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Environmental and economic assessment of urban wastewater reclamation from ultrafiltration membrane-based tertiary treatment: Effect of seasonal dynamic

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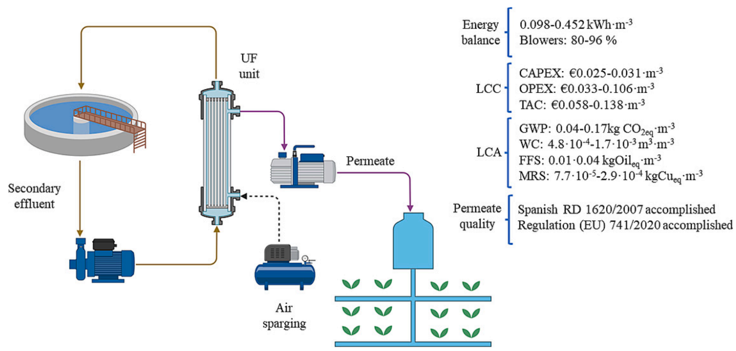
HIGHLIGHTS

- Full-scale UF plant shows energy consumption in the 0.098–0.452 kWh·m^{-3} range.
- CAPEX was estimated at €0.025–0.031·m^{-3} with membranes as major contributor (66–69 %).
- OPEX account for €0.033–0.106·m^{-3} being energy cost the main expenditure (66–79 %).
- LCA showed energy balance as most influential factor on the environmental impact.
- UF process allowed accomplishing regulation statements for agriculture reuse.

GRAPHICAL ABSTRACT



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ABSTRACT

This study aimed to assess the environmental and economic performance of an ultrafiltration (UF) tertiary treatment of effluent from an urban wastewater treatment facility. Data from a UF demonstration plant composed of commercially available equipment, including industrial hollow-fiber membranes was used to project a full-scale facility. The results from the demonstration plant recommended different ranges of trans-membrane fluxes and sparging air demands under summer and winter conditions to prevent excessive fouling. The energy balance of the full-scale facility would be 0.308 ± 0.112 kWh·m^{-3} in summer and 0.140 ± 0.040 kWh·m^{-3} in winter, with blowers' being the main consumers (86–93 %). CAPEX accounted for €0.030 ± 0.002·m^{-3} in summer and €0.027 ± 0.002·m^{-3} in winter and membrane acquisition represented 66–69 % of the investment cost. Energy expenditure was the major OPEX cost (66–79 %), with a total operating cost of €0.077 ± 0.023·m^{-3} and €0.042 ± 0.008·m^{-3} in summer and winter, respectively. The final average value obtained for the TAC was €0.107 m^{-3} in summer and €0.068 m^{-3} in winter. The environmental assessment confirmed optimizing energy consumption and membrane requirements as the main factors influencing environmental sustainability. Specifically, summer and winter emissions of 0.079–0.175 and 0.043–0.079 kgCO_{2eq}·m^{-3} (Global warming potential); 8.1 · 10^{-4}–1.7 · 10^{-3} and 4.8 · 10^{-3}–8.1 · 10^{-3} m^3·m^{-3} (water consumption); 0.019–0.041 and 0.010–0.019 kg oil$_{eq}$·m^{-3} (fossil fuel scarcity); and 1.4 · 10^{-4}–2.9 · 10^{-4} and 7.7 · 10^{-4}–1.4 · 10^{-4} kg Cu$_{eq}$·m^{-3}

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(mineral resource scarcity) were calculated, respectively. The obtained permeate quality complied with the most stringent Spanish and EU regulations.

1. Introduction

The fundamental objective of providing access to an adequate supply of clean water involves serious challenges. Besides the growing pressure on the conventional water resources (de Châtel, 2018), driven by an ever-increasing world population (Mekonnen and Hoekstra, 2016), the effects of global warming must be added, which will affect precipitation patterns, cause more droughts and modify the reserves of this basic resource (Samantaray et al., 2022). The consequences of these effects are already being seen worldwide (Rodell et al., 2018), e.g. according to the Water Exploitation Index (calculated as total water abstraction per year compared to long-term renewable resources) published by the European Environment Agency (EEA.). Belgium, Bulgaria, Cyprus, Germany, Italy, Malta, and Spain already exceed this indicator by 20 %, meaning that they are in danger of facing water shortages (European Environment Agency, 2023).

In addition to saving water and promoting its efficient use, it is imperative to explore alternative water sources to guarantee access to water in the context of economic, environmental and social sustainability (Aldaya et al., 2019). The increasing use of reclaimed water is particularly important, especially in arid and semi-arid regions and this water source has become a vector of change and innovation in the water sector. Wastewater treatment plants (WWTP) are also being converted into water resource recovery facilities (WRRF) and can be considered as a valuable reserve of resources of energy, nutrients and reclaimed water (Qadir et al., 2020).

Despite the advantages of using wastewater's embedded resources, there are also legal, economic, environmental and social conditions that hinder the spread of this practice. For instance, there is generally no economic incentive for farmers to change their irrigation source due to inadequate financial mechanisms and the low cost of conventional water sources, except for scarcity or legal obligations (Foglia et al., 2023). In the legal field, the EU Directive 91/271/EEC on the treatment of urban wastewater recommends water reuse when appropriate. However, its definition of "discharge" and the permitted maximum nutrient concentrations in environmentally sensitive areas prohibited direct reuse for fertigation without removing the nutrients. The different regulations, recommendations and guides for the reuse of water in agriculture in the EU countries were also a legal barrier. This situation has been clearly improved by Regulation (EU) 741/2020, which lays down minimum requirements for water reuse. This regulation better harmonizes EU water reuse practices and is expected to promote widespread reuse (Radini et al., 2023). In addition, the EU agreement on the proposal for a directive on urban wastewater treatment (Cartuyvels, 2024) could also contribute to ensure the legal security of water reuse.

The challenges in the environmental field are primarily associated with the proper control of effluent quality, including emerging contaminants such as microplastics, antibiotic-resistant bacteria (ARB) and genes (ARG), and the adequate use of the nutrients in the wastewater that prevent adding excessive fertilizers to the soil and soil salinization (Rahman et al., 2022). The extra cost and technical complexity of the additional treatments required to meet the current standards and the foreseeable limits on emerging contaminants in the updated wastewater treatment directive could be a further hindrance, together with the preparation, implementation and compliance with mandatory risk management plans for any reuse project that requires administrative authorization, as stated in Regulation (EU) 741/2020 (European Commission, 2020). It is therefore necessary to implement robust, simple and affordable systems for the efficient recovery of wastewater.

Membrane-based technologies can be considered an effective system for reclaimed water production due to its robustness and efficiency in terms of removing microbiological elements and solids (Yang et al., 2021). Moreover, these systems have the potential for wastewater nutrient recovery as can potentially represent a preliminary step for the implementation of fertigation (Robles et al., 2020). For instance, direct membrane filtration (DMF) is a technology that maximizes resource recovery from wastewater by installing a membrane filtration system before the secondary treatment stage (Lateef et al., 2013). Indeed, secondary treatment could be even avoided when treating low-strength wastewater. High quality reclaimed water can be produced while recovering the particulate matter from wastewater for later valorization in the form of energy and biosolids via anaerobic digestion (Sanchis-Perucho et al., 2024). Additionally, DMF minimizes spatial requirements for wastewater treatment and recovery, as only requires the installation of compact membrane modules. MBR combines the biological treatment designed to remove organic matter, nitrogen and phosphorus with filtration to ensure the production of a stable and excellent effluent quality (Krzeminski et al., 2017b). For this reason, this system is well established as secondary treatment technology, favoring water recovery for different purposes (e.g., irrigation, aquifer recharge, etc.) (Krzeminski et al., 2017a). On the other hand, AnMBR is a well-established technology that reduces sludge production, eliminates the need for aeration and produces methane. This technology produces effluent of suitable quality to be used for fertigation depending on the nutrient discharge limits. The solid-free effluent could also be sent to other nutrient recovery units (Robles et al., 2021).

Hence, high-quality reclaimed water can be produced from different membrane-based technologies for specific applications by selecting an appropriate membrane pore size, thereby enhancing the versatility of the technology. This is aligned with the fit-for-purpose approach that has been established in Europe with the new EU Regulation 2020/741, which proposes to apply reclaimed waters with different levels of quality according to their final use (Radini et al., 2023).

Several studies have proven that ultrafiltration (UF) effluents meet European standards for not sensible environments (Sanchis-Perucho et al., 2024). Other membrane-based technologies such as nanofiltration and reverse osmosis are able to remove contaminants of emerging concern, but their operating costs are significantly higher than those of UF, being this not feasible to treat wastewater (Foglia et al., 2023). Moreover, the performance of UF technology does not depend on the water quality, does not require the addition of chemicals or other substances or generates by-products, such as UV, ozonation or chlorination (Bray et al., 2021). However, membrane fouling is still a challenge that reduces filtration productivity and therefore the environmental and economic performance of UF as well (Gong et al., 2019; Lei et al., 2021; Li et al., 2023; Qasim et al., 2018). Nonetheless, in some UF applications, cake layer and gel layer development could contribute to improve selective retention of some pollutants. In this sense, previous studies showed that the application of UF to direct membrane filtration led to a reduction in fouling thanks to the protection of the membrane pores by forming a fouling layer that partially prevented pore blocking from colloids and soluble substances such as SMP, which could enter these membrane channels and reduce their permeability (Sanchis-Perucho et al., 2024).

Also, energy demand associated to membrane operation represents a challenge for its wider adoption as water reclamation technology. Pérez et al. (2022) evaluated the performance of a protype equipped with UF and reverse osmosis modules for water reclamation for industrial reuse and estimated a pumping energy consumption of 0.44 $kWh \cdot m^{-3}$ for the UF step. Grzegorzek et al. (2023) reported an energy range of 0.1–0.2 $kWh \cdot m^{-3}$ for UF treatment. Compared to these values, conventional treatments usually show lower energy requirements, e.g., Grzegorzek et al. (2023) reported 0.015–0.064 kWh·m^{-3} for a combination of filtration+UV and 0.03–0.15 kWh·m^{-3} for ozonization, while Longo et al. (2016) reported 0.048–0.117 kWh·m^{-3} for filtration + UV. However, UF has the advantage of achieving higher effluent quality in terms of particulate contaminants than conventional treatments, which improves the overall benefit of this technology. In addition, the increase in the contribution of renewable energy to the energy mix expected in the coming years would result in a reduction in both the energy costs and carbon footprint of the UF.

Optimizing permeate productivity and reducing fouling are therefore imperative to enhance the UF process competitiveness. It should be noted that the studies cited above evaluated tertiary treatments under specific conditions, regardless of possible seasonal WWTP variations. To the best of our knowledge, no study has yet been published on a seasonal analysis of not only technical issues but also of the wastewater characteristics relevant to long-term membrane operations.

In this context, the present study assessed the technical, economic and environmental feasibility of UF membranes as tertiary treatment for urban wastewater recovery, considering the seasonal dynamics of water quality. For this, a UF system that treated effluent from a WWTP in Valencia, Spain (treatment flow rate of 14,000 m^3·d^{-1}), was assessed considering the energy footprint, life cycle costing (LCC) and life cycle assessment (LCA). The data used was obtained from a continuous one-year run of a UF lab-scale prototype. Effluent quality was also characterized to confirm whether or not it met the standards of Regulation (EU) 2020/741 and the Spanish Royal Decree N° 1620/2007.

This work's main contributions include the operation of a UF demonstration plant for tertiary treatment under different seasonal conditions, whose results provided recommendations to reduce fouling. The study also included an upscaling projection to an industrial UF facility, with estimates of energetic, economic, and environmental performance, while UF technology was validated as suitable for producing high-quality effluent in compliance with the strictest agricultural water reuse regulations.

2. Material and methods

2.1. UF prototype

A prototype UF plant was designed and installed for post-treating effluent from a medium-size (50800-p.e.; 14,000 m^3·d^{-1}) urban WWTP in Valencia (Spain), mainly based on primary sedimentation and an activated sludge system for nutrient removal composed of biological reactors and secondary clarifiers.

The UF consisted of two 15-L membrane tanks (MT1 and MT2), each containing a 2.5-m high hollow-fiber bundle (KMS Puron® Koch Membrane Systems) with a membrane area of 3.44 m^2 each and average pore size of 0.03 μm. One bundle was used for continuous operation and the other was kept in reserve for use during membrane cleaning or system maintenance. The plant also had a 30-L clean-in-place tank (CIP) to store permeate for membrane backwashing as required. A flow diagram of the prototype UF plant is shown in Fig. 1.

A feed pump supplied the system with a flow rate of between 110 and 210 L·h^{-1} and the permeate pump filtered at a flow rate in the range of 95–135 L·h^{-1}, corresponding to a transmembrane flux (J) of 27.6–39.2 LMH. Any tank overflow was recycled to a secondary clarifier in the WWTP. To reduce cake layer formation, air was sparged by a blower with a flow rate range of 0.75–4.50 Nm^3·h^{-1}, corresponding to a sparging air demand per m^2 of membrane (SAD$_m$) of 0.2–1.3 Nm^3·m^{-2}·h^{-1}.

Filtration was controlled by an instrumentation, control and automation (ICA) system based on sensor-transmitters and actuators connected to a programmable logic controller (PLC), an automation system and a Supervisory Control and Data Acquisition (SCADA) under the control philosophy proposed by Robles et al. (2012a,b).

The operation of the prototype for one year provided data on performance, energy consumption and effluent quality, membrane behavior, fouling propensity and the chemical reagents required for cleaning. Since the data were obtained during the operation of industrial

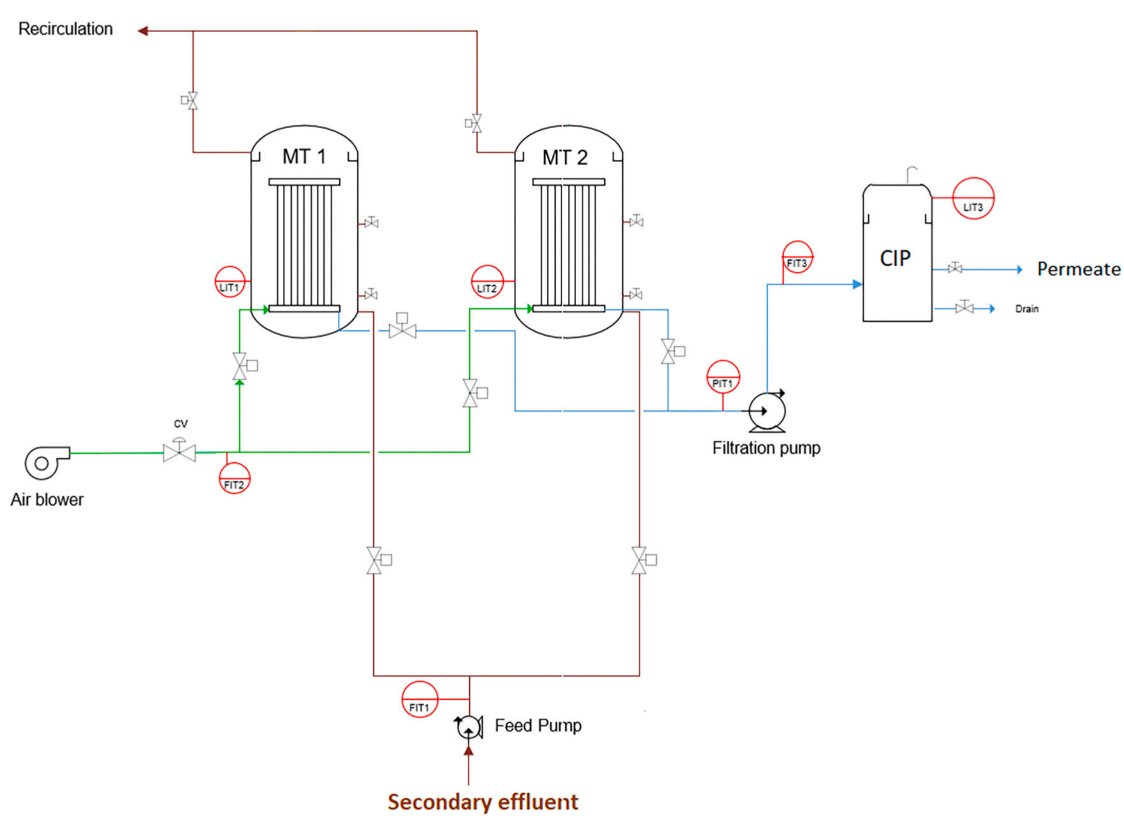

Fig. 1. Flow diagram of the UF prototype for tertiary treatment.

equipment (pumps, blowers, membrane) in ambient conditions with real secondary effluent, the information was considered be robust and can be used as the basis to design an industrial-scale plant to treat the 14,000 $m^3 \cdot d^{-1}$ of the WWTP containing the UF prototype.

2.2. Operational periods and conditions

The goal of the UF prototype was to achieve maximum pollutant retention and produce a high-quality effluent suitable for agricultural reuse at a reasonable energy and financial cost. Different operating conditions were evaluated (see Section 3) such as varying J and SAD_m according to the following aims:

- To achieve cost efficiency, the membrane filtration system was operated at high permeate fluxes (in the range 28–39 LMH) to reduce membrane requirements and lower the associated acquisition costs.
- To obtain optimum membrane sparging, the system operated at the SAD_m with the minimum fouling propensity and energy consumption. A wide SAD_m range (0.2–1.3 $Nm^3 \cdot m^{-2} \cdot h^{-1}$) was tested to determine the influence on UF performance with secondary effluent, as used in a previous study (González et al., 2023).
- To control membrane fouling and avoid over-consumption of chemicals for membrane cleaning.

3 summer and 3 winter periods were selected for further evaluation (Table 1). Different operating ranges were selected according to the season evaluated: summer (April–October, water temperature of 22–30 °C), winter (November– March, water temperature of 16–19 °C). Filtration was operated in cycles with alternate filtration, relaxation, and backwashing stages, with the filtration operating mode set as follows: a combination of filtration–relaxation (F–R) cycles of 180 s (150 s filtration followed by 30 s relaxation) and 60 s of backwashing every 5 F–R cycles.

When the TMP reached the maximum-security value of 0.6 bar (based on the supplier's information), the operation was stopped and the experiment was concluded. The experiment durations provided valuable information for estimating both membrane chemical cleaning frequency and membrane lifespan.

To evaluate filtration performance, the gross 20 °C-standardized transmembrane flux ($J_{20,g}$; $L \cdot h^{-1} \cdot m^{-2}$; LMH;), the fouling rate (FR; $mbar \cdot min^{-1}$), and the SAD_m ($Nm^3_{air} \cdot h^{-1} \cdot m^2$) were calculated by means on Eqs. (1), (2), and (3), respectively, as proposed by Robles et al. (2012a,b), based on the on-line monitored data:

$$J_{20} = J \cdot e^{-0.0239 \cdot (T-20)} \tag{1}$$

$$FR = \frac{n \cdot \sum_{1}^{n} (TMP_i \cdot t_i) + \sum_{1}^{n} TMP_i \cdot \sum_{1}^{n} t_i}{n \cdot \sum_{1}^{n} TMP_i^2 - \left(\sum_{i}^{n} TMP_i \right)^2} \tag{2}$$

$$SAD_m = \frac{Q_{air}}{A_m} \tag{3}$$

where T is the water temperature (°C), n is the total number of data monitored, TMP_i is the transmembrane pressures (mbar) at time t_i (min); Q_{air} represents the air flow for membrane scouring ($m^3 \cdot h^{-1}$) and A_m is the membrane surface area (m^2).

Table 1
Selected operating conditions of the UF prototype for tertiary treatment.

		$J_{20\ gross}$	SAD_m	T	Duration	FR
		LMH	$Nm^3 \cdot m^{-2} \cdot h^{-1}$	°C	d	$mbar \cdot d^{-1}$
Summer	*S1*	27.6	0.73	29.6	20.3	22.5 ± 5.6
	S2	27.6	0.44	28.0	16.0	
	S5	33.4	0.28	24.7	32.8	
Winter	*W1*	39.2	0.73	21.9	55.3	9.7 ± 1.7
	W2	33.4	0.44	18.6	52.6	
	W6	39.2	0.44	18.3	42.9	

2.3. Analytical method

2.3.1. Water quality analysis

Total and soluble chemical oxygen demand (tCOD and sCOD) were determined according to Standard Methods (American Public Health Association, American Water Works Association, 2012) 5220-COD-C and 522-COD-D, respectively. Ammonium (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N) and phosphate (PO_4-P) were analyzed according to Standard Methods (APHA, 2012): 4500-NH3-G, 4500-NO2-B, 4500-NO3-H and 4500-P-F, respectively, using an automatic analyzer (Smartchem 200, Westco Scientific Instruments, Westco). Volatile suspended solids (VSS) concentration was also measured in duplicate, according to method 2540 E of the Standard Methods (American Public Health Association, American Water Works Association, 2012).

2.3.2. Microbiology

The presence of *E. coli* was quantitatively determined by the standard method for enumeration of *E. coli* β-glucuronidase positive, following the UNE-EN ISO 9308-1:2014: 100 mL of the sample were filtered through a 0.45 μm nitrocellulose membrane (47 mm diameter Microfil©). The membrane was then placed on a Chromocult® coliform agar plate and incubated at 36 ± 2 °C for 24 h, after which the number of positive (blue-green) colonies on the plate was counted. The result was the number of *E. coli* in the samples capable of form a colony during the incubation period, expressed in colony forming units per 100 mL (UFC/100 mL).

2.3.3. Heavy metals

The Spanish regulation on agricultural reuse of water is included in Royal Decree 1620/2007, which, among other parameters, limits the concentration values of boron (B), arsenic (As), beryllium (Be), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), manganese (Mn), molybdenum (Mo), nickel (Ni), selenium (Se) and vanadium (V). These analyses were carried out on an Agilent 7900 inductively coupled plasma mass spectrometer (ICP-MS).

2.4. Energy assessment

The instrumentation used in the UF prototype acquired, among others, the following data: membrane tank influent flow, temperature, TMP, permeate flow, turbidity and air flows for membrane scouring supplied by the blower. As the over-dimensioned equipment in the prototype plant prevented directly measuring the energy requirements of the UF tertiary treatment, the theoretical energy consumption of pumps and blowers was calculated from the equations proposed in Judd and Judd (2006) and the energy model described by Pretel et al. (2016). Adiabatic indexes and blower and pump efficiencies were estimated from the standard values for these components. Further details can be found in the Supplementary Material.

2.5. Life cycle costing

The LCC method was applied to assess the economic viability of the projected UF full-scale tertiary treatment, based on the performance data obtained from the UF prototype. The calculated total annual cost (TAC) included capital and operating expenses (CAPEX and OPEX, respectively), according to the required inputs (building work, energy and chemical consumption, equipment acquisition and replacement, etc.) with 1 m^3 of treated water as the functional unit. Since only internal costs were considered, the applied LCC can be defined as conventional.

The TAC ($€ \cdot m^{-3}$) of the whole process was calculated by means of

CAPEX and OPEX, as proposed by Ferrer et al. (2015) (Eq. (4)):

$$TAC = \frac{r\,(1+r)^t}{(1+r)^t - 1}\cdot CAPEX + OPEX \tag{4}$$

where r is the annual discount rate and t is the depreciation period in years.

2.6. Life cycle assessment

Environmental performance was assessed by LCA according to ISO 14040:2006. The life cycle inventory (LCI) of individual materials and processes was based on Ecoinvent 3.4 and Industry data 2.0 databases accessed via SimaPro v9.1.1. ReCiPe 2016 and a hierarchical approach was selected as the life cycle impact assessment (LCIA) method (also accessed via SimaPro v9.1.1). The ReCiPe2016 method was selected for its global scope when calculating characterization factors and its wide use in LCA studies, while the hierarchical approach was chosen for its scientific consensus on the time horizon of impacts (100 years). The SimaPro software is also among the most used and referenced in scientific literature.

3. Results and discussion

3.1. UF prototype performance

The results obtained from operating the UF pilot plant (Section 2.2) were used to project a full-scale UF tertiary treatment for 14,000 $m^3 \cdot d^{-1}$. WWTP secondary effluent showed non-significant differences in COD concentrations in summer (77 ± 25 $mg \cdot L^{-1}$) and winter (70 ± 44 $mg \cdot L^{-1}$). There were no significant differences in terms of TSS (12 ± 5 $mg \cdot L^{-1}$ in summer and 28 ± 21 $mg \cdot L^{-1}$ in winter) and turbidity (5 ± 2 NTU in summer and 18 ± 17 NTU in winter) (see Table 2) as a result of some bulking and foaming episodes detected in the WWTP in winter, with higher TSS flocs in winter.

The average filtration fouling rates were higher in summer (22.5 ± 5.6 $mbar \cdot d^{-1}$) than in winter (9.7 ± 1.7 $mbar \cdot d^{-1}$), despite the higher winter transmembrane fluxes (J of 33.4–39.2 LMH) than in summer (J of 27.6–33.9 LMH). As influent TSS was lower in summer than in winter, the fouling was not attributed to a cake layer formation but to pore blocking, as found in similar UF systems treating non-biological liquors (Sanchis-Perucho et al., 2023a,b). These results suggest that the UF system should be operated under different strategies to cope with diverse types of fouling. To avoid excessive fouling, J in the range of 28–33 LMH and SAD_m ranges of 0.4–0.6 $Nm^3 \cdot m^{-2} \cdot h^{-1}$ can be recommended in summer, whereas a value of 33–39 LMH and SAD_m of 0.3–0.4 $Nm^3 \cdot m^{-2} \cdot h^{-1}$ would be preferable in winter. Further details on WWTP effluent quality and filtration performance can be found in the Supplementary Material.

3.2. Full-scale UF treatment

The ambient and operating conditions considered for the projection of the yearly operation of the full-scale UF tertiary plant are summarized in Table 3.

Table 3
Selected operating conditions of the projected full-scale UF unit.

		J	SAD_m	T
		LMH	$Nm^3 \cdot m^{-2} \cdot h^{-1}$	°C
Summer	*Sa*	28	0.40	26.0
	Sb	28	0.60	26.0
	Sc	33	0.40	26.0
	Sd	33	0.60	26.0
Winter	*Wa*	33	0.30	18.0
	Wb	33	0.40	18.0
	Wc	39	0.30	18.0
	Wd	39	0.40	18.0

3.2.1. Economic and environmental assessments

The following subsections describe the implementation, development and results of the financial and environmental assessment based in accordance with ISO 14040:2006.

3.2.1.1. Goal and scope. The objective of LCC and LCA was to study the economic and environmental performance of the projected full-scale UF tertiary treatment plant in summer and winter to identify the operating variables that could help reduce total costs and its impact on the carbon footprint and resource depletion. The construction, operation and dismantling phases were considered with a facility lifespan of 20 years.

The LCC consisted of calculating the TAC based on CAPEX and OPEX and including building work, energy and chemical consumption, equipment acquisition and replacement, etc. The main considerations were as follows:

CAPEX

- Depreciation: A depreciation period of 20 years for buildings and 10 years for equipment was used to calculate the TAC at an annual discount rate of 3 % for the former and 5 % for the latter.

- Membrane tank: The membrane tank volume was estimated according to the dimensions of a commercial membrane unit (KMS Puron® Koch Membrane Systems; 0.03-μm pore size, filtration area of 3.44 m^2).

- Pipelines: The maximum flow rate of liquid in the pipes was set to 1.5 $m \cdot s^{-1}$ to calculate the pipe diameter, thus preventing excessive losses of mechanical energy.

- Equipment: included acquisition of blowers, pumps and UF membranes. The cost of equipment was estimated from the capacity based on flow rates and the unit costs provided by the manufacturers.

OPEX

- Power requirements: both the energy and power tariffs were included in the appraisals, as proposed in Somenergia.coop (2022).

Table 2
Characteristics of the influent to UF prototype for tertiary treatment.

	TSS $mg \cdot L^{-1}$				tCOD $mg \cdot L^{-1}$				tCOD:TSS mg tCOD·mg^{-1} TSS				Turbidity NTU				pH			
	Mean	±	SD	n	Mean	±	SD	n	Mean	±	SD	n	Mean	±	SD	n	Mean	±	SD	n
Summer	12[a]	±	5	15	77	±	25	22	5.6[a]	±	1.7	18	5[a]	±	2	30	7.7	±	0.2	30
Winter	28[a]	±	21	30	70	±	44	52	2.6[a]	±	1.2	48	18[a]	±	17	60	7.8	±	0.3	60

n: number of samples.
[a] Showed statistically significant differences.

- Chemical reagents for membrane cleaning: According to experimental data, membrane chemical cleaning was considered every 1.5 months; sodium hypochlorite and citric acid were considered for chemical cleaning at a dose of 2000 ppm of both sodium hypochlorite and citric acid, while the contact time was 6 h.

- Membrane replacement: the lifespan of the UF membrane was calculated from the maximum total contact with chlorine permissible before replacement: 500,000 ppm-hours according to the supplier or a maximum lifespan of 10 years.

- Equipment replacement and maintenance: Blower and pump life was based on the total working hours recommended by the manufacturers, with a maximum of 10 years.

- Building work and maintenance: maintenance was a percentage of the associated CAPEX, according to the technical specifications for general WWTP structures (Inyges Consultores S.L., 2015)

- Labor costs: labor costs were not included, since no extra staff were required to operate the full-scale UF. The UF membrane process is highly automated and maintenance is low, with a reduced frequency of chemical membrane cleaning.

Details of unit costs and the correlation functions used to assess the CAPEX and OPEX can be found in Table S3 (Supplementary Material).

With regard to the LCA, the environmental indicators used were energy, chemical reagents, construction materials and effluent quality. In the construction and dismantling phases, the environmental loads were calculated based on the materials used: i.e. steel, iron, concrete, polyester and polypropylene. In the operating phase, the following aspects were included in the LCA: i) filtration performance, ii) energy consumption, iii) chemicals consumption, iv) maintenance and replacement of equipment; and iv) effluent quality. The ReCiPe2016 method with a hierarchical approach was applied as LCIA and global warming potential, water consumption, fossil resource scarcity and mineral resource scarcity were selected for the evaluation of the carbon footprint and potential resource depletion associated with UF.

The common functional unit of 1 m^3 of treated water was selected for both LCC and LCA to adequately represent the process of reclaiming water for agriculture.

3.2.1.2. Inventory analysis. The LCI of individual materials and processes is shown in Table 4:

Table 4
Life cycle inventory.

Construction materials, kg·m^{-3}	Chromium steel *(1 kg Steel, chromium steel 18/8 {GLO}| market for | APOS, S (Ecoinvent 3))*
	Cast iron (*1 kg Cast iron {GLO}| market for | APOS, S (Ecoinvent 3)*)
	Polyester resin (*1 kg Polyester resin, unsaturated {GLO}| market for | APOS, S (Ecoinvent 3)*)
	Polypropylene (*1 kg Polypropylene, granulate {GLO}| market for | APOS, S (Ecoinvent 3)*)
	PVC *(1 kg PVC pipe E (Industry data 2.0))*
	Transport *(1 tkm Transport, freight, lorry 16–32 metric ton, EURO6 {GLO}| market for | APOS, S (Ecoinvent 3))*
Energy consumption, kWh·m^{-3}, *(1 kWh Electricity, medium voltage {ES}| market for | APOS, S (Ecoinvent 3))*	
Reagent consumption, kg·m^{-3}	NaOCl *(1 kg Sodium hypochlorite, without water, in 15 % solution state {GLO}| market for | APOS, S (Ecoinvent 3))*
	Citric acid *(1 kg Citric acid {GLO}| market for | APOS, S (Ecoinvent 3))*
	Transport *(1 tkm Transport, freight, lorry 3.5–7.5 metric ton, EURO6 {GLO}| market for | APOS, S (Ecoinvent 3))*
UV tertiary treatment, unit	*1 p Ultraviolet lamp {GLO}| market for | APOS, S (Ecoinvent 3)*

The LCI was compiled from the Ecoinvent 3.0 and Industry Data 2.0 databases. Ecoinvent is updated regularly and contains approximately 18,000 processes under different themes like transport, energy, material production, agriculture, etc. All processes are available as unit- and system-processes and all processes are documented in detail. Industry Data includes over 300 datasets for a wide range of industries collected by their own associations, which guarantees accurate and reliable data. The main materials and energy flows are shown in Table 5.

The total energy consumption (TEC) of the projected full-scale UF unit was calculated considering: i) blower consumption for membrane scouring; ii) feeding pump; iii) permeate pump; and iv) backwash for membrane cleaning for summer (S) and winter (W) ambient conditions. The different J and SAD_m applied led to four sub-scenarios for each season, i.e., Sa to Sd and Wa to Wb.

The results are shown in Fig. 2 and Table S4 in the Supplementary Material. Summer TEC was between 0.193 (Sc) and 0.452 kWh·m^{-3} (Sb) (0.308 ± 0.112 kWh·m^{-3} on average); while in winter maximum TEC was for Wb (0.191 kWh·m^{-3}) and the lowest TEC was for Wc (0.098 kWh·m^{-3}). Regarding the energy contributions to TEC, the membrane blowers, which accounted for between 80 % (Wc) and 96 % (Sb), were the biggest energy expenditure. Feeding (2–10 %) and permeate (2–9 %) consumed much less, while backwashing was negligible (<1 %). Any efforts to reduce the energy expenditure should thus focus on membrane scouring. A more appropriate filtration strategy should be adopted to guarantee significant transmembrane fluxes, which would eventually reduce the membrane surface requirements, lead to smaller SAD_m and reduced energy consumption (see Fig. 2). Further research is needed for an in-depth study of other possible strategies to reduce fouling, such as modifying the filtration-relaxation cycles, increasing solids concentration in the membrane tanks, etc. (Sanchis-Perucho et al., 2023b).

By way of comparison, Melgarejo et al. (2016) evaluated the energy costs of water reclamation in the Rincón de León WWTP (Treatment flow: 52,000 m^3·d^{-1}; Valencia, Spain), which combined different tertiary treatment processes: coagulation + flocculation + filtration (CFF), UF, ultraviolet disinfection (UV) and reverse osmosis (RO). The results on energy consumptions were 0.047 kWh·m^{-3} for CFF; 0.236 kWh·m^{-3} for UF;0.056 kWh·m^{-3} for UV and 0.869 kWh·m^{-3} for RO. The energy consumption was therefore lower than the summer scenarios (0.308 ± 0.112 kWh·m^{-3}) but higher than winter scenarios (0.140 ± 0.040 kWh·m^{-3}). Akhoundi and Nazif (2018) applied a multi-criteria analysis to assess the sustainability of wastewater reuse alternatives in a WWTP in the south of Tehran (Iran) which treated 450,000 m^{-3}·d^{-1}. This study reported energy expenditures of 0.219 kWh·m^{-3}, 0.2 kWh·m^{-3}, 0.066 kWh·m^{-3} and 1.5 kWh·m^{-3} for coagulation + flocculation, UF, UV and RO, respectively. In this case the UF results were 58 % lower than the average energy consumption obtained in summer and 42 % higher than in winter. It should be noted that both examples used an upstream treatment process before UF. A similar approach was tested in a previous study using the UF system applied in the present study but preceded by sand filtration. The results were not considered satisfactory as membrane performance was not significantly improved but sand filtration increased the total energy consumption (González et al., 2023).

3.2.1.3. Life cycle costing. A financial evaluation was conducted based on the results of the long-term UF prototype according to the LCC method proposed in Section 2.5, including the four sub-scenarios considered (Sa to Sd and Wa to Wd, respectively), representing a cost range for a full-scale UF unit operated for a year.

3.2.1.3.1. Capital expenditure. Fig. 3a and Table S5 in the Supplementary Material show the CAPEX results obtained for the projected full-scale UF unit. Membrane acquisition appeared as the main contributor to CAPEX, accounting for 67–71 % in summer and 64–68 % in winter. Sub-scenarios with low J led to higher CAPEX, since the membrane requirements depend on filtration productivity. For example, CAPEX in Sa (J of 28 LMH) was €0.031·m^{-3} while Sc (J of 33 LMH)

Table 5
Energy and material.

Phase	Flow	Unit	Scenarios							
			Summer				Winter			
			Sa	Sb	Sc	Sd	Wa	Wb	Wc	Wd
Construction	Polypropylene	kg·m^{-3}	4.63E-05	4.63E-05	4.44E-05	4.44E-05	4.44E-05	4.44E-05	3.23E-05	3.23E-05
	Chromium steel	kg·m^{-3}	2.38E-05	4.00E-05	1.90E-05	2.62E-05	8.97E-06	1.47E-05	6.90E-06	1.10E-05
	Cast iron	kg·m^{-3}	2.65E-06	4.45E-06	2.11E-06	3.39E-06	1.43E-06	2.07E-06	1.24E-06	1.69E-06
	Polyester	kg·m^{-3}	3.56E-04	3.56E-04	3.02E-04	3.02E-04	3.02E-04	3.02E-04	2.56E-04	2.56E-04
	PVC	kg·m^{-3}	4.63E-05	4.63E-05	4.44E-05	4.44E-05	4.44E-05	4.44E-05	3.23E-05	3.23E-05
	Transport	tkm·m^{-3}	9.51E-06	9.87E-06	8.24E-06	8.50E-06	8.11E-06	8.23E-06	6.66E-06	6.75E-06
Operation	Energy	kWh·m^{-3}	2.55E-01	4.52E-01	1.93E-01	3.34E-01	1.22E-01	1.91E-01	9.83E-02	1.48E-01
	Sodium hypochlorite	kg·m^{-3}	1.51E-03	1.51E-03	1.28E-03	1.28E-03	1.28E-03	1.28E-03	1.09E-03	1.09E-03
	Citric Acid	kg·m^{-3}	1.51E-04	1.51E-04	1.28E-04	1.28E-04	1.28E-04	1.28E-04	1.09E-04	1.09E-04
	Chromium steel	kg·m^{-3}	5.50E-05	9.57E-05	9.57E-05	4.25E-05	7.14E-05	2.77E-05	4.20E-05	2.30E-05
	Cast iron	kg·m^{-3}	6.11E-06	1.06E-05	4.72E-06	7.94E-06	3.08E-06	4.67E-06	2.55E-06	3.68E-06
	Polyester	kg·m^{-3}	3.56E-04	3.56E-04	3.02E-04	3.02E-04	3.02E-04	3.02E-04	2.56E-04	2.56E-04
	Transport	tkm·m^{-3}	4.16E-05	4.25E-05	3.51E-05	3.57E-05	3.48E-05	3.51E-05	2.96E-05	2.98E-05
Demolition	Steel and Iron	kg·m^{-3}	8.76E-05	1.51E-04	6.83E-05	1.13E-04	4.51E-05	6.74E-05	3.79E-05	5.38E-05
	Polyester	kg·m^{-3}	7.12E-04	7.12E-04	6.04E-04	6.04E-04	6.04E-04	6.04E-04	5.11E-04	5.11E-04
	PVC	kg·m^{-3}	4.63E-05	4.63E-05	4.44E-05	4.44E-05	4.44E-05	4.44E-05	3.23E-05	3.23E-05
	Polypropylene	kg·m^{-3}	4.63E-05	4.63E-05	4.44E-05	4.44E-05	4.44E-05	4.44E-05	3.23E-05	3.23E-05
	Transport	tkm·m^{-3}	2.68E-05	2.87E-05	2.28E-05	2.42E-05	2.21E-05	2.28E-05	1.84E-05	1.89E-05

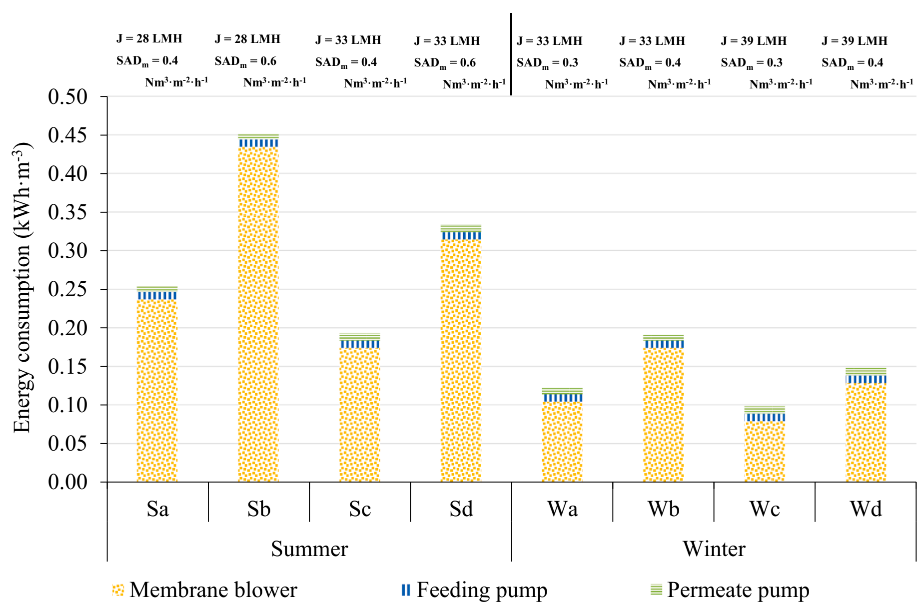

Fig. 2. Total energy consumption of the projected full-scale UF unit.

obtained €0.028·m^{-3}. On average, the summer scenarios produced a CAPEX of €0.030 ± 0.002·m^{-3} and winter scenarios €0.027 ± 0.002·m^{-3}. The largest investment cost was for membranes, €0.021 ± 0.002·m^{-3} and €0.018 ± 0.001·m^{-3} in summer and winter, respectively. Permeate pumps and feeding pumps accounted for €0.003 ± 0.000·m^{-3} both in summer and winter scenarios (11–12 %), while the blowers' contribution was calculated as €0.002 ± 0.000·m^{-3} (8 %). The SAD_m and J applied in this study did not therefore lead to significant differences in the CAPEX results for pumps and blowers. The cost of pipes and membrane tanks was negligible (≈1 % both).

These CAPEX results show that the membranes determine the investment cost, so that improving the results implies: i) working with filtration strategies with high transmembrane flows in winter (high fouling rates in summer); and/or ii) a reduction in membrane costs due to market trends or possible recycling of used membranes (Rodríguez-Sáez et al., 2022). A reduction of 10 % in the cost per m^2 of membrane would reduce its contribution to CAPEX to €0.019 and €0.016·m^{-3} under summer and winter conditions, which represents savings of 10 % and 11 %, respectively.

Finally, although an economic evaluation of CAPEX was carried out for the different summer and winter operating conditions, the effective implementation of a UF system at an industrial level should be evaluated by the most unfavorable CAPEX, i.e. the economic viability must be based on the most conservative scenario to guarantee that the investment cost of the implemented system corresponds to a process capable of fulfilling the treatment requirements all year round.

3.2.1.3.2. Operating costs. As can be seen in Fig. 3b and Table S6 of the supplementary material, OPEX varied in summer between €0.053 (Sc) and €0.106·m^{-3} (Sb), while operating costs were between €0.033 and €0.053·m^{-3} in winter (Wc and Wb, respectively). TEC was the main contributor to OPEX, with an average cost of €0.062 ± 0.022·m^{-3} in summer and €0.028 ± 0.008·m^{-3} in winter (79 ± 6 % and 66 ± 6 %, respectively). UF membrane replacement was also a significant OPEX cost and accounted for €0.008 ± 0.001·m^{-3} in summer and €0.007 ± 0.001·m^{-3} winter (11 ± 3 % and 17 ± 3 %, respectively). Chemical cleaning of membranes (2–3 %), permeate pump and feeding pump

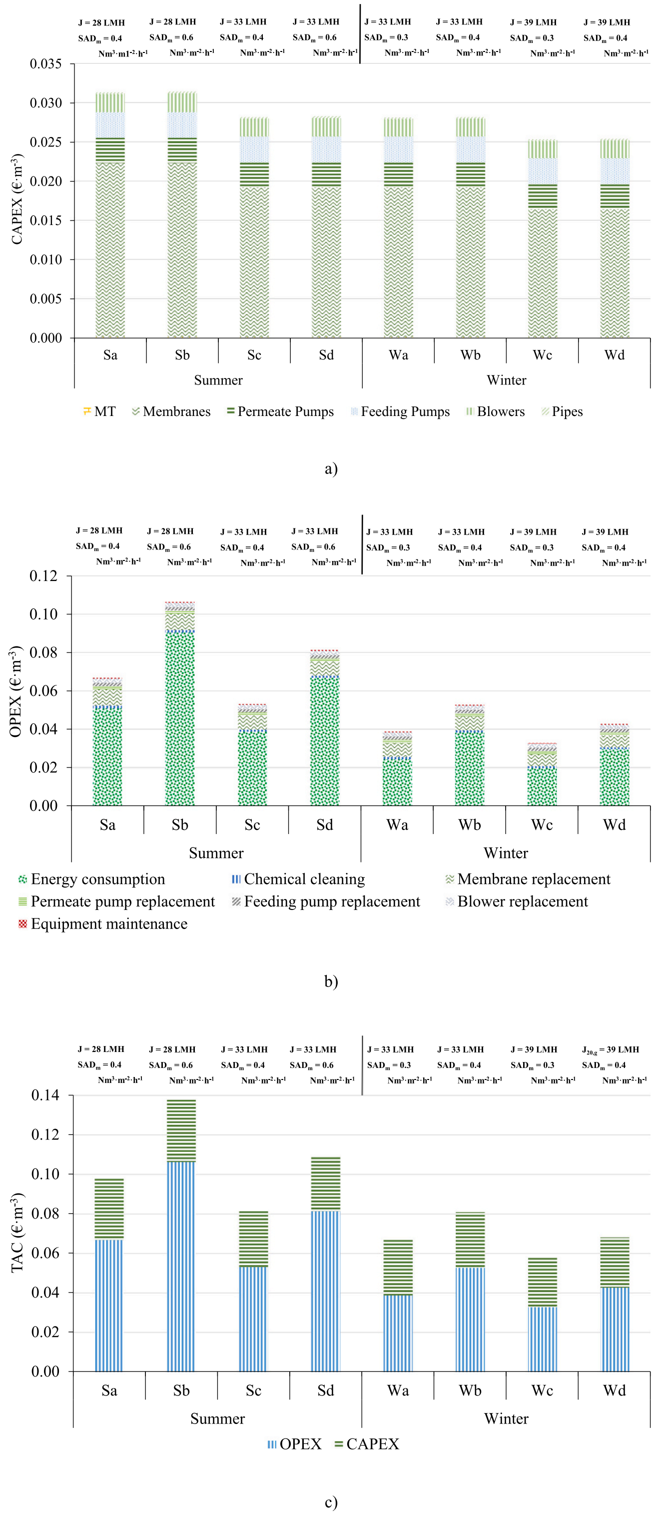

Fig. 3. Economic assessment of the projected full-scale UF unit: a) CAPEX; b) OPEX; and c) TAC.

replacements (2–4 % both), and blower replacement (3–5 %) were minor contributions to OPEX, while equipment maintenance was negligible (≈1 %).

Reducing OPEX means reducing the aeration requirements for membrane scouring, e.g. reducing SAD_m between Sb and Sa (from 0.6 to 0.4 $Nm^3 \cdot m^2 \cdot h^{-1}$) lowered OPEX from €0.106·m^{-3} to €0.067·m^{-3}, a 37 %. However, reducing SAD_m can increase fouling, which eventually increases the requirement for reagents for chemical cleaning, so that optimizing aeration should include the complete filtration process and the interrelationships of the different factors.

The operating costs of the different tertiary treatments (CFF, UF, UV and RO) can be found in Melgarejo et al. (2016). The results for OPEX, once corrected with the euro zone inflation rate until 2024, were: CFF of €0.017·m^{-3}; UF of €0.042·m^{-3}; UV of €0.009·m^{-3}; and RO of €0.262·m^{-3}. The UF result is in the range of that obtained in this study in the winter scenario (€0.033–0.053·m^{-3}). OPEX was also calculated by Akhoundi and Nazif (2018) (WWTP which treats 450,000 $m^{-3} \cdot d^{-1}$), who obtained the following operating costs (corrected with euro zone inflation rate): Coagulation-flocculation of €0.075–0.109·m^{-3}, UF of €0.176–0.688·m^{-3}, UV of €0.007·m^{-3} and RO of €0.223–0.352·m^{-3}. This range for UF is significantly higher than the highest OPEX obtained in the present study, €0.106·m^{-3} in sub-scenario Sb. Jodar-Abellan et al. (2019) reported an operating costs range of €0.06–0.11·m^{-3} (corrected with euro zone inflation rate) for tertiary treatment processes based on microfiltration, UF, RO and reversible electrodialysis, similar to the OPEX range in this study (€0.033–0.106·m^{-3}). Clem and de Mendonça (2022) assessed a combination of ozone and UF for tertiary treatment of wastewater and obtained an OPEX range of €0.300–0.922·m^{-3} (corrected with euro zone inflation rate), which is significantly higher than the operating costs in this study. Finally, Hernández-Chover et al. (2022) reported an operating cost of UF treatment for water reclamation of €0.197·m^{-3} (corrected for euro zone inflation rate). However, it is important to highlight that the OPEX results are strongly influenced by the items considered in the assessment (e.g., inclusion of labor cost), treatment capacity and the price of energy, so that comparisons between studies should be carefully conducted.

3.2.1.3.3. Total annualized cost. Fig. 3c shows that the TAC varied from €0.081·m^{-3} (Sc) to €0.138·m^{-3} in summer (Sb) while in winter it varied between €0.058·m^{-3} (Wc) and €0.081·m^{-3} (Wb). Summer TAC was an average of €0.107 ± 0.024·m^{-3} and 0.068 ± 0.009·m^{-3} in winter. OPEX was the largest TAC cost, accounting for 71 ± 5 % in summer and 61 ± 4 % in winter. Improving the financial viability of the technology thus involves setting the aeration to sustain significant transmembrane fluxes and reduce fouling to diminish the energy requirements for membrane scouring.

3.2.2. Life cycle assessment

The LCA outcome is shown in Fig. 4, where results for global warming potential, water consumption, fossil resource scarcity and mineral resource scarcity are depicted. Major direct environmental burdens were associated with the operating phase, with 95–99 % for global warming (0.04–0.17 kg $CO_{2eq} \cdot m^{-3}$), water consumption ($4.6 \cdot 10^{-4}$–$1.7 \cdot 10^{-3}$ $m^3 \cdot m^{-3}$) and fossil fuel scarcity (0.01–0.04 kg $Oil_{eq} \cdot m^{-3}$) and slightly lower for mineral resource scarcity, 90–92 % ($7.2 \cdot 10^{-5}$–$2.8 \cdot 10^{-4}$ kg $Cu_{eq} \cdot m^{-3}$). The construction phase had a small influence and was only significant for mineral resource scarcity (8–10 %), mainly related to steel production (>68 % of the construction phase load) and membrane acquisition (>14 %). Environmental benefits associated with the dismantling phase were negligible in all the categories evaluated (<3 %).

Energy consumption in the operating phase showed the highest influence of all the categories evaluated. Energy represents 86–96 % of the potential global warming impact, 69–88 % for water consumption, 85–95 % for fossil fuel scarcity and 64–76 for mineral resource scarcity.

As a way of comparison, assuming a tertiary treatment system for a similar flow rate using 12 UV radiation lamps with an energy consumption of 0.04–0.13 kWh·m^{-3} (according to Collivignarelli et al. (2021)) has a global warming potential of 0.01–0.05 kg $CO_{2eq} \cdot m^{-3}$, a water consumption impact of $1.3 \cdot 10^{-4}$–$4.2 \cdot 10^{-4}$ $m^3 \cdot m^{-3}$, a fossil resource depletion of $3.4 \cdot 10^{-3}$–$1.1 \cdot 10^{-2}$ kg $Oil_{eq} \cdot m^{-3}$ and mineral resource depletion of $1.9 \cdot 10^{-5}$–$6.1 \cdot 10^{-5}$ kg $Cu_{eq} \cdot m^{-3}$. Pintilie et al. (2016) applied LCA to determine the main environmental contributors to urban wastewater and water reclamation opportunities in Tarragona

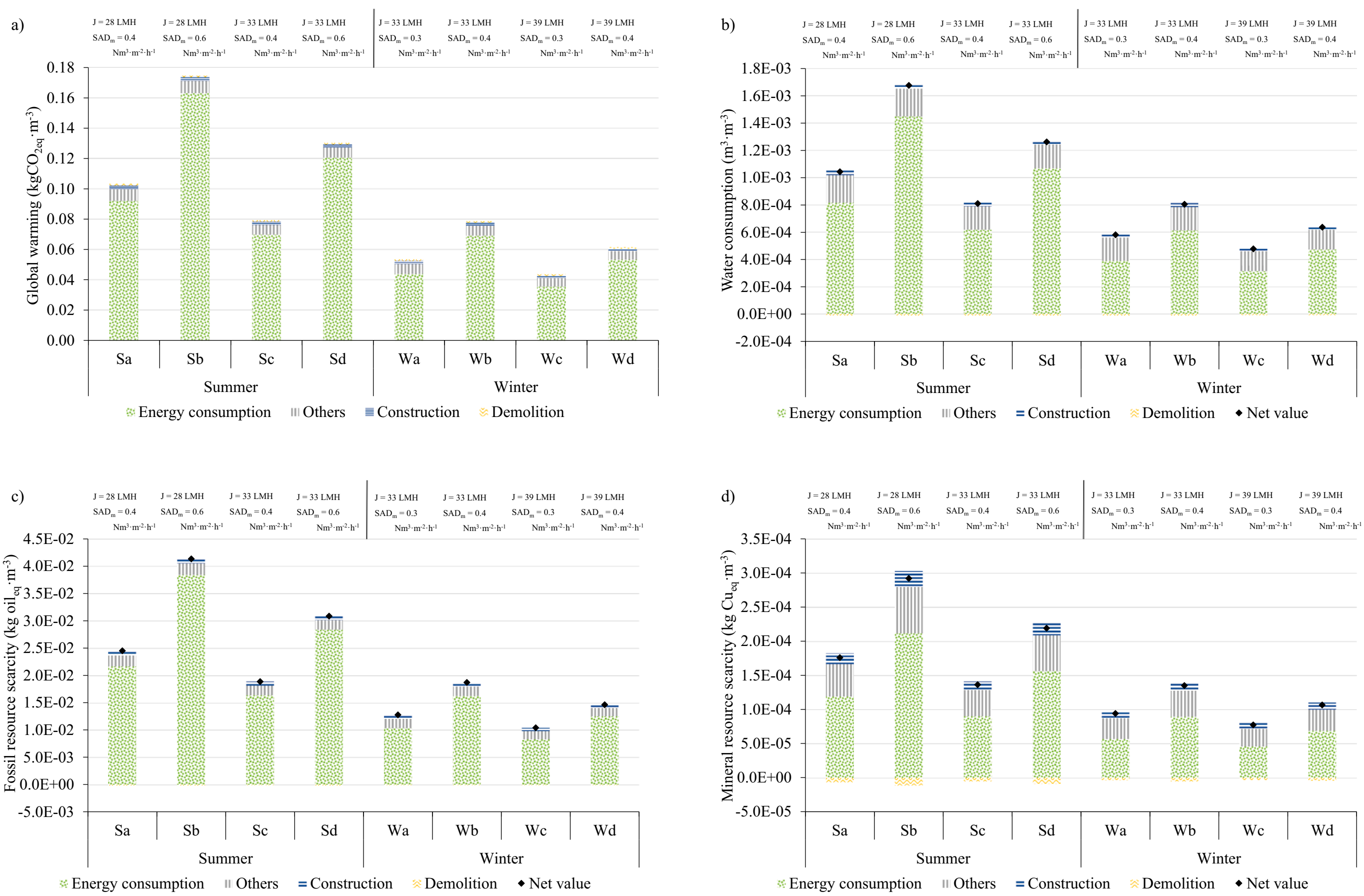

Fig. 4. Results of environmental assessment in both scenarios for a) global warming potential (GWP), b) water consumption (WC); c) fossil resource scarcity (FRS); and d) mineral resource scarcity (MRS).

WWTP (Spain). The authors defined two scenarios: i) direct discharge of the effluent; and ii) water reclamation for industrial reuse with a tertiary treatment that included coagulation-flocculation, microscreen filtration, two-stage sand filtration, reverse osmosis and UV treatment. The results showed a significant contribution to environmental loads in Scenario ii, for example climate change increased by $1.95 \cdot 10^{-1}$ $CO_{2eq} \cdot m^{-3}$, and fossil depletion by $4.85 \cdot 10^{-2}$ kg $Oil_{eq} \cdot m^{-3}$. However, including water reuse and potable water replacement in the scope reduced water to $-4.39 \cdot 10^{-1}$ $m^3 \cdot m^{-3}$.

Improving the environmental performance of UF membrane applications for tertiary treatment therefore necessarily involves reducing energy consumption. One possible option could be to increase the solids concentration in the membrane tanks. Previous studies on filtering raw wastewater showed that this approach favors the formation of sludge flocs inside the tanks, facilitating flux by creating a more porous cake layer (Sanchis-Perucho et al., 2023b). However, this would only be applicable when cake layer formation is mostly responsible for fouling (i.e. in winter). In summer, fouling seemed to be dominated by pore blocking, so that the operating strategy should be different. Including displacement of conventional water for irrigation, in line with Pintilie et al. (2016), would also improve the environmental profile of the tertiary treatment.

3.3. Effluent quality

The current legal framework for water reuse in Spain includes EU Regulation 2020/741 on the minimum requirements for water reuse (EU-WR) together with the Spanish Royal Decree 1620/2007 (S-RD). The EU-WR simply deals with agricultural reuse, while the S-RD also includes others (e.g. urban, industrial, recreational, and environmental uses). As the EU-WR has just started to be applied, the current Spanish legislation will soon be updated to adapt to EU regulations.

Both rules classify water qualities according to the irrigation method and water-crop contact. Both regulations are stricter when edible parts of the crop are in direct contact with reclaimed water, while water reuse for non-food crops is less restricted, following a fit-for-reuse approach (Radini et al., 2023). In this study, effluent quality was compared to the more stringent quality requirements of both regulations: “Crop irrigation with a water application system that allows direct contact of the reclaimed water with the edible parts for fresh human consumption” in the S-RD; and Quality Class A in the EU-WR suitable for “All food crops consumed raw in which the edible part is in direct contact with reclaimed water and root crops that are consumed raw”.

Table 6 shows the obtained microbiological and physico-chemical results for the effluent of the UF prototype together with the requirements of the EU-WR and S-RD. The sampling campaign comprised a total of 10 samples collected during the operation in spring. Microbiological analysis was limited to *E. coli* due to its smaller size. The absence of this bacteria in the effluent guarantees that other indicators present in the legislation and of larger dimensions cannot pass through the membrane either. This would be the case, for example, of Legionella ssp. and helminth eggs. In any case, the WWTP in this study is located in an environment with crops composed mainly of citrus and vegetables with localized or flooded irrigation. Therefore, according to the EU-WR, it would not be necessary to quantify legionella or eggs.

The analysis did not detect the presence of *E. coli* in the effluent immediately after the UF membranes. This is in line with other studies that described UF membranes as an effective barrier to pathogens, thanks to their pore size. Ferreiro et al. (2021) applied continuous UF (pore size of 0.02 μm) with and without a self-cleaning system to the

Table 6
Microbial and physicochemical characterization of the effluent.

Parameter	Unit	Effluent Mean ± SD			Regulation (EU) 2020/741	Spanish royal decree 1620/2007
E. coli	CFU·100 mL^{-1}	0	±	0	≤10	≤100
BOD_5	mg·L^{-1}	4.2	±	2.8	≤10	n.a.
SS	mg·L^{-1}	3	±	2	≤10	≤20
Turbidity	NTU	1.0	±	0.6	≤5	≤10
Conductivity	dS·m^{-1}	1.7	±	0.3	n.a.	≤3.0
Metals, transition metals and metalloids						
B	mg·L^{-1}	1.41E-01	±	4.08E-02	n.a.	≤0.5
As	mg·L^{-1}	8.26E-04	±	1.14E-04	n.a.	≤0.1
Be	mg·L^{-1}	2.90E-05	±	4.25E-05	n.a.	≤0.1
Cd	mg·L^{-1}	7.66E-05	±	5.97E-05	n.a.	≤0.01
Co	mg·L^{-1}	2.96E-04	±	1.16E-04	n.a.	≤0.05
Cr	mg·L^{-1}	2.70E-03	±	3.87E-03	n.a.	≤0.1
Cu	mg·L^{-1}	3.03E-02	±	2.90E-02	n.a.	≤0.2
Mn	mg·L^{-1}	1.61E-02	±	8.78E-03	n.a.	≤0.2
Mo	mg·L^{-1}	2.43E-03	±	9.87E-04	n.a.	≤0.01
Ni	mg·L^{-1}	3.70E-02	±	3.90E-02	n.a.	≤0.2
Se	mg·L^{-1}	9.64E-04	±	4.74E-04	n.a.	≤0.02
V	mg·L^{-1}	7.07E-04	±	3.75E-04	n.a.	≤0.1

effluent of the Galindo WWTP (Basque Country, Spain) and obtained 0.0 ± 0.0 and 0.7 ± 1.01 CFU 100 mL^{-1} for *E. coli*, respectively. Foglia et al. (2020) studied a treatment scheme based on UASB followed by AnMBR equipped with hollow-fiber UF modules (pore size of 0.03 μm) for the treatment of WWTP effluent in Falconara Marittima (Italy) and reported 1 CFU·100 mL^{-1} in the permeate, while (Yang et al., 2021)also confirmed UF (pore size of 0.02 μm) as a suitable method of meeting EU-WR standards for class A water quality.

However, other studies did find *E. coli* in UF permeate; Peña et al. (2019) analyzed the permeate of an AnMBR equipped with UF membranes with a result of 42 ± 48 CFU·100 mL^{-1}. As in this study, the permeate was stored in an open-air tank from which the samples were taken, so that bacteria re-growth was likely. Also, pore size was higher, 0.045 μm. Pérez et al. (2022) combined UF (pore size of 0.02 μm) and RO for water reclamation from secondary effluent at Vuelta Ostrera WWTP (Cantabria, Spain). *E. coli* in the permeate was in the range 0–649 CFU·100 mL^{-1}, with an average of 17 CFU·100 mL^{-1}. In this case, the UF permeate was stored in a tank that fed the RO or was used for backwashing and chemical cleaning. Although the study did not specify how the samples for microbiological analysis were collected, they were possibly obtained from an open-air storage tank. Bray et al. (2021) studied 200 kDa and 400 kDa UF membranes for disinfecting effluents at Gdansk-Wshód and Gdynia-Degórze WWPs (Poland) and reported concentrations of fecal coliforms, which includes *E. coli* in the range 5–20 CFU·mL^{-1} for 200 kDa membranes and 70–180 CFU·mL^{-1} for 400 kDa However, when explaining the presence of microorganisms in the permeate they hypothesized that it could have been due to leaks or breaks in the membrane. The above studies were mostly carried out under S-RD and not EU-WR requirements for qualities A and B. These findings highlight the importance of correct sampling when measuring pathogens, as well as efficient membrane operation and maintenance. It should also be remembered that EU regulations require appropriate water quality at the point of supply to the consumer, so that if bacteria re-growth is common in storage tanks, this is an issue that reclaimed water managers would have to deal with if this technology is implemented.

Regarding organic matter, BOD_5 is included as a parameter only in EU-WR and only for Quality Class A. while for the other quality classes, BOD_5 should not be over the limit imposed by the WWTD, i.e., 25 mg·L^{-1}. Average BOD_5 was 4.2 ± 2.8 mg·L^{-1}, which allows unrestricted use of effluents (BOD_5 < 10 mg·L^{-1}). As UF membranes retain particulate organic matter, only a fraction of soluble DBO_5 would pass through, which significantly improves the permeate quality. For instance, Ferreiro et al. (2021) obtained a BOD_5 of 4 ± 1 mg·L^{-1} and 2.9 mg·L^{-1} before and after UF, respectively. As conventional efficiently operated aerobic wastewater treatments tend to have low BOD_5 in their effluents, usually below 10 mg·L^{-1} (Tchobanoglous et al., 2014), this parameter should not be a bottleneck for water reuse.

TSS and turbidity also showed values below those established in the EU-WR and S-RD. Again, the presence of UF membrane guarantees good quality permeate in relation to these parameters, thanks to the control produced by a given pore size. Ferreiro et al. (2021) reported a reduction of TSS and turbidity from 3.4 ± 2.5 mg·L^{-1} and 9.5 ± 1.7 NTU (secondary effluent) to 0.06 mg·L^{-1} and 0.95 NTU, respectively. Pérez et al. (2022) obtained results for TSS and turbidity in the secondary effluent in the range of 0–30 mg·L^{-1} (average 10.7 mg·L^{-1}) and 1.12–26.3 NTU (average 6.62 NTU), respectively, whereas UF reduced them to the 0–6 mg·L^{-1} range (average 1.16 mg·L^{-1}) and 0.00–2.98 NTU (average 0.25 NTU).

Only the concentration thresholds for metals, transition metals and metalloids (MET) are explicitly included in the S-RD, although the EU-WR also refers to them as elements to be considered if there are significant risks to human health and the environment. These risks should be evaluated by the mandatory water reuse risk management plans (WRRMPs) (Radini et al., 2023). As shown in Table 6, MET concentrations clearly satisfy the maximum admissible concentrations, so that no significant risks from metal pollution would be expected if this wastewater is reused. These results are consistent with locations that make a small or negligible industrial contribution to WWTP influents. In the same way, conductivity only appears as a limiting parameter in the S-RD, with a maximum value of 3.0 dS·m^{-1}, which is significantly higher than the permeate conductivity in this study (1.7 ± 0.3 dS·m^{-1}).

Overall, the results obtained in this study demonstrate that tertiary UF treatment produces a high-quality effluent capable of accomplishing the stringent microbiological and physico-chemical requirements set by the Spanish and European regulations. On the other hand, the role of nutrient contents in reclaimed water effluents in Spain is unclear. As the Spanish Royal Decree 4/2023 does not consider reclaimed water as a discharge, treated wastewater could theoretically contain nutrients to be directly applied in fertigation. But in this case, nutrient addition to soils should be properly managed (Foglia et al., 2023). The consequences of these legal modifications should be considered and addressed by regulators and policy makers to guarantee the economic and environmentally sustainable quality of reclaimed water to promote water reuse and circular water management.

3.4. Alternative scenarios and future perspectives

Alternative scenarios could theoretically improve the feasibility of this UF system. For instance, the Spanish government recently issued a new regulation to boost water reuse (Royal Decree 4/2023, 2023) that provides an exception to the fee for discharge to surface water when treated wastewater is effectively reused. The potential benefits of this measure have already been pointed out in Jiménez-Benítez et al. (2020). The Spanish regulation sets a basic discharge fee of €0.01751·m^{-3} and a coefficient of increase or decrease proportional to the discharged water quality. Since tertiary UF treatment produces reclaimed class A water (European Commission, 2020), the savings in the discharge fee could reduce OPEX by up to an average of €0.05944 ± 0.02317·m^{-3} in summer and €0.0246 ± 0.0089·m^{-3} in winter, while the TAC would be between €0.08969 ± 0.02418·m^{-3} and €0.0512 ± 0.0910·m^{-3}, respectively.

Also, the nutrient content in the effluent can provide additional economic benefits as it can replace the use of mineral fertilizers in agriculture. Considering a nutrient concentration of 9 mg N·L^{-1} and 0.9 mg P·L^{-1} in the effluent (average annual values in the WWTP) and a fertilizer price for urea ($CO(NH_2)_2$) and superphosphate (P_20_5) of €0.295 and €0.435 per kg, respectively, the WWTP would discharge nutrients worth €34,728 per year. The optimal use of reclaimed water should therefore include the evaluation of the nutrient content, as the agricultural use of these substances would give the reused water a higher value. Furthermore, the lower treatment cost associated with the nutrient removal in the WWTP could be expected if their recovery or reuse is included in a comprehensive fertigation system, as their removal from the wastewater would no longer be desirable. This approach aligns with current European policies, which aim to boost the circular economy in the water sector, such as the Circular Economy Action Plan (European Commission, 2020). However, it should be remembered that boosting wastewater reuse in Europe requires more than technological implementations.

Finally, from a regulation perspective it should be noted that the definition of the point of supply is not made clear in the EU-WR and may be controversial. If the responsible organization for the treatment and reuse is different from the supplier to farmers (final consumer), the role and responsibilities of the different stakeholders must be clearly defined. This can be challenging in some EU member states where different sectors of the water cycle (wastewater treatment, potable water, etc.) are managed by national or local governments). This can create conflicts that hinder the application of water reuse schemes. For instance, current EU-WR legislation states that all the integrated systems involved in the wastewater reuse scheme must be described and considered in in the risk management plans, including external contributions. Many Italian water distribution systems use non-regulated surface water to irrigate crops and cannot be described or monitored. Hence, innovations must be applied not only from a technical perspective, but also innovative governance and policy-making tools are needed (Foglia et al., 2023). In this regard, it is expected that the new wastewater treatment directive (Cartuyvels, 2024) will boost the water reuse and adequately complement the regulation for agricultural use.

4. Conclusions

An UF demonstration plant for tertiary treatment was operated for water reclamation. J between 28 and 39 LMH and SAD_m in the 0.2–1.3 $Nm^3·m^{-2}·h^{-1}$ were applied during winter and summer ambient conditions and fouling rates were monitored to assess the treatment performance. The results obtained recommend operating under different conditions during the year to prevent excessive fouling propensity: J of 28–33 LMH and SAD_m of 0.4–0.6 $Nm^3·m^{-2}·h^{-1}$ in summer and J of 33–39 LMH and SAD_m of 0.3–0.4 $Nm^3·m^{-2}·h^{-1}$ in winter.

Based on the demonstration plant data, upscaling to an industrial UF facility was projected and its energy, economic and environmental profile was evaluated. On average, energy consumption of 0.308 ± 0.112 kWh·m^{-3} in summer and 0.140 ± 0.040 kWh·m^{-3} in winter were obtained, the blowers being the main contributors (93 % in summer and 86 % in winter). CAPEX was calculated as €0.030 ± 0.002·m^{-3} in summer and €0.027 ± 0.002·m^{-3} in winter. Membrane acquisitions accounted for the main contribution, 67–71 % in summer and 64–68 % in winter conditions. The expenditure associated with energy consumption was the major OPEX contributor (66–79 %), and total operating costs were calculated as €0.077 ± 0.023·m^{-3} and €0.042 ± 0.008·m^{-3} for summer and winter, respectively. A TAC of €0.107 ± 0.024·m^{-3} in summer and €0.068 ± 0.009·m^{-3} in winter were obtained from the CAPEX and OPEX, results.

The LCA results indicated the operation phase as the major contributor to the environmental burdens (over 95 % for global warming potential, water consumption and fossil resource scarcity), the construction phase only being significant in the mineral resource scarcity category (8–10 %). Contributions to global warming potential during 0.04–0.17 0.04–0.17 kg CO_{2eq}·m^{-3} (global warming potential), 4.6 · 10^{-4}–1.7 · 10^{-3} $m^3·m^{-3}$ (water consumption) 0.01–0.04 kg Oil_{eq}·m^{-3} (fossil fuel scarcity) and 7.2 · 10^{-5}–2.8 · 10^{-4} kg Cu_{eq}·m^{-3} (mineral resource scarcity). As the environmental impacts of UF are mainly related to energy consumption in all the categories evaluated, optimizing the filtration performance focused on reducing the energy requirements would directly improve the treatment's environmental sustainability. Including in the scope displacement of conventional water for irrigation is expected to reduce environmental loads, at least with regard to water consumption.

Hence, results show that considering seasonal variation in the membrane management, could help to improve the economic and environmental performance of the treatment, since energy sustainable transmembrane fluxes and energy needs differ. Therefore, seasonal variations should be identified and characterized, if any, in order to establish an appropriate operating strategy during the year.

Finally, the analysis of the effluents confirmed tertiary treatment based on UF technology as a suitable system to obtain a high-quality effluent, as it fulfilled the most stringent requirements for agriculture water reuse in both the Spanish and European Regulation on water reclamation for agriculture.

Future work should focus on scaling up to industrial applications to validate these findings, exploring process optimization to reduce fouling, and investigating better strategies to improve environmental and economic behavior of ultrafiltration (UF) tertiary treatment. In this sense, optimization should address: i) aeration to prevent fouling while reducing energy consumption; ii) increasing permeate flux to reduce membrane requirements; iii) and the duration and frequency of F-R and backwash cycles, to improve the productivity of the filtration process.

CRediT authorship contribution statement

A. Jiménez-Benítez: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation. **J. González-Camejo:** Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation. **V. Sandoval-García:** Writing – original draft, Methodology, Investigation, Data curation. **A. Robles:** Writing – review & editing, Validation, Supervision, Methodology, Investigation, Formal analysis. **N. Martí:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization. **A. Seco:** Writing – review & editing, Validation, Supervision, Methodology.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.177273.

Data availability

Data will be made available on request.

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