UNIVERSIDADE FEDERAL DE MINAS GERAIS
Escola de Engenharia
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Luiza de Barros Grossi

DISPOSURE OF END OF LIFE REVERSE OSMOSIS MEMBRANES IN BRAZIL: WASTE ESTIMATION, LIFE-CYCLE ASSESSMENT AND MANAGEMENT ALTERNATIVES

Belo Horizonte
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Orientadora: Míriam Cristina Santos Amaral Moravia

Coorientadora: Lisête Celina Lange

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RESUMO

O aumento expressivo no uso da osmose inversa como alternativa para dessalinização fez surgir uma preocupação com a quantidade de resíduo gerado ao final do processo. Isso porque a vida útil dos módulos de membranas de osmose inversa é limitada, e após o tempo de uso, eles são dispostos em aterros sanitários. No entanto, tendo em vista preceitos de economia circular e melhor aproveitamento de recursos, é importante avaliar outras alternativas de destinação para este resíduo. Neste sentido, esta tese buscou estimar e prever a quantidade de elementos de osmose inversa descartados no Brasil e seus impactos ambientais, econômicos e sociais, além de propor um caminho de transição para melhoria da gestão deste resíduo. Para a primeira etapa de estimativa e previsão, foram utilizados, respectivamente, os modelos de intervalo de tempo (*time-step*) e ARIMA. Com a metodologia proposta foi possível estimar uma geração de 900 toneladas entre 2016 e 2019, e prever uma geração de 1.800 toneladas de resíduo até 2024. Além disso, observou-se também que a taxa de disposição de membranas no Brasil (em elementos / m^3.dia) é o dobro do reportado para Espanha e Austrália. Para a avaliação dos potenciais impactos ambientais, foi realizada uma análise de ciclo de vida (ACV) utilizando a base de dados *Ecoinvent 3.8 Cut-Off* e o programa *Open LCA*. Foi possível observar que a reutilização do módulo após a limpeza e sua conversão para a membrana de ultrafiltração foram as opções mais ambientalmente amigáveis, que contribuem inclusive reduzindo o impacto ambiental da produção das membranas. Adicionalmente, foi realizada também uma avaliação econômica e social. Para a primeira, utilizou-se os custos estimados na literatura para cada uma das opções de disposição e, para a segunda, três opções de disposição foram categorizadas de acordo com o proposto no *Guideline* do Programa Ambiental das Nações Unidas (2009). No aspecto econômico, a reciclagem das partes da membrana demonstrou o menor custo (U$ 0,36/módulo reciclado), menor ainda do que a disposição em aterro. Para os aspectos sociais, reúso e conversão obtiveram melhores resultados, sendo a reciclagem a terceira melhor colocada na avaliação. A diferença se deu principalmente pela expectativa de geração de trabalhos formais no reuso/conversão. Em termos de propostas de transição, foi realizada uma abordagem multinível para caracterizar o cenário atual de governança deste resíduo. Ainda, as opções de destinação foram caracterizadas pelo seu grau de disrupção quando comparadas à disposição em aterro sanitário. Foi observado que o cenário atual conta com poucos atores e que para uma transição para um cenário mais sustentável – no qual as membranas são transformadas para uma segunda vida ou têm seus materiais reciclados (sem disposição direta em aterro), faz-se necessário desenvolver atores-chaves do processo, como empresas capazes de realizar o tratamento necessário para adequação dos elementos. Por fim, durante a transição, conflitos com empresas que realizam a destinação em aterros e até mesmo com fabricantes de membranas têm potencial de existir, o que reforça a necessidade de elaboração de um plano de negócios que busque fomentar parcerias com estes atores.

Palavras-chave: Descarte de membranas. Osmose inversa. Análise de ciclo de vida. Avaliação socioeconômica. Gerenciamento de resíduos.

ABSTRACT

Desalination has become an integral part of water management worldwide. It is also growing in Brazil and its utilization causes environmental impacts. Among them, the disposure of end-of-life reverse osmosis (EoL-RO) elements can be highlighted, since the membrane lifespan is limited and, after use, it is normally disposed of in sanitary landfills. However, to bring desalination closer to sustainable principles, it is important to assess the disposition option for this waste, given its potentialities. To this, this thesis aimed at quantifying and forecasting the volume of EoL-RO membranes expected in Brazil, their associated environmental and socio economic impacts and to propose transition pathways towards more sustainable scenarios for waste management. To the first step of estimation and forecasting, the time-step and ARIMA models were applied, respectively. It was possible to estimate a generation of 900 tons of EoL-RO elements between 2016-2019 and predict a generation of 1,800 tons until 2024. It was also observed that the membrane disposition rate in Brazil (elements / m^3.day) is as double as what is reported to Spain and Australia. To assess the potential environmental impacts, a life-cycle assessment (LCA) was performed using Ecoinvent 3.8 Cut-Off and Open LCA. It was possible to notice that reuse and chemical conversion to the ultrafiltration (UF) membrane presented the most environmentally friendly option, reducing the overall impact of the membrane life-cycle. In addition, a socio economic evaluation was also performed. To the economic aspect, estimated costs in the literature were used to compare the EoL disposition options. To the social aspects, reuse, chemical conversion to the UF membrane and recycling of parts were classified according to the United Nations Guideline. For economics, recycling of parts presented the lowest cost (U$ 0.36/element), even lower than landfill disposure. For social aspects, reuse and chemical conversion were better classified. As for transition pathways towards more sustainable scenarios, the multi-level perspective (MLP) was used to characterize the current waste management scenario. Additionally, the EoL disposition options were classified according to their disruption degree from the standard solution (sanitary landfill). It was observed that the current disposure scenario is composed by few actors and that a transition towards more sustainable scenarios will increase complexity within the institutional arrangement. To this, key-actors need to be developed so they can enter the process, which might lead to a very disruptive path. Finally, during transition, few potential conflicts were highlighted, such as the eventual resistance from landfill disposers to end their contracts. Hence, the suggestion is that entering actors, such as companies that perform adjustments in the elements so they can have a second life, come up with a business plan that strengthens partnerships rather than be seen as competition.

Keywords: End-of-life reverse osmosis elements. Life-cycle assessment. Socioeconomic analysis. Waste management.

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS, ACRONYMS AND SYMBOLS

ABRELPE – Brazilian Association of Public Cleaning and Special Waste Companies

ACF - Autocorrelation Function

AD – Anderson-Darling test

AIC - Akaike's Information Criterion

ARIMA - Auto Regressive Integrated Moving Average

ARMA - Auto Regressive Moving Average

CAGR – Compound Annual Growth Rate

CE – Circular Economy

CNI - Industry National Confederation

ED – Electrodialysis

EIO – Environmental input-output

EoL – End-of-Life

FR_{di} – Flow rate estimated for a given diameter interval

GPD – Gallons per day

ΔIC_{ind} – Gradient of industrial installed capacity between two subsequent years

IE – Industrial Ecology

IM_{di} – Number of imported membranes at the given interval

IM_{ind} – Total of membranes imported for industrial use

INPI – National Institute of Industrial Research

IPCA – Consumer Price Index

IPCC – Intergovernmental Panel on Climate Change

IPEA – Economic Applied Research Institute

IS – Industrial Symbiosis

IOA – Input-Output Analysis

LCA – Life-Cycle Assessment

LCIA – Life-Cycle Impact Assessment

MED – Multiple-effect Desalination

MF - Microfiltration

MFA – Material Flow Analysis

MLP – Multi Layer Perspective

MMA – Ministry of Environment

MNCMR – National Movement of the Recyclable Materials Collectors

MSF – Multi-stage Flash Desalination

MVP – Minimum Viable Product

NF – Nanofiltration

NGO – Non-governmental organization

PACF – Partial Autocorrelation Function

PAD – *Programa Água Doce*

PET – Polyester

PNRS – Solid Waste National Police

POC – Proof of Concept

RO – Reverse Osmosis

S_{di} – Stock for a given diameter interval

S-LCA – Social Life-cycle assessment

SWRO – Seawater Reverse Osmosis

TFC – Thin Film Composite

UF – Ultrafiltration

UMS – Urban metabolism studies

UN – United Nations

UNEP – Uniter Nations Environmental Program

SETAC - Society of Environmental Toxicology and Chemistry

W_{di} – Unitary weight estimated for a given diameter interval

TABLE OF CONTENTS

CHAPTER 1

INTRODUCTION

1.1 Background and justification

Recognizing the ecological limits of the planet is fundamental. Resources are not finite, and therefore we should walk away from the standard "take – make – consume – dispose" linear model. This change in paradigm is leading society towards understanding processes from a system's perspective, aiming at bringing sustainable practices to industrial processes. By promoting studies that think of closed-the-loop strategies, it might be possible to improve the efficiency of resources management and use, aiming at achieving a better balance between society, environment and economy.

Amid the industrial processes, desalination is increasingly becoming an integral part of water management strategies of several countries worldwide. This can be attributed to the rapid growth of population, water pollution, continuing industrialization, and poor resources management, that is expected to leave half of the urban population living in regions with severe water stress by 2050 (He et a.l, 2021). Among the technologies that can be used to desalination, reverse osmosis (RO) is the most preferable one (Cherif and Belhadj, 2018). This is due to the fact that RO presents high selectivity and flow rate while demanding a relatively small construction area. As a result, projections indicate that RO desalination is growing at a Compound Annual Growth Rate (CAGR) of 11.31%, forecasted for 2022-2026 (GlobeNewswire, 2022). In terms of water supply, Gao et al. (2017) estimate that around 1.76 billion people will use desalted water by 2050.

Despite not being highlighted as a water scarce region, the water uneven distribution is impacting the adoption of reverse osmosis in Brazil. According to Markets and Markets (2018), the country is most likely to follow the increasing tendency of using RO. Among the main RO projects already implemented or foreseen for Brazil in the next couple of years, both local and industrial ventures can be highlighted. For the first, the Brazilian government has been increasingly putting efforts towards decentralized brackish water treatment in semi-arid regions using RO. The Programa Água Doce (PAD), as it is called, intends to guarantee permanent access to potable water through the sustainable use of groundwater, incorporating environmental and social care in the management of desalination systems. Nowadays, PAD has implemented 864 desalination systems, supplying water for 214,000 people (Brasil, 2021a). In addition to decentralized systems, centralized desalination plants are also increasing in the country. For the next years, it is expected the construction of a seawater RO (SWRO) plant in Fortaleza - CE with a capacity of 86,400 m^3/day, and in Tubarão – ES, a 12,000 m^3/day plant is starting to be comissioned by ArcelorMittal (Vialli, 2018; ArcelorMittal, 2022). Furthermore, desalination plants for other industrial processes are a reality in Brazil, and Petrobras alone possesses an installed capacity of 988,000 m^3/day (Eke et al., 2020).

Even with its robustness, as during any other processes, RO membranes are subjected to adverse conditions and natural wear that hinders their utilization. To mitigate such wear and increase the membrane lifespan, it is possible to invest in proper operating conditions, adjust feed pretreatment, and promote adequate cleaning cycles (Li and Elimelech, 2004; Guo et al., 2015). However, several authors report that the membrane lifespan is still kept between 5 to 10 years (Senán-Salinas et al., 2021).

Combining RO expansion, limited membrane lifespan and a linear concept of production, the disposure of RO membranes is increasing worldwide. According to Senán-Salinas et al. (2019), the desalination sector will be responsible to discard 2 million RO elements until 2025. Considering the average weight of 13 – 15 kg per element (Lawler et al., 2015), around 30,000 tonnes are expected to be discarded by the aforementioned year. Due to their thin film composition that consists of polymeric layers, end-of-life (EoL) RO elements are classified as inert residue and, given the waste hierarchy, other options that increase the materials lifetime and take better advantage of the resources invested should be investigated. Among the alternatives, energy recovery, recycling and conversion to other types of membranes are already taking place worlwide (MemRe, 2022; Membrane Services, 2022, WaterSurplus, 2022). However, most of EoL-RO membranes are still disposed of in landfills especially in countries where these initiatives are not yet well developed.

By assessing waste management alternatives, it might be possible to promote sustainable practices within the desalination sector. In this sense, Lawler et al. (2015) assessed the associated impacts of six different destinations: disposure in landfills, incineration, use in syngas fabrication, co processing, material recycling, and direct reuse / chemical conversion to UF membranes. To this, the authors performed a comparative LCA for membrane manufacturing and the aforementioned end-of-life options. They did not find any significant difference between the manufacturing of 8” and 16” elements. The authors also observed that, as expected, direct reuse is the most environmentally friendly destination. However, they highlighted the impact of transportation on the viability of recycling in the Australian context (Lawler et al., 2015). Similar findings regarding transportation were reported by Senán Salinas et al. (2020, 2021) in Spain. Additionally, they also observed that the economic feasibility of recycling depends on a proper characterization of the residue.

These works are undoubtedly valuable contributions to studying waste management in the desalination sector. However, there are crucial regional differences that ask for a more detailed investigation of waste production that should be taken into consideration for other countries. The first of them relies on the maturity of national markets to control and invest in proper operating conditions for RO plants. This impacts directly on the membrane lifetime, which can result in misleading results towards waste estimation. Secondly, the economic and environmental impacts from an activity depend upon local factors, and in Brazil, recycling is also directly linked to social aspects. In addition, institutional frameworks such as regulations also play a major role in waste management, and should be taken into consideration when assessing impacts and potentialities.

In this sense, Pauliuk el al. (2017) highlight that to properly estimate the environmental and economic impacts of waste and the contribution of byproducts to sustainable development, it is necessary to extrapolate studies to distinct socioeconomic realities. In addition, it is important to understand how governance is made to propose meaningful transition pathways towards more sustainable scenarios. Hence, considering the importance of waste assessment for designing meaningful policies, regulations, and strategies to deal with the issue, this thesis aimed at providing an analysis of EoL-RO membrane disposition in Brazil. To this, this thesis presented the following hypotheses, that were tested in each of its three main Chapters (3,4, and 5):

I. There is a meaningful EoL-RO elements waste generation in Brazil that has been neglected so far;

II. Direct reuse after cleaning and chemical conversion to the UF membrane represent the most promising solution from an environmental, economic and social standpoint for EoL-RO elements;
III. The interaction of actors that comprise the socio-technical arrangement of managing EoL-RO membranes is still not well registered and its characterization is crucial to propose meaningful transition pathways towards sustainable management.

Hopefully, the overall results of this work will provide opportunities towards better understanding EoL-RO elements' circumstances and support further strategies for dealing with this waste.

1.2 Research goals

1.2.1 Main goal

This thesis aimed at integrating estimation models, life-cycle assessment and transition pathways to evaluate the environmental, economic and social impact from the disposure of end-of-life reverse osmosis membranes in Brazil.

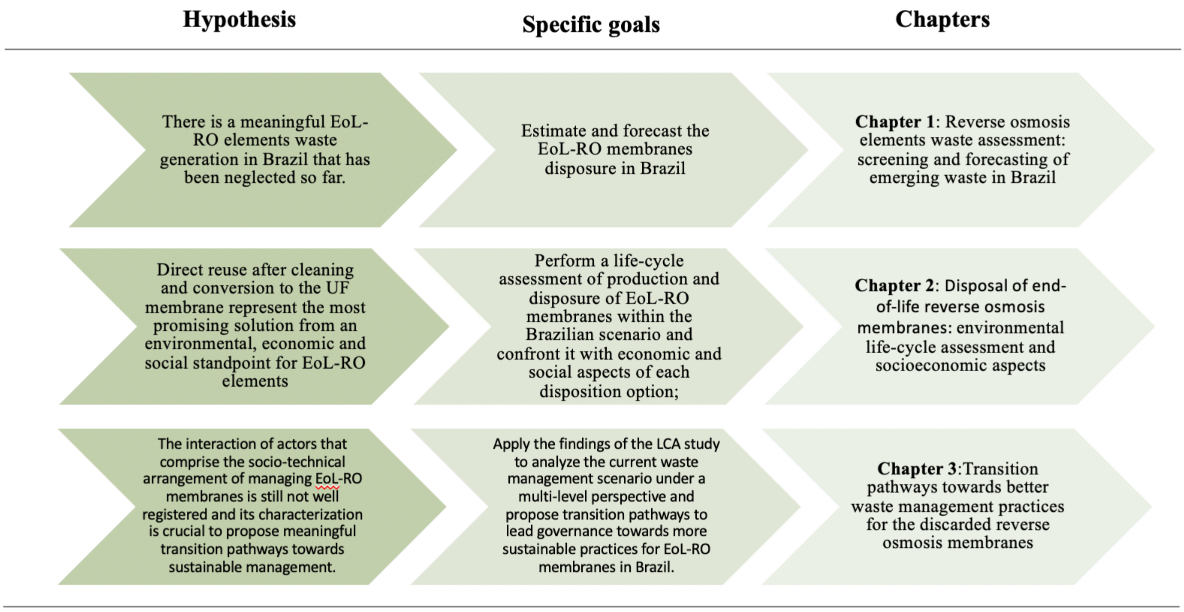
1.2.2 Specific goals

This thesis presented three specific goals:

- Estimate and forecast the EoL-RO membranes disposure in Brazil;
- Perform a life-cycle assessment of production and disposure of EoL-RO membranes within the Brazilian scenario and confront it with economic and social aspects of each disposition option;
- Apply the findings of the LCA study to analyze the current waste management scenario under a multi-level perspective and propose transition pathways to lead governance towards more sustainable practices for EoL-RO membranes in Brazil.

Each of the hypotheses stated in Section 1.1 were answered by one specific chapter, as summarized in Figure 1.1.

Figure 1.1 – Thesis hypotheses and chapters



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1.3 Document structure

This PhD thesis is divided into 6 chapters: Chapter 1 is an introduction and contains a theme presentation, objectives and the document structure; Chapter 2 presents the literature review; Chapter 3 addresses the estimation and forecast of the number of reverse osmosis membranes disposed in Brazil; Chapter 4 comprehends the LCA, economic and social aspects of each of the disposition options; Chapter 5 analyzes the governance scenario and proposes transition pathways towards more sustainable practices for EoL-RO membranes waste management. Finally, Chapter 6 contains the final considerations and suggestions for future work.

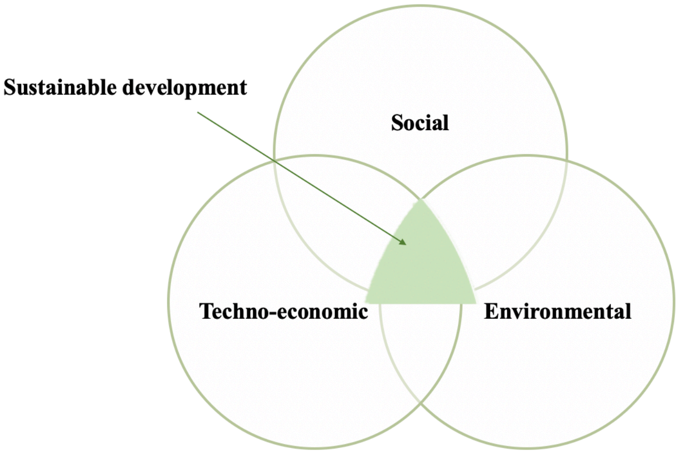
CHAPTER 2

LITERATURE REVIEW

2.1 Sustainability and Industrial Ecology

According to Jackson (2010), sustainability is the art of living well, within the ecological limits of the planet. To Jackson (2010), living well can be interpreted in two senses: (i) living a life with decent comfort, security and dignity and (ii) not living it at the expense of others. By analyzing such statements, it becomes clear that sustainability is not limited to environmental aspects. It is a transdisciplinary subject that comprises at least environmental, economic and social aspects, as shown in Figure 2.1 (Clift and Druckman, 2016).

Figure 2.1 - Sustainability and sustainable development



Source: Adapted from Clift and Druckman (2016)

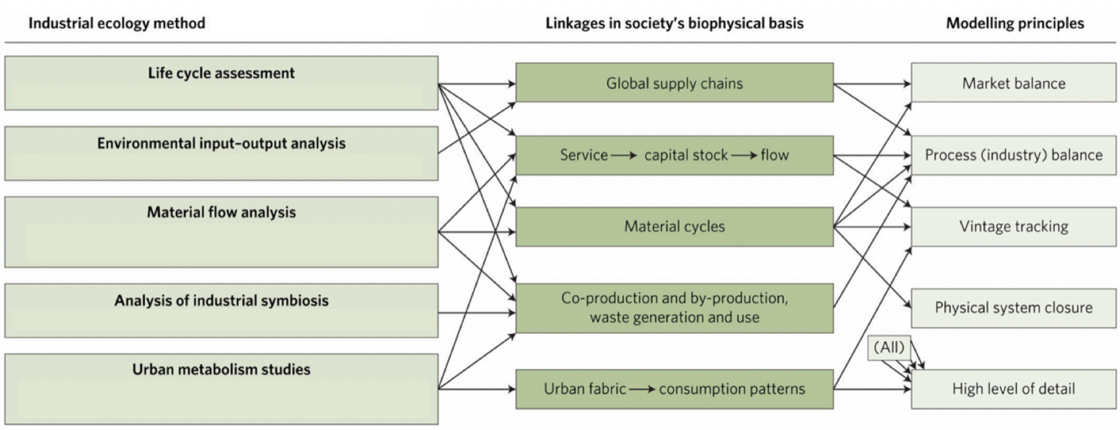
Given its complexity, to propose meaningful changes that move us towards a sustainable society might be a challenge. To address these questions, studies regarding sustainability are currently gathered under the field of industrial ecology (IE). According to White (1994), this is a field *that studies the flows of materials and energy in industrial and consumer activities, the effects of these flows on the environment, and the influences of economic, political, regulatory and social factors on the flow, use and transformation of resources*. The term brings an analogy between industrial and natural ecosystems, suggesting that nature should be used as inspiration when designing more sustainable industrial processes (Ehrenfeld, 1997).

2.1.1 Industrial Ecology tools

To assist decision-making stakeholders that pursue sustainability, several industrial ecology tools were developed. Their scope is to quantitatively analyze specific links in the biophysical basis of the society (Pauliuk et al., 2016). By bio, one can understand the transformation of natural resources into materials and products (Fischer-Kowalski and Weisz, 1999), and by physical services related to thermal comfort, mobility and their associated emissions (Pauliuk et al., 2016).

As determinants for sustainability, IE researchers highlight the following linkages: global supply chains and their impact (Hellweg et al., 2014; Wiedmann et al., 2015); the relationship between service, capital stock and flow (Pauliuk et al., 2014); material cycles (Müller et al., 2006; Graedel et al., 2013), co-production, by-production and waste processing (Lenzen et al., 2014); and the link between urban fabric and consumption patterns (Kennedy et al., 2015). In addition, Graedel and Lifset (2016) highlights five main tools for decision supporting within the industrial ecology context: Life-cycle Analysis (LCA), Environmental input-output (EIO), Material flow analysis (MFA), Industrial symbiosis (IS) and Urban metabolism studies (UMS). These tools can be used to micro (company level), meso (industrial parks) and macro scale (city, country, regional, global-level), depending on the system boundaries in which they are applied (Al-Thani and Al-Ansari, 2021). Their linkage with the biophysical basis can be found in Figure 2.2 and a brief overview of IE main methods is presented in the following sections.

Figure 2.2 - Main industrial ecology descriptive and assessment methods



Source: Adapted from Pauliuk et al. (2017)

2.1.1.1 Input – output analysis (IOA)

Input – output analysis intends to quantify the transactions that occur between different industrial sectors (Greadel and Liftset, 2016). It was first proposed to economic studies by Leontief, but model extensions use IOA to analyze social and environmental aspects (Nagashima et al., 2017; Corona et al., 2016). In IOA, input-output tables are used to organize and structure data. The tables describe in detail the structure with information about output, intermediate and final demand (Heihsel et al., 2019).

Extensions towards environmental data and enhancements in the methodology have led IOA to present meaningful contributions to the field of industrial ecology. They are used, for example, to analyze waste and management strategies (Ruiz-Peñalver et al., 2019; Li et al., 2019) and in combination with other methods to assess the water-energy-food nexus (Feng et al., 2019; Deng et al., 2020).

2.1.1.2 Industrial Symbiosis

The principle of industrial symbiosis is to promote, in the industrial context, the association of different processes that can build a relationship that is beneficial to all of them. According to Chertow (2000), it *engages traditionally separate entities in a collective approach to competitive advantage involving physical exchange of materials, energy, water, and byproducts.*

It is very common to associate industrial symbiosis with eco-industrial parks, where sharing resources and knowledge often leads to social, environmental and economic gains (Liu et al., 2018). An example of a successful association happens in Kalundborg, Denmark. Several studies were performed and the evaluation of its impact in the environment, society and economy helped to validate methods and structures. This allowed industrial symbiosis practices to spread worldwide, and such initiatives can be found in China, Thailand, Morocco and Nigeria (Neves et al., 2020).

2.1.1.3 Urban metabolism studies

Worldwide, more than 55 % of the population live in urban areas (Our World in Data, 2020). This agglomeration in a built environment implies several environmental, social and economic changes in the natural environment, which are often complex. Hence, to better understand these scenarios, urban metabolism studies aim to analyze the city by taking into consideration the continuous exchange of matter and energy between the environment and its inhabitants (Kennedy, 2016). From the industrial ecology perspective, it seeks to understand how cities can achieve a more sustainable functioning. By analyzing the literature, Céspedes and Morales – Pinzón (2016) concluded that tangible dimensions of urban metabolism, such as the flow of materials and energy have been studied to a bigger extent, while intangible aspects remained neglected. Hence, it is important to address these gaps in order to provide a more holistic understanding of complex organisms such as cities.

2.1.1.4 Material flow analysis

Material Flow Analysis is a methodology that intends to understand the flow of materials through interlinked processes (Allesch and Brunner, 2016). It is based on the mass conservation law, focusing on sources, sink and pathways of materials. Since it provides a comprehensive overview of the material system and its interaction with the surroundings, it is commonly applied to several areas such as resource, environmental, and waste management (Brunner and Rechberger, 2016). As an attempt to standardize MFA, the Practical Handbook of MFA (Brunner and Rechberger, 2016) presented a list of terminologies, as shown in Table 2.1.

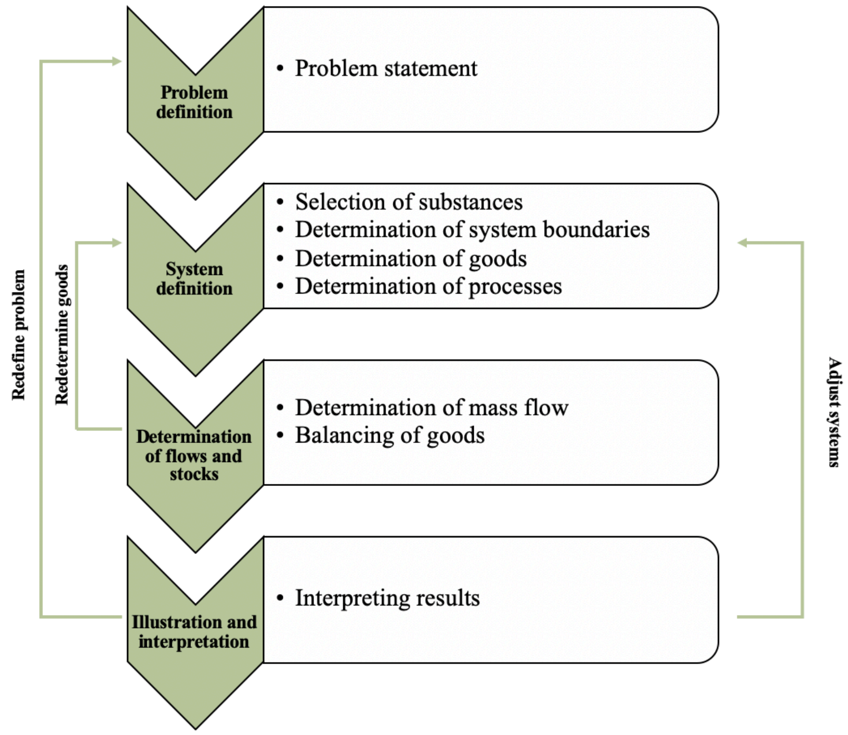
Table 2.1 - Terminology of MFA

Term	Description
Substances	Any element or compound made of uniform units (atoms or molecules)
Goods	Entities that possess economic value (positive or negative) and are made of one or more substances
Materials	Umbrella term that designates both substances and goods
Processes	Transformation, transport or storage of materials
Flows	Mass flow rate
Transfer coefficients	Describe the partitioning of materials in a process
System	Limited by boundaries in time and space. It is the actual object of study

Source: Adapted from from Brunner and Rechberger (2016)

MFA is an iterative process. It starts by defining the relevant system boundaries, processes and goods to be evaluated. Then, mass flows are calculated and allowed to reach substance flow and estimate uncertainty. Given the difficulty of the estimated date, the analysis starts with rough information and can be refined during the process, which originates from the iteration process. An overview of stages can be seen in Figure 2.3.

Figure 2.3 - Procedures for MFA



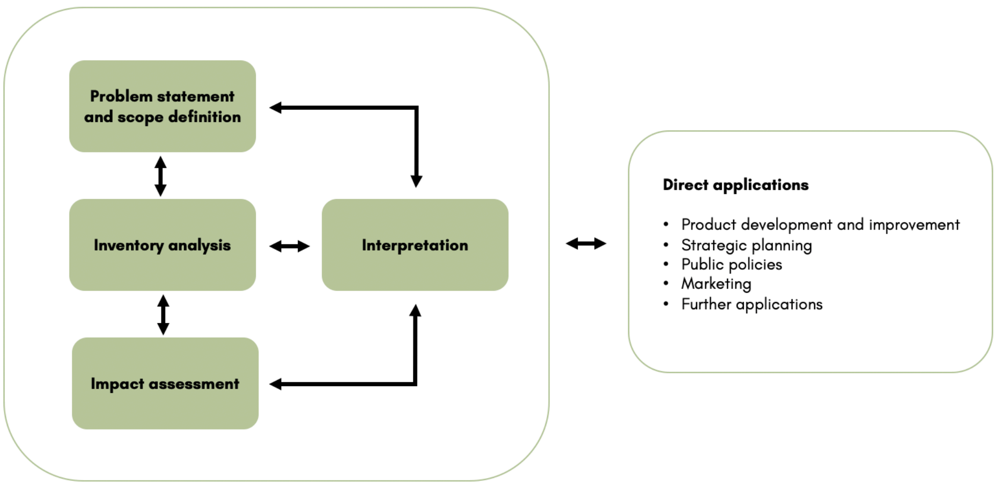
Source: Adapted from Brunner and Rechberger (2016)

2.1.1.5 Life-cycle assessment

Life-cycle assessment seeks to assess the environmental impact of a product, process or activity during its entire lifetime (Chau, Leung and Ng, 2015). To this, it quantifies the amount of materials and energy uses and releases to the environment. The first LCA studies are dated from the 1960s. Between 1970 – 1990, LCA studies were largely applied, but no common framework nor methodology was used. As a consequence, findings diverged even when the object of study was the same.

As a way to make LCA studies comparable and the results more meaningful, the International Organization for Standardization (ISO) published LCA Standards (ISO 14040, 2006; ISO 14044, 2006). They focus on technical and organizational aspects of LCA, describing principles and frameworks to LCA stages, that include but are not restricted to: definition of the goal and scope, life-cycle inventory, life-cycle impact assessment (LCIA), life-cycle interpretation, reporting, and interpreting results. In other words, after defining the object of study, it is necessary to define the spatial extent (boundaries) and the time period of the analysis. Secondly, an inventory of relevant material and energy inputs and releases is put together and an evaluation of potential impacts is developed. A summary of the main phases is shown in Figure 2.4.

Figure 2.4 - LCA main phases



Source: Adapted from ISO 14040 (ABNT, 2009)

An important step in LCIA is to define which method will be employed to perform comparisons. According to Hauschild et al. (2012), the potential impact of each inventory flow is quantitatively calculated using a characterization model. The overall impact calculation is then performed taking into account the units of measure of each impact category, for example, the emission of CO_2 equivalent of each of the flows. Another point that should be highlighted is that they can be classified according to their categories level: midpoint and endpoint. The main difference between these two classifications is that midpoint categories focus on single environmental problems, such as ozone depletion and acidification. Endpoint categories, on the other hand, show the impact on higher aggregation levels, as the effect on human health, for example. As a consequence of the aggregation level, endpoint categories increase uncertainty when compared to midpoint approaches. However, they also simplify the interpretation of LCA and might help subsidize decision-making processes.

Furthermore, to evaluate uncertainty, Monte Carlo simulation is widely applied to LCA. It consists of running repeated simulations using inputs randomly chosen from a specified range. By using this technique, it is possible to to estimate the possible outcomes of an uncertain event. Using Monte Carlo in LCIA experiments might help diminish eventual biases during the evaluation.

In addition to assessing the environmental impacts, social life-cycle assessment (S-LCA) has attracted increasing attention over the last few years. According to Norris et al. (2014), S-LCA is *a technique for collecting, analyzing and communicating information about the social conditions and impacts associated with production and consumption.* It focuses on negative and positive socio-economic impacts of processes, products and services during its life-cycle (Fausi et al., 2019). In other words, it intends to support decisions by providing insights related to changes in the life of key stakeholders (such as workers, consumers, and society) linked to the assessed product.

In 2009, the United Nations Environment Program (UNEP) and the Society for Environmental Toxicology and Chemistry's (SETAC) launched a Guideline for S-LCA, which aimed at setting a common background for research in the area

(UNEP/SETAC, 2019). The main steps are the same as defined for environmental LCA (E-LCA), differing from the focus given on the product description and impact categories. In addition, the *Methodological Sheets* (UNEP/SETAC, 2013) complements the Guidelines by presenting a hands-on tool for performing the S-LCA.

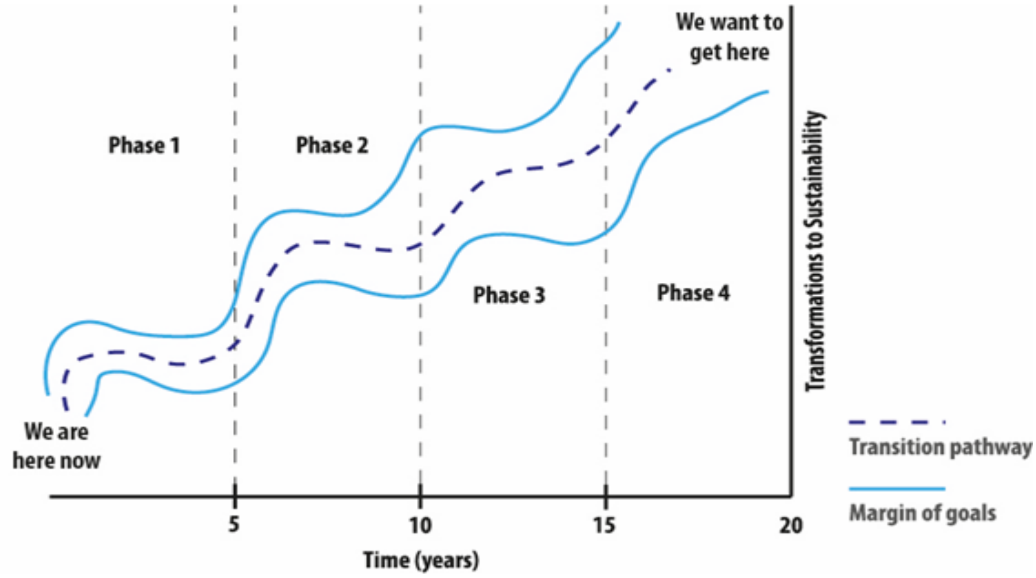
LCA is very helpful for learning about the object of research, the ways it affects the three dimensions of sustainability and to compare alternative approaches to solve a common situation (Lior, 2017). However, it suffers from a number of uncertainties that might arise from data availability and quality, scenario and model uncertainty (Lloyd and Ries, 2007). In the Brazilian context, LCA still presents another challenge to its application. Since the vast majority of LCIA models were developed in European countries, USA, and Canada, countries outside those regions might encounter difficulties in producing LCIA results with specific characteristics (Mendes, Bueno and Ometto, 2013). This is due to the fact that LCA brings intrinsic particularities of the places for which they have been evaluated, and might not correspond to other realities. Hence, as a way to overcome this issue, Piekarski et al. (2012) recommended to Brazil a comparison between the results of different methods to understand whether they converge to the same result or not.

2.1.1.6 Scenario building and IE tools

Transformations are often complex, dynamic, and multidimensional. They have been extensively studied (Streeck and Thelen, 2005; Walker, 2004; Folke, 2006), and among the theorization, the transition theory seeks to understand how niche-level activities result in socio-technical transitions. This is done by analyzing respective niches, social-technical regimes, and landscapes (Rotmans, 2005; Geels and Schot, 2007; Kemp et al., 2007). To this, scenario building can be used as a tool for future-oriented thinking and to assess decision-making by providing different contexts. Scenario building helps to comprehend how stakeholders might interact and the possible consequences of defined actions.

Scenarios can be divided into explorative and quantitative. The first one is defined by Van Notten et al. (2003) as multiple plausible futures described in words, numbers and/or images. Instead of providing a rather linear sense of decision-making processes, it intends to recognize and explore uncertainties of the situation with multiple stakeholders (Van der Sluijs, 2005). However, explorative scenarios by itself are not plans. They should be accompanied by transition pathways (Figure 2.5); that suggests how actors can move from a current position to a desirable future. To this, it is necessary to map what contexts should be taken into consideration and how they might evolve, both due to external factors and internal changes (Vervoot, Helfgott, Lord, 2016).

Figure 2.5 - Transition pathways towards a more sustainable scenario for end-of-life reverse osmosis membranes waste management



Source: Adapted from Vervoot, Helfgott, Lord (2016)

Quantitative scenarios, on the other hand, are based on simulations that can reduce biases from scenarios generated by groups of stakeholders, as well as present counter-intuitive effects not originally imagined by the participants (Schoemaker, 1993; Vervoot, Helfgott, Lord, 2016). However, since models often possess their own assumptions about the system (Vervoot, Helfgott, Lord, 2016), it is important to embrace uncertainty (provided by explorative scenarios) in order to avoid simplistic analysis.

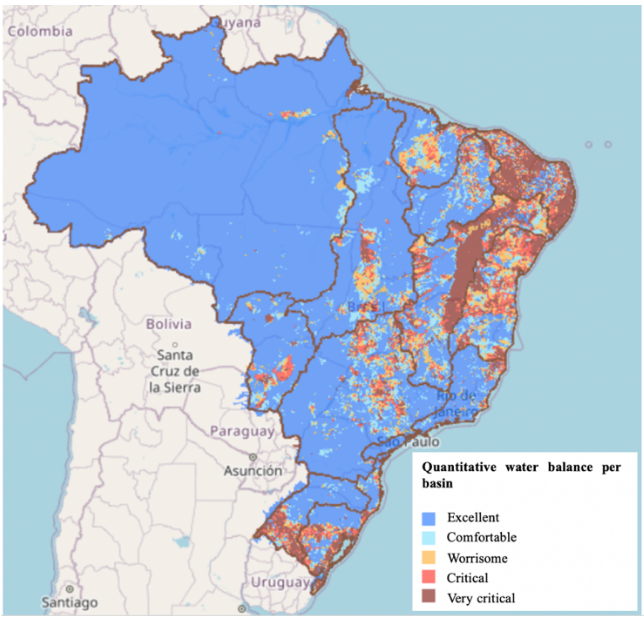
In the context of quantitative scenarios, industry ecology methods and models can be used as tools to study complex industrial systems and help build these scenarios. They provide detailed and quantitative information of material and energy stocks and flow, supply chains, and environmental impacts in a wide spectrum of spatial, temporal and organizational scales (Pauliuk and Hertwich, 2016). By complementing quantitative scenarios sustained by IE tools with explorative scenarios, it might be possible to provide a better forecast for transformation towards a more sustainable reality.

2.2 Water scarcity and desalination

From 3% of fresh water available in the world (97% is saline), only 0.5% is accessible in the form of surface and groundwater (Gude and Nirmalakhandan, 2009). It is important to highlight, however, that according to the United Nations (UN, 2021) there is not a water shortage as such, but individual regions that need to address the challenges provided by the water stress. Still according to the Organization, water scarcity is related to physical shortage or scarcity in the access due to lack of infrastructure or due to the failure of institutions in providing a regular supply (UN, 2021). This indicates that improvements in the water management – which includes the use of adequate technologies to properly treat water – is an important strategy to tackle the challenge of water for all.

Despite not being highlighted in the international scenario as a water-scarce region, the uneven water distribution in Brazil is leading to water conflicts in several regions, as shown in Figure 2.6. In Brazil, 45 % of the population lives in coastal areas, having access to only 3% of Brazil's fresh water (Jacobi, Cibim e Leão, 2015). In addition, it is expected that around 41 million people will suffer from water scarcity in the Brazilian semi-arid region and its surroundings due to an increase in water demand (ANA, 2020). Furthermore, according to the Intergovernmental Panel on Climate Change (IPCC) and the Ministry of Environment (MMA), the semi-arid region might be the most affected by climate change, reducing even more its fresh water availability (IPCC, 2007; MMA, 2007).

Figure 2.6 – Water uneven distribution in Brazil



Source: ANA (2020)

On the other hand, according to Gude (2017), many of the drought-affected communities have access to brackish water or seawater, which make them suitable for desalination. Because of that, desalination has become an integral part of water management strategies of several countries around the world, and as consequence, the global value of the desalination market is estimated to worth USD 32 billion until 2025 (Adroit Market Research, 2020). Still according to the Adroit Market Research (2020), domestic and industrial use are the main drivers for desalination growth.

Historically, desalination has been preferentially performed using changing-phase technologies such as multi-stage flash distillation, multi effect distillation, and thermal vapor compression (Ali et al., 2018). These processes, however, are energy-intensive since they depend on thermal energy mainly obtained from fossil fuels, and exhibit high capital and operational costs (Micale et al., 2009; Youssef et al., 2014). Advances in membrane processes have made them most promising options due to their energy-efficiency (when compared to other techniques), ease of automation and operation, and plant compactness (Qasim et al., 2018; Werber et al., 2016). Among membrane processes, reverse osmosis is the leading technology for water desalination, with a CAGR, estimated around 10 % for the period of 2018 – 2025 (Adroit Market Research, 2020). In this scenario, Brazil was pointed out as an emergent desalination market along with India and China (Market and Markets, 2018).

Reverse osmosis is a pressure-driven-membrane-based process that uses a semipermeable membrane to purify water. Currently, around 69% of desalinated water produced in the world comes from RO plants (Eke et al., 2020), and its market share is growing continuously. Several coastal areas that did not need desalination before are now integrating SWRO to its plans for water supply. This is the case of California, Texas, China, southeast Asia, and parts of Europe (Pacific Institute, 2012; Texas Water Development Board, 2016; Water World, 2013a; Water World, 2013b; Anderson et al., 2008).

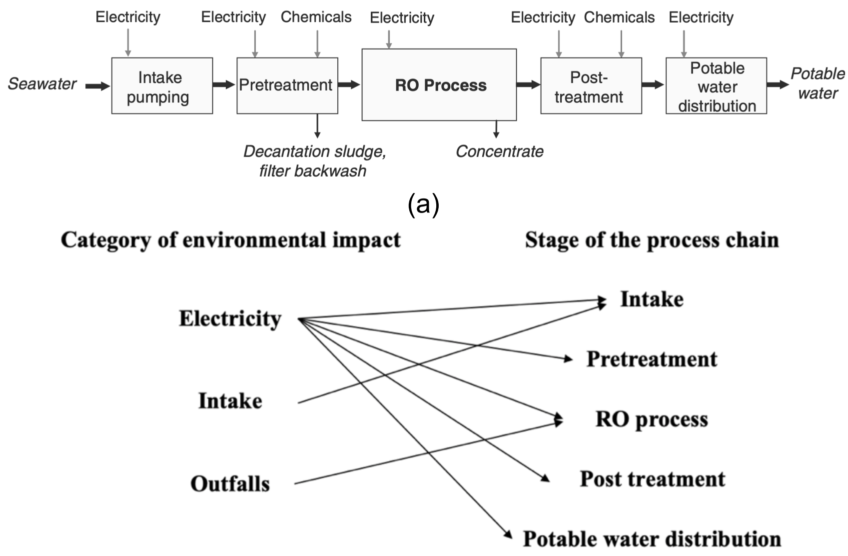
In addition to centralized water treatment facilities, membrane processes also present a great potential to compose decentralized water treatment systems. By treating water locally, several issues related to construction, maintenance, and water transportation are avoided. Such approach is especially interesting in remote areas with lack of infrastructure and no access to fresh water, where decentralized water treatment has been reported as both technical and economical alternatives to deliver potable water (Glueckstern, 1999; Schäfer et al., 2014; Peter-Varbanets et al., 2009). What makes membrane processes particularly interesting to decentralized water treatment are its intrinsic characteristics such as robustness, compactness, and conjugation with renewable energy systems such as those that use the sunlight and the wind power. Such systems have been currently tested under different conditions and approaches, as summarized by Ghaffour et al. (2014).

2.2.1 Environmental impacts associated with SWRO and BWRO facilities

Even though less energy-intensive than other desalination processes, RO still consumes a significant amount of energy and discharges brine and chemicals. These ‘byproducts’ cause negative environmental impacts which may affect marine life, greenhouse and gas emissions (Lattemann and Höpner, 2008), and soil salinity.

To study the reverse osmosis process chain (Figure 2.7a) from an ecological perspective is mandatory to understand and mitigate potential environmental impacts that originate from it. According to Missima and Maliva (2018), the environmental impacts of a seawater SWRO can be divided into three main categories: intake, outfalls, and energy consumption. A relationship of these categories with stages in the RO process chain is shown in Figure 2.7b.

Figure 2.7 – (a) Process chain of a RO desalination plant and (b) Relationship between categories of RO environmental impact and stages of the process



(b)

Source: Adapted from Vince et al. (2008)

Intake refers to the way the water catchment is done, and its environmental impacts are related to impingement and entrainment of marine organisms, construction, and facility operation (Maliva and Missimer, 2012). To environmentally assess intake strategies, Abdulrahman and Mackey (2019) used LCA to compare open intake pretreatment (standard approach) and subsurface intake of two plants located in the Arabian Gulf. They observed that subsurface intake led to a significant reduction in energy consumption, presenting lower environmental impacts across all the assessed categories. The authors also highlighted that the intake strategy needs to be investigated to specific sites, since it impacts on chemical dosage during pretreatment, which significantly contributes to several environmental impact categories.

Regarding outfalls, it is composed mainly by the discharge of brine, which might bring the following environmental impacts: increased salinity in water bodies or localized marine communities close to the brine discharge point, damages from construction, maintenance, and aesthetic issues (Maliva and Missimer, 2012). Brine salinity is considered the main parameter that impacts the marine environment, since it possesses 1.5 – 2 times the feed water salinity (Lattermann, Höpner, 2008; Tarnacki et al., 2012). However, a proper outfall location can decrease its environmental impacts, as shown by Kress, Gertner, and Shoham-Frider (2020). In their study, the authors have monitored the environmental impacts of outfalls from two mega size seawater RO desalination plants in Eastern Mediterranean. They found that the discharge has shown almost no impact on seawater quality, except near the bottom, where excess salinity $\geq$ 1%, ranging in a plume bigger than those initially forecasted by dispersion models. The authors do highlight, however, that 6 years is a short-term study that might not represent a steady-state situation (Kress, Gertner, and Shoham-Frider 2020).

As for energy consumption, the main environmental impact from it arises from greenhouse gas emissions that contribute to global warming. In this sense, several studies have tried to quantify and compare RO gas emissions to other desalination technologies, as performed by Muñoz and Fernández-Alba (2008), Liu et al. (2015), Setiawan et al. (2009), and Goosen et al. (2014). The authors found that with the development of novel membrane materials, RO is continuously improving its energy efficiency. As for other variables that affect environmental impacts related to energy consumption, feed water salinity and the source of energy can be highlighted.

In addition to environmental evaluation of each part of a RO chain, holistic assessments have already been conducted comparing the technology to other alternatives. Garfí et al. (2016), for example, used LCA to compare RO in both point-of-use and at water treatment plant to bottled water (glass and plastic) and conventional treatment. The authors found that water treated by domestic reverse osmosis filters was the most environmentally friendly solution for improving organoleptic characteristics of water, being from 8 – 19 times cheaper than bottled water. When comparing desalination options (MSF, MED, and RO) using

LCA, Raluy, Serra and Uche (2005a) also pointed out that RO is clearly the desalination technology with lower environmental load due to its lower energy requirement when compared to thermal processes. Raluy et al. (2005b) also compared the associated impacts of RO and a big hydraulic project of water catchment in other river basins, taking into account the entire potable water system in their assessment. They found that despite being slightly higher than the hydraulic project, the environmental impacts associated with RO desalination would follow a tendency of decrease in the upcoming years due to the rapid transformation in RO technologies. As a consequence, investing in RO desalination might actually bring environmental gains for water management in Spain.

2.2.2 End-of-life reverse osmosis membranes

Although studied to a lesser extent, reverse osmosis also generates environmental impacts by producing solid waste. During filtration, the accumulation of unwanted materials on the membrane surface or inside its pores reduces the permeate flux and the membrane selectivity. Even applying fouling mitigation actions – such as feed pretreatment, cleaning cycles and proper operational conditions, at some point, membrane replacement is still inevitable (Schäfer, Fane and Waite, 2005; Sanmartino, Khayet and García-Payo, 2017).

Reverse osmosis, as most technologies, follows a linear pattern of ‘take – make – consume – dispose’. RO membranes, for example, are made in central facilities that use fossil fuels, and discarded in landfills after five to ten years of use (Landaburu-Aguirre et al., 2016). Currently, once they reach the end of their life, EoL-RO modules are disposed of in landfills, and classified as inert waste. Seeing the spot that desalination and, most particularly, RO is occupying in terms of water management worldwide, it is mandatory to seek alternatives that mitigate its associated environmental impacts. This evaluation should be carried out to the entire process chain, including the EoL-RO membranes waste management.

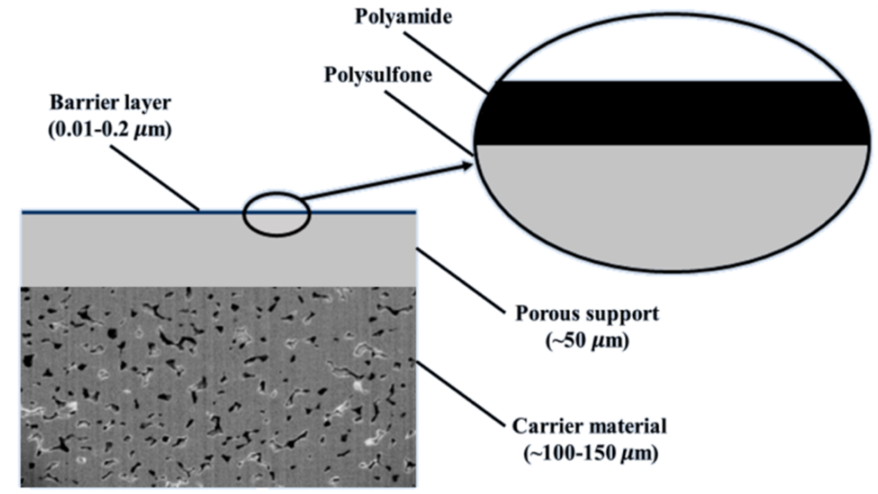
When thinking about a more sustainable approach for waste management, waste prevention through the ‘Reduce, Reuse and Recycle’ (3R) rule has attracted broad attention worldwide and will be further discussed in Section 2.3. In the EoL-

RO membranes management context, the first R, Reduction, can be achieved by (i) developing membrane modules that present bigger permeability, (ii) less propensity to fouling, and (iii) improve operational conditions. By aiming at these three strategies, one can diminish the amount of RO modules used to produce a certain amount of water and increase the membrane lifespan. Such strategies are widely reported in the literature, and have been summarized by Hailemariam et al. (2020) and Qasim et al. (2019).

By Reusing (second R), one can infer the direct use of discarded EoL-RO membranes for other purposes. As an example, membranes that are discarded from highly demanding processes, such as those from pharmaceuticals, can be directly reused in less demanding processes, like wastewater treatment. Reuse is normally recommended over recycling since it is associated with lower environmental impacts (Lawler et al., 2012). Nevertheless, few studies report EoL-RO direct reuse (Prince et al., 2011; Ould et al., 2010). In these two investigations, the authors found that even by decreasing RO membrane selectivity, it was possible to attain results similar to those found for nanofiltration (NF) membranes, with a salt rejection of 96 %. These results have inspired the commercialization of second hand RO membranes worldwide by companies such as *WaterSurplus* (USA), *Perry Process* (UK), and *MemRe* (Germany). Although preferred over recycling, it is important to highlight that direct reuse of RO membranes needs to be properly assessed. To this, one should have access to a detailed report of the membrane's performance during its first lifetime, which includes information regarding feed water, permeability, integrity, and rejection (Pontié et al., 2005; Lawler et al., 2012). By analyzing its previous use, it is possible to develop effective chemical cleaning procedures that will allow its safe reuse to other purposes.

As for the third R, Recycling, it can be interpreted as a reuse after a conversion of the primary material. In the case of RO membranes, what makes recycling a potential alternative is their asymmetric form. Nowadays, RO membranes are made of thin film composite (TFC), consisting of three layers: (i) a polyester web for structural support, (ii) a micro-porous interlayer of polysulfonic polymer, and (iii) an ultra-thin barrier, normally made of aromatic polyamide that actually provides RO selectivity, as shown in Figure 2.8 (Yang et al., 2019; Lee, Arnot, and Mattia, 2011).

Figure 2.8 – TFC structure



Source: Yang et al. (2019)

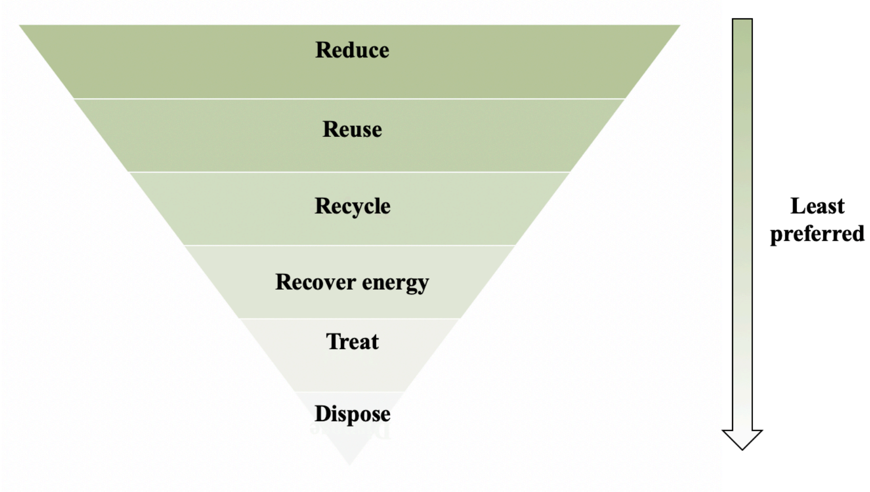
Recycling of EoL-RO membranes consists of removing the upper aromatic layer and exposing the micro-porous polysulfonic one. This removal is normally performed exposing the membrane to strong oxidants, such as sodium hypochlorite, potassium permanganate, and hydrogen peroxide, for example (Coutinho de Paula, Gomes, and Amaral, 2017; García Pacheco et al., 2015). Depending on the extent of the exposure, it is possible to produce structures with permeability and selectivity compared to those found for NF (Morón Lopes et al., 2019), UF, and MF membranes (Coutinho de Paula and Amaral, 2017). The chemical attack can be used for both direct and indirect recycling. In the first case, the chemical oxidant is directly circulated within the module, and the membranes are converted without any physical change in the module structure (Lejarazu-Laranaga et al., 2020). However, when the physical integrity of the EoL-RO membranes is excessively damaged, indirect recycling might be a more attractive option (García-Pacheco et al., 2017). In this case, the module is disassembled and the membrane removed for recycling. This allows the reassembly of the membrane sheets for other purposes and configurations, while also permitting the recovery of the spacer.

EoL-RO membranes have been successfully converted into UF and MF membranes, with both laboratory and pilot scale applications (Coutinho de Paula, Gomes and Amaral, Coutinho de Paula et al., 2018; 2017; García-Pacheco et al., 2018). Other applications that have been recently investigated are the use of the micro-porous structure combined with bioreactor to treat wastewater (Oliveira, 2020) and as support for the development of novel membranes. Lejarazu-Laranaga et al. (2020) used it to fabricate electrodialysis (ED) membranes using phase-inversion. The authors obtained a permselectivity of 87 % when compared to commercial ED membranes. Moradi et al. (2020) also transformed recycled EoL-RO membranes NF membranes by polyelectrolyte layer-by-layer deposition. The fabricated membranes exceeded water permeability and salt rejection of the commercial ones. In addition to the 3R options to waste management, other less noble destinations should also be highlighted: energy recovery, treatment, and disposure.

2.3 Waste management strategies

The waste hierarchy is a preferential order to manage waste that aims to reduce environmental impacts (Figure 2.9). It prioritizes reduce, reuse, recycle over energy recovery treatment and, at last, landfill disposition (Hultman, Corvellec, 2012; Overcash, 2002). In 2015, the European Union defended the role of waste hierarchy as a way to get valuable materials back to economy and help ‘closing the loop’ within the framework of circular economy (European Union Commission, 2015), and in 2016, the strategy was included in the 12th Sustainable Development Goal of the UN 2030 Agenda for Sustainable Development (UN, 2016). In the US, despite not being directly referred to at the Solid Waste Disposal Act of 2002 (US Congress, 2002), it is frequently assumed that the Resource Conservation and Recovery Act outlines the hierarchy (Ewijk and Stegemann, 2016).

Figure 2.9 –Waste hierarchy



Source: Adapted from EPA (2022)

Waste hierarchy is also used in countries outside the US – Europe route. In China, for example, the Environmental Protection Law enacted in 1989 stipulated that *producers should use high resource utilization equipment and process with less pollution, and the waste should be recycled and utilized comprehensively* (Liu et al., 2017). In 2009, the country implemented the Circular Economy Law, that makes reference directly to the waste hierarchy through promoting the 3R rule. In India, several policies were developed between 2015 and 2016 to encourage waste management practices in the scope of circular economy (CE): Hazardous and Other Wastes (Management and Transboundary Movement) Rules, 2015, Plastic Waste Management Rules, 2016, E-Waste Management Rules, 2016, Solid Waste Management Rules, 2016, and Construction and Demolition waste Management Rules, 2016. They were summarized by Priyardarshini and Abhilash (2020). Mentions of waste management hierarchy and circular economy principles are also been addressed by other developing countries (Ngan et al., 2019) and specific considerations regarding Brazil are presented in Section 2.3.2.

Despite being used as a baseline for several waste management guidelines worldwide, the actual implementation of waste hierarchy stumbles upon practical issues. According to Ewijk and Stegemann (2016), waste hierarchy does not necessarily lead to dematerialization, and specific difficulties includes the following: (i) poor policy support at prevention, since it is not under control of waste managers; (ii) lenience towards options at the bottom of the hierarchy and lack of incentives to promote the top options, and (iii) lack of guidance for choosing between the levels and for making trade-offs between waste disposal and other sectors such as energy and transport. Furthermore, the authors argue that policies are often vague or uncertain about the criteria to be followed. For instance, the Scottish government refers to “exhausting” the top priority options without describing what exhausting might mean in practical terms (Scottish Government, 2013). The European Waste Framework Directive, states that the hierarchy should be applied as a priority order. However, it also highlights that the waste policy is expected to favor the practical application (EC, 2008).

In addition to practical issues, Ewijk and Stegemann (2016) also highlight that the waste hierarchy does not always indicate the most environmentally friendly option nor reduces natural resources use. For example, Finnveden et al. (2015) showed that landfilling of solid waste might be more interesting than incineration as transport distances become bigger. Nevertheless, what is interesting about the waste hierarchy is that it stimulates incremental changes within the scope of waste management (Ewijk and Stegemann, 2016). This helps define the direction of changes, which can promote improvements regardless of the starting point.

2.3.1 Application of industrial ecology tools in waste management

Monitoring the progress and outcomes of actions related to CE and waste management is of fundamental importance to their implementation. However, the difficulty of creating common indicators to measure its performance is often highlighted in the literature. In their work, Price and Joseph (2000) stresses that the most widespread indicator for waste hierarchy is recycling rate, since it promotes a straightforward quantification and it is rather simple to quantify when compared to prevention and reuse strategies (Pires and Martinho, 2019). However, the concept of recycling rate has some limitations. At first, it can present definitions rather different amongst countries (Haupt, Vadengo, and Hellweg, 2017). In addition, by defining the recycling rate itself it is not sufficient to define the rate of other disposition options, such as treatment and landfill, since all of them can be high at the same time (Iacovidou et al., 2017).

According to Kalmykova, Sadagopan, and Rosado, 2018, CE is mainly grounded in management of physical resources, eco-efficiency, and waste prevention / management through waste hierarchy. This indicates that monitoring physical flows is of substantial importance to the process. Kalmykova, Sadagopan, and Rosado (2018) highlights that there are three main approaches to address the issue: MFA, IOA, and emergy analysis. Among them, MFA is the most mature method, and might provide meaningful insights regarding environmental impacts when associated with LCA (Goldstein et al., 2013; Lavers et al., 2017). This is due to the that fact that by combining MFA and LCA, it is possible to identify priority areas for waste policy evaluation (Redlingshöfer, Barles, and Wiesz, 2020; Turner, Williams, and Kemp, 2016; Padeyanda et al., 2016).

MFA and LCA have been widely used in the waste management context (Redlingshöfer, Barles, and Wiesz, 2020). In recent review about e-waste, for example, Islam and Huda (2019) reported some interesting features regarding the subject, in which the concentration of work dedicated to estimation of e-waste in the literature review, the need for standardization of national data to perform MFA, and the good outcomes of integrating MFA and LCA can be highlighted. Allesh and Brunner (2015) also performed a literature review about MFA on a series of waste types, such as food, plastics, and solid waste. The authors also highlight the use of the MFA analysis as an inventory for LCA. Regarding the use of these tools, it is also reported that the main bottleneck consists in difficulty in obtaining accurate data (Laurent et al., 2014a, Laurent et al., 2014b). This leads to the estimation of flows, that need therefore to be analyzed in terms of their uncertainty, in order present meaningful answers.

2.3.2 Brazilian Scenario

In Brazil, the main regulatory framework for waste management is the *Política Nacional de Resíduos Sólidos - PNRS* (Solid Waste Nacional Policy). It was enacted by the Law nº 12.305/2010 and it is also based on the waste hierarchy aforementioned. The *PNRS* presents guidelines that aim at encouraging a less linear structure, including, for example, mandatory reverse logistics for specific products and waste collection for recycling (Brasil, 2010a).

A point that deserves attention in the PNRS scope is the statement that the responsibility for the product life-cycle is shared amongst producers and consumers. This is a major difference between Brazilian and international regulation, since in Europe, for example, the responsibility lies upon the producer, and directly affects waste management (Azevedo, 2015). Another particularity in Brazil that is actually shared among other developing countries is the substantial amount of informal collection and recycling (IPEA, 2016; Kalmykova, Sadagopan, and Rosado, 2018). Among recycling materials in Brazil, aluminum cans deserve attention due to their high recycling rate, estimated by the *Associação Brasileira de Alumínio* (Brazilian Aluminum Association) in 97.6% in 2019, followed by paper recycling, estimated in 63,4 % (Associação Brasileira dos Aparistas, 2019). The commitment to recycling in Brazil is directly linked to the aggregate value of residues, which leads to poorer recycling rates to materials such as glass.

The organizational structure of recycling in Brazil is pulverized and arises from self-organization. Born from a social-economical demand for money, the major part of it is performed informally. In 2016, the *Instituto Econômico de Pesquisa Aplicada* – IPEA (Applied Research Institute) launched a book that describes the situation of waste collectors in Brazil. It brings a review of social, economic and environmental aspects of recycling and collectors, where the *Movimento Nacional dos Catadores de Materiais Recicláveis* (National Movement of Recyclable Materials) estimated that 800,000 collectors in Brazil (IPEA, 2016). In fact, Wirth and Oliveira (2016) highlights that, on the contrary of other countries from which Brazil took inspiration for its own waste policies, collectors and association of collects are key stakeholders for waste management. This importance is recognized at Law nº 12.305/2010, that states that the work from associations and collectors should be prioritized when hiring services for public cleaning (Brasil, 2010a).

This particularity in Brazil's waste management scenario brings challenges to the development of studies in the area. First, the lack of centralized information regarding waste management strategies difficult meaningful data acquisition that could subsidize studies and policies. This can be seen not only in primary data but also in the incipience of national inventories. As a consequence, several national papers that apply MFA and LCA, for example, utilize international databases. However, even being applicable to the national context it is very important to consider cultural and environmental differences when performing such studies. Another challenge that needs to be addressed is the intrinsic relationship between recycling and social economic aspects in Brazil. This makes mandatory an extension of environmental assessment impacts, that must take these other dimensions into the evaluation. This last challenge actually goes into accordance with what is currently being discussed in the industrial ecology context as a way to integrate areas and promote more holistic perspectives that might help promote meaningful changes towards sustainable practices.

CHAPTER 3

REVERSE OSMOSIS ELEMENTS WASTE ASSESSMENT: SCREENING AND FORECASTING OF EMERGING WASTE IN BRAZIL

3.1 Introduction

From 3 % of fresh water available in the world, only 0.5 % is accessible in the form of surface and groundwater (Dudgeon, 2020). The rapid growth of population, water pollution, continuing industrialization, and poor resources management is expected to leave half of the people living in regions with severe water stress by 2030 (UN, 2021). According to the United Nations (2021), water scarcity is related to physical shortage or scarcity in the access due to lack of infrastructure or due to the failure of institutions in providing a regular supply. This indicates that improvements in the water management – which includes the use of adequate technologies to properly treat water – is an important strategy to tackle the challenge of water for all.

According to Gude (2017), many of the drought-affected communities have access to brackish water or seawater, which make them suitable for desalination. Because of that, desalination has become an integral part of water management strategies of several countries around the world. Currently, around 69 % of desalinated water produced in the world comes from RO plants (Eke et al., 2020). Even though less energy-intensive than other desalination processes, RO still generates environmental impacts. Among them, the production of solid waste is studied at a lesser extent. Even so, previous estimation of industrial end-of-life (EoL) RO membranes disposure for developed countries estimated that the desalination sector will be responsible for discarding around two million of EoL-RO membrane worldwide modules until 2025 (Sénan-Salinas et al., 2020). Estimating the average weight of an 8-inch-diameter RO element between 13 - 15 kg (Lawler et al., 2015), it is expected that around 30,000 tonnes will be generated by 2025. Lawler et al. (2015) also estimated EoL-RO membranes disposure in Australia, reaching 800 tonnes per year, and Sénan-Salinas et al., (2021) estimated a RO element stock of 1,045,280 for a particular region in Spain (that concentrates 42 % of the desalination capacity of the country). However, both authors approached the issue by defining that only 8-inch elements were used for industrial purposes and fixed a lifespan regardless of the membrane use. In addition, the fixed lifetime was different in each of the studies (Sénan-Salinas et al., 2021; Lawler et al., 2015).

The environmental impacts of RO elements start in their fabrication. They are made of polymers that come from oil, being normally fabricated in central facilities that use fossil fuels (Landaburu-Aguirre et al., 2016). In addition, they are distributed using diverse transport media that are also responsible for emitting greenhouse gases. Furthermore, when discarded, they normally end up in landfills, the least desired option in the waste hierarchy. Hence, to bring desalination closer to more sustainable principles, it is mandatory to include the environmental impacts of RO elements in the desalination radar, since they seem to follow a linear pattern of ‘take – make – consume – dispose’ worldwide (Sénan-Salinas, 2021, Lawler et al., 2012).

The initial EoL-RO elements estimation performed by Sénan-Salinas et al. (2020) and Lawler et al. (2015) are valuable contributions to studying waste management in the desalination sector. However, there are some regional differences that ask for a more detailed investigation of waste production and should be taken into consideration. The first of them relies on the maturity of national markets to control and invest in proper operating conditions for RO plants. As a consequence, membrane lifetime can change much more than what is currently reported in the literature, which can result in misleading results towards waste estimation. Secondly, despite being a good initial approximation, to account for only 8” RO elements might also affect the overall results of waste estimation. This is due to the fact that 4” elements are also used for lighter industrial application (water and wastewater treatment), which might contribute to the overall waste estimation. This latter difference can be even more important where decentralized RO water treatment is encouraged, since they tend to use smaller membrane elements (Lakeh et al., 2017; Brazil, 2015). This is the case of Brazil, for example, where the Federal Government holds a program of promoting RO desalination to supply communities in the semi-arid region of the country (Brasil, 2015).

To tackle these challenges and provide more accurate results, waste estimation models might be good alternatives, since they already take into consideration the difficulty of obtaining reliable data on waste generation. In Brazil, for example, the RO market in Brazil is barely mapped. In addition, when it comes to waste generation, there is no research market and industries are not prone to collaborate with waste information for studies. Hence, an alternative for researchers is to rely on waste estimation models to understand the size of the problem. From that it will be possible to propose realistic waste alternatives that might lead towards more sustainable practices.

Several models have already been developed for waste assessment, and they normally depend (i) on the type of waste that is being investigated and (ii) the particularities of the chosen system boundaries (for example developed and developing countries). In general, the models estimate waste using input data of sales (Kawai and Tasaki, 2016), import, and estimation per capita (Luo et al., 2019). Then, they relate these inputs with lifetime, surveys or stock data. When the average lifespan is known, for example, a possibility widely used to avoid fixing lifespan is to apply the Weibull distribution. This function models the lifespan profile by defining that the probability that a product sold in a determined year (y_s) is discarded in subsequent year (y_w) is calculated as a function of two parameters: a time dependent shape parameter (α_{ys}) and a scale parameter (β_{ys}) (Polák and Drápalová, 2012). In the distribution, α_{ys} determines the rate at which the failure rate decreases over time and β_{ys} determines the average product lifetime (Peeters et al., 2015). The Weibull distribution has been widely used due to its higher analytical tractability and produces the best fit for most products (Walk, 2009; Peeters et al., 2015). However, to use the model, it is necessary first to define such parameters using related data and defining an average lifespan, as shown by Mahmoudi, Huda and Behnia (2019).

When the lifetime is variable and information is not available through surveys (or they are not representative), another possibility is to address the issue from a macro perspective. According to Araújo et al., (2012), the time-step model is a possibility to overcome the lifespan issue by defining waste from a macro mass balance. It is applied for market products whose lifetime is variable, as for example, non-mature markets. In this model, the number of products that enter the market and do not reach new customers will be used to replace products previously in stock, which will then be discarded. This model, despite being relatively simple, brings an analysis that is especially interesting where there is a lack of information regarding lifespan data, such as in this work.

To estimate waste streams is critical to design effective policies, regulations and strategies that lead to a circular economy (Paiano, 2015; Domínguez and Geyer, 2017). In addition, as important as estimating them, waste forecasting is also a powerful tool to foresee problems and take actions preventively (Paiano, Lagioia and Cataldo, 2013). To forecast waste, the Auto Regressive Integrated Moving Average (ARIMA) model was used in this work. The ARIMA(p,d,q) is an application of the Auto Regressive Moving Average that eliminates non-stationary behavior (Box and Jenkins, 1976). It combines autoregressive and moving average models. In the first, the variable of interest is forecasted using a linear combination of past values, while the last uses past forecast errors in a regression-like model. In addition, 'I' for the integrated refers to transformations that are eventually needed to fit the series into a stationary pattern. Each of these terms are related to a parameter, i.e, in ARIMA(p,d,q), p is the autoregressive term, d is the degree of differencing, and q relates to the moving average order. ARIMA is widely used for time series forecasting in several areas due to its adaptability towards time series distribution (Niu et al., 2021).

Considering the importance of waste estimation for designing meaningful policies, regulations, and strategies to deal with the issue, this paper provides an insightful analysis of EoL-RO membranes disposition in Brazil. Given the lack of data and mapping of membrane lifespan and use, the waste stream was estimated as a function of the imported material and stocks for a period of 2016 – 2019, since 2020 was very atypical due to COVID-19. After that, an ARIMA model was built to forecast waste by 2024. It is anticipated that the results of this chapter will provide opportunities towards better understanding EoL-RO element's circumstances and support further strategies for dealing with this waste. It also acts as a database for the development of life-cycle analysis and investigation of possibilities to approach the desalination market of more sustainable practices regarding its solid waste. The proposed methodology can be easily applied to other countries that mainly import membranes or possess accurate information regarding membrane selling.

3.2 Materials and methods

The evaluation of the Brazilian EoL-RO membranes waste stream based on data import of RO membrane modules can provide clear insights for key stakeholders to design a proactive waste management, taking into account possibilities of reuse, recycling, recovery, and disposition. To this, an estimation of the annual and cumulative waste generation was carried out. Since there is no meaningful membrane fabrication in Brazil, import data from RO elements is a very good approximation of the number of RO membrane modules that enter the Brazilian market.

3.2.1 Import data exploratory analysis

Import RO membrane data from 2016 - 2020 was retrieved from *Penta Transaction Database*. This timeframe was defined since only the last 5 years possessed detailed information regarding RO import. Under the NCM code 8421.99.91, data cleaning was performed to remove products that were not RO elements despite being imported as such. At the end, a total of approximately 215.000 data entries were used to estimate RO membranes that reached the Brazilian market in the aforementioned period. To associate the membrane importation to its weight, the import product description was used to classify membranes according to their diameter, length, supplier, and flow rate. To this, modules product sheets were used, since the product description normally presented the membrane code. Since the formatted data sheet contained more than 200,000 lines, an exploratory data analysis (EDA) was performed using *Google Colab* with the programming language Python and the libraries described in Table 3.1.

Table 3.1 - Libraries used to perform EDA

Library	Pandas	Numpy	Matplotlib	Seaborn	Scipy
Description	Data frame manipulation	Mathematical functions and operations	Graphical visualization	Graphical visualization	Used for numerical routine, including statistics

Source: Elaborated by the author (2022)

The aforementioned libraries were used to investigate any significant correlation between the following parameters: import year, membrane flow rate, and diameter using Spearman's correlation at a confidence interval of 95 %. After that, the import quantity and weight were plotted against the membrane diameter distribution to understand the impact in overall waste for each of the categories.

3.2.2 Waste estimation

To estimate waste, two main variables need to be defined for a given interval i: (i) membrane flow rate (FR_i) and (ii) membrane weight (W_i). To this, data was categorized in terms of membrane diameter since it presented a strong correlation with flow rate and this designation facilitated comparison with the literature. Then, the density function was plotted for both unitary weight and flow rate, to understand how data was distributed. The distribution was classified and the values of mean, median, and mode were compared to define the most appropriate unitary weight and flow rate for each element diameter interval.

After establishing both unitary weight and flow rate, waste estimation was performed for the industrial category, since it most affected the overall weight. To this, the time-step model was used because of the following reasons: (i) it does not depend on the product lifespan (and in Brazil this might vary due to lack of proper operational conditions); (ii) there is no previous estimation of sufficient data to meaningfully extrapolate EoL-RO membranes generation in Brazil. The time-step model is defined according to Equation $(Generation\ of\ waste)y = sales\ in\ year\ y - (stock\ in\ year\ _y - stock\ year_{y-1})$ (3.1) (Araújo et al., 2012).

For a given year $\boldsymbol{y}$,

$$(Generation\ of\ waste)_y = sales\ in\ year\ y - (stock\ in\ year\ _y - stock\ year_{y-1}) \quad (3.1)$$

In this case, since local production is virtually zero, $\boldsymbol{sales\ in\ year_y}$ corresponds to importation, and $\boldsymbol{stock_y}$ is the number of membranes in use. In addition, since the most impacting category for waste production (in tonnes) are membranes for industrial use, industrial stock was defined from the RO industrial installed capacity in Brazil, retrieved from International Desalination Water Security Handbooks (2017, 2018, 2020). To transform installed capacity into membrane quantity, a weighted average was used as shown in Equation 3.2.

For a certain membrane diameter interval d_i,

$$S_{di} = \left(\frac{\Delta IC_{ind}\ x\ IM_{di}}{FR_{di}\ x\ IM_{ind}}\right) W_{di} \quad (3.2)$$

Where S_{di} is the stock for a given diameter interval, ΔIC_{ind}, the gradient of industrial installed capacity between two subsequent years, IM_{di}, the number of imported membranes at the given interval, IM_{ind} is the total of membranes imported for industrial use, FR_{di} and W_{di} are equal to the flow rate and unitary weight estimated for a given diameter interval. In addition, a sensitivity analysis was performed to understand the impacts of the estimated unitary flow rate FR_{di} and weight FR_{di} and approaching industrial use as composed by 8-inch diameter elements only on the overall estimation.

3.2.3 Waste forecasting

After estimating the annual waste stream, data was split to months according to the import pattern. This allowed to spread data in time so the forecasting could be performed. To apply the ARIMA(p, d, q) model, waste data was first plotted as a function of time and its behavior was assessed. The overall methodology consisted of four major steps: (i) Identify if the series is non-stationary; (ii) If necessary, transform to eliminate any eventual non-stationarity (define d); (iii) Obtain the ARMA model of the transformed series after estimating (p, q); (iv) Forecast and undo previous transformations (Navarro-Esbrí, Diamadopoulos and Ginestar, 2002).

To evaluate if the series was stationary or not, the augmented Dickey-Fuller (ADF) test was used, since it tests the null hypothesis that a unit root is present in an autoregressive model. The null hypothesis of non-stationary was performed using 1 %, 5 % and 10 % significance levels. Being the series non-stationary, it is possible to transform it by differencing the raw observations until the stationarity is reached, as shown in Equation 3.3 (Navarro-Esbrí, Diamadopoulos and Ginestar, 2002).

$$x_t = \nabla^d y_t = \nabla\ (\nabla\left(\ldots^d \nabla(y_t)\right)) \quad (3.3)$$

In Equation 3.3 ∇ is the lag-1 difference operator ($\nabla y_t = y_t - y_{t-1}$). To help identifying the differentiation order, the autocorrelation function (ACF) plot was used. Normally, a stationary function presents a quick decay for moderate and large lags. If data presents a slowly decaying positive ACF, it should be differentiated (Brockwell and Davis, 1996).

The next step was to obtain the ARMA (p, q) model orders of the transformed series. To this, the Akaike's Information Criterion (AIC), defined in Equation 3.4, was used.

$$AIC\ (p, q) = -2\log(L) + 2(p + q + k + 1) \tag{3.4}$$

In Equation 3.4, L is the likelihood of the data, p and q are the orders of the ARIMA model and $k = 1$ if $c = 0$ or $k = 0$ if $c \neq 0$. AIC defines, p and q by minimizing the value of the AIC function. It is important to highlight that it is not used to select the order of differencing, since differencing data will change its likelihood. As a consequence, AIC models with different differencing orders cannot be compared among each other.

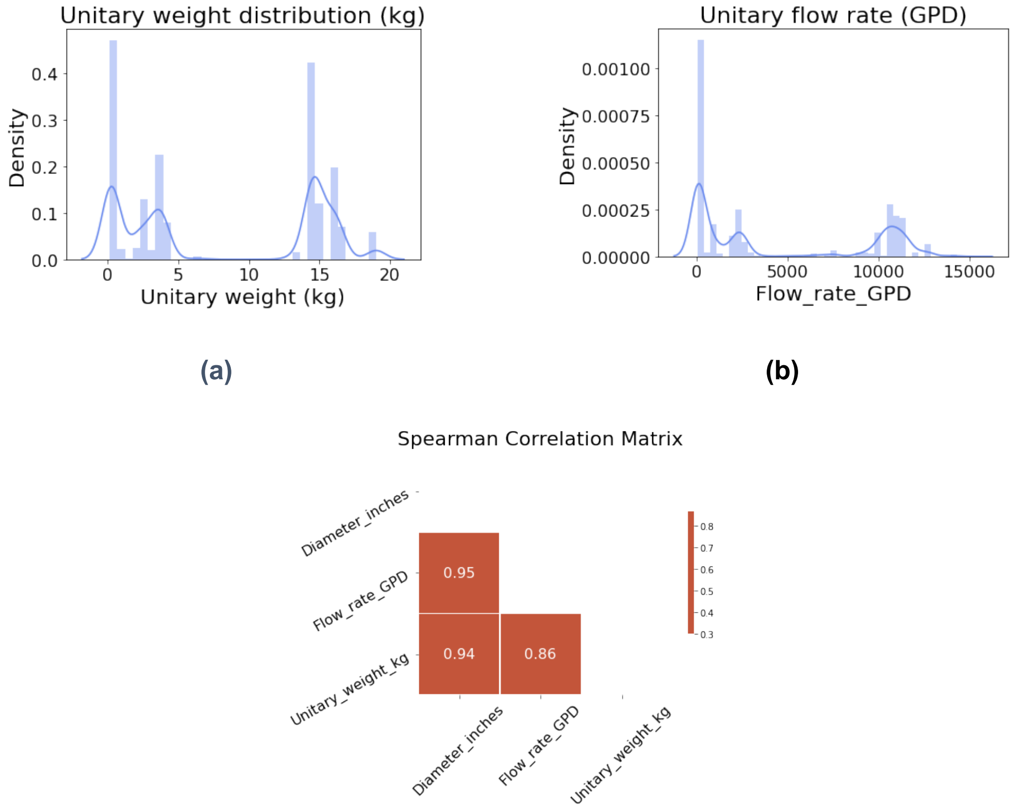
After defining p and q, an ARIMA model was estimated, and to understand if it could actually be used, the residuals were tested. In this case, Ljung-box test was applied exclusively to the residuals, testing the null hypothesis that there is no auto-correlation amongst them (Ljung and Box, 1978). Hence, if the p-value > 0.05, the residuals are not correlated, the model can be used. Using the best model, forecast was performed until 2025. The implementation was done in Phyton using the libraries described in Table 3.1 plus statsmodels.

3.3 Results

3.3.1 Exploratory import data analysis

To understand any significant correlation between importation year, membrane diameter, flow rate (GPD), and unitary waste (kg), a Spearman correlation matrix was plotted, as shown in Figure 3.1. It was seen that there is a strong correlation between membrane diameter and both flow rate (0.95) and unitary weight (0.94), at a confidence interval of 95 % (p-value < 0.05). They follow a non-parametric distribution trend and the greater the membrane diameter, the greater the weight and the flow rate.

Figure 3.1 – Non-parametric distribution for (a) unitary weight and (b) flow rate and (c) Spearman's correlation matrix index for the assessed variables



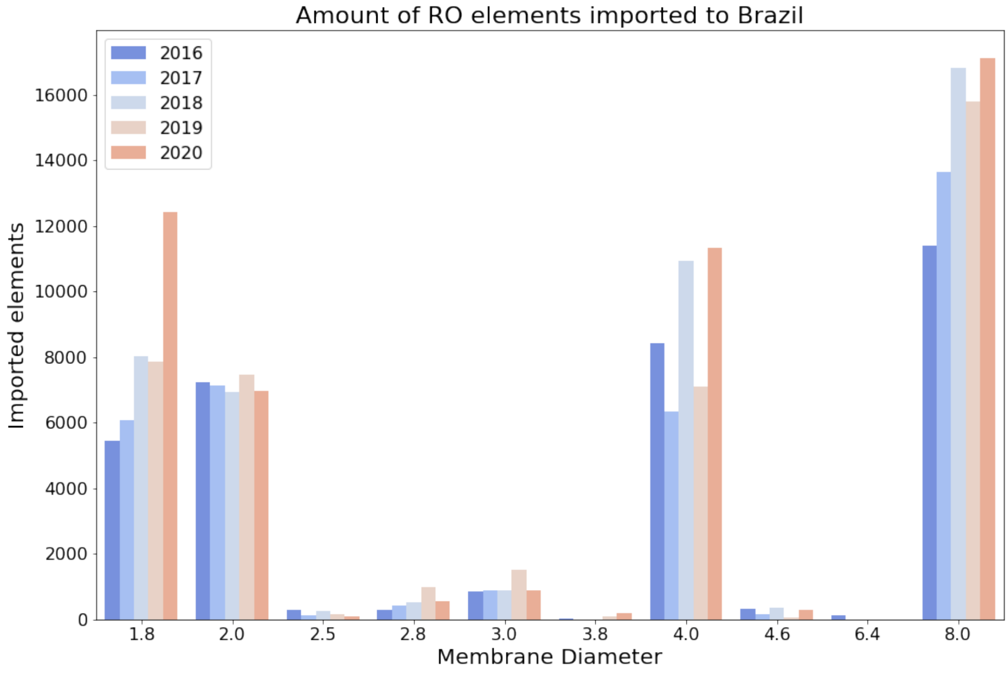
(a) **(b)**

(c)

Source: Elaborated by the author (2022)

Given the strong correlation between flow rate and diameter, categorizing membranes in terms of each of them will be equally meaningful. In this case, membrane diameter was selected as standard categorization to facilitate comparison with the literature. Regarding the amount of membrane elements imported to Brazil, Figure 3.2 shows that the 8-inch-diameter membrane is the most imported type, followed by 1.8, 2.0, and 4.0 inches. When taking into consideration the categorization between domestic, small commercial and industrial uses (Filmtec, 2021a; Filmtec, 2021b; Filmtec, 2021c; Filmtec, 2021d), it becomes clear that, in Brazil, RO is used at its best to industrial applications. However, domestic use is increasing over the years due to the popularization of RO point-of-use filters, as reported by *Statista Forecast* (Statista, 2019).

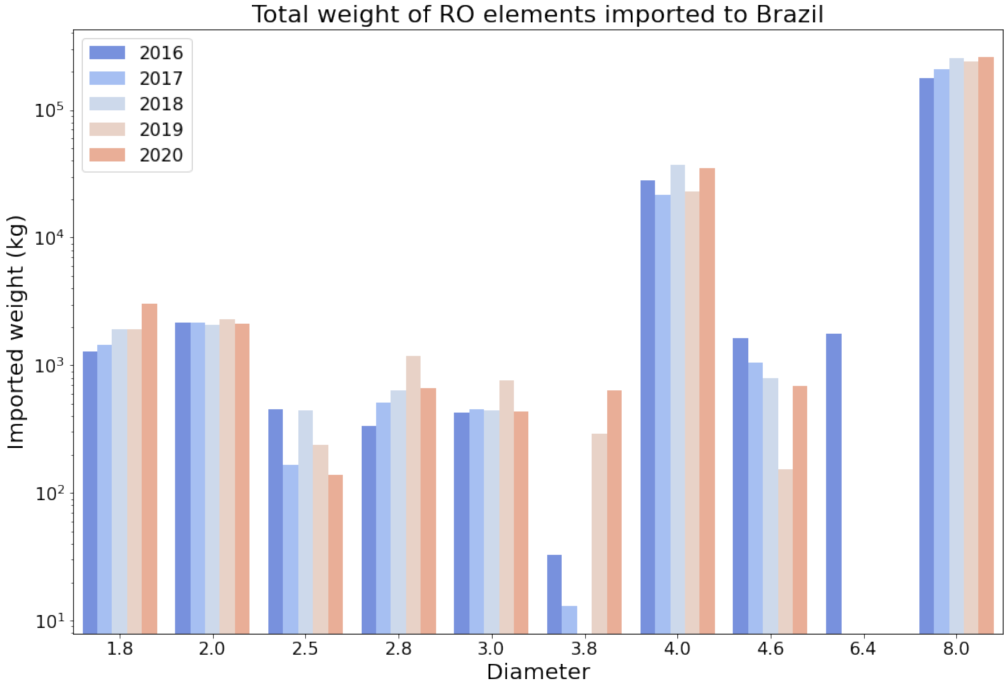
Figure 3.2 – Membrane categorization according to the diameter



Source: Elaborated by the author (2022)

Despite competing in terms of imported amount, when comparing the impact of each category in terms of weight, Figure 3.3 shows that the potential waste generated by point-of-use and small commercial RO elements in less than 1 % of what might be produced during industrial applications. This is due to the fact that membrane datasheets frequently report that 1.8” and 2.0” RO modules weigh around 0.2 kg, while 4.0” and 8” weight 2.7 and 15 kg, respectively (Filmtec, 2021a; Filmtec, 2021e). This means that despite the sum modules sold for small commercial use / domestic use is quite high, their impact on import weight (and consequently, waste) is low. In addition, point-of-use and small commercial elements stock are much harder to track since they are widely decentralized. Hence, in this work, the waste estimation was narrowed down exclusively for industrial use (i.e., membrane diameter between 3.8 and 8 inches), since it represents the potential biggest part of generated waste and membrane stock is possible to be estimated.

Figure 3.3 – Membrane categorization according to the diameter Imported weight according to the membrane diameter and its use



Source: Elaborated by the author (2022)

3.3.2 Waste estimation

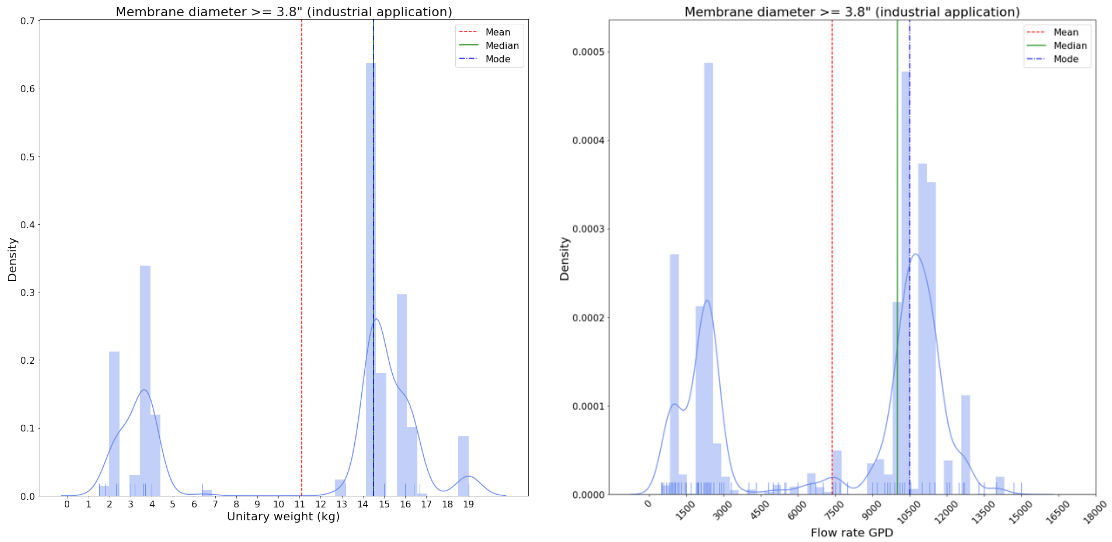
As seen in Section 3.3.1, modules used for industrial purposes are the most imported type and account for the major amount of membrane waste. This is due to the fact that the weight of small commercial and industrial use RO modules is much smaller than the industrial ones (Figure 3.3). As a result, waste estimation was performed taking into consideration this category.

3.3.2.1 Definition of flow rate and unitary weight values

To estimate industrial RO membrane module stocks, data was transformed from installed capacity (m^3/day) into unities (and then, weight) using the most appropriate flow rate and weight estimator. To this, the density function of flow rate and weight of imported modules was used (Figure 3.4). For both flow rate and weight, Figure 3.4 clearly shows that using any statistical parameters such as mean, mode or median for the entire interval is not representative, since data is gathered along two distinct peaks. Hence, to bring a more accurate estimation, industrial use was divided into two intervals for a given membrane diameter $\boldsymbol{d_i}$:

(i) $3.8 \leq d_i \geq 4.6$; and (ii) $6.4 \leq d_i \geq 8.0$. Hence, four values were estimated ($FR_{3.8 \leq d_i \geq 4.6}$, $W_{3.8 \leq d_i \geq 4.6}$, $FR_{6.4 \leq d_i \geq 8.0}$, and $W_{6.4 \leq d_i \geq 8.0}$).

Figure 3.4 – Density function for weight (a) and flow rate (b) of membranes with industrial use

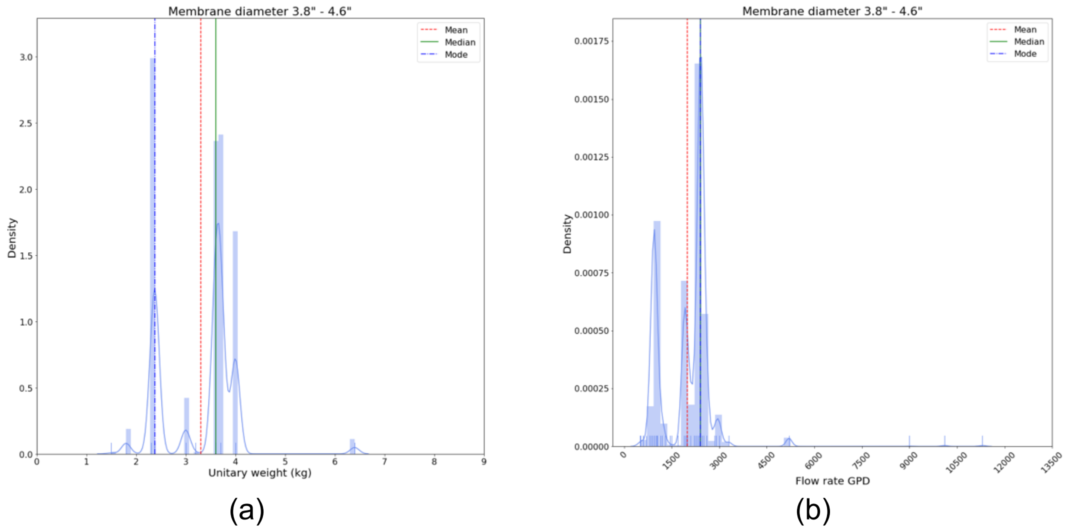


(a) **(b)**

Source: Elaborated by the author (2022)

To estimate $FR_{3.8 \leq d_i \geq 4.6}$ and $W_{3.8 \leq d_i \geq 4.6}$, the density function of the interval was plotted, as shown in Figure 3.5. For membrane weight (Figure 3.5a), the mode, median and mean presented a variation of 27 %. This can be justified by the variety of membranes and suppliers that fabricate membranes within this interval. However, since for flow rate (Figure 3.5b), these parameters became closer (variation of 17 %), being the mode coincident to the median, it can be inferred that a membrane can weigh more or less while presenting the same performance in terms of flow rate.

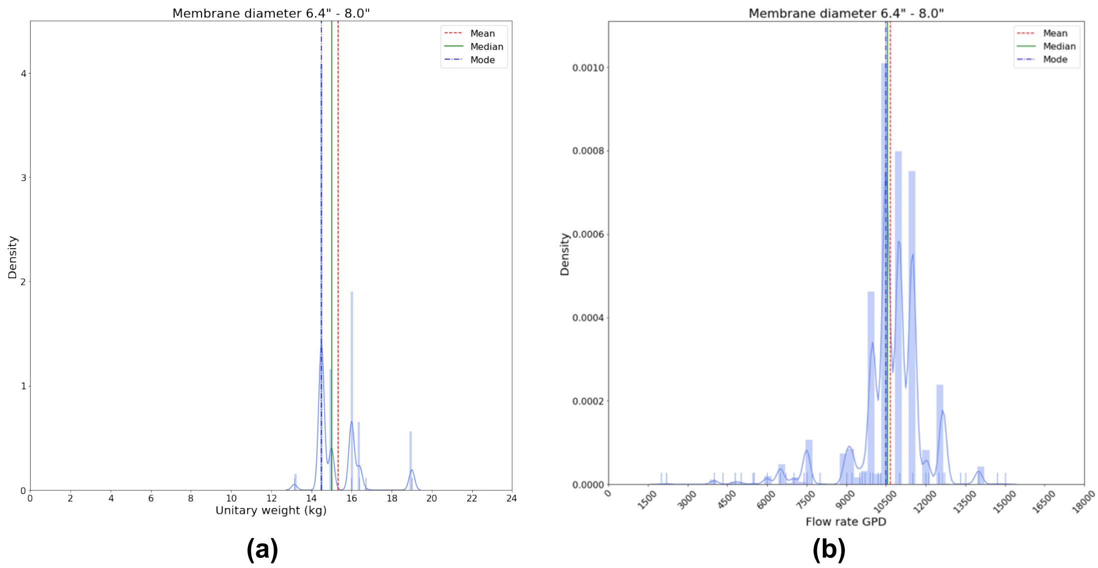
Figure 3.5 – Density function for weight (a) and flow rate (b) of membranes within the $3.8 \leq d_i \geq 4.6$ interval



Source: Elaborated by the author (2022)

For the second interval ($6.4 \leq d_i \geq 8.0$), the variation between mean, mode and median was only 5 % (Figure 3.6). This can be attributed to the lower membrane diversity within the market. Since there are fewer suppliers, the membranes are more uniform.

Figure 3.6 – Density function for weight (a) and flow rate (b) of membranes within the $6.4 \leq d_i \geq 8.0$ interval



Source: Elaborated by the author (2022)

Given the results, Table 3.2 summarizes the statistical values that were used to help transform stocks into elements weight. It is noteworthy that the variations were assessed through a sensitivity analysis.

Table 3.2 - Values used to transform installed capacity into weight

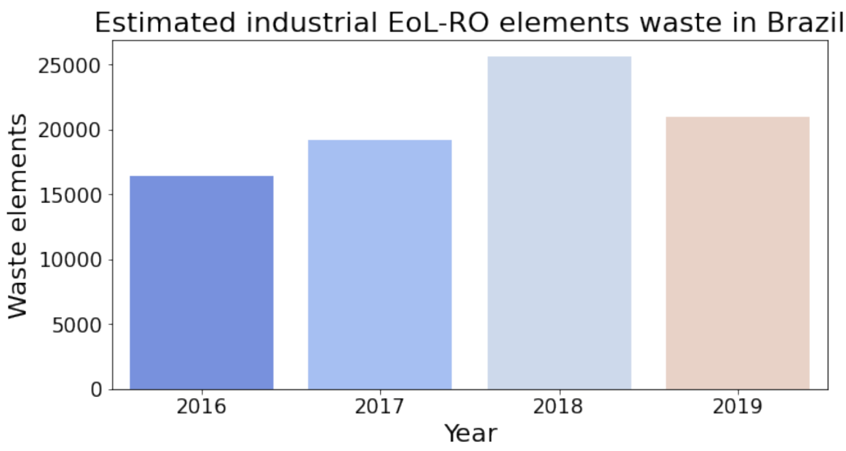
Parameter	$\mathbf{3.8 \leq d_i \geq 4.6}$	$\mathbf{6.4 \leq d_i \geq 8.0}$
Flow rate (GPD)	$\boldsymbol{FR}_{3.8 \leq d_i \geq 4.6} = 2{,}400$	$\boldsymbol{FR}_{6.4 \leq d_i \geq 8.0} = 10{,}500$
Unitary weight	$\boldsymbol{W}_{3.8 \leq d_i \geq 4.6} = 3.3$	$\boldsymbol{W}_{6.4 \leq d_i \geq 8.0} = 15$

Source: Elaborated by the author (2022)

3.3.2.2 Waste estimation

To estimate waste, the time-step method was used. To divide stocks into the $3.8 \leq d \geq 4.6$ and $6.4 \leq d \geq 8.0$ intervals, data was split in terms of the imported trend, as shown in Equation 3.2. In this case, the $3.8 \leq d_i \geq 4.6$ was estimated to contribute with 10.2 % of the overall installed capacity, while 89.2 % of the new installed plants possessed a diameter ranging from $6.4 \leq d_i \geq 8.0$. It is important to highlight that membrane fouling was not taken into account for the waste estimation. Nevertheless, the average weight reached in this work is similar to what was already reported in the literature by Lawler et al. (2012). Yearly estimation (in kg) is shown in Figure 3.7, and it can be seen that there was a tendency of growth until 2018. However, the time series is too short to understand whether 2019 would be an outline year or not.

Figure 3.7 – EoL-RO membranes industrial modules waste estimation



Source: Elaborated by the author (2022)

A local sensitivity analysis was performed to understand the impact of unitary weight and flow rate in the overall waste estimation, as well as to understand the variations of approaching industrial desalination of being performed with 8-inch elements only. By putting waste as a function of flow rate and weight, it becomes clear that it is directly proportional to the unitary weight and inversely proportional to flow rate. Hence, the maximum and minimum for the sensitivity analysis are a combination of minimum flow rate and maximum weight and maximum flow rate and minimum weight, respectively. As seen in Table 3.3, results show that the model’s deviation is around 9 % by both using intervals of by defining membrane diameter at 8 inches. In addition, when comparing the use of 8-inch elements only to the intervals, the variation is 3 %. This latter is justified by the dominance of 8-inch elements in the Brazilian market, both in terms of imported elements and weight. Hence, for Brazil, to standardize the industrial market as 8” element also is a fair approximation, as it was for Australia and Spain (Lawler et al., 2015, Senán-Salinas et al., 2021).

Table 3.3 - Sensitivity analysis for the waste estimation

Membrane diameter	**Accumulated waste (ton)**	**Minimum flow rate (GPD) and maximum weight (kg)**	**Minimum accumulated Waste (ton)**	**Maximum flow rate (GPD) and minimum weight (kg)**	**Maximum accumulated Waste (ton)**
Predefined intervals	889.3	$FR_{3.8 \leq d_i \geq 4.6}$ = 1,995 $FR_{6.4 \leq d_i \geq 8.0}$ = 10,500 $W_{3.8 \leq d_i \geq 4.6}$ = 3.6 $W_{6.4 \leq d_i \geq 8.0}$ = 15.3	864.55	$FR_{3.8 \leq d_i \geq 4.6}$ = 2,400 $FR_{6.4 \leq d_i \geq 8.0}$ = 10,654 $W_{3.8 \leq d_i \geq 4.6}$ = 2.4 $W_{6.4 \leq d_i \geq 8.0}$ = 14. 5	974.5
8 – inch only	901.6	$FR_{di=8}$ = 10,500 $W_{di=8}$ = 15.3	888.6	$FR_{di=8}$ = 10,654 $W_{di=8}$ = 14.5	951.3

Source: Elaborated by the author (2022)

When comparing the overall results found for Brazil to previous studies, it can be seen that the weight of EoL-RO elements generated annually (Figure 3.7) is lower than what was found for Australia and Spain (Lawler et al., 2015; Senán-Salinas et al., 2018). However, it is important to highlight that these countries invested in RO desalination as a water management strategy (MIT Technology Review, 2021, El Saliby et al., 2009;). In Brazil, on the other hand, desalination using RO is relatively new, with records starting from 2010 (IDA, 2016), and mostly used for industrial purposes (Eke et al., 2020). As a consequence, the installed volume in these countries is not directly compared to Brazil at the moment. However, as indicated by Markets and Markets (2018), desalination in Brazil are increasing over the years, and has even been encouraged for addressing the water issue in semi-arid regions by the Ministry of Science, Technology, Innovation and Communication (Brasil, 2021a).

To facilitate comparison and bring meaningful results, the waste generation was equalized to the estimated installed capacity to define a disposure rate (Table 3.4). It is important to highlight that Spain and Australia presented the same disposure rate due to the fact that the authors used the same methodology to estimate waste in their countries (Lawler at al., 2012; Senán-Salinas et al., 2021). The methodology proposed by Lawler et al (2012), assumes that 100 elements are installed for each 1000 m3/day of installed capacity and a mean service life of RO modules of 10 years, that weight 15 kg each (Lawler et al., 2012).

By equalizing waste generation, what was actually seen is that the disposure rate in Brazil is much higher than seen in these countries, and it can be inferred that the average lifespan for Brazil can be estimated as half as the lifespan reported worldwide. One possibility to justify this difference is the probable lack of efforts towards studying and optimizing national operations. Since in Brazil desalination is used to treat water / wastewater within the industrial process but water is not the final product, the operation of RO plants ends up neglected, since the main effort is normally directed to improvement within the products and / or production expansion. In this context, membranes might actually lose their standard performance prematurely or be discarded before time.

Table 3.4 - Estimated desalination capacity in Brazil, Australia, and Spain (2019)

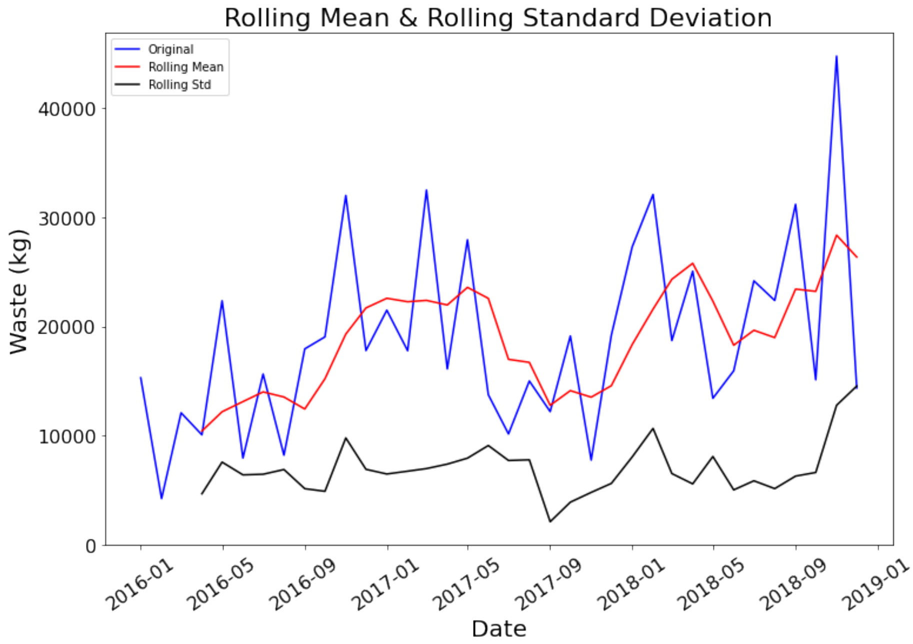
Country	**Disposure rate (RO element / $m^3.d^{-1}$)**
Australia	0,10
Spain	0,10
Brazil	0.19

Source: Elaborated by the author (2022)

3.3.3 Waste forecasting

Following the implementations steps proposed in Section 3.2.3, data was first plotted as a function of time along with its respective rolling mean and rolling standard deviation. To build the ARIMA model with more accuracy, the analysis was narrowed by data between 2016 – 2018, since 2019 could be considered a turning point. This resulted in a relatively small sample, despite being reported as enough to promote forecast with meaningful results (Hyndman and Athanasopoulos, 2021). Nevertheless, when using the AIC, both parameters and the amount of noise are taken into considerations, which allows a more accurate estimation. Hence, data was plotted and, as shown in Figure 3.8, the time series presented a slightly increasing tendency, since the rolling statistics did not seem to be constant over time. To confirm, the ADF test was performed at confidence intervals of 1%, 5%, and 10%.

Figure 3.8 – EoL-RO elements waste generation function




```
Results of Dickey-Fuller Test:
Test Statistic                 -2.333121
p-value                         0.161508
#Lags Used                      1.000000
Number of Observations Used    34.000000
Critical Value (1%)            -3.639224
Critical Value (5%)            -2.951230
Critical Value (10%)           -2.614447
dtype: float64
```

Source: Elaborated by the author (2022)

As observed, the p-value was found bigger than 0.05 (Figure 3.8), which means that the null hypothesis of non-stationarity could not be rejected. Hence, to transform the time series to a stationary series, the function was differenced by one order ($\boldsymbol{d} = \mathbf{1}$). The ADF test was performed for the differenced function and confirmed the stationary of the transformed time series.

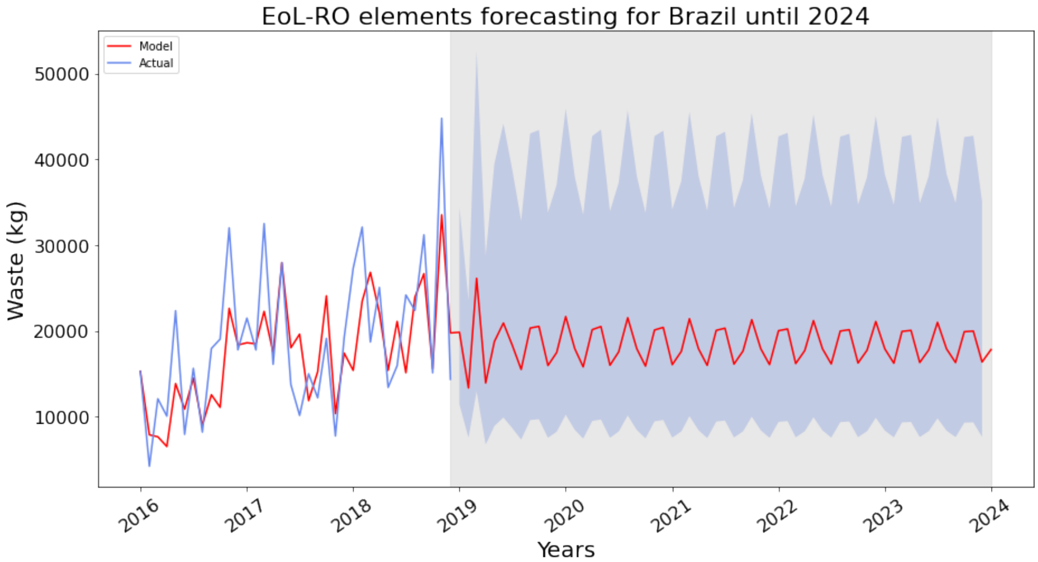
Hereinafter, the AIC was used by writing a function that tested a combination of $\boldsymbol{p}$ and $\boldsymbol{q}$ varying from 1 to 8. The orderassociated with the best model (i.e., the lowest AIC value) is ARIMA (3,1,5), which means an order of 3 for autoregressive terms, 1 for differencing and 5 for moving average terms. Hence, the ARIMA model could be written as shown in Equation 3.5.

$$y'_t = -1.004\, y'_{t-1} - 1.235\, y'_{t-2} - 0.553\, y'_{t-3} + 0.258\, \epsilon_{t-1} + 1.014\, \epsilon_{t-2} - 1.038\, \epsilon_{t-2} - 0.260\, \epsilon_{t-4} - 0.970\, \epsilon_{t-5} \quad (3.5)$$

In Equation 3.5, *epsilon* is noise with a variance of 0.071. To confirm the model accuracy in forecasting, i.e., that *epsilon* did not present autocorrelation, the residuals were evaluated according to the Ljung-box test. Since the p-values are above 0.05, the null hypothesis of no autocorrelation cannot be rejected. Hence, residuals are considered white noise.

Given the results, the model was considered ready for use and was implemented to forecast the RO waste generation until 2024 (Figure 3.9). A relatively small timeframe was used due to the fact that since the model depends on previous values to predict the next (autoregressive), it loses its accuracy by increasing the timeframe. Nevertheless, even with all its limitations, the model is a fair starting point, and forecasted that around 1,800 tonnes of EoL-RO elements will be discarded in Brazil by 2024.

Figure 3.9 – EoL-RO forecasting until 2024. The confidence interval is shown in light blue



Source: Elaborated by the author (2022)

3.3.4 Prospective alternatives for EoL-RO elements management in Brazil

Given the high disposure rate and the waste hierarchy, several actions should be taken to at least approximate the Brazilian productivity to what is experienced in other countries, thus diminishing the estimation of RO waste over the years. As a first hierarchy level, prevention could be achieved by setting better controls towards waste generation. This might discourage waste production, which will ultimately lead to setting more accurate operational standards to RO desalination plants in the country.

Another possibility is that considering that RO membranes are discarded before time, membrane reuse might be favored after proper cleaning procedures for the discarded elements. According to Lawler et al. (2015) and -Salinas et al. (2019), direct reuse is the most environmentally friendly alternative to EoL-RO elements. However, the authors highlight that it is important to have track on the membrane previous use to define proper cleaning procedures.

Following the waste hierarchy, EoL- RO elements transformation to microporous membranes should also be investigated. By removing the selective layer through an oxidative process, the porous structure is exposed, and the membrane acts as an ultrafiltration (UF) / microfiltration (MF) structure (Coutinho de Paula, Gomes and Amaral, 2017, Senán Salinas et al., 2019). The process is relatively simple and in Brazil UF presents a great application potential to treat surface water and improve the drinking water quality. The application has already been successfully tested at pilot scale (Coutinho de Paula et al., 2017). The structure can be also employed to wastewater treatment (Oliveira et al., 2020) and act as support layer to other membrane types (Lejarazu-Larrañaga et al., 2020).

Despite the potentialities, both reuse and recycling need to be further assessed in terms of the infrastructure needed to adequate EoL-RO elements. To these two alternatives, one category that should be highlighted is the impact of transporting EoL RO elements. Brazil has a continental dimension, and transportation is done mainly by trucks (Potter, 2018). In this context, Sénan-Salinas (2021) highlighted that depending on the distance, emissions related to transportation are the most impacting category for choosing an appropriate disposure of the RO elements. As a prospective exercise, in Brazil, the southeast is the main region when it comes for desalination use. Despite being relatively small when compared to Brazil, the southeast alone has almost double of the size of the entire Spain. This might bring additional challenges to centralized EoL-RO membranes options in Brazil, when they need to be further investigated. Still regarding the transportation matter, it is important to remember that both RO and UF membranes are imported to Brazil from countries such as the United States, China, and Germany. This means that the environmental impact from bringing those elements from their original country to Brazil should also be taken into consideration when assessing the entire picture of RO and UF use, reuse and disposal impacts within the country.

In addition to challenges towards transportation, the lack of institutional encouragement towards reverse logistics also hinders membrane reuse and / or recycling. In Brazil, the main regulatory framework for waste management is the *PNRS*, that defines that the responsibility for the product life-cycle is shared amongst producers and consumers. This statement directly affects waste management, since there is no institutional obligation of collecting old membranes, which makes its centralized collection difficult.

Another point that should be mentioned is that recycling modules parts might also pose as an interesting alternative in Brazil. This is because the recyclability of EoL-RO elements by weight is estimated at 63 %, being mostly composed of polyester – PET (Sénan-Salinas, 2021, Lawler et al., 2015). In Brazil, PET recycling is increasing over the years, being directly linked to a social-economical demand for money. The intrinsic relationship between recycling and social economic aspects in Brazil makes mandatory the inclusion of socioeconomic aspects when performing an impact assessment. This last challenge actually goes into accordance with what is currently being discussed in the industrial ecology context as a way to integrate areas and promote more holistic perspectives that might help promote meaningful changes towards sustainable practices.

3.4 Chapter conclusion

An estimation of industrial EoL-RO elements disposure and forecasting was carried out for Brazil. To this, the time-step model was used by comparing the amount imported to the difference in installed capacity from 2016 – 2019. It was seen that approximating industrial as supplied by 8-inch elements only is a good approximation of the Brazilian market, with deviations around 2%. By equalizing the disposure with the installed capacity, it was seen that the disposure rate (in elements / m^3.day) in Brazil is two times higher, i.e., the average lifespan in Brazil is below what is reported worldwide. This indicates that operating conditions are not optimized in the country, and/or membranes are currently being disposed before their standard lifespan. In addition, a waste forecasting was performed using the ARIMA model, estimating a total generation of 1,800 tons until 2025. The high disposition rate and the accumulated waste suggest that the EoL-RO elements management in Brazil should be better assessed whether by improving membrane operating conditions or defining standard procedures for membrane reuse.

CHAPTER 4

DISPOSURE OF END-OF-LIFE REVERSE OSMOSIS MEMBRANES: ENVIRONMENTAL LIFE-CYCLE ASSESSMENT AND SOCIOECONOMIC ASPECTS

4.1 Introduction

Following a linear pattern of ‘take – make – consume – dispose’, reverse osmosis (RO) membranes are expected to be discarded in sanitary landfills after five to ten years of use (Landaburu-Aguirre et al., 2016). However, regional differences regarding membrane management might have a significant impact on membrane lifetime and can reduce by half the lifetime normally reported. This would ultimately lead to a waste estimation even bigger that what was initially proposed by Landaburu-Aguirre et al (2016). However, end-of-life reverse osmosis (EoL-RO) membranes are valuable waste that can be used for several applications rather than be disposed of in sanitary landfills.

Considering a more sustainable approach, the waste hierarchy suggests a preferential order to manage waste that aims at reducing environmental impacts. It prioritizes reduce, reuse, recycle over energy recovery treatment and, at last, landfill disposition (Hultman, Corvellec, 2012; Overcash, 2002). In 2015, the European Union defended the role of waste hierarchy to get valuable materials back to the economy and help ‘closing the loop’ within the framework of the circular economy (European Union Commission, 2015). In 2016, the strategy was included in the 12th Sustainable Development Goal of the UN 2030 Agenda for Sustainable Development (UN, 2016). Nowadays, waste hierarchy and circular economy permeates several waste management policies worldwide, such as those from the global north. In addition, Brazil, China, India, and further developing countries also have legal frameworks that encourage these concepts (Van Ewijk and Stegemann, 2016; Liu et al., 2017; Priyardarshini and Abhilash, 2020; Ngan et al., 2019).

Applying the waste hierarchy to the EoL-RO membranes management context, reduction can be achieved by (i) developing membrane modules that present bigger permeability, (ii) less propensity to fouling, and (iii) improving operational conditions. By aiming at these three strategies, one can diminish the number of RO modules used to produce a certain amount of water and increase the membrane lifespan. Such strategies are widely reported in the literature and have been summarized by Hailemariam et al. (2020) and Qasim et al. (2019).

By reusing, one can infer the direct use of discarded EoL-RO membranes for other purposes after proper cleaning. When doing so, the membrane lifetime is increased, and the second-life element can substitute the purchase of new ones. Few studies report EoL-RO direct reuse (Prince et al., 2011; Souza-Chaves et al., 2022), but the available ones concluded that even by decreasing the RO membrane selectivity, it was still possible to attain results similar of those found for nanofiltration (NF) membranes. These findings reinforce that it is technically possible to provide a second lifetime for the RO membrane for less demanding processes but still meet high-quality permeate, if properly addressed.

As for recycling, it can be divided into two subcategories for this application. First, the element itself can be reused after a chemical conversion of the RO membrane. Converting EoL-RO membranes consists of removing the upper aromatic layer and exposing the micro-porous polysulfonic one. This removal is normally performed exposing the membrane to strong oxidants, such as sodium hypochlorite, potassium permanganate, and hydrogen peroxide, for example (Coutinho de Paula, Gomes, and Amaral, 2017; García Pacheco et al., 2015). Chemical conversion methodologies are wide, and EoL-RO membranes have been successfully converted at both laboratory and pilot scale applications (Coutinho de Paula et al., 2018; 2017; García-Pacheco et al., 2018, Senán-Salinas et al., 2019).

Other applications have also been recently investigated, such as the use of the micro-porous structure combined with bioreactor to treat wastewater (Oliveira et al., 2020) and as support layer for the development of novel membranes. Lejarazu-Laranaga et al. (2020) used it to fabricate electrodialysis (ED) membranes using phase-inversion, and Contreras-Martínez et al. (2021) transformed EoL-RO membranes into membrane distillation after proper treatment. Finally, they have also been investigated to act as anion-exchange membranes (Pompa-Pernía, 2021). All these applications presented successful results towards permeability and selectivity and are strategies that add even more value to such waste.

The second interpretation from recycling is recovering each material that composes the membrane element individually, i.e., the recycling of parts. A RO element is mostly composed of recyclable parts since it is mainly made of polymers. This option is not only environmentally addressed but has also an intrinsic relationship with socio economic issues. In Brazil, for example, the commitment to recycling is directly linked to the aggregate value of residues, since it was born from a social-economical demand for money and, according to ABRELPE (2021), polymers are the second most valuable recycled material in Brazil. Still, according to the association, it represents 18% of the collected material but is responsible for 50% of the reduction in emission of CO2equivalent amongst the recycled materials (ABRELPE, 2021).

When thinking about more sustainable practices in the industry sector, it becomes of great importance to quantify, compare, and manage impacts, since each option has its advantages and drawbacks. Life-cycle Assessment (LCA) is a tool that quantifies the number of materials and energy uses and releases to the environment. LCA is very helpful for learning about the object of research, the ways it affects the three dimensions of sustainability and to compare alternative approaches to solve a common situation (Lior, 2017).

As previous work using LCA for EoL-RO membrane management, Lawler et al. (2015) assessed the associated impacts of six different end-of-life destinations in Australia. To this, the authors performed a comparative LCA for membrane manufacturing and end-of-life options. The authors found that reuse and chemical conversion were the most environmentally friendly options. In addition, regardless of the transportation distance, reusing the membrane for one year was more environmentally favorable than landfill disposure. In a next study, Senán Salinas et al. (2019) investigated two pilot designs for converting EoL-RO membranes into NF and UF membranes. The authors reported that the most environmentally interesting option was transforming brackish water RO membranes into NF and UF membranes and estimated an associated cost of € 54.5–73.75/module. Next, they projected a virtually conversion facility in Spain and assessed environmental and economic impacts of different locations (Senán Salinas et al, 2021). On the contrary of Lawler et al (2015), these authors noticed that long transportation distances could be limited by their contribution to climate change (Senán Salinas et al, 2021).

Despite all standardizations, LCA brings intrinsic particularities of the places for which they have been evaluated. During the inventory construction, several elements are regionalized and, consequently, findings from one country might diverge from others. In this assessed case, for example, most of the Brazilian landfills provide biological treatment for the leachate, which might not be suitable when thinking about the biodegradability of this stream (Costa et al., 2019). Therefore, leachate is more prone to cause environmental impacts and results would differ from other countries. In addition, life-cycle impact assessment (LCIA) methods were initially developed in northern countries to their own reality, which might make their direct application worldwide difficult (Mendes, Bueno and Ometto, 2013).

To help overcome these issues, this chapter intends to expand knowledge towards EoL Ro membrane waste management beyond the reality of developing countries. To this, a LCIA was performed for the following EoL-RO membrane disposition options: reuse, chemical conversion, recycling of parts, energy recovery by incineration, and landfill, in Brazil. The inventory was built considering Brazilian particularities in recycling, energy recovery and landfill disposure, as well as nationally developed methodologies for membrane reuse and chemical conversion into UF elements (Coutinho de Paula et al, 2017). It is important to highlight that these methodologies were previously assessed by Coutinho de Paula (2018) and presented a smaller cost when compared to those reported for Senán-Salinas et al (2021). Environmental impacts for the Brazilian context were compared to existing literature to understand eventual differences, and socioeconomic aspects were evaluated to assess potentialities of these disposals. Finally, the environmental and socioeconomic findings were integrated and discussed in terms of socio-impact, considering what recycling and encouraging other economic activities might represent in Brazil. It is important to highlight that since Brazil shares several environmental and social aspects with other developing countries, the findings of this research will hopefully help broaden knowledge for other countries outside the northern route.

4.2 Material and methods

4.2.1 Life-cycle assessment

This life-cycle assessment intends to understand the environmental impacts associated with different disposition options of EoL-RO elements and to compare the results to the acquisition of novel membranes in the Brazilian context. The LCA study followed the ISO 14040-44 guidelines (ISO, 2006a; ISO, 2006b) and is composed by four main stages: (i) goal and scope definition; (ii) life-cycle inventory; (iii) life-cycle impact assessment; and (iv) interpretation. Models were generated using Open LCA, and the inventory is described in Section 4.2.1.4. In addition, a cost-effectiveness analysis was held to understand economic impacts of each disposition option.

4.2.1.1 Goal and scope definition

The goal of this LCA was to assess which are the environmental impacts of waste management options for EoL-RO elements. The study is directed to policy makers, companies that use RO membranes and researchers that work with membrane reuse / conversion. It aims to answer the following question: Which is the best end-of-life option for RO membranes in Brazil from an environment and resources consumption standpoint?

In addition to the primary goal, special focus was given to the impact of transportation for both new and reused/converted membranes. Membrane manufacturing was also taken into consideration to better understand the eventual benefits of reuse and chemical conversion of EoL-RO membranes. In addition, to compare disposition options to novel membranes, membrane manufacturing (in white) was also assessed. The use phase was not considered in this study, being assumed that all membranes reach disposition at the same conditions.

As LCA boundaries, membranes used in Brazil were taken as reference. They are considered to be imported from the United States, used and disposed locally. Since membrane manufacturing is performed outside Brazil, data from the USA was used to assess the impacts related to this stage. To evaluate the disposure, on the other hand, data from Brazil was used and is further described in Section 4.2.1.4.

4.2.1.2 Declared unit

As seeing in the overall results of waste estimation, to define industrial use as supplied by 8-inch RO elements is a good approximation for Brazil, as shown in Chapter 3. Hence, the declared unit for the LCA was defined as a standard 8” Thin Film Composite (TFC) RO membrane module. The element is composed by a polyester (PET) base, a polysulfone (Psf) support layer and a polyamide (PA) active layer. The SW30HR-380 element, from Dupont, was chosen to represent the declared unit. It has a dry weight of 13.75 kg, and its main characteristics are shown in Table 4.1 (Filmtech, 2022).

Table 4.1 - SH30HR-380 element main characteristics

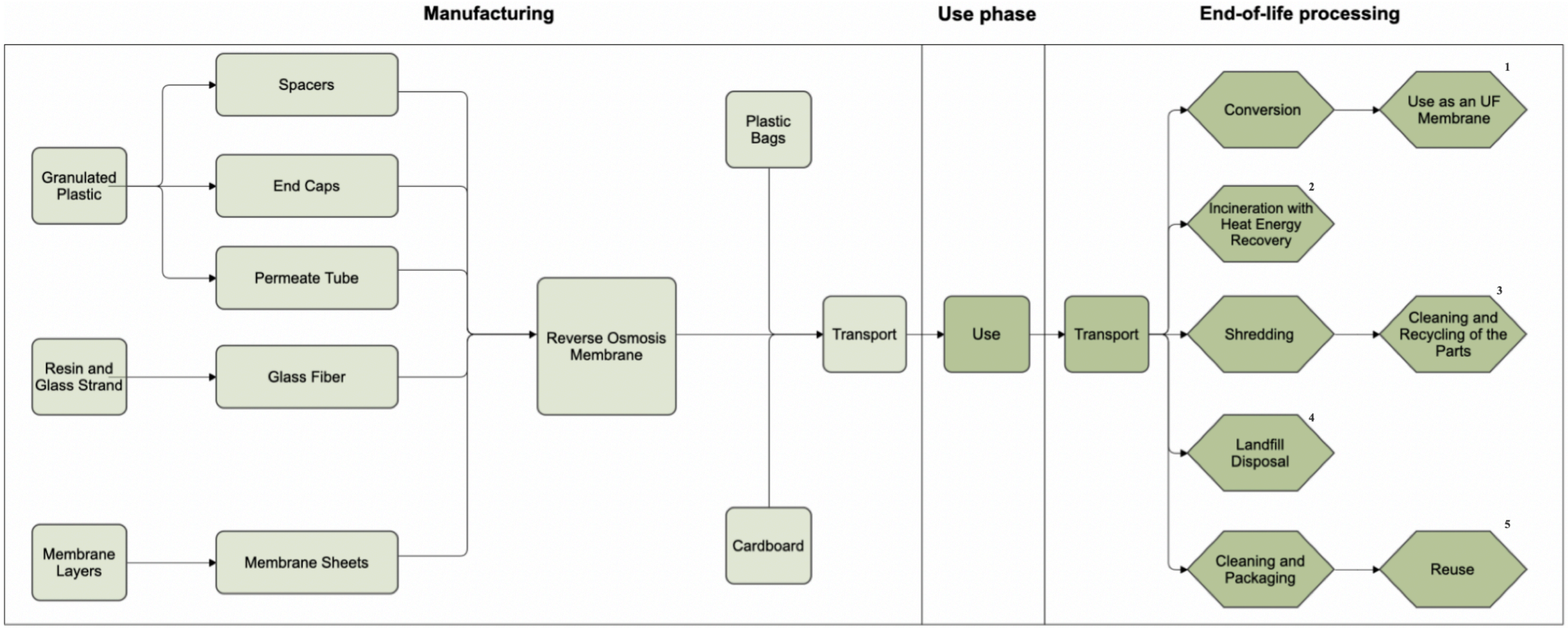
Characteristic	Amount	Unit
Dry weight	13.75	kg
Active area	35.3	m^2
Maximum operating pressure	69	bar
Standard operating pressure	55	bar

Source: Elaborated by the author (2022)

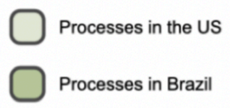
4.2.1.3 Model description

The primary goal of this study was to determine which end-of-life option will be most environmentally friendly in the Brazilian context. To this, Figure 4.1 presents a simplified flow diagram for membrane manufacturing and disposition options.

Figure 4.1 – Flow diagram for membrane manufacturing and disposition options



[1] Conversion procedure: Exposure to NaClO 300,000 ppm.h-1 (Coutinho de Paula, 2017). It also includes transporting, cleaning with NaOH 0.1% and HCl 0.2% and packing
[2] Incineration to energy recovery
[3] Membrane disassemble and processing for recycling, considering Appendix I
[4] Current alternative in Brazil, considered the baseline scenario for comparison
[5] Direct reuse of RO after transporting, cleaning procedure with NaOH 0.1% and HCl 0.2% (Coutinho de Paula, 2017) and packing



Source: Elaborated by the author (2022)

4.2.1.4 Life-cycle inventory

To build the life-cycle inventory, primary data was retrieved from *Ecoinvent 3.8 Cut-off*, considering the Brazilian reality and experimental data reported by Coutinho de Paula et al., 2017. The inventory is presented in table form, in Appendix 1.

<u>RO Membrane production and transportation</u>

Membrane production inventory was built using the SW30HR-380 Membrane Module Production, retrieved from *Ecoinvent 3.8 Cut-off*. The membrane module is considered to be produced in Minnesota (USA) and sent to Brazil through land and water (Appendix 1). Transport flows were added to the standard module production to account their impacts, as well as the flows associated with packaging.

<u>Direct reuse and chemical conversion to the UF membrane</u>

For membrane cleaning (for reuse) and chemical conversion, data was obtained primarily from extensive related research (Coutinho de Paula et al., 2017). For reuse, only membrane housing and cleaning were considered, while for chemical conversion, inventory was built adding the oxidation stage, as stated in Table 4.2. It is important to highlight that an offset was created to allow comparison between the virgin and the reuse/converted membranes. In the offset, the reused and converted membrane lifespan was set to 1/3 of the virgin ones.

Table 4.2 - Stages for membrane reuse and chemical conversion

Stage	Description
Housing after disposure	Membrane is individually submerged in sodium bisulfite (1% w/w), stored in a sealed plastic bag. Solution volume was set at 1.5 L (Coutinho de Paula et al., 2017)
Element cleaning	Cleaning was performed in two stages: • NaOH 0.1% w/w solution • Citric Acid 0.2 % w/w solution Since cleaning solutions can be used more than once, the specific volume for each element was set to 300 mL (Coutinho de Paula et al., 2017)
Oxidative treatment (Chemical Conversion)	Oxidative treatment was performed using NaClO 15% w/w solution, keeping the 300 mL for each element, since the solution is also reusable (Coutinho de Paula et al., 2017)
Treatment of the oxidant solution	The oxidant solution was considered to be neutralized using sodium sulfite, keeping the stoichiometric proportion for 300 mL of NaClO, as shown in Appendix 1.

Source: Elaborated by the author (2022)

In addition to the process itself, both membrane product systems for direct reuse and chemical conversion also took in consideration aspects related to packaging and transportation (set at a range of 50 km from the using center), as shown in Appendix 1.

<u>Material recycling</u>

Material recycling was evaluated using the Opposite Direct Approach, where recyclable materials are negative inputs in the process, corresponding to 67% of the membrane element. In addition, non-recyclable materials were either considered to be correctly treated (waste glass, corresponding to 14 % of the material breakdown) or sent to the closest landfill at a 50 km range (19 % of the element). Details are described in Appendix 1.

<u>Incineration and landfill disposure</u>

To incineration, it was considered the transport from the using site to the closest incinerator, at a range of 50 km and the material energy recovery, where 76% of the material was burned and the rest was sent to the landfill. For landfill disposal, the product system also consisted of transporting the membrane elements to the nearby landfill within 50 km range.

4.2.1.5 Life-cycle impact assessment (LCIA)

Method selection

To perform the LCIA, Open LCA was used and six different methods were first compared. This stage was added because the vast majority of LCIA models were developed in European countries, USA, and Canada. This might prevent countries outside those regions from producing LCIA results with specific characteristics. Hence, a way to diminish such biases is to test which methods converge to the same result.

To this, ReCiPe 2016, IMPACT 2002+, IMPACT World +, EDIP 2003, CML-IA, and ICLD Midpoint 2011+ were compared in terms of common categories. To evaluate if there was any significant difference between the methods, the Monte Carlo simulation (1000 runs) was applied to each of the methods, creating distribution functions for each of them. Functions were plotted in box-plot form for exploratory data analysis (EDA), where the Shapiro-Wilk test verified the non-parametric distribution. After, they were pairwise compared using the non-parametric Anderson-Darling test (AD Test) at a confidence level of 95%.

The AD test was chosen because it verifies if a sample data comes from a specific population. It is a modification of the Kolmogorov-Smirnov (KS), where it makes use of the specific distribution in calculating critical values (Information Technology Laboratory, 2017). In other words, the AD test compares values along the whole domain, while the KS test uses the cumulative distribution at the point of maximum distance. Hence, results from the AD test are more sensitive and, according to Arshad et al (2003), it is the most powerful empirical distribution function test. EDA and statistical tests were performed using Python and the Sci-Py library.

LCIA experiments

The chosen method was used to perform LCIA experiments at Open LCA. The EoL-RO disposition options were compared using the Monte Carlo simulation, with a total of 1000 runs. Results were compared in terms of relative impact related to the RO membrane production and its current disposition option, landfill. In addition, those end-of-life scenarios that generate another membrane lifetime (reuse and chemical conversion) were put against the use of novel RO and UF membranes, respectively. To this, the lifespan ratio of the second-life membranes was kept at 1/3. Additionally, a sensitivity analysis was carried out using the Monte Carlo simulation to understand eventual variation in the results.

4.2.2 Socioeconomic analysis

An economic analysis was held to compare the assessed disposition options. For the cost estimation of membrane chemical conversion and reuse, the methodology described in Coutinho de Paula et a (2018) was used. For the first one, it was assumed the same cost as described by the authors, since the recycling/ chemical conversion methodology used in this work is based on their experimental findings. There was, however, a correction of value according to the Brazilian inflation rate from 2017 to 2022 using the consumer price index (IPCA). For reuse, the chemical conversion phase itself – that comprises the use of hydrochloric acid 37% and its neutralization with sodium thiosulfate – was not taken into account. However, other associated costs were kept, since reuse would also need facility and personnel. Again, the final value was corrected using IPCA, estimated at 1.31 from 2017 – 2022 (Brazilian Central Bank, 2022).

For recycling, incineration and landfill, costs were obtained from ABRELPE (2015). For all these disposition options, only operational costs were taken into consideration, since the number of elements disposed would not demand the construction of new facilities. Since costs are normally estimated by ABRELPE considering the number of people that would be assisted, they were transformed to U$/ RO element whenever necessary. To this, it was considered values for a population of 100,000 people and the generation of 1.0 kg/day per person, from which 60% was considered recyclable (ABRELPE, 2021). In addition, it was considered that 225 tons of RO membrane were discarded in Brazil per year (Chapter 3). All costs were corrected using IPCA, estimated at 1.46 from 2015 – 2022 (Brazilian Central Bank, 2022).

To enrich economic and environmental findings, the three more attractive disposition options were classified according to their potential positive impact on the categories described by UNEP/SETAC for social impact (2009), as shown in Table 4.3. Each subcategory was classified from 1-6, where 1 was considered great impact; 2 – good; 3 – average; 4 – small; 5 – very small; and 6 – not applicable. After that, the categories were weighted according to the number of subcategories and classified as the sum of each subcategory potential impacts, considering the scale from 1-6.

Table 4.3 – Subcategories to social assessment

Categories	*Subcategories*	
Worker	· Freedom of Association and Collective Bargaining	· Forced Labor
	· Child Labor	· Equal opportunities/Discrimination Health and Safety
	· Fair Salary	· Social Benefits/Social Security
	· Working Hours	
Consumer	· Health & Safety	· Transparency
	· Feedback Mechanism	· End of life responsibility
	· Consumer Privacy	
Local community	· Access to material resources	· Respect of indigenous rights
	· Access to immaterial resources	· Community engagement
	· Delocalization and Migration	· Local employment
	· Cultural Heritage	· Secure living conditions
	· Safe & healthy living conditions	
Society	· Public commitments to sustainability issues	· Technology development
	· Contribution to economic development	· Corruption
	· Prevention & mitigation of armed conflicts	
Value chain actors not including consumers	· Fair competition	· Supplier relationships
	· Promoting social responsibility	· Respect of intellectual property rights

Source: Adapted from UNEP (2009)

4.3 Results and discussion

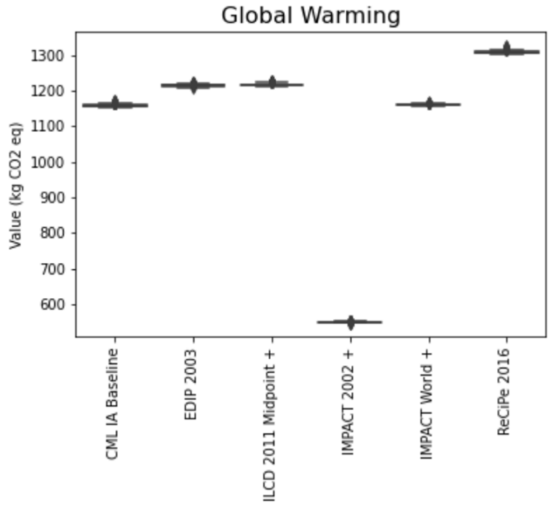
4.3.1 Environmental LCA

4.3.1.1 Method selection

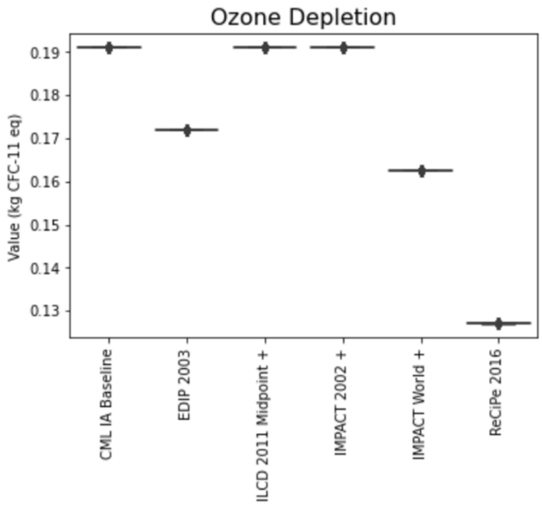
As a first step to define the LCIA method to be used, the tested ones were compared in terms of common categories. Despite not having the exact same name, by comparing their description and unit of reference, it was seen that ozone layer depletion and global warming were present across all six methods. The EDA performed a comparison of the global warming category, where all 6 distributions did not follow a normal distribution, as confirmed by the Shapiro-Wilk test. For that reason, further statistical tests were applied considering non-parametric distribution.

Samples were then plotted in box-plot form, as shown in Figure 4.2a. It was possible to notice that apart from Impact 2002+, at first the results seemed to converge, the reason why the Anderson-Darling test was applied to the remaining group. However, the visual comparison was not confirmed by the statistical test, and the null hypothesis of the samples being originated from the same population was rejected. The AD test was also used for pairwise comparison, and the null hypothesis was continuously being rejected. This means that, for the global warming category, each method presented a different impact result, and the analysis was not able to indicate alone which method(s) could lead to better results.

Figure 4.2 – Box plot representation of the analyzed distribution functions for (a) the global warming category and (b) ozone depletion



(a)



(b)

Source: Elaborated by the author (2022)

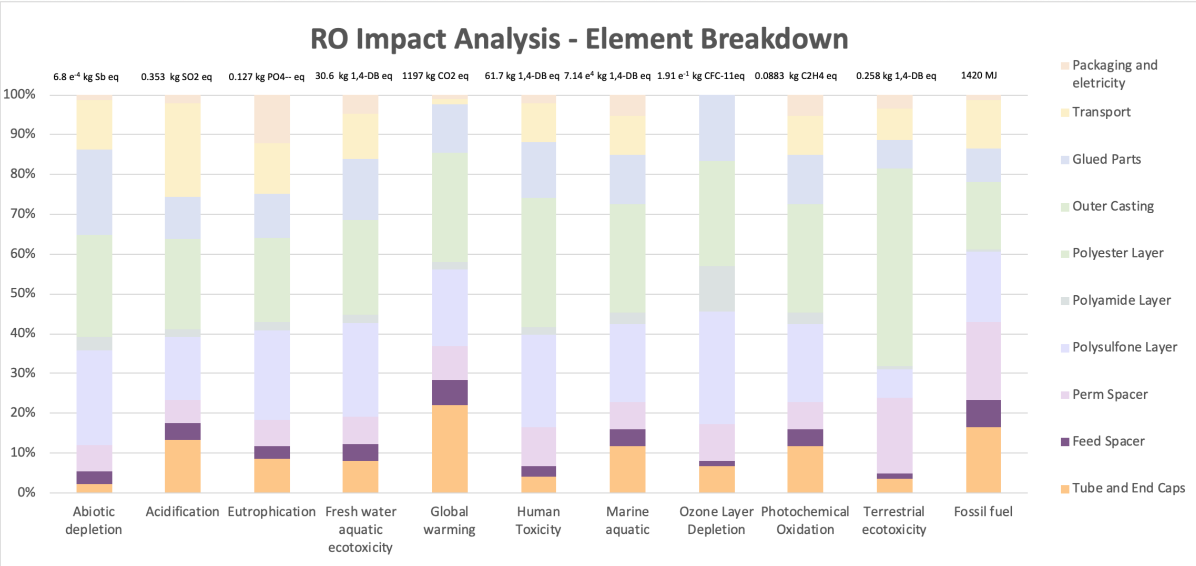
The same methodology was applied to the ozone depletion category. In this case, the boxplot showed that CML, Impact 2002+ and ILCD seemed to converge to the same population, as shown in Figure 4.2b. This was statistically confirmed by the AD-test, where the null-hypothesis was confirmed at a confidence level of 95%. This means that using either of the tests, to this impact category, would lead to the same result. However, when crossing information between both categories – global warming and ozone depletion, it was possible to notice that Impact 2002+ presented a higher deviation from the other two (ILCD and CML) for global warming, narrowing down the selection. In this case, since CML IA Baseline was one of the methods recommended by Mendes, Bueno and Ometto (2013) for the Brazilian context, it was chosen to perform further experiments.

4.3.1.2 LCIA experiments

<u>RO and UF membrane manufacturing</u>

To better understand eventual benefits from the studied end-of-life options, it was first necessary to investigate the impacts associated with the membrane manufacturing. Figure 4.3 presents the relative impact of the RO membrane manufacturing in terms of their components. It was possible to notice that membrane manufacturing itself presented the major environmental impact across all categories, being the biggest share related to membrane sheets (42% combining all layers). Transport, on the other hand, presented an overall impact that represents an average of 10% across the categories, despite the distance between Minnesota and São Paulo. These findings are in accordance with what was reported by Lawler (2015), where even with a greater travelled distance (from the USA to Australia), membrane sheets remained the main environmental impact.

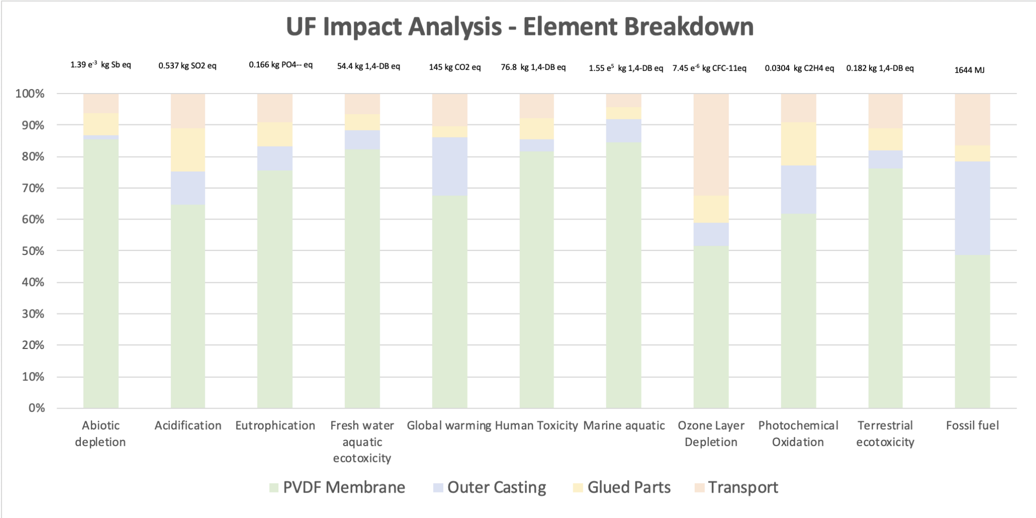
Figure 4.3 – Relative impact breakdown in terms of components for one RO membrane



Source: Elaborated by the author (2022)

Since one of the end-of-life scenarios corresponds to converting EoL-RO membranes into UF elements, the associated impact of the UF membrane was also investigated (Figure 4.4). In this case, the PVDF membrane was most responsible for the associated environmental impacts, corresponding to approximately 80% across all categories. In this sense, it is interesting to highlight that solvent has little impact on the overall PVDF production, which is in accordance with what was reported by Prézélus et al (2022). Transport was also kept around 10% across all categories, as seen for the RO membrane, except for ozone layer depletion.

Figure 4.4 – Relative impact breakdown in terms of components for one UF membrane



Source: Elaborated by the author (2022)

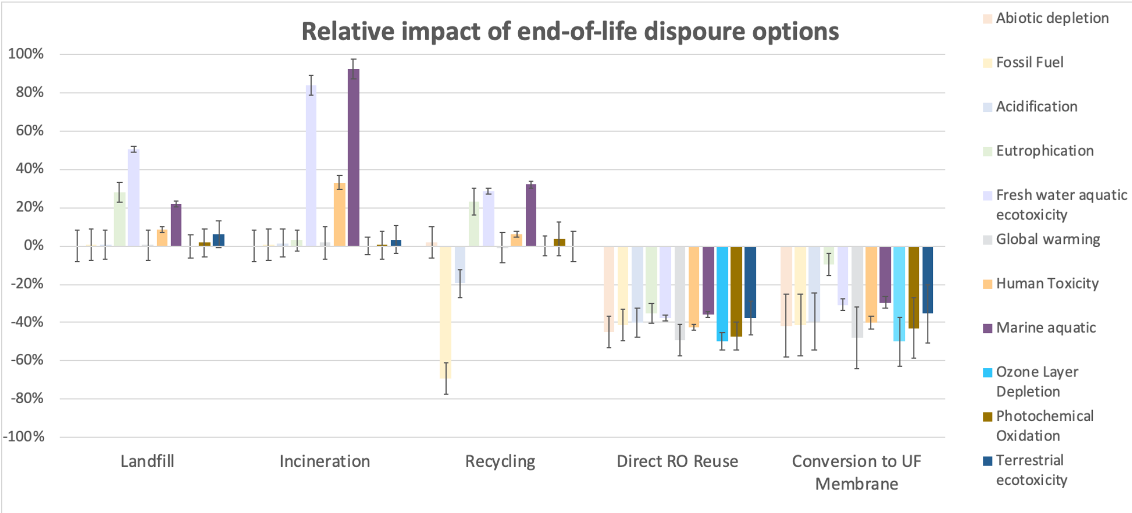
For RO, uncertainty analysis for all categories was kept between 3% - 16%, being the lowest ones related to Freshwater Aquatic Ecotoxicity, Marine Ecotoxicity, and Human Toxicity. For UF, uncertainty varied from 3% - 15%, being the lowest results also found for the same categories previously described. For both manufacturing processes, uncertainty was considered very low, considering the number of layers within the model. In fact, when compared to what was found by Lawler et al. (2015), it was possible to notice a reduction of the deviation and an inversion of accuracy, since they found the highest uncertainty levels for ecotoxicity, marine eutrophication, and human toxicity.

<u>End-of-life scenarios: EoL options</u>

End-of-life options for the RO membrane were first compared in terms of their relative impact in two scenarios: against (i) the fabrication of a novel RO membrane; and (ii) the disposure of an element to the landfill – current application in Brazil. In this section, this assessment counted only with the impacts related to the end-of-life option itself. That means, for example, that the impact associated with the disposal in those end-of-life scenarios that provide a second lifetime (recycling and chemical conversion) were not considered at this moment. They are further discussed in the following sections.

Results for this scenario are shown in Figure 4.5, where it was possible to notice that reuse and chemical conversion to the UF membrane presented environmental gains across all categories when compared to the membrane fabrication and disposition into the landfill. Their results were actually very close, since the major difference between both processes is the oxidation step. However, since it was considered that the oxidant solution was reused before being disposed of, the environmental impact of the oxidation solution was not relevant in the overall scenario.

Figure 4.5 – Relative impact of end-of-life scenarios compared to the production of a novel membrane



Source: Elaborated by the author (2022)

As for landfill, it presented the smaller uncertainty from Monte Carlo (around 3%). Also, it increased environmental burdens to the RO membrane life-cycle, especially for the following categories: freshwater aquatic ecotoxicity, eutrophication, and marine aquatic ecotoxicity. The strong impact into aquatic-related categories might be associated with the generation of leachate. If not properly collected and treated, leachate causes severe environmental impacts during its decomposition by releasing organic matter, trace metals and recalcitrant components, reaching both ground and surface and waters (Costa et al., 2019; Renou et al., 2008). In Brazil, this scenario is particularly augmented, since approximately 15% of the Brazilian waste is still incorrectly disposed of into open dumps (ABRELPE, 2021). In addition, leachate treatment based on biological technologies is widely applied in the country. However, this technique is not indicated for old and low biodegradable leachate, and this might contribute to spreading contaminants to the water bodies (Costa et al., 2019).

Regarding incineration, the most prominent impact categories are measured in terms of 1,4 dichlorobenzene equivalent, which can be associated with the possibility of releasing trace metals during the burning process (Kleib et al., 2021). On the other hand, it is important to highlight that the impact associated with the emission of greenhouse gases is diminished in this process, since the energy recovery that comes from the burned elements offsets other fossil fuels consumption (Lawler et al., 2015).

For recycling parts, environmental gains are considerably low when compared to reuse and chemical conversion since several components are not easily recyclable. In fact, approximately 20 % of the membrane module was considered to be sent to the landfill and 10%, related to waste glass, burned. Consequently, the same categories that presented a major environmental burden for landfill and incineration are also seen recycling but reduced due to its relationship with the waste weight. The recyclable part, on the other hand, was mainly focused on the polymers that compose the element. The greatest environmental gain came from recycling polyester since it is the major contributor for impact when fabricating the RO membrane.

Still in Figure 4.5, when comparing the EoL disposition options to the one currently used – landfill, it becomes clear that all of them presented environmental gains. This means that it is mandatory to of think other possibilities to RO discarded elements. In addition, it is important to highlight that the sensitivity analysis showed that results varied from 3 – 16% across all categories, being the lowest variation to Freshwater Ecotoxicity, Marine Aquatic Ecotoxicity, and Human Toxicity.

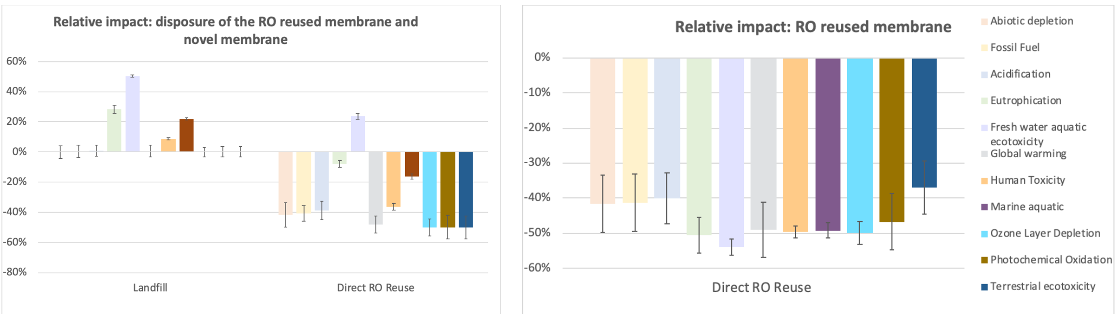
<u>RO membrane lifecycle: Reuse</u>

In addition to assessing the environmental impacts of different EoL themselves, it is also important to take into consideration that some of the assessed options can provide the element a second lifetime. Hence, by investigating their overall impact – including their final disposure after being used for the second time – is important to better understand their environmental gains when compared to the use of virgin membranes.

To this, Figure 4.6a presents the relative impact of a reused RO membrane in terms of the impacts associated with the production of a novel RO membrane into the landfill. To make comparisons possible, it is important to reinforce that the membrane offset defined is still applied.

By adding the final destination of the membrane in the RO element reuse process, the associated environmental impacts increase, especially for the following categories: eutrophication, freshwater aquatic ecotoxicity, and marine aquatic ecotoxicity. These are the exact same categories in which landfill disposal presents the greatest impact due to reasons discussed in Section 4.3.1.2. Still, giving the RO membrane a second lifetime significantly reduces its overall environmental impacts, especially when compared to the lifetime of a novel membrane that is not reclaimed (Figure 4.6b).

Figure 4.6 – Relative impact considering (a) the entire reused RO membrane lifecycle and (b) the entire reused RO membrane lifecycle, compared to the landfill disposure of a novel membrane



(a) ***(b)***

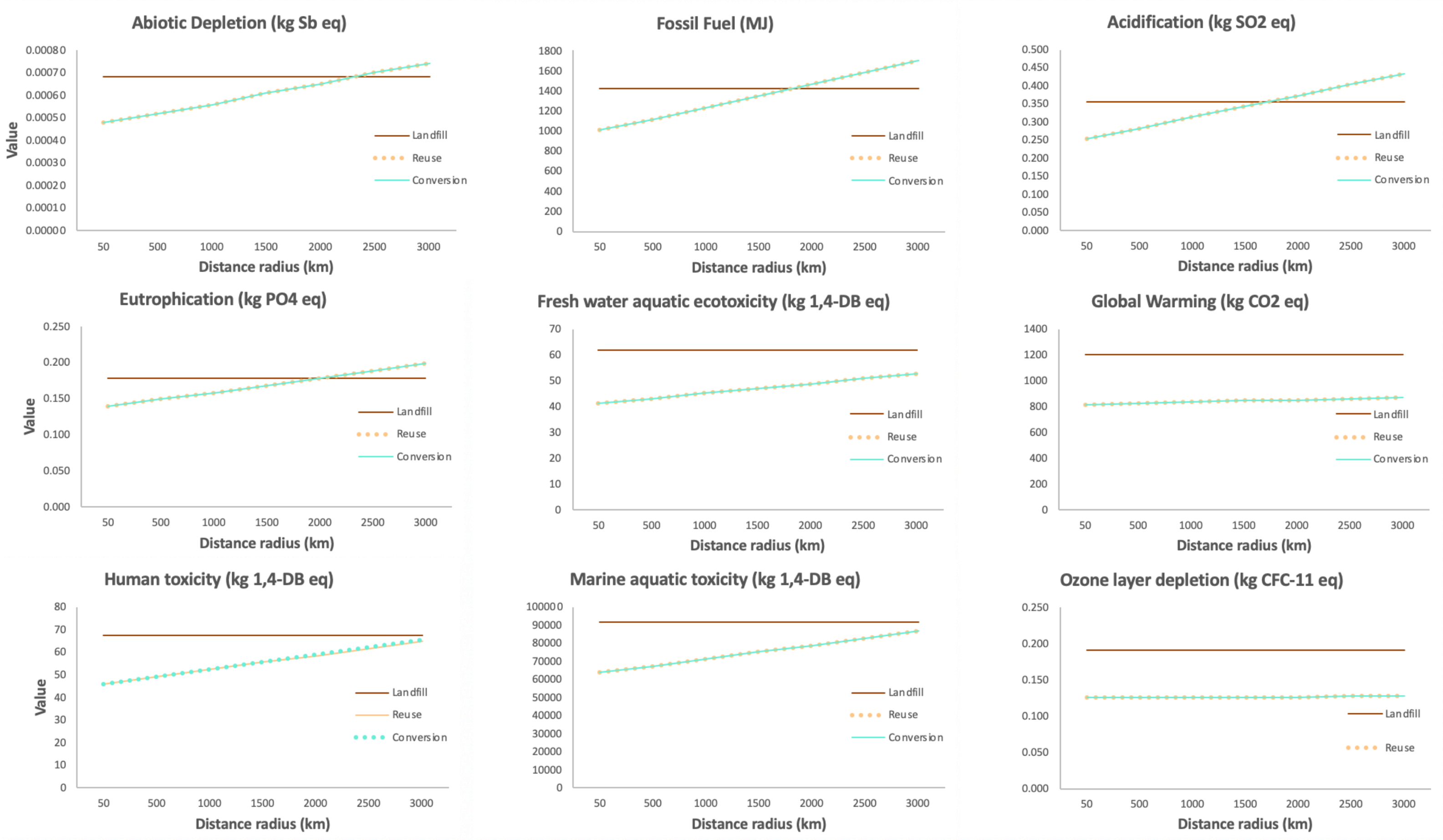
Source: Elaborated by the author (2022)

4.3.1.3 Transportation

Despite not presenting at first a prominent impact for the EoL disposition options, transport has a big potential for environmental burden. In Brazil this is particularly augmented, since it is the 5th biggest country in the world and majorly interconnected by roads. Hence, to address this question, the travelled distance for cleaning/reusing or converting a RO membrane was assessed. Transportation radius of 50, 500, 1,000, 1,500, 2,000 and 3,000 km were simulated using Open LCA and results are shown in Figure 4.7. At the threshold value of 1,500 km, it was noticed that transportation starts to negatively impact on the overall environmental gain from recycling or conversion, outcoming the environmental burden of the landfill disposal for 4 out of 9 categories. These findings converge to those reported by Lawler at al. (2015), where the authors noticed that using a second-life membrane for one year would be enough to promote environmental gain, regardless of the transported distance.

When comparing this distance to the reverse osmosis plants distribution in Brazil (Figure 4.8), it is possible to notice that this distance is already enough to cover half of the Brazilian territory, in the most densely populated region (91%). That means the potential of installing reuse/conversion facilities in Brazil is huge. For Region 2 (Figure 4.8), which represents the south-southern axis, both second-life reverses osmosis membranes and converted UF elements have significant application. Since 56.7 % of the population is concentrated in this axis (Agência Brasil de Comunicação, 2019), water and wastewater treatment are under pressure. In addition, most of the water treatment is done by physicochemical treatment, which does not address issues regarding emerging pollutants, such as pharmaceuticals. In 2019, Reis et al. (2019) assessed 12 samples collected in water treatment plants (WTP) in Brazil and noticed that 18 out of 28 investigated pharmaceuticals were present in drinking water and were not totally removed by standard treatment. On the other hand, several authors have investigated the potential of RO and NF of removing these components (Foureaux et al., 2019; Couto et al., 2020; Heo et al., 2020). Therefore, a potential alternative for reused RO membranes or their chemical conversion for NF, for example, is to complement the current treatment, hence increasing water quality.

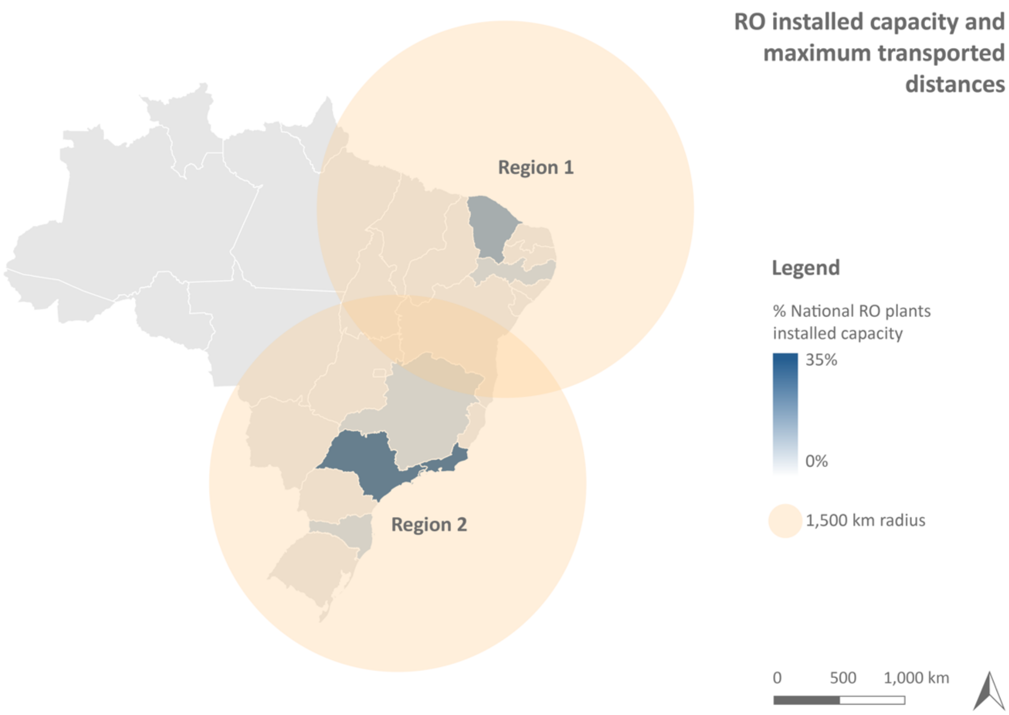
Figure 4.7 – Impact of distances on the overall environmental impact



Source: Elaborated by the author (2022)

Another possibility is to use converted RO-UF membranes as a substitute for the coagulation-flocculation stage. This would increase WTP robustness when treating surface water, since a physical barrier would be added to the process. Converted membranes have already been successfully tested to this application, as reported by Coutinho de Paula et al., 2017 and Moreira et al., 2022. They can also be potentially used for water in agriculture, a sector that is responsible for approximately 62 % of the water demand in Brazil (ANA, 2019) and lacks accessible technologies to water treatment. These are only a few examples of potential applications in the water sector, but these membranes can also be easily applied to domestic and industrial wastewater treatment.

Figure 4.8 –1,500 km radius from the main industrial parks that possess reverse osmosis plants in Brazil



Source: Elaborated by the author (2022)

In addition to all possibilities mentioned for the South-Southern axis, Region 1 (Northern) still presents further applications. According to the National Water and Sanitation Agency – ANA (2020), it is expected that around 41 million people will suffer from water scarcity in the Brazilian semi-arid region and its surroundings due to an increase in water demand (ANA, 2020). On the other hand, a less explored water resource in the region is underground water, since 70% of it has a salinity percentage that requires desalination before consumption (Feitosa and Diniz, 2011). Hence, second-life RO membranes or even converted RO to NF membranes might pose as alternatives to treat underground brackish water at lower costs. This application has already been investigated by Pacheco et al. (2018), where the authors found NF converted membranes presented similar performance as novel NF membranes, where desalination of brackish water was successfully attained. In fact, the Brazilian Federal Government has already encouraged low-scale desalination plants within the region (Brasil, 2012). Several improvements in this integrated solution have been made, and to reduce the associated costs to the element purchase by introducing second-life RO elements are in accordance to establishing more sustainable water security practices.

4.3.2 Socioeconomic assessment

Considering that sustainability is a multidimensional concept, to understand the socioeconomic viability of each of the EoL disposition options is also necessary. Starting with costs, Table 4.4 shows that landfill and recycling of parts have similar prices, around U$ 0.37/element. However, landfill disposal releases CO_2 eq to the environment. If the cost for capture and storage were considered on the overall cost, expenditure for this option would be around U$ 3.70, considering the capture cost estimated by the International CCS Knowledge Center (2022) at U$ 445/ton. Since recycling reduces CO_2 eq in the overall perspective, no additional cost for capture would need to be addressed, hence becoming an even more attractive solution. Finally, combining these results to the positive social aspects from recycling that will be further discussed, it does not make sense to consider landfill as an option for EoL-RO membranes disposition.

Table 4.4 – Associated costs to dispose / recover 1 RO element

EoL disposition option	Subcategory	Opex (U$/element)
Landfill	-	**0.38**
Incineration	-	**2.99**
Recycling of parts	-	**0.36**
Chemical conversion to the UF membrane	Capital cost amortization	2.76
	Water and chemicals	2.65
	Energy	0.14
	Labor	4.11
	Maintenance	0.50
	Total	**10.16**
Reuse as RO membrane	Capital cost amortization	2.10
	Water and chemicals	1.44
	Energy	0.11
	Labor	3.14
	Maintenance	0.39
	Total	**7.18**

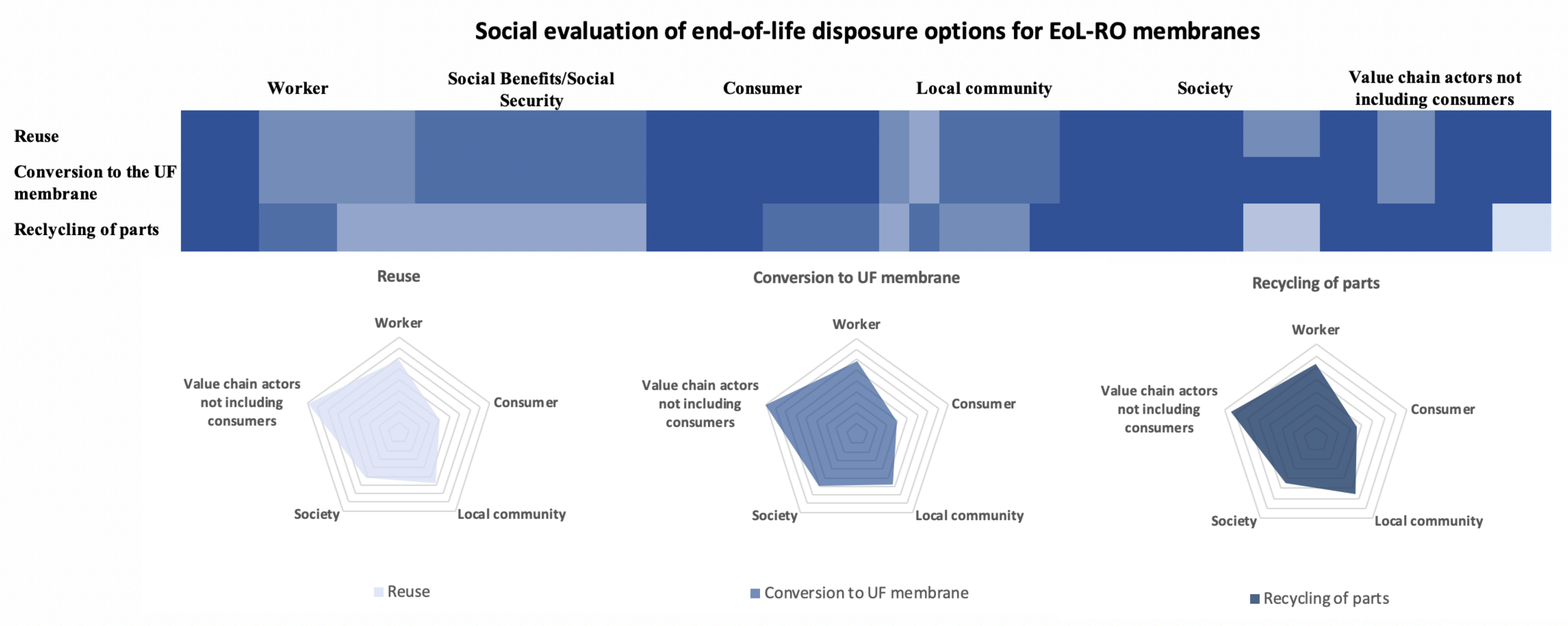
Source: Elaborated by the author (2022)

Despite presenting a far better alternative than landfill disposal, the associated environmental impacts to recycling of parts still are bigger than those seen for reuse and chemical conversion to the UF membrane (Figure 4.5). In addition, recycling has also few drawbacks that are not in the scope of the presented LCIA. If mechanically recycled, many polymers reduce their ductility. This means that the polyester is quickly transformed into lower-value products that can no longer be recycled (Lamberti et al., 2020). For polyester (the membrane main recyclable component), for example, the ductibility shrinks from 310 to 2.9% in only three recycling cycles (Payne and Jones, 2021), limiting its use. Hence, considering a longer timeframe, recycling has limited potential in terms of CO_2 eq capturing, since downcycling impairs closing the loop for these materials.

Regarding the social aspects (Figure 4.9), it was possible to notice that recycling of parts diverged more from the other two options in terms of social benefits, since it is not normally registered under a formal job with their associated social securities. In fact, recycling in Brazil is pulverized and arises from self-organization, where the major part of it is performed informally (IPEA,2016). Another category that it diverged corresponds to value chain actors, since the technology for recycling is well established and does not directly contribute to the development of novel technologies.

Additionally, since chemical conversion and reuse were considered to benefit from the same organizational structure, a company that employs people securing their social rights, their social impacts are rather similar. The main difference corresponded to the subcategory technology development (in Society). This is due to the fact that the possibilities for the converted membranes are bigger, as they can serve as a support layer for the fabrication of novel membranes. Still, despite presenting a bigger cost, chemical conversion to UF presents better social benefits, especially because it opens possibilities to the development of novel technologies.

Figure 4.9 – Heat and radar maps for the social evaluation performance



Source: Elaborated by the author (2022)

4.3.3 Potentialities and limitations of reusing and converting RO membranes

Despite its evident environmental gains, direct reuse and chemical conversion to UF possess some limitations. For both applications, it is important to notice that the membrane lifetime is significantly reduced during the second life. This is due to the fact that during its operation, the membrane suffers from irreversible fouling that remains even after cleaning procedures (Jepsen et al., 2018). In addition, the membrane selective layer also decreases its efficiency over time, which impairs the permeate quality.

Operationally speaking, the membrane lifetime is highly dependable on the operation conditions and the expected permeate quality. However, for very restricted processes, RO membranes can still be disposed of in conditions that could still easily meet other desalination applications. To this, one should have access to a detailed report of the membrane's performance during its first lifetime, which includes information regarding feed water, permeability, integrity, and rejection (Pontié et al., 2005; Lawler et al., 2012). By analyzing its previous use, it is possible to develop effective chemical cleaning procedures that will allow its safe reuse to other purposes.

As examples of closing-the-loop strategies for RO membrane reuse, one could cite that an element disposed of by the pharmaceutical industry might be cleaned and reused for tertiary wastewater treatment, water for irrigation, cattle, and even human consumption. In Brazil and other developing countries such a strategy is a very interesting way to promote advances in technologies used for water and wastewater treatment at an affordable price. This is due to the fact that most membrane elements are imported to these countries and they face several problems to reach a decentralized market and wastewater applications. The possibility to reuse discarded membranes from the industry, however, poses as a good opportunity to enter these market niches, especially if accompanied by an encouraging environment, with proper regulation and economic incentives.

When it comes to converting RO to UF membranes, the element configuration also needs to be taken into consideration. This is due to the fact that most RO elements are in a spiral wound configuration that is very sensitive to particulate. Hence, to act as an UF module, for example, it would be necessary to either (i) restrict its use to lower turbidity feed / reduce even more its lifetime or (ii) disassemble the element and reassemble the membrane into other configurations, such as plate and frame. This last option, however, needs further assessment on environmental and economic impacts to better understand the practical feasibility of it. Even with these drawbacks, the applicability of the discarded EoL RO membrane is very wide and needs to be assessed in terms of market feasibility.

Potentialities of EoL RO membrane are wide, and hopefully, using these closing-the-loop strategies may contribute to a more rational waste management in this area. Also, it might broaden membrane processes application since these alternatives allow other processes to benefit from a technology that under standard conditions would not be affordable.

4.4 Chapter conclusion

An environmental assessment was carried out to compare five EoL disposition options for RO the membrane. To this, the Open LCA was used to perform a life-cycle assessment. First, the comparison between six impact methods led to the choice of CMI-IA Baseline, since amongst the converged results, this method was the only one recommended for Brazil. During experiments, it was seen that the most environmentally friendly options were reuse and chemical conversion to the UF. Additionally, transportation distance radius for reusing/converting the RO membrane varied from 50 – 3,000 km, and at the threshold value of 1,500 km, 4 out of 9 categories a presented bigger environmental burden than disposing the membrane into the landfill. However, this distance is already enough to cover half of the Brazilian territory, which comprehends more than 90% of the country's population. Furthermore, the environmental assessment was complemented with a socioeconomic evaluation. Regarding the economic aspect, recycling of parts presented the lowest overall cost, but in terms of workers, it achieved a lower score than reuse and chemical conversion. This could be explained by the informal means that recycling is currently practiced in Brazil. As for reuse and chemical conversion, it is expected that formal jobs would be created to address these options, hence guaranteeing social security. Finally, when comparing results from all these dimensions, it becomes clear that there is a lot of room to improve sustainability in the desalination sector. By encouraging other disposition options beyond landfill, one can achieve environmental, social, and economic gains at the same time, using technologies that are already consolidated and can be used at industrial scale.

CHAPTER 5

TRANSITION PATHWAYS TOWARDS BETTER WASTE MANAGEMENT PRACTICES FOR THE DISCARDED REVERSE OSMOSIS MEMBRANES

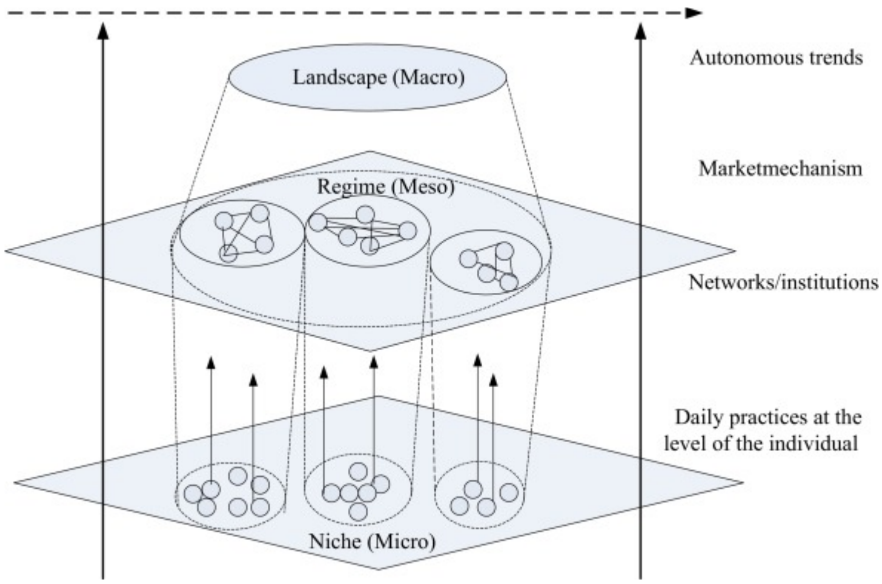
5.1 Introduction

As reported by Sénan-Salinas et al., 2020, desalination using reverse osmosis membranes will be responsible for discarding at least two million RO elements until 2025. Nevertheless, considering particularities in operational standards worldwide, this number is expected to be even bigger for developing countries. RO elements are made of high technology polymers, which brings interesting opportunities other than landfill disposition, such as direct reuse and chemical conversion to less selective membranes (UF, MF). However, apart from the technology needed to take advantage of what is currently called waste, it is also important to assess how governance is made and possibilities towards better waste management, aiming at more sustainable scenarios.

It is widely known that governance is central to shape transformations towards sustainability. Efforts to bring about sustainability surpass technology development itself and are deeply political, since these actions will mostly likely promote gains and losses, affecting actors in different ways (Meadowcroft, 2011; Patterson et al., 2017). Either potential outcome will arise from a series of major, interdependent changes in each of the three main elements normally discussed in governance: actors, technologies and institutions (Geels, 2004).

Socio-technical systems can be assessed through the Multi-Level Perspective (MLP), by analyzing the interaction of social factors or technological innovation within an analytical framework (Wu et al., 2021). Normally, it is applied by projecting future scenarios based on past performances or case-study analyses of socio-technical transitions (Foxton et al., 2010). Scenario building helps to comprehend how stakeholders might interact and the possible consequences of defined actions. The MLP divides the assessment into three levels: *niche* (micro-scale), system (meso-scale) and landscape (macro-scale), as shown in Figure 5.1 (Smith, Voss, Grin, 2010). The general multi-level transition is built upon *niche*-innovations that gradually create internal momentum; these *niche*-innovations, aligned with landscape changes, create pressure on the system and regime. This ends up creating a destabilization of the regime and consequent windows of opportunity for change (Geels, 2019).

Figure 5.1 – MLP on socio-technical transition



Source: Wu at al. (2021)

Since existing socio-technical systems are a combination of patterns, technologies, policies, cultural discourses, and infrastructures created in the last decades, innovations promoted by incumbent actors are normally incremental due to lock-in mechanisms (Geels, 2019; Klitkou, 2015). Hence, transitions last several decades and can be better assessed using transition pathways. According to Rosenbloom (2017), transition pathways can be defined as continuous changes of socio-technical systems configuration, under processes of political contestation. Although many transition pathways suggest a sequence of events, they are not deterministic (Geels and Shot, 2007). They can incorporate aspects of other paths and exhibit mixed characteristics in the end mixed characteristics (Geels, 2010; Geels, 2011). This happens because transitions are often shaped by the circumstances in which they are developed, and might bring different outcomes depending on the interaction of actors, technologies and institutions (Hansen and Coenen, 2015; Foxon, 2013). In this context, transition literature highlight that very often incumbent actors intend to slow down changes, since it might affect their assets and business model.

To help analyze transition pathways, Lindberg, Markard and Andersen (2019) proposed two dimensions to reflect potential conflicts within the way: the degree of environmental sustainability and the degree of disruption. The first one is related to whether actions are stronger or weaker to solve environmental issues, while the second discusses if the proposed pathway includes a profound change in the socio-technical arrangement. As for the degree of disruption, it is interesting to highlight that solutions that deviate from standard arrangements might bring radical innovations to socio-technical systems (Geels, 2019). According to Kemp (1998), they tend to emerge in small *niches* at the periphery of existing systems, and are normally developed by entrepreneurs, start-ups, or other outsiders. These small *niches* provide shelter so these ideas can be nurtured without the intervention of the mainstream market, so they can further be the seed of sustainable innovations (Schot and Geels, 2008).

Characteristics regarding the degree of sustainability and disruption of the proposed changes might then help understand how actors would act in each of the proposed pathways. This is due to the fact that less disruptive strategies can maintain a certain socio-technical order well established, while highly disruptive strategies ask for newly structural organization. Hence, using this two-dimension analysis improves understanding the behavior of stakeholders, which might help comprehend in which of them changes will be more or less meaningful. By comparing new proposed situations with the *status quo*, it is then possible to identify where and how to act. For instance, in the membrane context, if converting the EoL-RO membrane into UF elements presents itself as a highly disruptive sustainability transition, it is possible to identify the main challenges for each of the involved stakeholders and define priority aspects that need to be further developed (for example, evaluate if it is necessary to improve the legal framework that regulates such activity or even the degree of technological maturity of the proposed solutions).

Seeing this discussion, this chapter proposed, from an MLP perspective, transition pathways towards more sustainable practices in the EoL-RO waste management context, based on the main findings from the LCA study of Chapter 4. This is an important contribution to the field since, according to Gillard (2016) and Geels (2019), transition pathways normally assume that green innovations are often positive. Hence, they do not assess eventual impacts and outcomes of these transitions in terms of actual sustainability, which is normally left to life-cycle analysts. By integrating LCA and transition pathways, this chapter not only intends assess changes for an actual greener scenario in the future but also promote the association between two areas of study.

To this, a scenario of ending landfill disposal and sending all EoL-RO elements to reuse/conversion and recycling was proposed by 2050. The EoL disposition options evaluated in Chapter 4 were classified according to their level of disruption (Lindberg, Markard and Andersen, 2019). Also, a transition pathway containing four phases was proposed and assessed in terms of actors and their prospective conflicts, maturity level of technologies and institutions, considering a system level perspective within MLP and the level of disruption of each phase. By analyzing the scenario, expectations are to shed some light on the current waste management practices and which are the main potentialities and drawbacks from a governance perspective that might shape transformations in EoL-RO membrane waste management.

5.2 Material and methods

5.2.1 Legal background and characterization of the current scenario

As a first step to understand the background of the EoL RO membranes waste governance, a literature review regarding waste management in Brazil was carried out. The main national legal framework was highlighted and included environmental laws, procedures, and financial incentives. State legislation regarding waste management from São Paulo, Rio de Janeiro, Minas Gerais, and Espírito Santo, located in the region with the biggest concentration of EoL RO membranes were also reviewed.

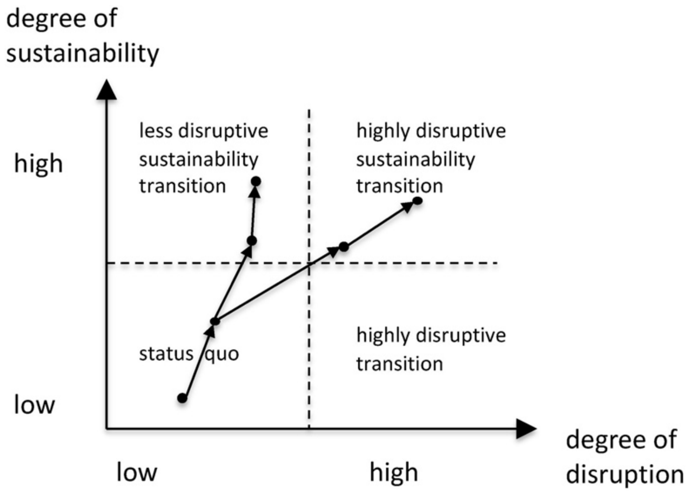
In addition, the current scenario of EoL-RO membrane waste management at a system-scale was described in terms of mechanisms and actors. This involved pointing out who is responsible for the waste disposure and how it is currently done. Furthermore, the main actors involved in the current waste management strategy, as well as novel stakeholders that might be needed to achieve more sustainable scenarios were mapped and their current activities in the waste management chain were highlighted.

5.2.2 Classification of the disposure waste options

After characterizing the current situation, the disposure waste options assessed in Chapter 4 were classified according to the degree of sustainability and disruption from *status quo* (Figure 5.2). To this, they were pairwise compared and classified as more environmentally sustainable or not, according to the results from the LCA study

(Chapter 4). Additionally, they were also compared in terms of their degree of disruption. To this, aspects of the socio-technical arrangement were taken into account. The most disruptive option was considered the one that required the creation of stakeholders that do not even exist in a structured form and further development of technologies, while the less disruptive was closer to the current arrangement and did not demand novel technologies.

Figure 5.2 – Dimensions to analyze potential conflicts in transition pathways



Source: Lindberg, Markard and Andersen (2019)

5.2.3 Scenario building and transition pathways

To propose transition pathways towards more sustainable practices for EoL RO membrane waste management, reuse / conversion and recycling of parts were taken as final disposition references. Landfill disposure, the most common destination in Brazil, was taken as a starting point (*status quo*). To help map win, losses, challenges and how actors interact within the process, a theoretical proposition of percentages of waste destination for each disposition option was made, as shown in Table 5.1.

Table 5.1. was built upon the following premises: seeing the results from the LCA study, it is urgent to walk away from landfill disposal and they need to initiate as soon as possible. However, reuse and chemical conversion are initiatives that are not yet well-structured in Brazil. On the other hand, recycling of materials is a well-established technology, that even demanding adjustments to receive EoL-RO elements, was considered to be closer to what can be done in the next few years. Hence, reuse / chemical conversion was virtually considered as 0% during Phases 1 and 2 because

they would be seeding during this time, while recycling would increasingly be receiving EoL-RO elements until they are not directly sent to the landfill anymore. Of course, since the membrane element is not 100% recyclable, part of it would end up in the landfill, but only after being sorted by the recycling cooperatives.

Table 5.1 – Progressive increase to assess transition pathways towards more sustainable practices in the studied context for the Brazilian scenario

Context	% of residues sent to specific destination		
	Reuse / conversion	**Recycling of parts**	**Landfill disposure**
Current scenario in Brazil	0%	0%	100 %
Phase 1	0%	50%	50%
Phase 2	0%	100%	0%
Phase 3	20%	80%	0%
Phase 4 Final scenario	80 %	20%	0%

Source: Elaborated by the author (2022)

After being properly developed, reuse / conversion would enter the market in a more structured form during Phases 3 and 4, where there would be a decrease in recycling of parts. For all phases, main challenges, wins and losses were discussed, as well as the role of each of the existing actors.

In this proposed pathway, it is important to highlight that the main goal of this chapter is to map and understand how actors interact and would probably react when defining sustainable goals to EoL-RO waste management. Additionally, as stated by Geels (2010, 2011), transition pathways actually taken are normally a mix of several possibilities that are shaped according to reality. Hence, from an *ex-ante* perspective, this chapter aims at proposing a reflexive exercise regarding the environmental governance of EoL-RO elements. The focus is to initiate discussion towards the subject by mapping and highlighting eventual conflicts rather than actually defining a transition pathway, once this latter needs the participation of other actors to be properly built. So, the transition proposed in Table 5.1. serves as a background to this discussion.

5.3 Legal background

In Brazil, the main regulatory framework for waste management is the *Política Nacional de Resíduos Sólidos - PNRS* (Solid Waste Nacional Policy). It was enacted by the Law nº 12.305/2010 and it is also based on the waste hierarchy. The *PNRS* presents guidelines that aim at encouraging a less linear structure, including, for example, mandatory reverse logistics for specific products (products whose packaging is classified as dangerous – such as those from pesticides, batteries, tires, oil and its packaging, lamps and electronic devices) and waste collection for recycling (Brasil, 2010a).

The PNRS generally encourages sustainability, indicates that solutions must pursue eco efficiency, highlights the importance of environmental impact assessment and circular economy. It also emphasizes the National Plan for Solid Waste as a major guideline to understand the current scenario and establish targets to promote more integrated waste management practices. Finally, another point that deserves attention is the fact that the responsibility for the product life-cycle is shared amongst producers and consumers (Brasil, 2010a). This is a major difference between Brazilian and international regulation, since in Europe, for example, the responsibility lies upon the producer, and directly affects waste management (Azevedo, 2015).

Constructing the shared responsibility concept was one of the major challenges when defining PNRS. According to Almeida and Gomes (2010), the *Confederação Nacional da Indústria* (CNI – Industry National Confederation) suggested the approach, arguing that if the responsibility for waste was diluted into the value chain, so would be the costs, and that the European model has already proved to be expensive for the industry. NGO’s agreed with the industry sector, arguing that this would be an opportunity to include the association of collectors of recyclable material in the producing chain (Almeida and Gomes, 2010).

Another particularity in Brazil, shared with other developing countries is the substantial amount of informal collection and recycling (IPEA, 2016; Kalmykova, Sadagopan, and Rosado, 2018). The commitment to recycling in the country is directly linked to the aggregate value of residues. This importance is actually also recognized at Law nº 12.305/2010, that states that the work from associations and collectors should be prioritized when hiring services for public cleaning (Brasil, 2010a). In addition, still in 2010 the Decree nº 7.405 established a program to fund and invest in cooperatives and associations funded by collectors (Brasil, 2010b).

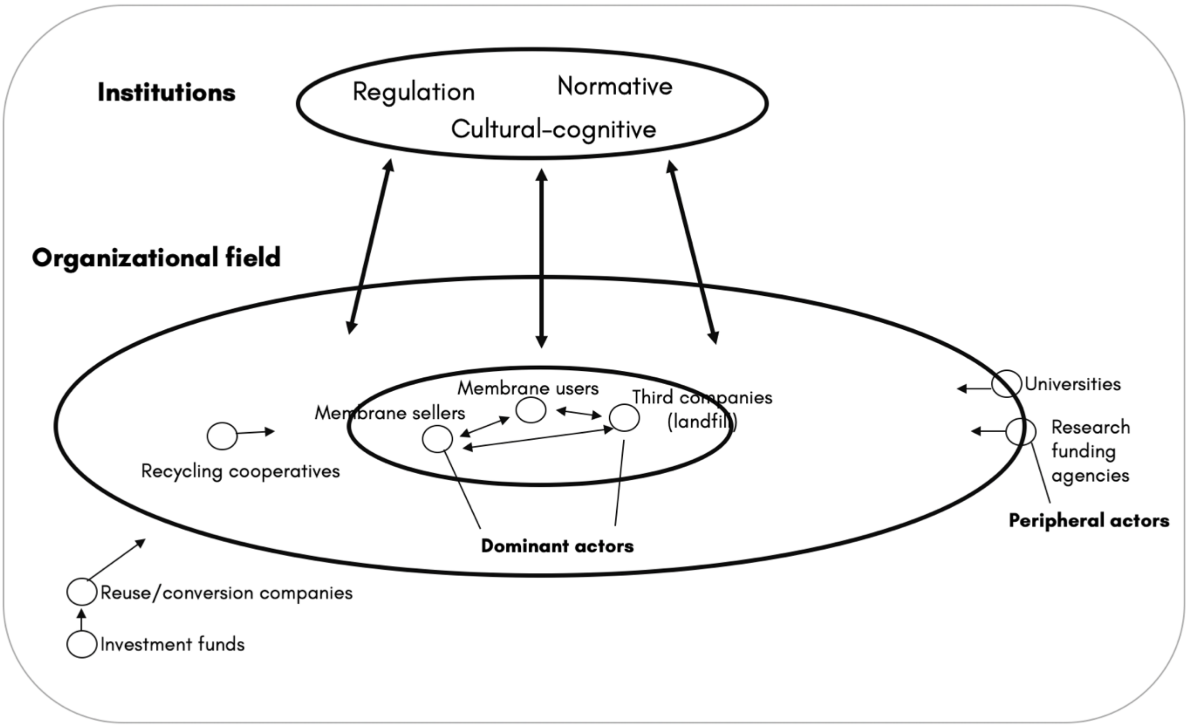
Besides the general guidelines stated in Law 12.305, before disposing of waste it is first necessary to classify it, and to do so, one should follow the NBR 10.004:2004 (ABNT, 2004). It defines procedures for classification and specific characteristics that will classify waste as dangerous, non-inert, or inert. Each type of residue has its own disposure rules. For the EoL-RO membrane, since it is classified as inert, it can be disposed of in the landfill (Lawler et al., 2015). Other regulatory rules that complement the PNRS and must be followed for industrial waste are NBR 12.235 (ABNT, 1992) – Dangerous Residues Storage, NBR 11.174 (ABNT, 1990) – Inert and Non-inert Residues Storage, NBR 13.221 – Terrestrial Transportation of Dangerous Residues (ABNT, 2021), and Law 6.938 – National Policy of the Environment (Brasil, 1981). Furthermore, in 2021 a law that allows a deduction of taxes from companies that invest in recycling was also enacted (Brasil, 2021b).

In addition to the national guidelines, each state has its own particularities. Considering the ones that comprise the Southeast region, the most densely populated and with the biggest amount of industries that use RO membranes (São Paulo, Rio de Janeiro, Minas Gerais, and Espírito Santo), it is possible to draw a parallel. All of states have their own State Policy of Solid Residues (ALESP, 2005; ALERJ, 2003, ALMG, 2009, ALES, 2009) that date before the enactment of the Law nº 12.305/2010 and the discussion they present are based on the same sustainable principles stated by the national law. Most probably they inspired the national law and therefore are aligned with each other.

5.4 Current scenario characterization

While proposing transition pathways, it is important to describe from where changes will start, which also involves understanding the main actors and their organization (Figure 5.3). In the EoL RO management context, the membrane elements are currently bought from industry users that might contract additional assistance from the membrane sellers for operation and maintenance of the industrial plants. When operating controls demonstrate that the membrane is not performing accordingly, it is substituted, and the end-of-life element is discarded. Normally, this disposition is made by a third company in the industry sector. Since EoL RO elements are considered as inert material, they are expected to be collected and disposed of in regulated sanitary landfills, and these third companies normally perform transportation and disposition.

Figure 5.3 – Current organization of the socio-technical system considering landfill disposure



Source: Adapted from Geels (2020)

Parallel to what is currently done, several researches have been conducted to define better waste management strategies. As far as the author is concerned, they are at laboratory and small pilot scale in Brazil, mostly performed by Universities and funded by funding agencies.

Additionally, other actors that are still not involved in this particular waste management but are prospective ones should be highlighted: (i) cooperatives of waste collectors and recycling; (ii) companies involved in preparing the EoL membranes for their second life; and (iii) investors of new business model of reusing / converting RO membranes. It is relevant to note that this last actor might actually arise from the membrane sellers themselves, since these second-life membranes will also be commercialized. However, in this study, it will be considered an extra stakeholder. Finally, the government also plays a relevant role in waste management, since it is responsible for defining goals and incentives to membrane recycling. A summary of the actor's main attribution and current status is shown in Table 5.2.

Table 5.2 – Actors involved in the EoL RO membrane waste management and their role

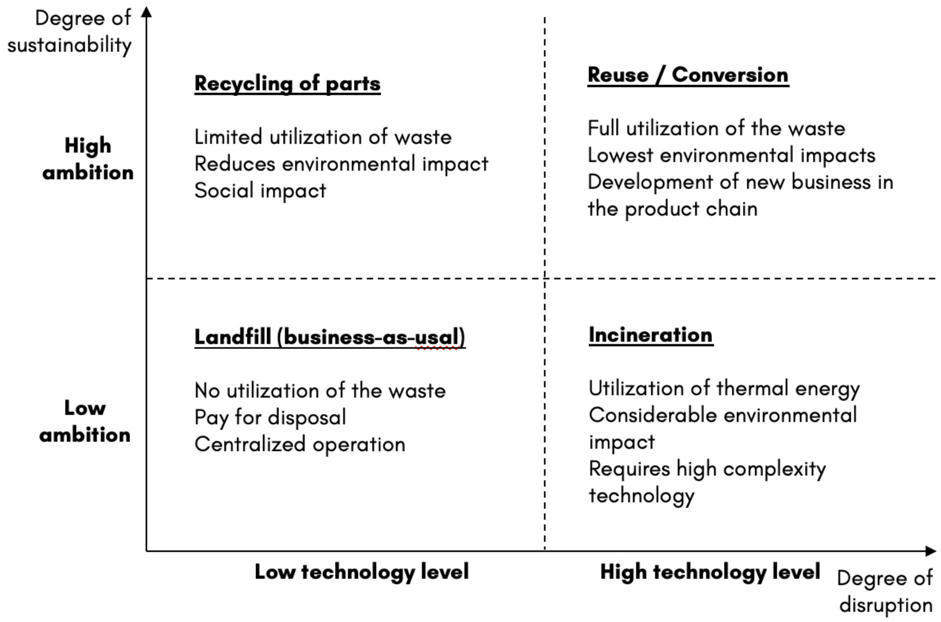
Actor	Main attributions	Current status
Membrane seller	Membrane import and supplying; Technical assistance and even operation, if contracted.	RO and UF membrane supplier.
Membrane buyer/ user	Membrane purchase; Operation and maintenance of the plant; Contract for supply and disposure of the elements.	Main user of the UF / RO elements; Purchase of novel membranes; Disposure in the landfill through third companies.
Cooperative of waste collectors and recycling	Collection and recycling of the elements; Pricing the service; Disposure of the non-recyclable parts into the landfill through third companies.	Do not participate into the process.
Company that performs landfill disposure	Transport and disposure of waste to the landfill; Pricing the service; Maintenance of the landfill / payment for disposure;	Contracted by the membrane buyer to dispose 100% of the membrane elements.
Companies that prepare the element for its second lifespan	In Brazil, such initiatives are still in their very beginning and are not know; Still it was not possible to find initiatives from membrane manufacturers towards this opportunity.	Do not participate into the process.
Universities	Research regarding the technical feasibility of reusing / converting EoL RO membranes.	Have already published their work in laboratory and pilot scale (Coutinho de Paula et al., 2017; Moreira et al., 2022; Moreira et al., 2021).
Funding agencies and investors	Support for research regarding the feasibility of reusing / converting EoL RO membranes.	Support of past and ongoing research regarding the theme.
Government	Establishment of legal background to promote more sustainable practices.	There is no direct legal background for this theme but it is regulated by Laws nº 12.305, 6.938 and states laws highlight in Section 5.3.

Source: Elaborated by the author (2022)

5.5 Classification of EoL disposition options

As stated in the methodology section, a more sustainable scenario is planned to be achieved by 2050, and it was broken down into in 4 main phases (Table 5.1). The goal is to walk from the current situation to the reuse/conversion of 80% of the membrane elements. 20% will be considered to be sent to recycling, since many of them might present damages that hinders their second life. To help assess paths, the current disposition options were classified in terms of their environmental sustainability and degree of disruption, as shown in Figure 5.4.

Figure 5.4 – Classification of end-of-life disposition options for the RO membrane in terms of their degree of sustainability and disruption



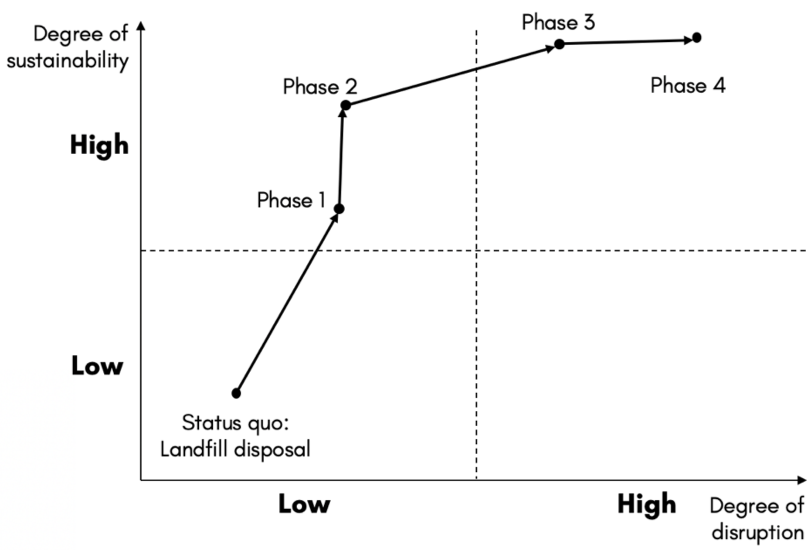
Source: Adapted from Lindberg, Markard and Andersen (2019)

To build this classification, findings from Chapter 4 were taken into account, where reuse and chemical conversion were found the friendliest solution from the environmental perspective, followed by recycling of parts. Incineration, on the other hand, did not present environmental gains when compared do business-as-usual, hence being classified as low ambition for environmental sustainability. Regarding the degree of disruption, reuse/conversion requires a new business model, in which new companies or spinoffs from current actors are needed to adequate the elements for their second-life. In addition, the technology developed so far is still in bench and pilot scale in Brazil, and needs to be further assessed in terms of its *Minimum Viable Product* (Coutinho de Paula et al., 2017, Moreira et al., 2022). As for incineration, since it requires a technological structure that is more complex than physical transformations of recycling, it was also classified as high technology level. Recycling of materials was defined as low technology level, since it is a well-established activity and does not require complex equipment such as incineration.

5.6 Transition pathway

A transition pathway composed of 4 phases was proposed as a background to assess understanding the main challenges and conflicts for bringing EoL-RO elements waste management closer to sustainability. Phases were proposed according to the current level of maturity of each EoL option, considering both technology and institutional aspects (Figure 5.5). Hence, phases 1 and 2 propose a transition from landfill disposal to the recycling of parts, which already represents an expressive environmental gain (as stated in Chapter 4) while not needing big organizational disruptions within the institutional scenario. For big organizational disruptions, one can understand that it was considered that the recycling cooperatives already exist, as well as their physical infrastructure, including organized in a national centralized form. This, however, does not mean that there would not be challenges to be addressed, such as training for membrane disassemble, diagnosis, waste sorting and processing, and are further discussed in Section 5.6.1. As for phases 3 and 4, new stakeholders will also take part in the chain bringing new technologies and institutional arrangements. Each phase and their respective sustainability / disruption degree are further discussed in Sections 5.6.1 and 5.6.2.

Figure 5.5 – Transition phases classified according to their level of disruption and environmental sustainability



Source: Adapted from Lindberg, Markard and Andersen (2019)

5.6.1 Phases 1 and 2: from *status quo* to 100% of recycling materials

In phases 1 and 2, recycling of materials is supposed to be implemented, and at the end of phase 2, the main goal is to send 100% of the membrane elements to recycling, where the cooperatives would be responsible for collecting, sorting and directing waste. This justifies the straight arrow connecting phases 1 and 2 in the upper left quadrant of Figure 5.5, which means that (i) an improvement in environmental sustainability from moving from landfill to phase 1 (beginning of recycling); and (ii) a straight line increasing sustainability by augmenting the amount of recycled material while using the same technology (Phases 1 to 2). It is important to highlight though that 100% of elements sent to recycling does not mean that the elements will not reach the landfill as they are not 100% recyclable, but they will pass through sorting before being discarded.

During this transition, the main affected actors are the membrane user, the recycling cooperatives and the company that performs landfill disposure. For the membrane sellers, changing the destination in this case does not affect them directly, but can be seen as a good strategy to walk towards more sustainable scenarios without much disruption in the institutional order. In addition, since recycling is a well-established technology for polymers – which represent 63 % of the membrane element, it does not bring any technological disruption to the waste management chain either. Table 5.3 presents a summary of the main actors directly involved, their current position and what is expected from them during phases 1 and 2.

Table 5.3 – Actors involved in phases 1 and 2, their current status and prospective changes

Actor	Current status	Phase 1: Prospective changes	Phase 2: Prospective changes
Membrane seller	Membrane supplier; Consultancy for operation and maintenance (if in contract).	To recycle material elements might be seen as an improvement in the product chain without affecting business-as-usual for selling. However, membrane sellers can help training cooperatives so they can learn how to actually recycle the elements.	
Membrane buyer/ user	Membrane purchase; Operation and maintenance; Disposure in landfill through third company.	Articulation with local cooperative of waste collectors; Reduction of contract with landfill disposure companies.	Reinforcement of partnership with local cooperative of waste collectors; End of contract with landfill disposure companies.
Cooperative of waste collectors and recycling	Do not participate in the current product chain.	Pricing the new service; Preparation to receive and process the novel material; Defining partnership with companies responsible for landfill disposure for the non-recyclable material.	Increase of processing capacity; Optimization of processes and prices; Increase partnership with companies for landfill disposure of non-recyclable material.
Company that performs landfill disposure	Transport and disposure in landfill.	Reduction of direct contract with membrane user companies; Search for partnership with recycling cooperatives.	End of contracts with membrane user companies (regarding disposure of elements); Strengthen partnership with recycling cooperatives to receive more non-recyclable material
Government	Supporting laws that encourage the recycling chain (Law 12.305 and Decree 7.405/2010)	Increasing/encouraging investment in recycling cooperatives and membrane sellers / users to support recycling strategies	

Source: Elaborated by the author (2022)

As main challenges in this transition, it is important to highlight the role of membrane users. Since they are responsible for hiring third companies to promote the disposition, they would be responsible for shifting from landfill to recycling cooperatives or companies. Since cooperatives have been explicitly cited in PNSR, the scenario took into account this structure to perform recycling. In this context, it is worth mentioning the structure of these cooperatives. They were born from self-organization of collectors and nowadays are nationally connected through the *Movimento Nacional dos Catadores de Materiais Recicláveis* (MNCMR – National Movement of the Recyclable Materials Collectors). The increase in their organization is helping them achieve more influence to bargain prices and national recognition, which leads to specific legislations, like the Decree nº 7.405/2010 and their articulation with other movements (Brasil, 2010b, IPEA, 2017). Despite improvements of the last decades, recycling is still mostly performed informally. Additionally, even cooperatives struggle with labor rights and management infrastructure, which can be challenges that need to be addressed when shifting the waste destination.

The relative informality in which recycling is still performed in Brazil also brings another challenge to the EoL-RO element's waste management during several recycling phases. First, membranes are products used in diverse industrial contexts and can present different types of contamination. This fact needs to be carefully addressed to avoid exposing recyclers to these materials. Secondly, they are not well-known for these niches such as domestic recyclable waste. Hence, sorting and segregation of these materials need to be taught, and this knowledge is still restricted to membrane producers, few users and scholars. In fact, when recycling in cooperatives to what is currently reported for membrane recycling worldwide, a major difference is observed. Recycling is currently performed by means of structured companies, not cooperatives, such as MemRe (Germany) and WaterSurplus (USA). They present a deep knowledge of membrane processes and use them to perform services such as proper disposition, autopsy and even reuse (MemRe, 2022; WaterSurplus, 2022).

Recycling in Brazil has another perspective from developed countries, and might be performed differently from what is currently reported. However, to promote recycling taking into account the Brazilian background, it is necessary not only to direct waste to recycling, but also to provide training and support so the cooperatives can actually act.

This can be even promoted by membrane fabricants, since encouraging recycling can be seen as a social technology with institutional incentives.

Another point that should be highlighted is that by decreasing the amount of waste sent to the landfill, companies that work in this niche might end up losing their market. Even still receiving the non-recyclable part, they will most probably lose revenues, and eventual interference of these companies will also need to be addressed when assessing their relationship with the membrane users. This type of conflict has been reported in the literature, and Patterson et al (2017) highlight that as a consequence, these actors work to difficult eventual chances, the reason why incremental changes might also be difficult to achieve. In this situation involving winners and losers, it is worth mentioning that policies can support and protect niche innovations or constrain incumbent technologies (Lindberg, Markard and Andersen, 2019). However, policies currently established in Brazil are still not enough to influence transformation. Despite encouraging recycling and cooperatives, the lack of actual improvement in the collector's conditions and specific goals towards walking away from landfill disposal impairs an effective pressure under the current scenario, which might difficult transition.

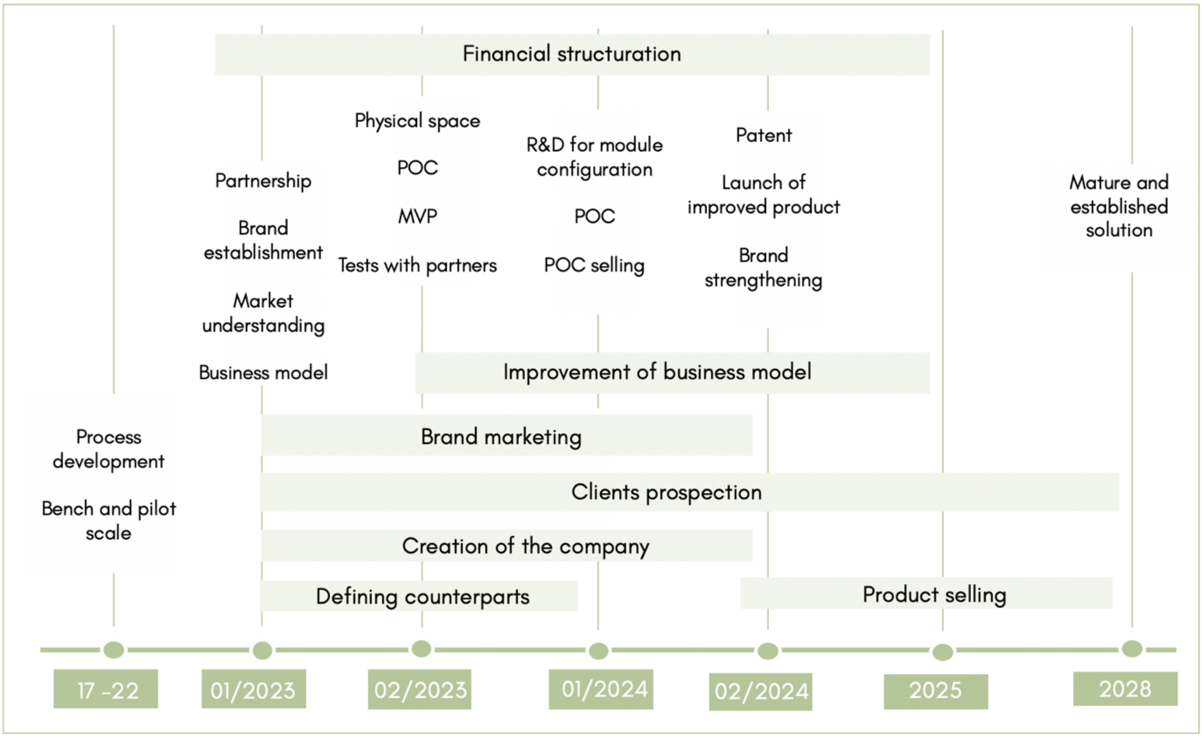
Despite not explicitly participating ins phases 1 and 2, the timeframe in which these phases are developed also need to be explored in parallel. This is due to the fact that the next step – to give the elements a second lifespan – counts with disruptive aspects both in technological and institutional arrangements. Over the last decade, membrane chemical conversion has been addressed in terms of empirical research for several research groups worldwide (Contreras Martínez, 2021; Coutinho de Paula et al., 2017; Lawler et al., 2013; Sahuquillo et al., 2015; Rosa, 2012; Soice et al., 2004). In addition, few commercial endeavors started in the business. In the USA, for example, WaterSurplus buys EoL-RO membranes and performs their chemical conversion to further commercialization, while in Japan, Organo Corp has patents regarding recycling and regenerating RO membrane elements (Organo Corp, 2003). In Brazil, when searching at the INPI (National Institute of Industrial Research) platform, there is no national patent registered regarding the process, and the national research is at bench and pilot scale. Hence, it is important that third-party stakeholders start developing the Proof of Concept (POC) and the Minimum Viable Product (MVP) of these technologies and deepen knowledge regarding market challenges and opportunities, to understand if they are commercially viable.

It is interesting to highlight that the creation of these new companies might fit to what was classified by Kemp (1998) as radical innovations. This can be justified by the current distance between what is actually performed in the market regarding the disposal of EoL elements and what is to be proposed by these start-ups. Additionally, Geels (2019) states that this type of innovation could form the seeds of sustainability transitions.

5.6.2 Phases 3 and 4: From recycling materials to second-life elements

To start transforming EoL-RO membranes into other membrane elements that can be reused, it is important to add another actor into the waste management chain: the membrane converters. They can be either a specialized company or a spinoff from membrane sellers/fabricants. In both cases, they would be responsible for selecting the elements that can be reused/converted, perform their transportation into the facility, perform transformations and resell these elements to the market. In addition, as stated in Section 5.6.1, they need to be seeded during the elaboration of phases 1 and 2, since the technology is still in bench and pilot scale in Brazil. Considering the time frame, Figure 5.6 presents standard steps normally required to transform technologies into business.

Figure 5.6 – Flowchart of reuse/conversion of the EoL-RO elements



Source: Elaborated by the author (2022)

Considering further possibilities of using EoL-RO membranes discussed in Chapter 4 and their maturity level in Brazil, it is also important to promote further research and development of novel chemical conversion techniques. Hence, Universities and Research and Development R&D divisions are key actors within the process and, as a consequence, so are funding agencies.

In terms of roles performed by the involved actors while transitioning to phases 3 and 4, it is worth mentioning that by adding other stakeholders in the process, others might bring resistance. Transforming EoL-RO membranes into UF membranes might first be seen as a possible reduction in revenues related to the sale of novel UF membranes, which cannot be seen as a good outcome for membrane sellers. The same can be inferred to second-life RO elements. Considering that the market of membrane fabricants is quite centralized, they can have a prominent influence within the chain, using strategies such as exclusive supplier or even collecting discarded modules when replacing for new ones. In addition, these companies can also participate into the maintenance of industrial plants, and considering that Brazil still presents limited specialized workforce in the sector, membrane users might encounter difficulties in operating their plants on their own when using different membranes.

Despite this setback, it is important to notice that by decreasing the value of second-hand UF membranes, novel market niches can be achieved and might not directly compete with novel elements, hence being possible their coexistence in the market. As an exercise to understand the size of such companies specialized in converting membranes, two scenarios can be simulated. In the first, let’s consider that 80% of the annual average disposition calculated in Chapter 2 (180 ton/year) would be converted into UF membranes. For an 8-inch membrane diameter weighing 13.75 kg, that would represent approximately 14.000 disposed modules/year. Considering a water flow rate for the converted module into UF of 0.022 $m^3/m^2.h$ and a membrane area of 37.2 m^2 (Filmtec, 2021), this represents an installed capacity of 10,650 m^3/h. If the membrane lifespan of converted membranes is of two years, that means that the maximum installed capacity of converted UF membranes would be approximately 20,000 m^3/h, since after commissioning new plants in the two first years, the newly converted membranes would replace the old ones.

When taking this installed capacity into consideration, there are several market niches that can benefit from it. The food and beverage industry, for example, is responsible for consuming 58.9% of the water used in the industrial sector in Brazil (ANA, 2019). Since the quality of their products depends on the water used, they normally present higher standards for water treatment, and also need to treat their effluents, estimated at 50% of what is input into the process (ANA, 2019). According to FIESP/DEPECON (2020), more than 95% of food and beverage companies are small and medium size, which means a water demand between 4.0 to 33 m^3/day and low investment for water and wastewater treatment. For them, novel membranes might not be economically viable, but reducing their costs and maintaining quality, such as proposed by second-life RO membranes can be a feasible solution. Considering this, the amount of discarded EoL-RO membranes would be able to serve around 15,000 fabrics with a higher water demand of 33 m^3/day, such as microbreweries.

Considering the installed capacity of key players in the membrane market, this market niche might not be worthy considering its fragmentation. However, it is still representative for new business model within the waste management chain and needs to be further assessed in terms of economics. At last, it is interesting to highlight that 70% of these companies are located in the southeast region of Brazil (ANA, 2019).

This means they are in the area where transportation does not negatively impact the overall environmental gains from reusing / recycling.

In addition to supplying second-life membrane elements, another point that needs to be further addressed by prospective parties is the technical assistance issue. As shown in Chapter 3, low maturity in the operation might lead to a decrease in membrane lifespan. Since these medium companies normally do not have specialized personnel, it might be interesting also to assist operations by providing this type of service.

Another stakeholder that would be impacted by this shift towards reusing membrane elements are the recycling cooperatives. In this case, they would receive the membrane elements after their second-life. This shifts their relationship with novel membrane buyers and establishes new ones. However, considering a fragmented prospective market to second life elements, logistics in collecting and treating this material might pose as a challenge. Finally, in the last years, the Brazilian government has increasingly promoted policies and financial incentives to start-ups dedicated to developing new technologies. As an example, Law 11.996 was enacted in 2005 (Brasil, 2005). The government and their partners have been trying to make Brazil attractive to this type of endeavor, and this might have a positive impact in transitioning during phases 3 and 4. A summary of the main stakeholders, their activities and prospective conflicts are shown in Table 5.4.

Table 5.4 – Actors involved in phases 3 and 4, their main roles and prospective conflicts

Actor	Phases 3 and 4: Main roles	Prospective conflicts
Membrane seller	Membrane supplier; Eventual service of operation and maintenance.	Might experience a slight decrease in novel membranes revenue, if the reused/converted elements reach the industrial market; Can adopt practices to reduce competition such as collecting EoL elements while replacing the new ones or providing operational assistance only to their own novel elements.
Membrane buyers/ users (novel membranes)	Reinforcement of partnership with local cooperative of waste collectors End of contract with landfill disposure companies. Purchase of second life membrane elements	Key stakeholder in the process. In these scenarios we have 2 membrane buyers
Membrane buyers/ users (second-life membranes)	Purchase of second life membrane elements; Disposure of second life elements through partnership with recycling cooperatives	Key stakeholder in the process. In these scenarios we have 2 membrane buyers
Cooperative of waste collectors and recycling	Increase of processing capacity Optimization of processes and prices Increase partnership with companies for landfill disposure of non-recyclable material	Changing partnership from novel membrane users to fragmented second-life membrane users.
Company that performs landfill disposure	Strengthen partnership with recycling cooperatives to receive more non-recyclable material	Do not directly take part in conflicts with other parties
Universities	Technology development and proof of concept; Formation of specialized workforce; Partnership with reuse-conversion companies	Do not directly take part in conflicts with other parties
Investors and funding agencies	Funding novel companies that aim at EoL-RO elements reuse and conversion	Do not directly take part in conflicts with other parties
Government	Supporting laws that encourage the recycling chain (Law 12.305 and Decree 7.405/2010)	

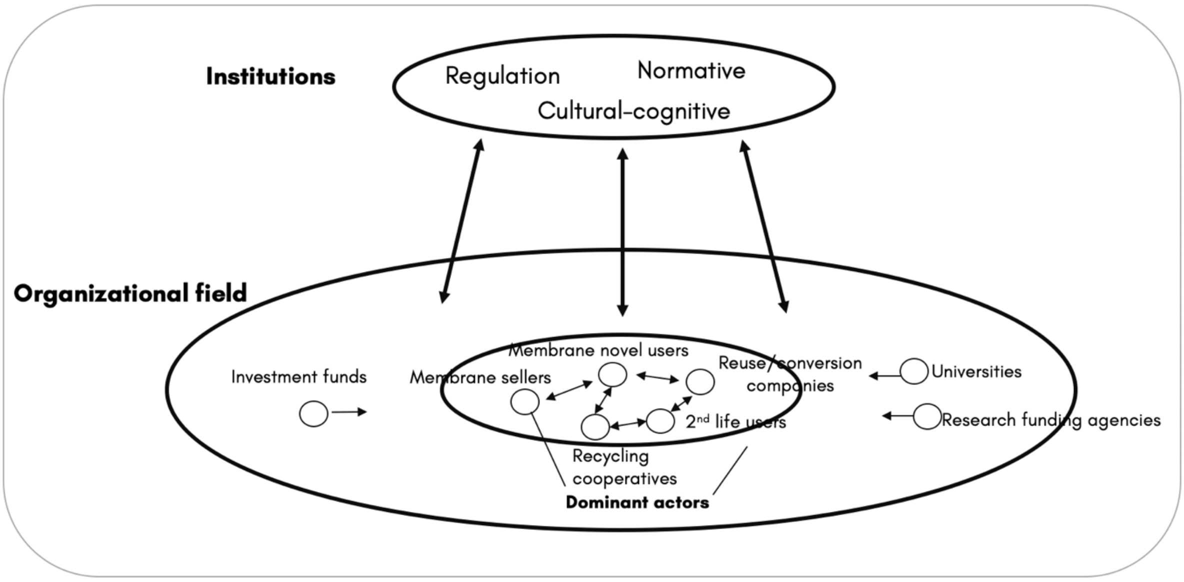
Actor	Phases 3 and 4: Main roles	Prospective conflicts
Reuse / conversion companies	Transportation and transformation of the EoL-RO membranes Supplier of second-life membrane elements other companies Eventual service of operation and maintenance	Should find complementary market niches when compared to those of novel membrane elements, in order to avoid direct conflict with novel membrane sellers

Source: Elaborated by the author (2022)

5.6.3 Final scenario

After the proposed transition, it is expected that the actors described in Table 5.4 engage in the process helping to close the loop. In this sense, Figure 5.7 illustrates the final expected scenario for EoL-RO waste management in terms of socio-technical structuration.

Figure 5.7 – Organization of the socio-technical system after the transition



Source: Adapted from Geels (2020)

When assessing the current and prospective socio-technical arrangement for bringing EoL-RO waste elements towards more sustainable practices, few important insights can be highlighted. First, the current structure of disposing of EoL-RO elements is composed of three main actors, counts with a very simple organization and does not pose as an environmentally sustainable scenario. Also, from an institutional standpoint, the Brazilian legal framework lacks in actually providing the means to achieve such alternatives, whether by not providing tangible targets for recycling or lack of investment, despite presenting guidelines that encourage sustainable development. On the other hand, relevant improvements were made to encourage new companies of technological basis to initiate in the market, which might help promote this closing-the-loop strategy.

As for the phases proposed for transition, it was possible to observe a few conflicts that require further attention when designing transitioning strategies:

- Phases 1 and 2: by decreasing the participation of landfill disposal in the picture, there would probably be a contest from the company that performs the disposition in the business-as-usual model. Aligned with the lack of management structure of recycling cooperatives and tangible legal restraints or encouragement towards more sustainable practices, to shift from *status quo* to recycling of materials will mostly depend upon the commitment of the membrane users. To this, other advantages such as fiscal and achievement of environmental certification might be important strategies to encourage these changes;
- Phases 3 and 4: the first challenge of reusing/converting EoL-RO elements is to come up with a business model that does not intend to compete with novel membrane fabricants/sellers. In this case, the answer might be aiming at more fragmented markets that still do not use membranes (smaller companies that need to treat water and wastewater). However, other challenges will arise from a fragmented market, such as those for logistics and assistance in maintenance, and these two might be crucial to understand the commercial viability of these solutions. In addition, there is a demand to continue evolving with the technologies already developed in bench and pilot scale to products, in order to achieve the market.

5.7 Conclusion

A transition pathway for managing EoL-RO elements towards more sustainable practices was proposed to understand win, losses and conflicts within the way. To this, the MLP for a system level was used to characterize and understand the current and prospective scenario in the assessed context. It was possible to see a rather simple organization for the current disposition and a need for further actors to enable reuse/recycling to take part of the waste management chain. Next, the disposition options were compared in terms of their environmental and institutional disruption, and reuse/recycling posed as the most environmentally friendly solution while requiring considerable chances in the institutional arrangement. As for the transition pathway itself, it was possible to divide it into two major moments. In the first one, it was proposed a transition from landfill disposal to recycling of parts, where it is worth to pay attention to how landfill disposers react when losing their market, especially in a context where there is no legal restriction for this type of disposal. In addition, management difficulties in recycling cooperatives might also pose a challenge to the implementation of this step. As for the second part, to start transforming and applying these elements to other purposes depends upon more profound changes in the current socio-technical system. As a consequence, they might encounter resistance from membrane sellers, recycling cooperatives and landfill disposers, the reason why it is mandatory to look for business models that avoid direct competition and promote partnership with these and other actors in the waste chain.

CHAPTER 6

FINAL CONSIDERATIONS

6.1 Conclusions

This thesis aimed at estimating, assessing the environmental and socioeconomic impact and and propose transition pathways to waste management towards more sustainable practices for EoL-RO membranes in Brazil. To this, an estimation and forecasting of the number of EoL-RO elements generated in Brazil was performed, the environmental impact through LCA and socioeconomic aspects of 5 different disposition options and membrane production was investigated and an analysis of the socio-technical waste management situation was developed.

For the estimation and forecasting, an 8-inch element weighing 13.75kg has proven to represent the average element disposed of in Brazil. In addition, the time-step model was used to estimate around 900 tons of elements generated between 2016-2019, forecasting through the ARIMA time series model 1,800 tons until 2024. Furthermore, the disposition rate in Brazil was found as double as other countries, which indicates a lower membrane lifespan. This difference was justified by the lack of operational maturity and an early membrane disposure.

Regarding environmental aspects, CML-IA Baseline was the method chosen to perform experiments after comparing 6 different methods. LCIA results converged from those seen for developing countries, where direct reuse and chemical conversion to the UF membrane posed as most environmentally friendly options. Transportation distances were also evaluated, and it was seen that half of the Brazilian territory can be covered without adding environmental burden to the overall process, which comprehends 90% of the Brazilian population. Additionally, recycling of parts presented itself as the most inexpensive alternative (U$0.36/element). However, on the social aspect, recycling was outreached by reuse and chemical conversion to the UF membrane, since these two would hopefully provide regular jobs with greater social security.

Finally, a transition pathway was proposed to understand conflicts between the actors in their different phases. It was possible to notice a current scenario with few actors, where the transition would require the development of new ones, as well as maturing technology, especially regarding the maturation of actors responsible for providing the EoL elements a second life. At last, challenges regarding winning and losers in the institutional changes also deserves attention, since the current legal background in Brazil does not practically restrain landfill disposal and the encouragement of recycling and the development of technology-based companies is still small.

6.2 Suggestion for future work

The following studies are suggested for future work:

Chapter 3:

- Whenever available information regarding import data and installed volume for RO within a sufficient timeframe, compare results for waste estimation using both the time-step model and the standard methodology of fixing the membrane lifetime. Hence, it will be possible to understand the gains of each methodology and propose a correction factor to relate them within the assessed case.

Chapter 4:

- Deepen the evaluation of social aspects through a S-LCA, performing strucured interviews with prospective stakeholders in order to better understand their social benefits;
- Perform LCA including the membrane use phase and simulate scenarios assessing the membrane intire life-cyle having more than three subsequent destination: novel membrane – reused after cleaning – chemical conversion to the UF membrane – recycling of parts.

Chapter 5:

- Confront the proposed transition pathways with the perspective of involved actors through workshops and/or interviews to capture understand the adherence of what was discussed.

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APPENDIX A – LIFE-CYCLE INVENTORY

Process	Process	Amount	Unit	Description
Landfill	**Inputs**			
	RO Module Waste	13,750	Kg	No longer used RO Module
	Transport	0,025	t*km	Transport in São Paulo State from the water plant to a sanitary landfill
	Outputs			
	Waste Fiber Glass	1,810	kg	Burning waste of Fiber Glass
	Landfill Waste	9,271	kg	Waste treatment of the rest of the materials on a sanitary landfill in SP - BR
	waste polypropylene	1,530	kg	Disposable polypropylene waste on a sanitary landfill in SP - BR
	waste polyurethane	1,139	kg	Disposable polyurethane waste
	Module Disposed in Landfill	**13,750**	**kg**	**Reference process, which should be the exact amount as the sum of the other processes in the output table, since they are all Waste Treatment Processes**
Recycling of parts	**Inputs**			
	Used Seawater Reverse Osmosis Module	13,750	kg	Disposable mixture of the RO Module
	ABS Recycling	-2,520	kg	Amount of acrylonitrile-butadiene-styrene copolymer separated to be recycled
	Polypropylene Recycling	-1,530	kg	Amount of Polypropylene polymer separated to be recycled
	Polyester Recycling	-5,384	kg	Amount of Polyester polymer separated to be recycled
	Outputs			
	Waste glass	1,810	kg	Fiber Glass Waste, disposable on a sanitary landfill in SP-BR
	Waste Polyurethane Seal	1,139	kg	Waste amount of polyurethane used in the production process
	Municipal Solid Waste	1,367	kg	Disposable waste on a sanitary landfill in SP - BR

Process	Process	Amount	Unit	Description
	Dissembled RO Element	**4,316**	**Kg**	**Reference process, which should be the exact amount as the sum of the other processes in the output table, since they are all Waste Treatment Processes**
Clinker Prod.	**Inputs**			
	RO Module Waste	13,750	kg	Used RO Module that is disposed
	Heat return	19,776	MJ/kg	Heat acquired from burning waste plastic
	Heat return	0,024	MJ/kg	Heat acquired from burning waste glass
	Heat return	3,218	MJ/kg	Heat acquired from burning municipal solid waste
	Outputs			
	Burnt Municipal Solid Waste	3,575		
	Burnt Waste Glass	1,788		
	Burnt Waste Plastic	8,388		
	Total Burnt Weight	**13,750**	**kg**	**Amount of burnt product in the incineration - Reference Process**
Direct reuse of RO Membrane	**Inputs**			
	EoL RO Module	9,167	Kg	Used RO Module - Named as a parameter on the OpenLCA app as Offset Module, so the unit won't show a weight value, but a name
	Sodium Hydrogen Sulfite	0,015	Kg	Membrane Chemical conversion
	Sodium Hydroxide	0,0003	Kg	Membrane Cleaning
	Citric Acid	0,001	Kg	Membrane Cleaning
	Water	5,800	L	Solvent

Process	Process	Amount	Unit	Description
	Transport	0.03	t*km	Transport of the membrane from the use point to the cleaning facility
	Package Box	1.32	Kg	Carboard box to pack the reusable membrane
	Polyethylene (Plastic Bag)	70,000	g	Packaging
	Eletricity	0,225	Kwh	Lawler at al., 2015
	Outputs			
	Sodium Hydrogen Sulfite	0,015	Kg	Membrane Conservation
	Sodium Citrate	0,0008	Kg	Neutralized waste
	Wastewater	5,800	L	
	Membrane Offset	.33	Unit	
	Reused RO Module	**9,167**	**Kg**	
Membrane Chemical conversion	**Inputs**			
	EoL RO Module	9,167	Kg	Used RO Module - Named as a parameter on the OpenLCA app as Offset Module, so the unit won't show a weight value, but a name
	Sodium Hydrogen Sulfite	0,015	Kg	Membrane Conservation
	Sodium Hydroxide	0,0003	Kg	Membrane Cleaning
	Citric Acid	0,001		Membrane Cleaning
	Sodium Hypochlorite, in 15% solution	0,002	Kg	Membrane Conversion - Oxidative treatment
	Sodium Sulfite	0,090	Kg	Neutralizer
	Water	5,800	L	Solvent
	Transport	0.03	t*km	Transport of the membrane from the use place to the conversion facility
	Package Box	1.32	Kg	Carboard box to pack the reusable membrane

Process	Process	Amount	Unit	Description
	Polyethylene (Plastic Bag)	70,000	g	Packaging
	Electricity	0,225	Kwh	Lawler et al., 2015
	Outputs			
	Sodium Chloride	0,001	Kg	
	Sodium Citrate	0,0008	Kg	Neutralized waste - Product of Citric Acid and Sodium Hydroxide
	Sodium Hydrogen Sulfite	0,015	Kg	Membrane Conservation
	Sodium Sulfate	0,003	Kg	Neutralized Waste - Product of the Sodium Hypochlorite with Sodium Sulfite
	Wastewater	5,800	L	
	Membrane Offset	.66	Unit	
	UF Membrane Converted	**9,167**	**Kg**	Main Reference Flow