Prospective Life Cycle Assessment and economic analysis of direct recycling of end-of-life reverse osmosis membranes based on Geographic Information Systems

Senán, J. and Blanco, A. and García-Pacheco, R. and Landaburu, J. and García-Calvo, E. *Prospective Life Cycle Assessment and economic analysis of direct recycling of end-of-life reverse osmosis membranes based on Geographic Information Systems.* Journal of Cleaner Production

https://doi.org/10.1016/j.jclepro.2020.124400

Highlights:

1- GIS-LCA provided reliability in the ex-ante logistics quantification
2- Environmental and economic results evidenced the potential viability of recycling
3- Transport of recycled modules should be limited within an international scale
4- Service Life Ratio identified the priority processes for eco-improvement
5- Economic viability is linked to improvements in waste characterisation

1.-Introduction

The current environmental multicrisis that humanity is facing demands to accelerate the transition towards Circular Economy (CE) (Martin Geissdoerfer et al., 2017). This transition requires an important effort in product rethinking and their life cycles, overcoming the actual linear life cycle model taken across all sectors and technologies (EMF, 2013). Nowadays, reverse osmosis (RO) is the most common technology used for desalination worldwide. Nevertheless, the RO membrane modules follow a linear life-cycle scheme. They are made in central facilities using fossil-based materials and then, distributed to desalination plants where they are discarded to landfills after five to ten years of service life (Landaburu-Aguirre et al., 2016). Landaburu-Aguirre et al. (2016) estimated that more than 840,000 end-of-life (EoL) RO modules are discarded per year globally. During the last five years, the RO market annual growth has been over 11, and it is forecasted to have sustained growth of 10.9% for the next years (Market Reports, 2017). Consequently, in 2025 more than two million of EoL-RO modules might be discarded. Within the framework of CE, different waste valorisation alternatives are being studied such as i) RO membrane reuse (García-Pacheco et al., 2020), ii) direct RO membrane recycling into nanofiltration (NF) and ultrafiltration (UF) membranes (Lawler et al., 2012; García-Pacheco et al., 2018; Landaburu-Aguirre et al., 2016), iii) biofilm-membrane reactors (Morón-López et al., 2019), or iv) electrodialysis (Lejarazu-Larrañaga et al., 2020). Among all the alternatives, the EoL-RO membrane direct recycling into NF and UF membranes is the most developed technology. Recycling processes had been tested at pilot scale (García-Pacheco et al., 2018a; IMDEA-Water and Valoriza-Water, 2016). Furthermore, recycled membranes have also been validated in desalination and wastewater treatment plants (García-Pacheco et al., 2018).

To the knowledge of the authors, few environmental analyses support this recycling alternative (Lawler et al., 2012; Senán-Salinas et al., 2019). Those studies aimed at evaluating the environmental outcome through Life Cycle Assessment (LCA). The first assessment performed by Lawler et al. (2016) developed a specific waste hierarchy for EoL-RO modules that coincided with the theoretical waste hierarchy of the Waste Framework Directive (WFD) (European Commission, 2010). Among the different EoL-RO potential alternatives, the recycling into UF was the second preferable option behind the RO reuse. Posteriorly, Senán-Salinas et al. (2019) analysed the recycling process into NF and UF, in this case, at pilot-scale, updating the Life Cycle Inventory (LCI) from the lab results obtained by Lawler et al. (2016). Furthermore, although Lawler et al (2016) studied the recycling of brackish water (BW) RO membrane models into UF membranes, Senán-Salinas et al (2019) introduced the recycling of seawater (SW) RO models and their recycling into NF. Additionally, in the substitutability factor, they introduced a ratio of permeabilities to correct the environmental accreditation with the performance relationship between the recycled membranes and the new ones. Nonetheless, this last study was constrained to a gate to gate study, considering, uniquely the recycling process. In this way, some processes as transport were out of scope. Regarding the membrane direct recycling, the transport was just analysed by Lawler et al. (2016). It assumed the transport distance between two desalination plants (2,480 km) representing the distance between the waste source and the end-user. It concluded that this distance allowed an environmental net accreditation. However, a sensitivity analysis evidenced an important relationship in the haulage distance with the service life of the recycled membranes (introduced in the substitutability factor).

Despite recycled membrane transport has been rarely considered in earlier studies of alternative EoL membrane management, it has been evidenced as one of the most relevant contributors to the overall environmental balance in several recycling activities of waste, such as construction and demolition (Blengini and Garbarino, 2010), plastic packaging (Perugini and Vivaldi, 2003), electric and electronic equipment (Islam and Huda, 2018), photovoltaic modules (Mahmoudi et al., 2019) or biopolymers (Hottle et al., 2017). In this sense, two main transport activities have been pointed out to be crucial. First, the reverse logistics related to the waste collection and post-consumer transport from the locations of the waste generation to the recycling location (Faraca et al., 2019). Second, the delivery or distribution of secondary products to end-users (Islam and Huda, 2018; Toniolo et al., 2013).

Geographic Information Systems (GIS) has been noticed as an ally for the transport quantification in reverse logistics and secondary products delivery. It provides reliability and accuracy to the analysis with specific on-site distances (Laurent et al., 2014; Ripa et al., 2017). For example, GIS has been defined as a fundamental tool for the case studies focused on construction and demolition waste recycling, where the reverse logistic has an elevated impact (Blengini and Garbarino, 2010; Göswein et al., 2018). GIS also has been used for geospatial waste characterisation and waste stock quantification (Blengini and Garbarino, 2010; Guo et al., 2018; Mastrucci et al., 2017). Regarding the distribution of secondary products, the main role of GIS is to provide spatial resolution. However, one of the most important limitations found is the identification of specific locations of end-users. Therefore, in many cases, distribution distances are directly assumed or based in assumed locations (Blengini and Garbarino, 2010; Faraca et al., 2019; Lawler et al., 2015). Other proposed goal of GIS-LCA nexus is establishing haulage distances for the distribution of secondary products associated with a hierarchy through LCA results (Vossberg et al., 2014).

For these reasons, GIS can overcome the issues of *ex-ante* logistics quantification in a prospective analysis of EoL-RO direct recycling. The nexus with LCA will provide an important degree of reliability in its quantification. The use of specific on-site information could reveal unexpected aspects and interactions as the role of the distribution of desalination plants in the location choice, waste stocks quantification per membrane model type (BWRO or SWRO) and so on. Until now, current knowledge has not clarified the relationship and influence of some of these aspects in conjunction with the availability of waste stocks per type or the variability in the performance of the recycled membranes. Moreover, the specific contribution of reverse logistics and recycled membranes distribution has not been analysed separately with the specific on-site distances. Finally, there is also an absolute knowledge gap from the economic point of view of on-site real projects. In this sense, economic analysis of the direct recycling process has been assessed at Senán-Salinas et al (2019) without including membrane transportation.

The aim of the present study is the environmental and economic assessment of a complete waste management system of EoL-RO membranes through direct recycling into NF and UF membranes. In particular, it is focused on clarifying the contribution of the transport of the EoL-RO modules from the reverse logistics to the distribution of recycled modules to the secondary end-users. In addition, different strategies of collection and recycling considering the modules type are analysed. Therefore, the herein proposes an entire site-specific GIS-based model for a prospective analysis of the innovative recycling system.

2.-Material and methods

2.1.-Case study: Segura’s watershed

Segura’s watershed (SWS) was chosen as the region for the study based on two criteria (Fig. 1). First, this region accumulates 42% of the total Spanish desalination capacity, which is the fourth country in the world with the highest installed desalination capacity (AEDYR, 2018). Thereby, it contains a potential stock of EoL-RO modules. Second, Segura´s watershed is one of the most important Spanish regions with water reusing capacity and desalinated water use for agriculture (Molina and Casañas, 2010; DESALDATA, 2012). Therefore, it contains an important quantity of potential end-users.

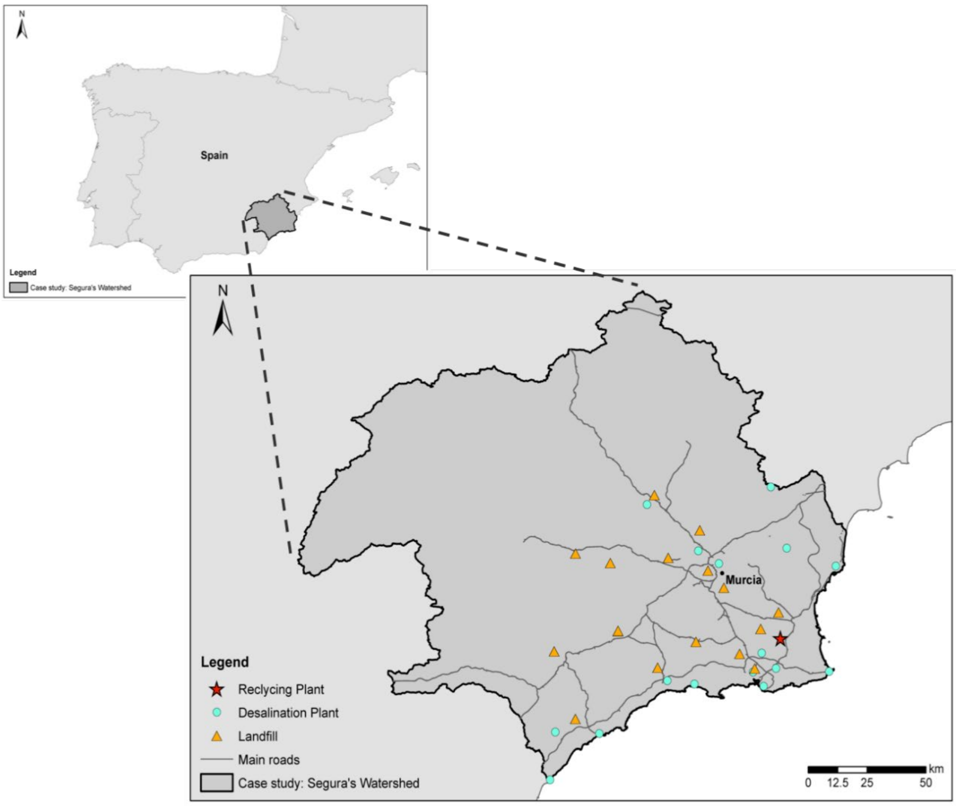

Fig. 1. - Location of the study region with the inventory of desalination plants and landfills. It also includes the location chosen for the potential full-scale recycling plant.

2.2.-GIS methodology

2.2.1.-Reverse logistics: stock quantification, recycling collection strategies and plant location choice.

EoL-RO stock quantification

The first use of GIS was the identification of the desalination facilities within the SWS found in the AEDYR database (AEDYR, 2015). The annual stock of EoL-RO modules generated in the SWS was estimated by assuming that 100 elements are installed for each 1,000 $m^3 \cdot day^{-1}$of installed capacity and mean service life of RO modules of 10 years (Lawler et al., 2012). Table 1 summarises the stock results.

The recyclability of the EoL-RO modules was estimated through the available distributions of weights and type of fouling. Based on the experience gained within the Life-TRANSFOMEM project, the modules with a weight higher than 25 kg were discarded (37.5% of SW modules and 12.5% of BW modules). Within those modules that the weight was below 25 kg, uniquely the modules with organic fouling and clay were considered potentially recyclable. According to membrane autopsy studies, these fouling types correspond to 71.9% of the RO modules in Europe (Darton and Fazel, 2001). Therefore, this coefficient (0.719) was assumed to determine the quantity of recyclable EoL-RO membranes due to the type of fouling. As a result, 4,433 SWRO and 366 BRO modules were considered as the annual recyclable membranes stock.

Table 1. - Number of facilities and estimated mean annual stock of EoL-RO membranes (AEDYR database 2015. SW: Seawater; BW: Brackish water

	SW	BW	Total
Number of facilities	20 (51%)	19 (49%)	39 (100%)
Total capacity installed ($m^3 \cdot day^{-1}$)	992,270	53,010	1,045,28
Estimated amount of RO elements installed	99,227 (94%)	5,301 (6%)	104,528 (100%)
Average number of RO elements installed per plant	4,961	279	2,680
Mean annual flow (modules·year^{-1})	9,923	530	10,453
Estimated recyclability of EoL-RO elements by weight (in %)	62.5	87.5	63.7
Estimated recyclability of EoL-RO elements by fouling type (in %)	71.9	71.9	71.9
Overall estimated recyclability (in %)	44.9	62.9	45.8
Approximate quantity of recyclable modules (modules·year^{-1})	4,433	366	4,799

Membrane collection strategies

For the present study, three strategies were analysed (Table 2) depending on i) the type of modules that are collected (BWRO or BWRO and SWRO called Com strategy, from complete) and ii) the target product of the recycling process (NF or UF membranes). The recycling of SWRO into UF membranes was excluded due to the low process performance of the recycled UF membranes for that specific brand used in the study (García-Pacheco et al., 2018).

Table 2.-The strategies based on i) the type of EoL-RO modules collected and ii) the target product.

Recycling strategy code	EoL-RO modules collected	Target products by EoL-RO type	Description
BW-NF	BWRO	NF	Only EoL-BWRO modules are collected and recycled into NF membranes
BW-UF	BWRO	UF	Only EoL-BWRO modules are collected and recycled into UF membranes
Com-NF	BWRO & SWRO	NF	Both BW and SW EoL-RO modules are collected and recycled into NF membranes

Plant location choice

The plant location defines the impact of the collection network, thereby, sustainability criteria are commonly used in its definition during the waste management planning phase (Faraca et al., 2019; Ripa et al., 2017). Therefore, for the present case, a location was chosen based on suitability and minimum transport impact. The suitability was analysed by a Multi-Criteria Evaluation (MCE) analysis using ArcGIS v10.3 that identified potential areas within the region (895 raster cells; cell size 100 m). It included restrictive criteria as land use, protected areas and conditioning as the slope. *K-means* cluster analysis was used to define four main regions. The closest raster cell to their centroids was considered representative of each region. Finally, for the four potential locations, distances to the desalination plants were estimated based on the shortest routes network analysis using the exported Google Earth roads in QGIS v3.8. Then, payload distance was estimated with the annual stock of the desalination facilities (Eq. 1). Among the estimated four locations, a location close to Cartagena was chosen as the recycling plant location due to the lowest payload distance.

$$Pd_{\,R.L} = \sum_{l \in L} \sum_{t \in T} d_l \cdot n_{tl} \cdot w_t \qquad \text{(Eq. 1)}$$

Where,

$Pd_{\,R.L}$: Total payload distance (in t·km) of the reverse logistics

d_l: Road distance (in km) from de desalination plant *l* to the recycling facility location.

$n_{t,l}$: Amount of EoL-RO modules (in items) by EoL-RO type *t* and plant *l*

w_t: Module weight (in tonne/module) by EoL-RO type *t*

L: the set of desalination plants

$T = \{BW, SW\}$: the set of EoL-RO types.

2.2.2.-Recycled products distribution into the end-markets

Regarding the distribution of recycled products, three different target market regions were defined using the recycling facility as the reference point: i) Regional, ii) Iberian, and iii) European. The geographical markets were constrained to the European continent, excluding islands (Fig. 2). In each of the regions studied, 1,000 uniformly distributed random points were generated among the space with ArcGIS 10.3, representing the locations of end-users. The haul distances were calculated multiplying the Euclidean distances from the location of the recycling facility to random points by a detour factor of 1.25 as recommended for continental long hauls (Zgonc et al., 2019).

2.2.3.-Distribution of commercial products into the end-market

The comparison between the distribution chains of the recycled membranes and the commercial ones was conducted. The membrane suppliers centralise the production of membranes in one or two facilities for worldwide distribution. Therefore, two main locations associated with commercial membrane production facilities were used for the comparison: European LANXESS® facility at Bitterfeld, Germany and American DOW FilmTec® facility at Edina, Minnesota, USA. For the distribution of the membranes from the European facility to the location of end-users was considered to be conducted by road and the distances were calculated using the same methodology as described for the recycled products. The distribution chain from the American facility was assumed to be different and consists in the next steps: i) in America, from the facility to the port the transport is conducted by road, ii) from the American port to the European port by shipping, and iii) once reaching the European port the transport to the target end-user is done by road. The port of New York-New Jersey was considered the most appropriate location for the distribution of the commercial membranes into the European market. The 20 most important European ports, as well as the port of Cartagena, were chosen as the most suitable receptor ports. (Fig. 2). The haulage distances between ports (American and Europeans) were estimated multiplying the Euclidean distance with a detour factor of 1.5 (Hine and Preston, 2018). Finally, the same methodology used for the recycled products was applied to estimate the inland distribution.

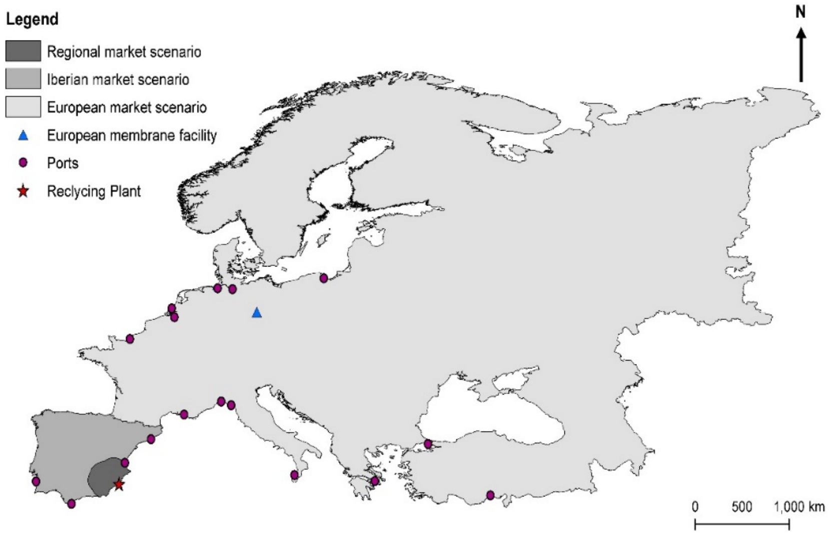

Fig. 2.- Market regions considered. It also includes the location of the recycling plant, the docks considered and the location of the European membrane facility.

2.3.-Life Cycle Assessment

2.3.1.-Goal and scope

LCA was implemented to evaluate the environmental impact of the recycling process and the distribution of recycled membranes. This study is *gravel to gate study*. It includes the processes from the EoL-RO collection to the distribution of recycled membranes to the end-users, including the characterisation and transformation processes and other transports (extended description in section 2.3.2). The functional unit (FU) considered was one EoL-RO standard membrane module (8 inches, active area of 37 m^2) recycled and transported to the location of the secondary user.

2.3.2.-System boundaries

Fig. 3 illustrates the system boundaries. The substitution approach involves the balance between the impact of the recycling processes and the offsetting impact of new membranes production (Eq 2). The recycling process includes the collection of EoL-RO modules, their characterisation, the transportation of discarded modules to the landfill, the transformation of recyclable modules through a free chlorine oxidation solution and the transportation of recycled modules to the end-users location within the three target market regions defined (Eq. 3). The membrane characterisation of both, EoL-RO and the recycled product, consists on the evaluation of the membrane performance (in terms of permeability and salt rejection) filtering BW during 1h as reported in (García-Pacheco et al., 2018). The transformation process, includes the oxidation of the modules via passive immersion in the NaClO solution (García-Pacheco et al., 2018), their washing, the treatment of the wastewaters and emissions to water as earlier reported in Senán-Salinas et al. (2019). Transport processes involved were considered by road with a truck as described in section 2.2.

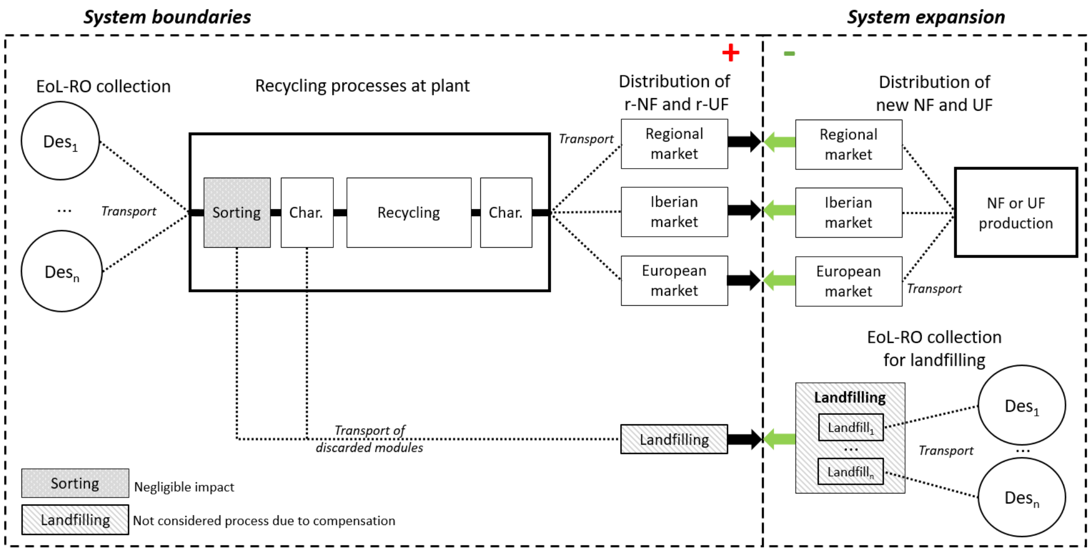

Fig. 3.-System boundaries and system expansion scheme with processes involved.

On the other side of the system expansion includes the production of the new module and its transport to the end-user location within the three market regions considered (Eq 4), and the avoided transport of the EoL-RO modules to landfill (Eq. 2). The avoided environmental impact by landfilling was not quantified because the recycling alternatives are considered *open loops,* where the landfilling is considered inevitable after their second life. The impact of commercial products was corrected by a substitutability factor that introduces the functionality equivalence described in Senán-Salinas et al. (2019). This factor included the active area and the permeability and the service life ratios (Eq 2).

$$I_{total} = I_{recycling} - I_{tr.land} - \left(I_{new} \cdot \left(\frac{SL_{rec}}{SL_{new}} \cdot \frac{A_{rec}}{A_{new}} \cdot \frac{L_{rec}}{L_{new}} \right) \right) \quad \text{(Eq. 2)}$$

$$I_{recycling} = I_{tr.col} + I_{char} + I_{rec.pro} + I_{tr.dist} + I_{tr.disc} \quad \text{(Eq.3)}$$

$$I_{new} = I_{prod.new} + I_{tr.new} \quad \text{(Eq.4)}$$

Where,

I_{total}: Impact balance based on the substitution approach of the direct membrane recycling process.

$I_{recycling}$: Impact of the recycling and distribution of a module since the EoL-RO location to the secondary end-user location.

$I_{new:}$: Impact of the production and distribution of a new (NF or UF) commercial module (system expansion).

$I_{tr.col.}$: Impact of the transport of EoL-RO modules collection since the desalination plants and the recycling plant.

I_{char}: Impact of the membrane module characterisation processes. It includes two types the characterisation of the EoL-RO module and the characterisation of the recycling product (as NF or UF) and two stages (before and after recycling process).

$I_{rec.proc}$: Impact of the recycling process. It includes the oxidation, washing, wastewater treatments and emissions to water.

$I_{tr.dist}$: Impact of the distribution of recycled modules from the recycling plant to the secondary end-users location.

$I_{tr.disc}$: Impact of the transport of discarded modules from the recycling plant to the landfill.

$I_{prod.new}$: Impact of the production of a new (NF or UF) commercial module at the plant.

$I_{tr.new}$: Impact of the transport of a new (NF or UF) commercial module from the production plant to the end-user location.

$I_{tr.land:}$: Impact of the original transport of EoL-RO modules to landfills (system expansion).

SL_{rec}: Membrane life span (years) of the recycled modules.

SL_{new}: Membrane life span (years) of the new (NF or UF) modules.

A_{rec} : Active surface membrane area (m^2) of the recycled modules.

A_{new}: Active surface membrane area (m^2) of the new (NF or UF) modules.

L_{rec} : Permeability ($dm^3 \cdot m^{-2} \cdot h^{-1} \cdot bar^{-1}$) of the recycled modules.

L_{new}: Permeability ($dm^3 \cdot m^{-2} \cdot h^{-1} \cdot bar^{-1}$) of the new (NF or UF) modules.

2.3.3.-Sensitivity assessments

The characterisation of all the modules has an important impact. It is expected from the trends in other recycling activities that the characterisation effort as well as the development of sampling protocols and predictive models would decrease due to the understanding of waste variability and representativeness. Therefore, a sensitivity assessment was performed with different characterisation ratios changing the orders of magnitude. The current practice at research stage was 1:1 (García-Pacheco et al., 2018). 1:10 represents a more realistic approach assuming that one out of ten modules would be characterised. The results of the scenarios 1:100 were similar to 1:10 thereby, they were excluded from the manuscript but they can be found in supplementary data in Appendix A.

2.3.4.-Life Cycle Inventory analysis

The most relevant part of the LCI developed in the present study was focused on the transport and recyclability of the modules. Table 3 summarises the relation and the module flows considering the recyclability rates of Section 2.2. The EoL-BWRO modules have a higher recyclability rate (63%) than EoL-SWRO modules (45%). Table 4 summarises the inventory associated with the logistics of the modules. The distribution of recycled modules is the most important transport process, although there are important differences in the distribution market regions. The impact in the reverse logistics in *Com* strategy is higher than in the BW due to the recyclability rates mentioned before. Also, EoL-SWRO modules are heavier than EoL-BWRO ones. Nevertheless, this difference comes from the original weight of the new modules (for BWRO 11.5-12 $kg \cdot module^{-1}$ and 14-16 $kg \cdot module^{-1}$ for SWRO). Lawler et al. (2015) included 37.24 t·km (2,480 km) for the distribution of the BWRO modules, which would consist of an intermediate value within the herein defined European market.

Table 3. - Number of modules in each flow by scenario relativized by FU. In parenthesis, the percentage of EoL-RO modules discarded.

Recycling strategy code	**BW**	**Com**
Modules recycled (FU)	**1**	**1**
EoL-RO modules collected	1.58	2.18
EoL-RO modules characterised-initial	1.39	1.39
Recycled modules characterised-final	1	1
Total EoL-RO modules discarded	0.58	1.18
Modules discarded by weight (>25kg)	0.19 (32%)	0.79 (67%)
Modules discarded due to low performance	0.39 (67%)	0.39 (33%)

The inventories of the recycling processes were collected and discussed in Senán-Salinas et al. (2019). Regarding the characterisation of the modules, only the energy use was taken into account. The energy use for characterisation processes (sum of pre and post-recycling process) depends on the concrete scenario and the performance of the recycled membranes (NF or UF), varying from 1.40·to 2.06 $kWh \cdot module^{-1}$ with a characterisation ratio of 1:1. It is important to note that this energy is ten times higher than the electricity used during the transformation and recycling processes (1.53 $kWh \cdot module^{-1}$). Detailed LCI including the background processes can be found in the supplementary material (Tables S1-S4).

Regarding the LCI of the offset products, Bonton et al., (2012) was used for NF membrane production and Ecoinvent process for UF membrane production. Background processes were regionalised based on the facility location (supplementary material of Appendix A). The substitutability ratios (or offset ratio) were the same as in Senán-Salinas et al. (2019): NF from BWRO 0.60±0.19, NF from SWRO 0.47±0.06 and UF from BWRO 0.99±0.9. They explain the differences between strategies. However, most important differences can be observed between BW-NF and BW-UF membranes because their weights differ. The weight of the spiral wound NF modules is 11.5 kg·module^{-1} with a membrane area of 37 m^2 (0.31 kg·m^{-2}). However, the weight of a hollow fibre UF module of 51 m^2 is 48 kg (0.94 kg·m^{-2}) (DOW, 2011). The offset transport impact was compared to the distribution of recycled membranes impact. The offset transport impact was higher in all the categories except for the European region, where the impact avoided is lower than the impact produced by the sum of the recycling transport processes.

Table 4. - LCI related to the transport (payload distance: in t·km, mean and standard deviation) in each process and scenario of the recycling supply chain and the commercial supply chain associated with the recycling processing and distribution of modules. RL-Reverse logistics; DW-discarded by weight transport to landfilling, DC-discarded by characterisation transport to landfilling, AV- Avoided transport to landfills.

(*) Results were corrected by the membrane surface area (37 m^2) and the permeability ratio.

		Transport impact (t·km·module^{-1})								
		Recycling supply chain (by road at the EU)					Commercial supply chain*			
							Facility at Europe	Facility at America		
Recycling strategy code	Market scenario	RL	D.W	D.C	AV	Distribution	By road at the EU	By road at the USA	By freight ship	By road at the EU
BW-NF	Regional	1.29±0.71	0.06±0.01	0.06±0	-0.48±0.21	3.24±1.3	-17.51±4.95	-14.7±4.07	-80.32±22.28	-1.28±0.69
	Iberian					10.73±3.54	-17.04±5.21	-14.7±4.07	-77.34±21.8	-2.8±1.71
	European					66.48±24.73	-17.05±9.67	-14.7±4.07	-88.02±26.36	-11.44±8.19
BW-UF	Regional	1.29±0.71	0.06±0.01	0.06±0	-0.48±0.21	3.24±1.3	-81.15±54.03	-68.23±45.59	372.74±249	-5.87±4.89
	Iberian					10.73±3.54	-79.24±54.24	-68.23±45.59	-358.98±241.01	-13.04±12.32
	European					66.48±24.73	-79.75±71.98	-68.23±45.59	-406.86±272.18	-54.16±56.97
Com-NF	Regional	2.54±1.26	0.14±0.13	0.05±0.04	-0.87±0.34	3.66±1.36	-12.86±0.75	-10.8±0.31	-59±1.74	-0.94±0.43
	Iberian					12.1±3.48	-12.55±1.78	-10.8±0.31	-56.78±2.8	-2.05±1.09
	European					74.99±24.98	-12.51±5.92	-10.8±0.31	-64.66±7.11	-8.4±5.33

2.3.5.-Life Cycle Impact Assessment method, software and databases

The LCA modelling was performed with OpenLCA v1.10 (opnlca.org) and R (R Core Team, 2018). For background data, Ecoinvent v3.4 was used (Wernet et al., 2016). Five categories of the ILCD-midpoint method v1.05 (available at nexus.openlca.org) were chosen due to its relevance in transport as Climate Change (GWP), Resource Depletion of fossil, minerals and non-removable resources (RD-f+m), Particulate Matter (PM), and Human Toxicity-non carcinogenic (HT-nc); and the EoL-RO recycling processes as Marine Eutrophication (ME) (Senán-Salinas et al., 2019) (Table 5). In addition, Monte Carlo analysis with 1,000 runs was performed for the introduction of the inventory variability related to the transportation (described in section 2.2), the substitutability factor as well as the uncertainty of background processes. A confidence interval of 95% was considered according to Guo and Murphy (2012).

Table 5. The ILCD-midpoint v1.0.5 method categories selected and the abbreviations used.

Categories of the ILCD-midpoint method	**Characterisation methods**	**Reference unit**	**Relevance in**
GWP	Climate change	kg CO_2 eq.	Modules transport
HT, nc	Human toxicity, non-cancer effects	CTUh	
PM	Particulate matter/Respiratory inorganics	kg $PM_{2.5}$ eq.	
RD, f+m	Resource depletion, mineral, fossils and renewables	kg Sb eq.	
ME	Marine eutrophication	kg N eq.	EoL-RO recycling process

2.3.6.-Normalisation and service life ratio

The difficulties to estimate the service life of recycled membranes and its comparison with commercial ones has been an issue in direct membrane recycling. Therefore, this issue was approached by using an indicator proposed by Senán-Salinas et al. (2019): the Service Life Ratio (SLR). Eq. 5 and 6 describe the adaptation of SLR ratio for the present case. In addition, the limiting SLR results (highest SLR value) were filtered for each location of secondary-users. In this sense, this SLR value can be understood as an internal normalisation procedure in which the system expansion impact serves as a reference within the study (Prado et al., 2017).

$$I_{recycling} - I_{tr.land} - \left(I_{new} \cdot \left(\frac{SL_{rec}}{SL_{new}} \cdot \frac{A_{rec}}{A_{new}} \cdot \frac{L_{rec}}{L_{new}}\right)\right) = 0 \quad \text{(Eq. 5)}$$

$$SLR = \frac{SL_{rec}}{SL_{new}} = \frac{I_{rec} - I_{tr.land}}{I_{new} \cdot \left(\frac{A_{rec}}{A_{new}} \cdot \frac{L_{rec}}{L_{new}}\right)} \quad \text{(Eq. 6)}$$

Where,

SLR: Service life ratio

2.4.-Economic analysis

The goal of the economic analysis was to evaluate the potential economic viability of a recycling facility. The business plan was based on the full cost recovery by the selling of the recycled modules (García-Pacheco et al., 2017). Boundaries in the economic study were identical to the LCA described in section 2.3.2 and Fig 3. Firstly, the projected recycling plant was sized by scaling-out the recycling pilot plant developed during the Life-TRANSFOMEM project. Then, a cost analysis was performed including the capital expenses (CAPEX) and operational expenses (OPEX). The costs related to the recycling pilots were obtained from Senán-Salinas et al. (2019). Similar inventories to the LCA described in section 2.3 were used to maintain the coherence. Regarding the transport costs, ACOTRAM 3.1 database was used (SMT, 2018). Finally, the price of the recycled modules was estimated for different payback periods (Eq. 7). Other details regarding the characterisation and transport cost are in the supplementary material of Appendix A.

$$P = \frac{\frac{I}{p}+C}{n} \qquad \text{(Eq. 7)}$$

P: Price of the recycled module (in euro·module^{-1})

I: Investment (in euro)

p: Payback period (in years)

C: Annual cost expenses (in euro·year^{-1})

n: number of modules produced

2.5.-Data analysis: software and tools

The R software was used for data wrangling (R Core Team, 2018) with R packages as *Tidyverse* (Wickham and R-Studio, 2019), *ggplot2* for graphics (Wickham, 2016) and *rgdal* for geographical data management (Bivand et al., 2019).

3.-Results and discussion

3.1.-Life Cycle Impact Assessment results

Fig.4 illustrates the contribution graph of the Life Cycle Impact Assessment (LCIA) results for Com-NF strategy. The rest of the strategies were not included because the contribution profiles are very similar. They can be found in Fig. S1 in the supplementary material in Appendix A. The most remarkable differences were found in the environmental contribution profiles due to the characterisation ratio. When the characterization ratio is 1:1, the characterisation processes are the main contributors in four of the five categories regardless of the distribution scenario. The reason is the high-energy use required for testing each module at high pressure (around 15 bar for NF and RO and 5 bar for UF tests) during an hour (already mentioned in LCI section 2.3.3). It has to be mentioned that this ratio represents the real ratio used according to the technology readiness achieved at the research stage. Otherwise, when the characterization ratio is 1:10, the contribution of the characterisation contribution reduces drastically, and the recycling processes appear to be the main contributors within Regional and Iberian regions. That means that improvements in the characterisation techniques and sampling protocols could reduce the environmental impact between 20 to 50%, respect to the current practice (research stage). Nevertheless, it highlights a necessary intermediate stage between the research stage and the full implementation. Focusing on other sectors practices, there is a constant effort to enhance and optimize their characterisation techniques (Chatterjee and Mazumder, 2020; Rada and Cioca, 2017). Most of them are focused on the identification of key parameters, the optimization of sampling protocols and the application of new characterisation techniques. In membrane recycling, one of the key elements is the waste variability of the EoL-RO membranes in terms of fouling and EoL performance (in terms of permeability and rejection). A deeper knowledge of waste variability will introduce reliability and representativeness in the sampling design as evidenced in other wastes (Pérot et al., 2020). In this case, these issues have not been addressed yet. It is important to note that ME is the category where the recycling processes have a major contribution regardless of the scenario. The main reason is the Nitrogen (N) emissions to water ($7.2 \cdot 10^{-3}$ kg·module^{-1}) coming from the membrane fouling content. Jeong et al., (2016) analysed the organic matter of different RO modules with different positions within a train. In terms of N content in the RO modules fouling, the figures oscillated from $1.60 \cdot 10^{-3}$ kg·module^{-1} to $6.15 \cdot 10^{-3}$ kg·module^{-1}, a lower range than the used for the LCI of the present study (Fig 4). Nonetheless, important quantitative and qualitative differences can be found due to the membrane position within the plant and between plants. Therefore, a part of the impact could vary from the fouling content, as it could happen with other substances or elements found also in autopsies of some modules. Previous research studies analysed the inorganic part of the fouling by inductively coupled plasma mass spectrometry (ICP-MS) and they indicated the presence of heavy metals as iron or chromium as well as other inorganic phosphorus as phosphates (García-Pacheco, 2017; Molina et al., 2018). Those substances could have an impact on some toxicity or eutrophication categories, respectively. Consequently, a deeper analysis of their influence should be studied in further research. In this sense, fouling distribution knowledge is aligned with the need for getting a better understanding of waste variability, coinciding with the variability knowledge mentioned above. Concerning the RD-m+f, the recycling process is the dominant process for the scenarios of Regional or Iberian membrane distributions. At the scenario of European distribution, the distribution process is the most important contributor in all the categories. Finally, the transport of the discarded modules to landfill does not have a relevant effect due to the short distance found between the recycling plant and the closest landfill.

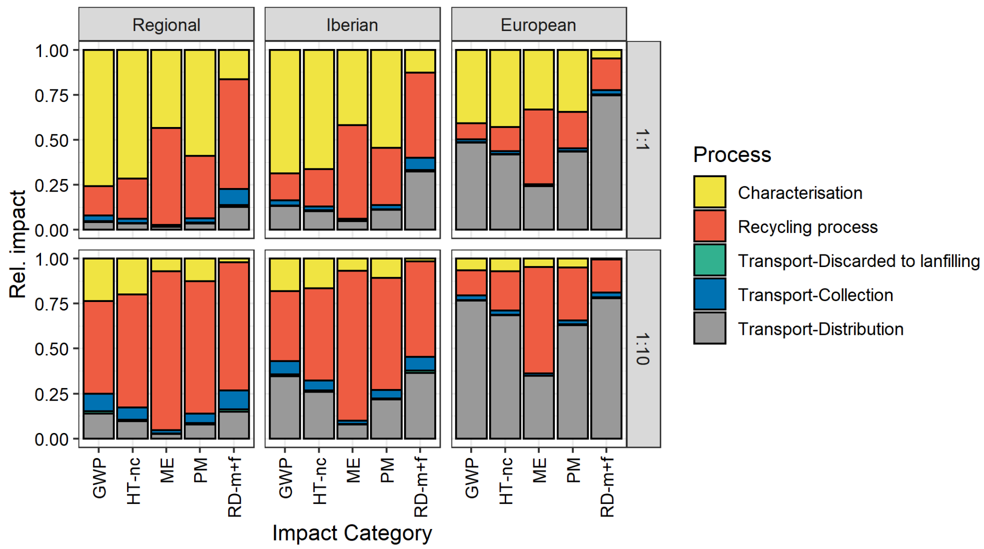

Fig. 4.- Contribution graph of recycling (defined as $I_{recycling}$) for Com-NF strategy in the five ILCD categories selected, including the three market regions (Regional, Iberian and European) and two different characterisation ratios (1:1 and 1:10).

The LCIA results are summarised in Fig. 5. The differences between recycling strategies (BW and Com) are minor. Indeed, the characterisation ratio and the target distribution regions are the main factors that generate differences among the LCIA results. In GWP, HT-nc, ME and PM categories, the characterisation ratio is the variable that affects the most due to its influence observed in Fig 4. In all the cases, the distribution into the European market has the highest impact due to the direct relation with road transport (Table 4 of section 2.3.4). The impact results with characterisation ratio 1:10 at Regional and Iberian market region are higher than the results obtained in our previous publication (Senán-Salinas et al., 2019). In particular, in the categories of GWP, and HT-nc the impact of characterisation and transport almost doubles the impact of the recycling processes by themselves. In the rest of the strategies the contribution of road transport of recycled modules distribution, as well as the characterisation process, have a greater impact. Categories as GWP and RD-m+f point out the relevance of the transport of recycled modules by road (the 74% of the overall impact in the European scenario and 1:10 characterization ratio). However, these results could turn out to be different if future studies contemplating other transport schemes such an intermodal train route (Zgonc et al., 2019) or other possibilities of near future such as post-fossil fuel societies (Ingrao C, et al., 2019). At the European distribution scenario, Lawler et al. (2015), obtained an impact of $1.40 \cdot 10^{1}$ kg CO_2-eq.$\cdot$module^{-1} with a distance of 2,480 km within the European range. Nonetheless, the contribution structure was different. The most remarkable dissimilarities were found when comparing with ME. Related to ME category, Lawler et al. (2015) calculated that the transport has an impact up to $6.70 \cdot 10^{-2}$ kg N-eq·module^{-1} , which is a higher value than the estimated within this study. In addition, Lawler et al. (2015) did not consider N emissions and characterisation processes. The main contributor to ME is the high consumption of NaClO due to the extrapolation of lab inventory results (Caduff et al., 2014). Unfortunately, further comparison with other studies is limited due to the lack of similar works. Besides, the are remarkable differences as a function of the impact methods employed. E.g. Lawler et al (2015) used the ReCiPe method for the assessment of EoL-BWRO membrane recycling into UF membrane, while in our work it is used ILCD. Therefore, only results obtained for GWP and ME categories are comparable.

Even though this on-site case study is limited to the geographical scope of Murcia province (Spain), some findings could be useful to other regions and countries. Concretely, for the delimitation of the secondary product's market as suggested in other sectors (Vossberg et al., 2014). In this case, the national market scale (within 1,200 km) seems a proper area to guarantee the net environmental impact through these limits (further discussed in section 3.3). Nonetheless, discrepancies in other regions could come from a major dispersion of the desalination plants, meaning a major impact in the contribution of EoL-RO collection. The dispersion between facilities will limit more the hauls distances for secondary products and therefore the recycling implementation.

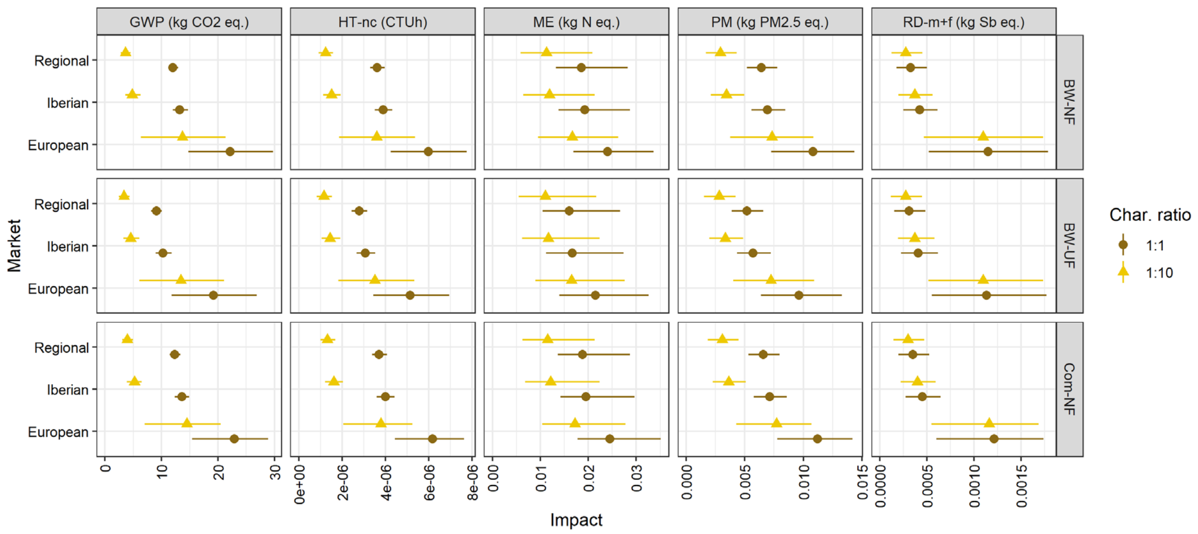

Fig. 5.-Life Cycle Impact Assessment results (mean and confidence interval of 95%) of the one recycled module at the secondary end-user location by characterisation ratio, recycling strategy and market scenario (defined as $I_{recycling}$).

Regarding the potential environmental credits in the different strategies, there are two components to be considered: i) the avoided transport of modules to landfill, which has an insignificant impact (Table 10); and ii) the production and distribution of commercial new products. Fig. 6 represents the results of the two commercial referenced distribution supply chains. The margin of one or two orders of magnitude found between the impact results (Fig. 5) and the potential environmental accreditation (Fig. 6) indicate an important potential benefit that will depend on the recycled membrane SL. The main factor that affects environmental accreditation is the permeability ratio. Recycled modules with permeabilities similar to the new produced modules have a ratio close to one as it is the case of BW-UF recycling strategy. Thus, a higher accreditation is expected (Senán-Salinas, J. et al 2019). The second main factor is the inventory of the new membrane production. The production of UF membranes has a higher impact than the production of NF membranes. This is due to due to the major quantities of polymeric materials involved in UF membrane production and the production impact of the materials. There are low differences between the locations of the commercial facilities even though there is a high payload distance by shipping (Table 4) in the distribution from the American facility. This is because of the low unitary impact of shipping resulting in an overall lower impact than truck transport, especially with long-hauls. (Fig. S2 in supplementary material).

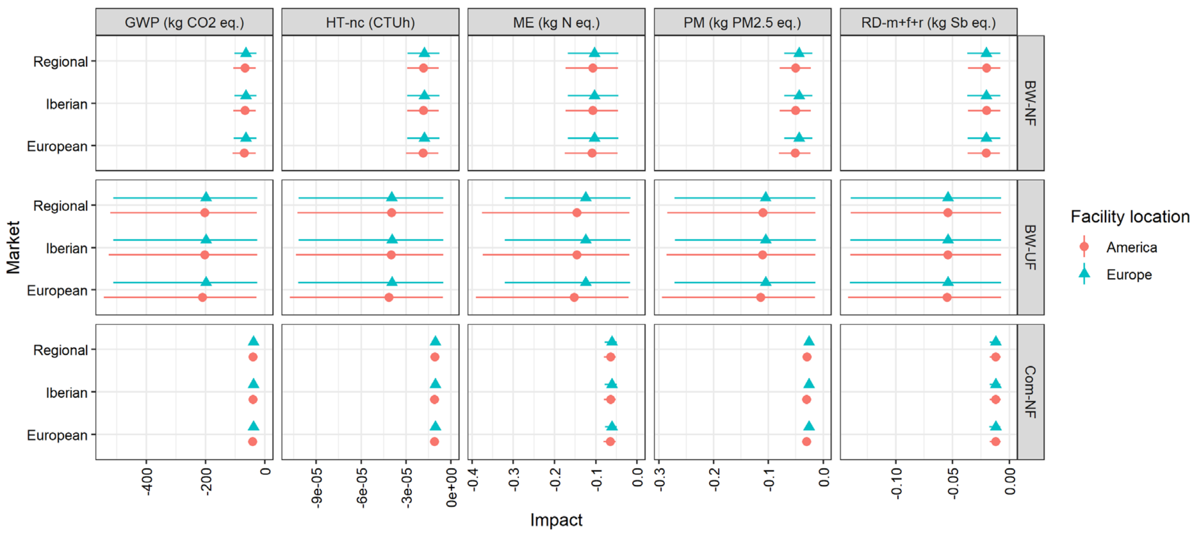

Fig. 6.-Life Cycle Impact Assessment results (mean and confidence interval of 95%) of the potential environmental accreditation of the production and distribution of one recycled module at the secondary end-user location due to substitution of commercial products production and distribution by recycling strategy, membrane manufacture provider and market scenario (defined as I_{new}).

3.2.-Service Life ratios

Table 6 summarises the SLR results for the different strategies and market regions limited by the most restrictive following precaution criteria. SLR values are ranged between 0.02 and 0.6, which means that the service life of the recycled membranes would range between 1.2 months and 6 years. This represents the minimum lifespan of the recycled membranes for its feasible reuse in water treatment processes. Validation of recycled membranes in pilot-scale for BW, urban wastewater and landfill leachate have demonstrated good performance with no significance decline up to one-year testing (García-Pacheco et al., 2020, 2018).

The SLR values are one order of magnitude higher than the ratios obtained by Senán-Salinas et al. (2019). The cause is the introduction of the membrane characterisation process and transport as part of the integral recycling process, which has a major impact than the environmental credits of the transport of commercial products. Nonetheless, as it was mentioned in the previous study, this low SLR remarks the potential use of the recycled membranes in harsh environments or applications where the lifespan of the membranes is significantly low (e.g. landfilling leachate treatment). Regarding the differences amongst the recycling strategies, Com-NF strategy has the most limiting SLR values (the highest values) due to the low environmental accreditation associated with the low permeability of SWRO recycled modules into NF membranes. In this sense, there is an important bias over the performance of SWRO recycled (further discussed in section 3.5). Nonetheless, the influence of the characterisation impact is the main cause of high SLR values.

Table 6. - SRL results (mean and standard deviation) for strategies and markets with a characterisation ratio of 1:1 and 1:10. (*) indicates results limited by the impact of the American location. The rest are limited by the distribution from a European facility.

			Categories of the ILCD-midpoint method				
Char. Ratio	**Recycling strategy code**	**Market scenario**	**GWP**	**HT-nc**	**Meu**	**PM**	**RD, m+f**
		European	0.38±0.17	0.37±0.17	0.26±0.13	0.27±0.12	0.07±0.03*
	BW-NF	Iberian	0.23±0.09	0.25±0.10	0.21±0.10	0.17±0.07	0.02±0.01*
		Regional	0.21±0.08	0.23±0.09	0.20±0.10	0.16±0.07	0.02±0.01*
		European	0.18±0.24	0.24±0.32	0.32±0.43	0.17±0.23	0.04±0.05
1:1	**BW-UF**	Iberian	0.10±0.13	0.14±0.19	0.25±0.34	0.10±0.14	0.01±0.02
		Regional	0.08±0.11	0.13±0.18	0.24±0.33	0.09±0.13	0.01±0.02*
		European	0.60±0.10	0.59±0.09	0.41±0.09	0.43±0.07	0.11±0.03*
	Com-NF	Iberian	0.36±0.03	0.38±0.04	0.32±0.08	0.27±0.04	0.04±0.01*
		Regional	0.33±0.03	0.36±0.04	0.31±0.08	0.25±0.03	0.03±0.01*
		European	0.24±0.12	0.23±0.11	0.18±0.10	0.18±0.09	0.06±0.03*
	BW-NF	Iberian	0.08±0.04	0.10±0.04	0.13±0.08	0.09±0.04	0.02±0.01*
		Regional	0.06±0.03	0.08±0.03	0.12±0.07	0.07±0.03	0.02±0.01*
		European	0.12±0.17	0.16±0.22	0.24±0.34	0.13±0.18	0.04±0.05
1:10	**BW-UF**	Iberian	0.04±0.06	0.07±0.09	0.17±0.25	0.06±0.09	0.01±0.02
		Regional	0.03±0.04	0.06±0.08	0.16±0.24	0.05±0.07	0.01±0.01*
		European	0.38±0.10	0.36±0.09	0.28±0.08	0.30±0.07	0.10±0.03*
	Com-NF	Iberian	0.14±0.02	0.16±0.03	0.20±0.07	0.14±0.03	0.04±0.01*
		Regional	0.11±0.02	0.13±0.02	0.19±0.07	0.12±0.03	0.03±0.01*

3.3.-Limiting Service Life ratios

The identification of the most limiting categories allows prioritizing the most important environmental issues. Therefore, it helps the definition of clear goals for future researchers. To obtain the most limiting results and figure out geographical relations, the maximum SLR value (thus the most limiting value) was filtered per end-user localization. The SLR results (in terms of values and categories) with characterisation ratios 1:1, and 1:10 are illustrated in Fig. 7. Firstly, there is a clear geographical relationship between the limiting values and categories with the proximity to the recycling facility location (Fig. 7a and Fig. 7b). In Regional and Iberian areas, SLR values are below 0.4 (with characterisation ratio of 1:1) which means that an environmental accreditation can be obtained when the membrane lifespans are over 2 and 4 years for UF and NF membranes, respectively (Fig 7a).

Within the Regional and Iberian target markets, the limiting categories: ME and HT-nc are associated to the recycling process or characterisation, depending on the proximity to the recycling plant and the characterisation ratio (Fig 7c and 7d). When the characterisation ratio is 1:1, the predominance of HT-nc in the regional and Iberian markets is due to the energy use during the characterisation processes (Fig 7c). Consequently, it seems interesting to put special effort on the following three aspects: i) to increase the knowledge regarding the waste variability, ii) to explore other potential characterisation techniques such as non-invasive in situ characterisation processes (Tan et al., 2017; Jiang et al., 2017)) and iii) gathering information regarding the potential recycling rates. Related to the characterisation techniques, Jiang et al. (2017) discussed an important number of characterisation techniques that could be feasible for EoL-RO characterisation, as well as environmentally friendly and cost-effective. One example is the electrical impedance spectroscopy used for organic matter and scaling fouling detection. Waste variability and characterisation have been common issues in different recycling sectors such as municipal waste or waste electric & electronic equipment (WEEE) (Clavreul et al., 2012; Islam et al., 2016). In the case of EoL-RO recycling, this topic has not been assessed as such. However, information regarding fouling distribution at the desalination plant and along the pressure tubes (Jeong et al., 2016) can be used as a baseline for this topic. It is important to note that during the present study we did not consider specific recyclability rates for each plant since they normally have installed a common brand and model. However, under the characterisation ratio 1:10, ME becomes dominant over Regional, Iberian, and west European areas (Fig. 5d). This significance (of the ME) is due to the Nitrogen emissions to water. The second environmental priority, according to SLR values within the recycling process is the impact in ME category, directly related to the nitrogen emissions. As mentioned before, in section 3.1, variability in N content varies between plants (due to the water quality) and the membrane position within the facilities. Therefore, to reduce the eutrophication potential associated with the EoL-RO membrane recycling processes, end-of-pipe treatments for nutrients recovery technology could be applied. In this way, secondary environmental credits could be achieved.

It is important to note that within the European region there are two main areas differenced by both SLR values and categories: Central-west Europe and East Europe (Fig. 7a and Fig 7b). Those regions will be divided by a haulage distance of 3,100-3,360 km from the recycling facility location. Within West Europe, membrane delivery by trucking seems environmentally viable. However, in the East European region, alternative transport routes are required because the limiting category is GWP that is related to transport by road. As it has been previously mentioned, one potential improvement could be the use of near-future alternatives post-fossil fuel and decarbonised transport mediums. Other solutions have also been proposed such as the transportable recycling plant for in situ recycling (Grimaud et al., 2018) that could be interesting for the self-use of recycled RO membranes in the same facility.

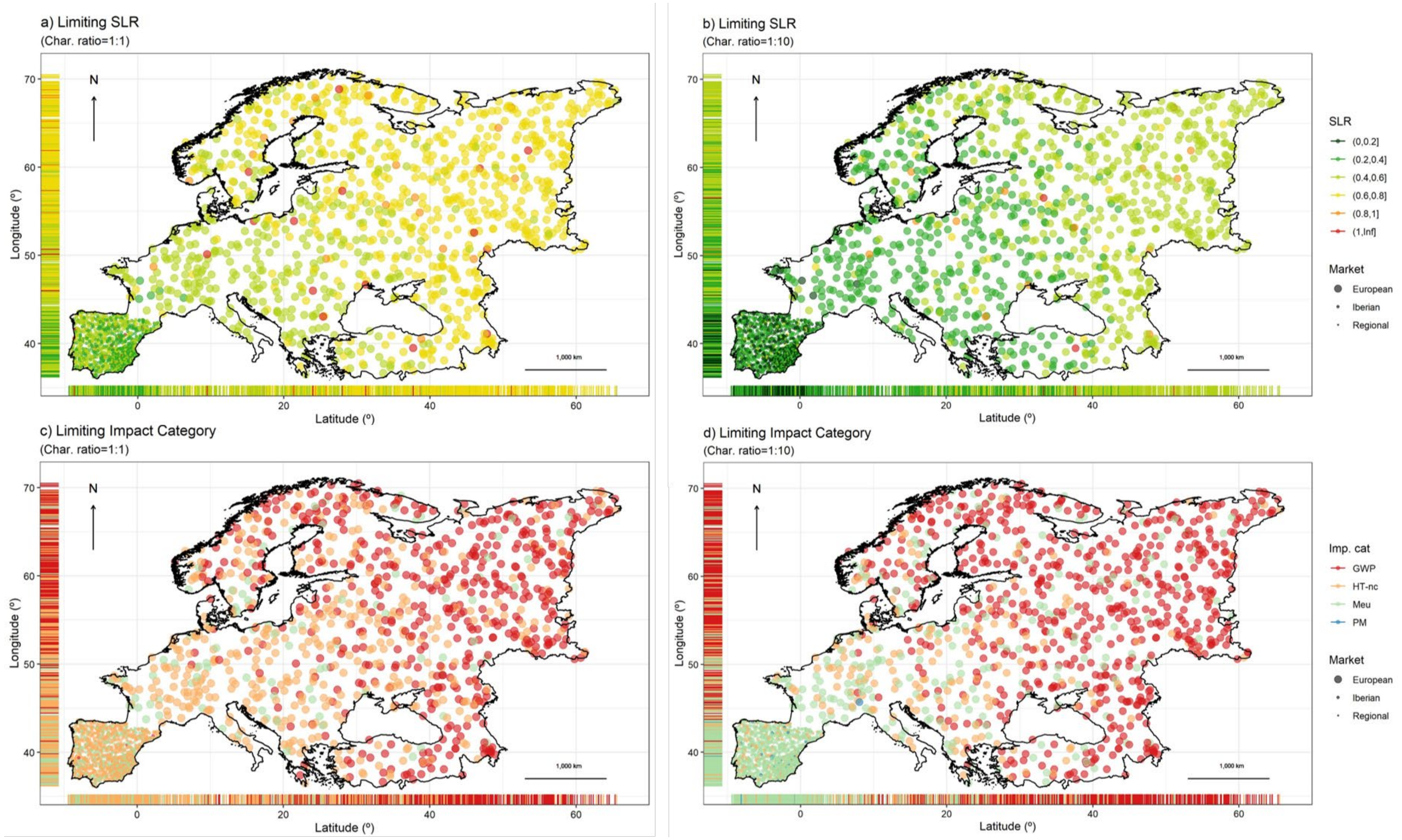

Fig. 7. - Geographical distribution of a) the highest SLR values (limiting), b) the limiting categories, c) scenarios with the highest SLR and d) commercial limiting (highest SLR, lower environmental accreditation). Results from characterisation ratios: 1:1 and 1:10.

3.4.-Economic analysis

The economic viability was measured by the minimum price required for different payback periods (Fig. 8). In all the cases studied, the prices do not differ importantly after 3 years. In concordance with LCA results, the most important variations depend on the characterisation ratio applied. Considering the membrane characterisation ratio of 1:1, the minimum selling price estimated is around 200-300 euros per module. More competitive module prices can be achieved when the recycling process includes a characterisation ratio of 1:10 at the Regional and Iberian distribution scenarios. Thus the total production cost of the recycled membrane would range of 75-100 and 45-80 euro per module for BW and Com recycling strategies, respectively. Comparing with commercial NF and UF modules, this price range seems to be competitive, since the price of new commercial membranes oscillates between 400 to 800 euros per module (García-Pacheco et al., 2017; Mendret et al., 2019). Nonetheless, prices could be reduced in the acquisition of important amounts of modules. Improvements in characterisation processes, such us in-situ methods at the collection point, would make the recycling alternative more attractive for its industrial implementation.

The BW recycling strategy has higher minimum selling prices due to the limited number of EoL-BWRO modules available annually and, hence, the under-usage of the equipment (Tables S5, S6 and S7). Previous results from Senán-Salinas et al. (2019) indicated that cost could be reduced to 24-30 euro per module if the number of recycled modules increase to 1,000 per year due. That is due to the greater distribution of CAPEX into a major amount of modules. The low contribution of reverse logistics in the environmental and economic results addresses the potential extension of the collection area of the plant, probably to an Iberian region, which could help to achieve the cited minimum flow of 1,000 modules per year. In this sense, GIS buffer analysis for the collection should be performed to ensure enough stock of EoL-RO for the economic sustainability of the plant (Coelho and Brito, 2013a, 2013b).

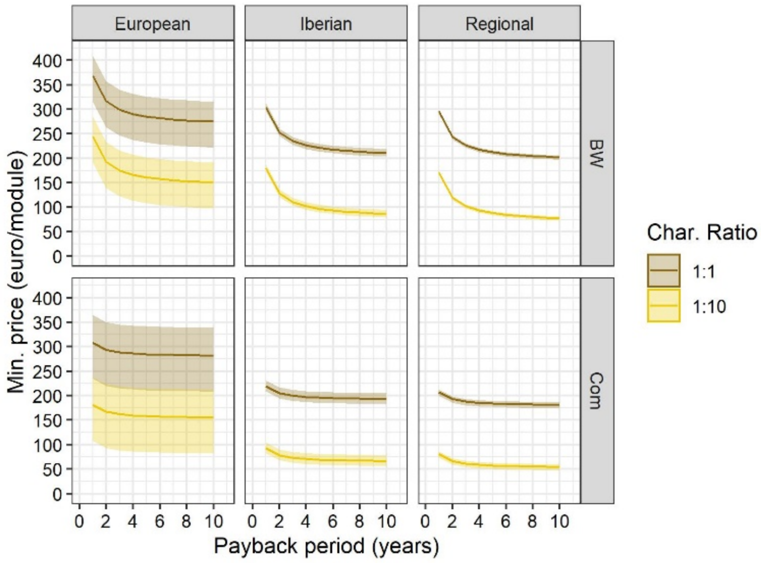

Fig. 8. - Minimum price (mean and confidence interval of 95%) required per module for the viability of the plant.

3.5.-Limitations of the study

The current study has to be considered as a preliminary assessment to identify opportunities and barriers of the potential implementation of a membrane recycling plant at full scale, with a capacity up to 4,500 membranes per year. The main limitations are methodological such as i) representativeness of the case study, ii) inventory limitations, and iii) approach.

Regarding the representativeness of the case study, it is important to note the dispersion of the desalination plants, their capacity and the type of water treated, which will limit the potential stock of EoL BWRO and SWRO membranes. Other cited investigations pointed a higher proportion of BWRO membranes than the found in the present study (Landaburu-Aguirre et al., 2016; Lattemann et al., 2010). Although delocalized BWRO has been pointed out as a more energy-efficient and eco-effective technology to resolve structural water scarcity, important investments are also being focused in SWRO technology with huge capacities as in north Africa o Arabia Saudi (Zarzo and Prats, 2018). Both trends will define the best strategy of direct RO recycling.

Regarding the inventory of this study, the main limitation is related to the assumptions made with the recyclability rates of the modules due to the previous author's experience (Life-Transfomem project). In this sense, membrane performances of 128 modules were extrapolated to a wider number (up to 4,500 modules). As mentioned among the study, this bias affects especially to the SWRO modules, precisely, the most extended module design in the market but the least module design tested within the project (only 2 SWRO membrane models vs 14 BWRO models). In this sense, wider experimentation with SWRO modules could result in important changes in its recyclability.

Finally, regarding the approach, the definition of a watershed or a specific region was a methodological choice that could have done differently by applying a dynamic buffer analysis (Coelho and Brito, 2013a, 2013b). This buffer analysis could analyse the trade-off of increasing the collection area for a reduction of the minimum selling prices with a major environmental impact in the waste collection in BW strategies. Furthermore, as mentioned before, the present approach was also constrained to logistics by the conventional fossil-oil transport system. However, the paradigm of transport is expected to be changed in the next years towards a post-fossil-oil system based on the electrification road transport systems (Navas-Anguita et al., 2018).

4.-Conclusions

In the present study, for the first time, a virtual recycling plant of EoL-RO direct membrane recycling was assessed from the environmental and economic point of view. Overall, GIS-LCA nexus could be one of the solutions to effectively prevent the impacts associated with the implementation of recycling strategies of EoL-RO membranes. The use of GIS for the modelling of reverse logistics and recycled modules distribution provided a high degree of reliability with on-site results. As a novelty, a new GIS methodology was proposed for the integration of the uncertainty in the transport quantification of recycled products into three different scale scenarios: Regional, Iberian and European. Moreover, the recycling impact was compared with the production and the distribution chains of new membrane modules considering two different facility locations within Europe and America. The main conclusions are:

- Results pointed out the great environmental potential of the membrane recycling process in the five midpoint impact categories studied (GWP, HT-nc, ME, PM and RD-f+m). However, the distribution of the recycled modules was concluded to be a limitation in the European scenario.

-The membrane performance characterisation within the recycling processes was also evidenced as one of the main contributors for almost all the categories and the economic study with the current practice state. Therefore, this study concludes that further research effort should be focused on the waste variability of EoL-RO modules as well as the development of efficient in-situ characterisation protocols at the collection point.

-Important environmental conclusions were obtained through the quantification of SLR, which allowed the identification of the limiting categories and their geographical distribution as a normalisation system. In all the cases, SLR values were below one, which means that the environmental impact of membrane recycling is lower than the impact obtained with the production of commercial membranes.

-The financial viability of a recycling plant was revealed for the regional and national scenario. However, it is linked to the existence of enough stock of EoL-RO modules and improvements in the membrane performance characterisation technologies.

Acknowledges

The authors acknowledge the financial support to the LIFE program for funding the TRANSFOMEM project (LIFE13 ENV/ES/000751), to the project INREMEM-CTM2015-65348-C2-1-R (MINECO/FEDER) and the project INREMEM 2.0-RTI2018-096042-B-C21 (MCIU/AEI/FEDER, UE) and the Madrid Community.

Appendix A. Online supplementary data

See supplementary material online

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Abbreviations

A: Area

AEDYR: Spanish Association of desalination water reusing

AV: Avoided transport to the landfill

BW: Brackish Water

CAPEX: Capital Expenditure

CE: Circular Economy

Com: Strategy where both BW and SW EoL-RO modules are collected

DC: Modules discarded during the characterisation

DW: Modules discarded by weight

EoL: End-of-life

EPR: Extended producer responsibility

FU: Functional unit

GIS: Geographic Information Systems

GWP: Climate change

HT, nc: Human toxicity, non-cancer effects

L: Permeability

LCA: Life Cycle Assessment

LCI: Life Cycle inventory

MCE: Multi-criteria evaluation

ME: Marine eutrophication

N: Nitrogen

NF: Nanofiltration

OPEX: Operational Expenditure

PM: Particulate matter/Respiratory inorganics

PRO: Producer responsibility organisation

PVDF: Polyvinylidene fluoride

RD, f+m: Resource depletion, mineral, fossils and renewables

RL: Reverse Logistics

RO: Reverse osmosis

SL: Service Life

SLR: Service Life Ratio

SW: Sea Water

SWS: Segura’s watershed

UF: Ultrafiltration

USA: United States of America

WEEE: Waste electric and electronic equipment

WFD: Waste Framework Directive (European Directive on Waste (2008/98/EC)