementation of this treatment scheme. In this regard, promising results have been reported for the process's energy balance, its carbon footprint and preliminary economic balance, with great improvements over conventional treatments. However, future in-depth research on the overall system energy balance and economic and life cycle is required to determine the potential advantages of applying DMF compared to other alternatives.

CRediT authorship contribution statement

José Ferrer: Supervision, Project administration, Funding acquisition. **Aurora Seco:** Supervision, Project administration, Funding acquisition. **Daniel Aguado:** Writing – review & editing, Supervision, Funding acquisition. **Pau Sanchis-Perucho:** Writing – original draft. **Angel Robles:** Writing – review & editing, Supervision, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

No data was used for the research described in the article.

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