Master's thesis

NTNU
Norwegian University of Science and Technology
Faculty of Engineering
Department of Energy and Process Engineering

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The environmental impacts of extreme water and wastewater decentralization: a life cycle assessment approach

Master's thesis in Industrial Ecology
Supervisor: Johan Berg Pettersen
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Abstract

Decentralized wastewater treatment and reuse emerges as an alternative approach to conventional water management with the potential to alleviate escalating pressures on the water cycle and secure future supply. This study performs a complete environmental assessment of a building-scale system capable of recycling urban wastewater on-site and treating rainwater to drinking water quality.

Material flow analysis and life cycle assessment are performed with the aim to quantify the impacts of extreme decentralization in a European setting and compare the environmental profile of the system with that of traditional, centralized treatment. The impact assessment is performed in Brightway2 following the ReCiPe (H) v1.03 methodology and covers both the construction and operation of the technology. The comparison with the centralized approach considers all infrastructure involved in water management. Four site-specific implementation scenarios are modelled to evaluate the sensitivity of the results to regional parameters.

The results confirm the environmental benefits of implementing decentralized water reuse systems in terms of resource recovery and life cycle impacts. The assessed reuse layout can double the nutrient and energy recovery potential of conventional treatment and outperforms the centralized approach in 10/15 of the studied ReCiPe midpoint indicators. The operation of the system dominates impacts to all environmental indicators and is greatly influenced by the switch of regional factors (esp. national electricity mixes), which contrastingly have a limited influence on the construction phase. The inclusion of rainwater capture and treatment equipment yields positive results in all studied locations. However, it must be accompanied by site-specific assessments as insufficient rainfall can cause avoided impacts from water recycling to be lower than those caused by installation of the necessary equipment.

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Keywords

Life Cycle Assessment; Decentralization; Sustainability; Water; Environmental engineering

Abbreviations

AnMBR – Anaerobic membrane bioreactor

BAU – Business as usual

BOD – Biological oxygen demand

BrW – Brown water

BW – Black water

COD – Chemical oxygen demand

EC – Electrochemical cell

EEA – European Environment Agency

EIB – European Investment Bank

FEP – Freshwater eutrophication potential

FETP – Freshwater ecotoxicity potential (FETP)

FFP – Fossil fuel potential

FPP – Flexible polypropylene

GHG – Greenhouse gas

GW – Grey water

GWP – Global warming potential

HDPE – High density polyethylene

HFMC – Hollow fibre membrane contactor

HRT – Hydraulic retention time

HTTP – Human ecotoxicity potential

IRP – Ionising radiation potential

ISO – International Organization for Standardization

LCA – Life cycle assessment

LOP – Agricultural land occupation

MABR – Membrane aerated biofilm reactor

MBR – Membrane bioreactor

MEP – Marine eutrophication potential

METP – Marine ecotoxicity potential

MFA – Material flow analysis

N – Nitrogen

NMVOC – Non-methane volatile organic compounds

ODP – Ozone depletion potential

ODS – Ozone-depleting substance

OFMSW – Organic fraction of municipal solid waste

OLAND – Oxygen-limited autotrophic nitrification and denitrification

OLR – Organic loading rate

P – Phosphorus

PDMS – Polydimethylsiloxane

PMFP – Particulate matter formation potential

POFP – Photochemical oxidant formation potential

PVC – Polyvinyl chloride

PVDF – Polyvinylidene fluoride

RBC – Rotating biological contactor

RO – Reverse osmosis

RW – Rainwater

SDGS – Sustainable development goals

SRT – Solids retention time

TAN – Total ammonia nitrogen

TAP – Terrestrial acidification potential (TAP)

TETP – Terrestrial ecotoxicity potential

TN – Total nitrogen

TP – Total phosphorus

TSS – Total suspended solids

UASB – Up-flow anaerobic sludge blanket

UD – Urine diversion

UF – Ultrafiltration

UN – United Nations

UV – Ultraviolet

VSS – Volatile suspended solids

VUV – Vacuum ultraviolet (treatment)

WCP – Eater consumption potential

WWTP – Wastewater treatment plant

YW – Yellow water

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1 Introduction

1.1 Background

During the last decades, growing pressures on freshwater systems have resulted in severe water scarcity and increasing supply uncertainties worldwide. Water contamination and resource depletion caused by the environmental crisis and a growing demand call for alternative water management systems that are resilient and contribute to development within a circular economy framework making optimized use of resources, as the business-as-usual approach to water management increasingly fails to tackle most rising issues related to the water crisis (Büttner et al., 2022; Ferreira et al., 2021; Narayanamoorthy et al., 2022; Wen et al., 2022).

It is widely discussed how potable water augmentation techniques will be necessary to secure supply. Centralized or decentralized reuse of wastewater and desalinization of seawater and groundwater are gaining traction as the main possible routes to increase water availability (Arden et al., 2021; Giammar et al., 2022; Goodwin et al., 2018; Jeffrey et al., 2022; Lahnsteiner et al., 2017; Tow et al., 2021).

This thesis focuses on wastewater, which is currently treated in centralized plants based on the activated sludge process. This centralized approach is associated with avoidable transport losses and costs, high energy demands, and low effectiveness in recovering the valuable nutrients and energy contained in wastewater (Garrido-Baserba et al., 2018; Rabaey et al., 2020). In fact, large volumetric losses of up to 55% of total water are associated with current water distribution networks, resulting in nearly 50 billion m^3 of potable water being lost annually worldwide (Ociepa et al., 2019). Besides, as for now, raw, mixed wastewater is collected with generally no separation of the different types of flows produced in municipalities, requiring large treatment volumes and diluting pollutants hindering their extraction.

While centralized wastewater reuse involves less deviation from the existing infrastructure and *modus operandi*, the decentralization of wastewater treatment offers high potential for numerous improvements. Decentralization aims to provide the possibility to eliminate transport losses and costs, to improve resource recovery, and to facilitate avoiding the blending of effluents with different characteristics, optimizing treatment (Khalkhali et al., 2021; Pikaar et al., 2020; Roefs et al., 2017; D. Zhang et al., 2021; Zhang et al., 2023). Off-grid systems could allow for wastewater recycling in urban environments providing on-site treatment and reuse. Extreme decentralization consisting of small-scale systems adapted to single buildings has the potential to facilitate the shift between current and future water treatment infrastructure, which could involve an integrated solution combining centralized and decentralized facilities (Keller, 2023). Regarding the specific technology required to achieve adequate quality for water recycling in buildings, layouts involving the source-separation of the different wastewater flows and urine diversion seem to offer the highest potential in cost-effectiveness and resource recovery among decentralized configurations (Garrido-Baserba et al., 2022) and are therefore the focus of this thesis.

Support has been shown for decentralization as a sustainable augmentation system which can be key in the transition to a water management scene capable of adhering to the United Nations' Sustainable Development Goals (SDGS) presented in the 2030 Agenda for Sustainable Development (Sadoff et al., 2020; United Nations, 2023). Water plays an important role in the UN'S SDGS as it is closely tied to equality, human and environmental health, and economic sustainability. Decentralization has a direct impact on goal 6 (clean water and sanitation), which is entirely dedicated to water, and goal 3 (good health and well-being). Besides, it has rather indirect, but clear effects on goals 10 (reduced inequalities), 11 (sustainable cities and communities), and 12 (responsible consumption and production), as, similarly to solar power, decentralized systems empower communities giving them the chance to reuse their own water, lowering consumption and allowing for the recycling of resources in cities (Rabaey et al., 2020).

However, the secure provision of safe water can contribute to virtually all SDGS. Access to clean water and nutrients can help SDG 1, which focuses on the reduction of poverty, and SDG 2, which aims to reduce hunger, by improving food security and quality. The recovery of biogas can provide a cleaner energy source, adhering to SGD 7 (affordable and clean energy). Additionally, the development of innovative technology, and the efficiency increase linked with the reduction in water consumption helps SDG 9, which relates to industry, innovation, and infrastructure. By providing a reduction of water use and ensuring proper treatment, the increase in the sustainability of wastewater treatment impacts goals 13, 14, and 15 related to climate action, life below water, and life on land, respectively. Finally, SDG 16 on peace, justice, and strong institutions can benefit from a decentralized approach to water treatment because of the reduction of social conflicts surrounding water use.

While interest on the environmental analysis and life cycle assessment of alternative water treatment technologies is growing, the impacts of specific complete systems (incl. decentralized systems) are still under-researched. Besides, the water and wastewater management system is a complex combination of pumping stations, pipeline networks, and water and wastewater treatment plants (Venkatesh, 2011), but environmental analyses are often limited to a fraction of these utilities (Rashid et al., 2023).

Thus, research and innovation in the water treatment scene is required to cover the needs of a growing population and ensure safe and sustainable water supply. Holistic assessments of alternative technologies such as decentralized systems considering not only economic and technical but also environmental and social aspects can help guide the water sector towards efficient solutions to the current water and environmental crises.

1.2 Research objectives and methodology

While in Garrido-Baserba et al. (2022), we discussed the assessment of the costs, final water quality, adaptation to buildings, and variability in feasibility according to climate and size of decentralized, water-reusing systems, extreme decentralization is not only about the economics, but rather presented as a highly sustainable alternative to traditional water management. Thus, the aim of this thesis is to extend the previous research on decentralization and assess the sustainability of its implementation in urban buildings, taking into account the totality of infrastructure involved in water management (incl. pipeline networks, treatment plants, and pumping stations).

In Garrido-Baserba et al. (2022), the techno-economic performance of five building-scale treatment layouts and their adaptation to six different types of housing of capacities ranging from 12 to 300 inhabitants in five different climatic zones was investigated. The five treatment trains were designed for resource recovery and wastewater recycling inside the buildings themselves, showed great economic and technical performance, and could potentially decrease non-potable water consumption in percentages from 74% to 100%. Economies of scale were detected causing the price of water produced by the decentralized systems to be lower than that of general water tariffs in Europe and the United States in buildings from 300 inhabitants. Potable water was proposed to be produced from rainwater, and demand coverage depended on the region's rainfall and decreased as housing density increased, as catchment area does not grow proportionally to population and demand. Rainwater capture is also studied in this thesis as an additional feature in decentralization aiming to optimize water use and complement the recycling system while potable reuse of wastewater still faces social and political challenges.

While the economic and technical aspects of the different layouts were evaluated with promising results, and despite the inherent sustainability-oriented nature of the system, the focus was on costs, and the environmental performance of the technology was not assessed. The need for intensive on-site treatment to make water suitable for reuse, with its associated construction and operation costs, could challenge the real environmental performance of the proposed systems. Thus, (1) an optimization for improved resource recovery and environmental performance and (2) a complete life cycle assessment of a combination of the most promising technologies in Garrido-Baserba et al. (2022) are proposed as the main objectives of this work.

The specific research questions that this thesis aims to answer are presented as follows:

RQ1. What is the resource recovery potential of innovative urban water management under the conditions of extreme decentralization?

RQ2. What are the environmental impacts of water and wastewater decentralization, and does resource recovery palliate the damages caused by treatment?

RQ3. How does the environmental performance of the proposed decentralized layout compare to that of conventional water supply and treatment, both in terms of resource recovery and environmental impacts?

RQ4. How do site-specific factors influence the environmental performance of the system?

RQ5. Is the addition of rainwater capture and treatment environmentally favourable?

In order to answer these questions, the methodology illustrated in Figure 1 is followed. Firstly, a specific case study is proposed involving the provision of water supply and wastewater treatment for 300 people during a year.

Besides considering the possibility to cover these demands through conventional water management, this study focuses on a building-size decentralized system. The proposed installation is based on a rearrangement of the most promising technologies in Garrido-Baserba et al. (2022) and consists of the necessary equipment to source-separate, treat, and reuse the wastewater produced in a residential building by the reference population of 300 inhabitants, on-site. It is defined in terms of treatment technology, pipeline network, building characteristics and context, water flows and quality, and additional equipment.

The design of the decentralized layout for its adaptation to the reference building, involving the and sizing of treatment processes, as well as additional tanks, sewers, and infrastructure, allows for the construction of an inventory with the materials, chemicals, and resources needed to construct and operate the system, and for the calculation of detailed mass balances for all treatment units and for the complete process. Dynamic simulation system SIMBA[#] developed by inCTRL is used to model the biological processes in the brown water and grey water lines to verify the correctness of the calculations and ensure flows and process units are studied accurately. With this data collected from treatment design and sizing, technology manufacturers, simulation results, and literature, the thesis relies on material flow analysis and life cycle assessment to conduct an environmental assessment. Both are well-established and standardized methodologies widely used for sustainability studies.

On the one hand, material flow analysis helps tackle RQ1 through the identification of resource loss in the system. It also ensures all calculations are coherent within the mostly closed-loop treatment trains. Life cycle assessment, on the other hand, helps quantify the impacts on the environment to answer RQ2. A process-based, attributional life cycle assessment is performed with Brightway2 in accordance with ISO 14044 and relying on Ecoinvent v3.9.1. (Wernet et al., 2016). A cradle-to-gate approach is taken where not only the impacts associated with operation are accounted for in the results, but also those related to energy production, the transport and treatment of waste, material investment, construction, and resource recovery. The definition of goal and scope for the life cycle assessment are further explained in the Life Cycle Assessment section as part of the standardized ISO methodology. ReCiPe (H) v1.03 is used as the assessment methodology.

The performance and impacts of the decentralized system are then compared with those of traditional treatment to answer RQ3 and identify the competitiveness of decentralization and its potential for being considered a relevant option towards sustainability in the water management scene.

Regarding RQ4, additional scenarios considering the implementation of the decentralized system in four cities within Europe (incl. Barcelona, Trondheim, Vienna, and Bucharest) are modelled. This comparison showcases how dependant the performance of the decentralized layout is on factors such as the composition of the electricity mix used, rainfall patterns, or regional fertilizer obtention routes. Finally, the site-specific scenarios allow to include rainwater capture and treatment according to local rainfall volumes and assess whether the inclusion of equipment for the utilization of rainwater is favourable in each case (RQ5).

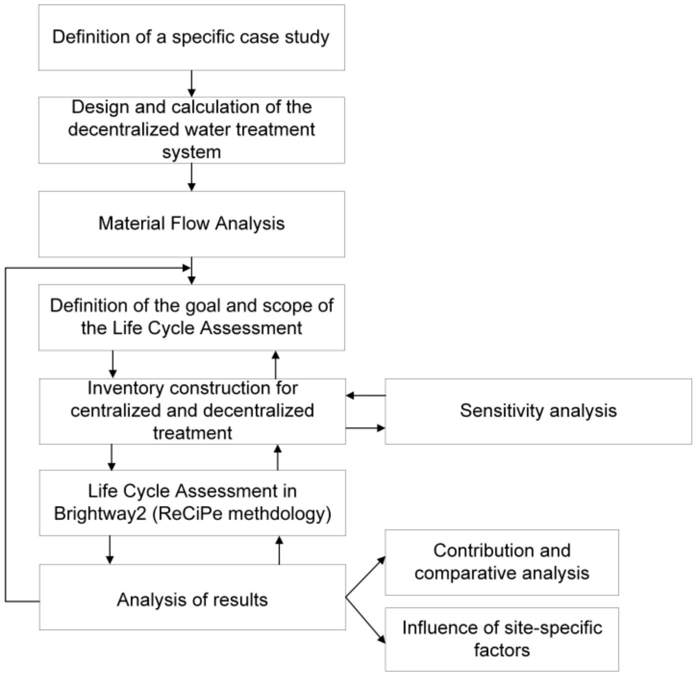

Figure 1: Simplified methodological approach followed to assess the environmental performance of water and wastewater decentralization.

2 Theory and methods

The challenge to provide water management to a reference population is taken as the basis of the thesis, which is achieved through centralized or decentralized treatment. While the centralized approach is based on conventional treatment, a specific combination of technologies and equipment is proposed to reach the same goal through a decentralized strategy. This section focuses on the report of the case, the technologies considered to be part of conventional treatment, and the complete description of the decentralized scenario.

2.1 Case description

A population of 300 inhabitants is taken as reference for the study, since the cost-effectiveness of decentralized systems is optimized from this population size (Garrido-Baserba et al., 2022). Thus, the modelling of centralized and decentralized treatment considers the need to provide water and wastewater treatment to the reference population during a year.

The scenarios modelled for the comparison between centralized and decentralized treatment are focused on a European setting. Therefore, all data relates to Europe, including water use and composition, building characteristics, and context data. For the environmental analysis, European input factors are used when available. In the cases where there is no regional data, global inputs are taken. Additionally, four site-specific cases are modelled representing the cities of (i) Barcelona (Spain), (ii) Trondheim (Norway), (iii) Vienna (Austria), and (iv) Bucharest (Romania). Consideration is taken for the regional electricity mixes, fertilizer production routes, and precipitation volumes in each country.

Daily water demand per capita is 15 L and 108 L for BW and GW, respectively, considering the use of toilets that allow the separation of urine and use less water per flush than regular toilets, which consume 40 L per flush (Rabaey et al., 2020). An average production of 1.5 L of YW per day and person is considered, and a recovery ratio of 70% is associated to the urine-diverting toilets (Lienert and Larsen, 2010). Urea hydrolysis is assumed for all non-collected urine, while source separated YW is hydrolysed during its treatment, enabling the controlled precipitation of phosphorus in the form of valuable Ca and Mg salts (mainly $Ca_3(PO_4)_2$). Rainfall patterns are considered for the site-specific scenarios to estimate the available potable water that the rainwater (RW) capture and treatment system is able to produce.

2.2 Centralized water treatment

Conventional treatment of urban wastewater consists of various stages and a range of technologies, mostly dependent on plant size and required effluent quality. Most medium to large-sized plants produce an effluent ready for discharge to natural water bodies, requiring water to undergo posterior treatment to meet potable quality requirements.

This approach requires large piping networks for wastewater collection, discharge to the environment after treatment, extraction of new water from water bodies, transport to potable water plants, and distribution to end users. In concordance to observed proximities of wastewater treatment plants (WWTPs) to city centres for various European cities (incl. Barcelona, Berlin, Bucharest, Girona, Madrid, Oslo, Paris, Rome, Stockholm, Trondheim, Vienna, and Warsaw) the European average scenario considers a distance to the nearest conventional centralized water treatment plant of 10 km. Distance from WWTPs to the nearest local freshwater systems where discharge takes place is considered to be 2 km (Shahmoradi and Isalou, 2013). Distance from the source of potable water to the potable water treatment plant is an average of 36 km (as observed for the cities listed previously for distances to WWTPs).

Regarding wastewater, once it reaches treatment plant, it is subjected to pre-treatment, usually involving screening and grit removal, and often complemented with the comminution of large waste. Afterwards, primary treatment consisting of physical and chemical separation processes including filtering, sedimentation, coagulation, flotation, and centrifugation mainly eliminates suspended solids, oils, and odours. Biological treatment is applied afterwards in secondary treatment, mainly through the aerobic activated sludge process. Primary and secondary sludges are combined and stabilized before disposal. Even though often not implemented, sludge treatment creates an opportunity for energy removal in the form of biogas through anaerobic digestion, as well as for nutrient recovery, to some extent. When effluent quality is not sufficient to meet legal standards, tertiary treatment is performed, providing additional nutrient removal with the use of membranes, biological treatment (e.g., enhanced biological phosphorus removal process), oxidation, or alternatives. Final disinfection with ozone, ultraviolet radiation, or chlorine helps ensure health safety. Figure 2 showcases the described treatment process. Drinking water treatment most commonly involves physical and chemical separation of pollutants, clarification, chemical disinfection, and filtration (Alver, 2019; Stackelberg et al., 2007).

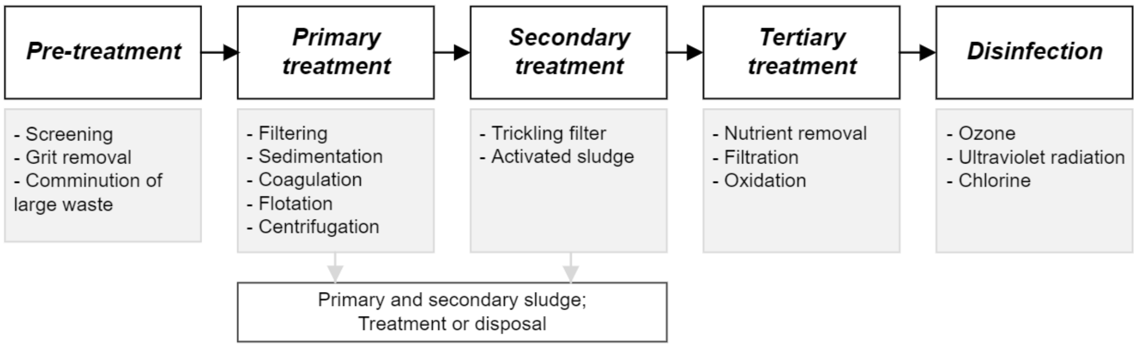

Figure 2: Flowchart of conventional wastewater treatment.

The combination of high energy consumptions, excessive chemical use and waste generation, high operational costs, aging infrastructure, and significant losses during water transport call for new ways to treat and manage water (Contzen et al., 2023; Kumar et al., 2022; Rabaey et al., 2020).

Besides, although highly discussed and despite vast advancements in research during the recent years, resource recovery is still poorly implemented in WWTPs (EEA, 2022; EIB, 2022; Kehrein et al., 2020; Radini et al., 2023). Nutrient (mainly nitrogen and phosphous) removal is widely predominant over recovery, as wastewater collection and treatment to meet effluent quality for discharge in the environment is prioritized (Yadav et al., 2021). Studies focusing on WWTPs worldwide reveal that only a small fraction of them include some variant of recovery technology, both in developed (Kehrein et al., 2020), and developing (Chrispim et al., 2020) regions.

Nutrients and energy are currently mostly recovered from primary and secondary sludge, and reused through land application (Mihelcic et al., 2011). However, sludge is often only dewatered and disposed or incinerated. Moreover, the nitrogen that is accumulated in the sludge after primary and secondary treatment is less than 40% of the influent load because of its dissipation as nitrogen gas to the atmosphere during biological treatment in the principal line (Ostermeyer et al., 2022).

Decentralization of wastewater treatment offers the opportunity to grow from the traditional approach water management and overcome its shortcomings, opening the door to direct water reuse and resource recovery, and increasing the sustainability of the process.

2.3 Decentralized water treatment

The approach to the definition and assessment of the studied extreme decentralization scenario is presented in this section, including details on the characteristics of the installation and the methodology for the development of the MFA and LCA.

2.3.1 Approach

A residential, urban building able to house the studied population of 300 inhabitants is taken as the baseline infrastructure where a decentralized water recycling system is implemented, including treatment units, tanks, and pipeline for the complete separation of the different wastewater flows. Black water (i.e., water from toilets containing higher amounts of pollutants, abbreviated to BW) is divided between urine (or yellow water, YW from this point onwards) and brown water (BrW, i.e., BW without urine). Grey water (i.e., water from appliances, showers, and sinks, or GW) is also separated at point of origin. Equipment is sized according to the building's needs.

An average European capacity of 2.4 inhabitants and a surface of 72 m^2 per dwelling are assumed, as well as a standard floor height of 3 m (Garrido-Baserba et al., 2022; Roefs et al., 2017). The baseline building consists of 12 floors with a total surface area of 750 m^2 including extra space for common areas and a basement destined for parking and storage of technology (see Figure 3).

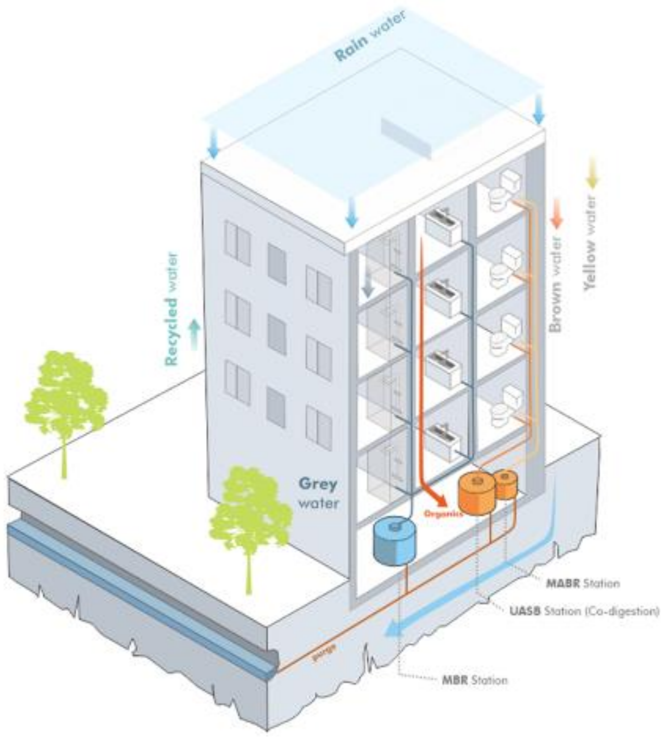

Figure 3: Concept figure showing 4 of the 12 floors of the building and the principal treatment units (adapted from Garrido-Baserba, 2023).

2.3.2 *Wastewater composition and final water quality*

Table 1 shows the compositions of the different wastewater flows used to calculate the treatment requirements of the building, as well as the influents to the BrW and GW treatment lines calculated considering daily constituent loads per capita and food waste composition in concordance with literature and the volumes presented in section 2.1.

Food waste is mixed with BrW and its composition is expressed in mg/g wet basis considering a water content of 76% (Dhadwal, 2020; Dhadwal et al., 2021; Gao et al., 2020, 2020; Garrido-Baserba et al., 2022; Kim et al., 2015; Kujawa-Roeleveld et al., 2006; Larsen et al., 2013; Lienert and Larsen, 2010; Rajagopal et al., 2013; Rossi et al., 2009; Sun et al., 2020).

Table 1: Composition of black water (BW) [mg/l], undiluted fresh urine (YW) considering urea hydrolisis [mg/l], GW [mg/l], the organic fraction of municipal solids waste (OFMSW) [mg/g wet basis], Brown water (BrW) from urine-diverting toilets, the mixture entering the BrW line combining BrW, non-recovered, non-hydrolysed YW, and OFMSW [mg/l], and the influent to the grey water (GW) line combing the treated effluent of the BrW line and GW [mg/l]. COD: chemical oxygen demand; BOD: biological oxygen demand, TN: total nitrogen; TAN: total ammonia nitrogen; TP: total phosphorus; TSS: total suspended solids; VSS: volatile suspended solids.

	BW	YW	GW	OFMSW	BrW	Influent BrW line	Influent GW line
COD	3,520.0	10,400.0	472.0	286.0	2,490.0	5,487.4	490.9
BOD	1,186.7	3,866.7	175.0	218.2	800.0	3,007.6	154.1
TN	666.7	8,800.0	8.0	5.9	80.0	391.3	18.7
TAN	600.0	463.0	3.0	0.8	72.0	90.9	6.8
TP	86.7	800.0	5.0	3.2	34.0	87.2	6.6
TSS	2,786.7	0.0	175.0	189.5	2,786.7	4,545.3	258.6
VSS	2,229.3	0.0	64.0	179.8	2,229.3	3,909.7	116.1

Variability is expected to be low as water that enters the system is solely produced in households. Deviations from the presented pollutant loads may occur due to exceptional events involving one or more of the households in the building (e.g., change in behaviour, changes in the amount of time spent in the household, or system failure). Influent equalisation tanks previous to treatment aim to reduce the variability in the composition of the flow entering the treatment train and homogenize pollutant concentrations. The highest uncertainty in the presented compositions is associated with the organic fraction of municipal solid waste (OFMSW), as it is directly related with diet and can therefore vary significantly depending on the region. The considered composition refers to a western diet.

Removal efficiencies and influent/effluent ratios widely influence final water composition and are therefore confirmed with SIMBA[#]. When simulation is not possible, data is obtained from reported values. Table 2 shows the final composition of water after leaving the treatment train, which is capable of meeting reuse standards of US EPA (EPA/600/R-12/618) and EU regulation 2020/741, and drinking water requirements by WHO (GDWQ, 2022), US EPA (SDWA,1974), and EU Directive 2020/2184.

Table 2: Effluent composition [mg/l] of water destined for reuse. COD: chemical oxygen demand; BOD: biological oxygen demand, TN: total nitrogen; TAN: total ammonia nitrogen; TP: total phosphorus; TSS: total suspended solids; VSS: volatile suspended solids.

	Effluent composition (mg/l)	Grey water reuse standards	Drinking water standards
COD	3.45	-	< 10 mg/l
BOD	0.18	< 10 mg/l	< 5 mg/l
TN	2.20	< 10 mg/l	-
TAN	0.03	-	< 1.5 mg/l
$N\text{-}NO_3$	0.85	-	< 50 mg/l
TP	0.04	< 0.05 mg/l	-
TSS	0.0	< 10 mg/l	-
VSS	0.0	-	-

Table 2 shows the most restrictive threshold values for each pollutant extracted from the cited legislation and standards. In the case of water reuse, those values related to urban and bathing applications are selected. Control of some pollutants is not prioritized in legislation regarding reuse or drinking water quality due to their low impact on the target applications. For example, phosphorus limits are usually not included in potable water legislation as its ingestion is not toxic for humans unless the levels of the pollutant in water are exceptionally high. However, its presence enhances microbial growth, which is strictly regulated in drinking water policies and addressed in the Health and safety section. Phosphorus can also cause grave environmental problems due to eutrophication. Therefore, the limit for phosphorus for reuse applications is stringent.

2.3.3 Overview of technology

The wastewater reuse system is composed of three treatment lines aimed at the treatment of the source-separated flows and diverted urine, as the specialized treatment of each type of wastewater creates the opportunity to reduce treatment sizes and costs, facilitate resource recovery, and increase overall sustainability (Garrido-Baserba et al., 2018; Lam et al., 2015; Larsen et al., 2013). BW is divided between BrW and YW, which receive treatment that targets nutrient and energy recovery. Once pollutant loads for BrW are reduced, the treated BrW is blended with GW collected from households and passed through aerobic treatment, filtration, and disinfection.

The technology involved treats BrW in a biogas-producing up-flow anaerobic sludge blanket (UASB) reactor and then directed towards equipment specifically designed to recover phosphorus from the flow and remove nitrogen to meet adequate effluent quality. Next, the treated brown water is blended with GW in a side-stream membrane bioreactor (MBR) where aerobic treatment helps ensure final water quality. The resulting effluent is subject to the final stage of treatment consisting of a reverse osmosis (RO) membrane and ultraviolet (UV) disinfection. Urine (i.e., yellow water, YW) is separated in the system due to its high nitrogen and phosphorus content, stabilization of the flow is necessary. An electrochemical cell (EC) coupled with a crystallizer where phosphate precipitation is carried out is proposed as the most sustainable option for urine stabilization, also providing phosphorus recovery. The hydrolysed effluent undergoes nitrification in a small-scale membrane aerated biofilm reactor (MABR), where liquid N-based fertilizer is produced. The treated BW and GW are recycled in the building for non-potable applications, and potable water demand is also reduced by the incorporation of a RW capture and sanitization installation consisting of a second combination of RO and UV treatment. The coverage of potable water demand by the building's inhabitants is dependent on the region's climate.

Figure 4 shows the flowchart of the described treatments. Note that auxiliary equipment (e.g., pumps, holding tanks and vessels, valves) is not represented. While the YW line maintains the same technologies described in the urine diversion scenario in Garrido-Baserba et al. (2022), the treatment trains for BrW and GW are based on the scenario which involved the most mature technologies, to ensure best final water quality and reliability in terms of sizing, costing, and performance. Besides, the BrW and RW lines are complemented with additional units, including a specific degassing membrane for biogas capture, and filtration and re-mineralization units for RW. All treatments are re-sized and re-calculated to adjust to the requirements of the proposed building and influent pollutant compositions in this thesis, as well as to obtain the data necessary for the construction of a life cycle inventory for the impact assessment. Auxiliary equipment including holding tanks, vessels, valves, and pumps is also incorporated.

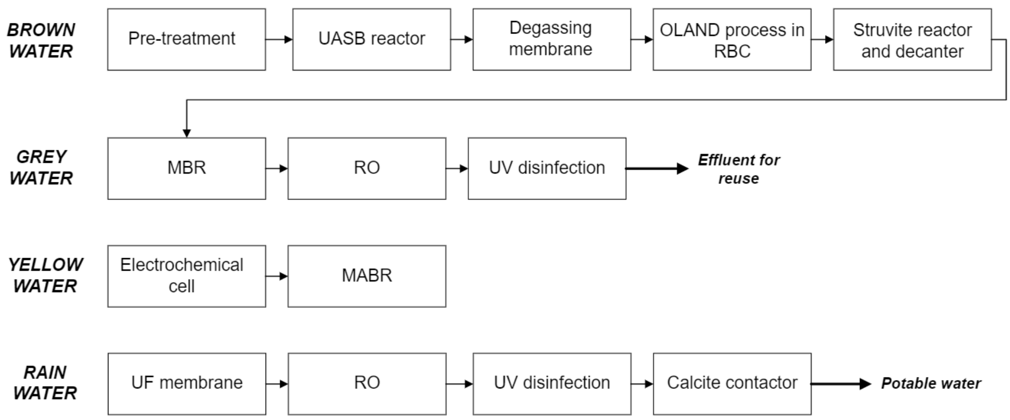

Figure 4: Simplified flowchart of the described treatments.

2.3.4 Brown water treatment line

The BrW line is characterized by more intensive treatment designed to deal with high pollutant concentrations and enable resource recovery. Firstly, anaerobic treatment is applied in a UASB reactor. This process is selected because of its suitability for decentralized systems given by its low excess sludge production, low construction costs, and reduced treatment volume, its flexibility and ability to decouple solids retention time (SRT) and hydraulic retention time (HRT), and its adaptation to a circular-economy framework in terms of its capacity for energy production from waste. Besides eliminating the need for energy-intensive aeration required in aerobic treatment, anaerobic processes recuperate the stored energy in organic matter in the form of methane (Capodaglio et al., 2017; Cecconet et al., 2022). Dissolved gas recovery from the effluent flow is performed to avoid greenhouse gas (GHG) emissions and resource loss as described in the Resource recovery section.

UASB reactors show robust COD elimination and are now a consolidated process in wastewater treatment, providing reliable and effective treatment for BrW (Cecconet et al., 2022). Co-digestion of OFMSW and BrW is proposed as a route for increasing organic matter loads in the influent of anaerobic digestion thus enhancing methane production (Adami et al., 2020; Gao et al., 2020; Ma et al., 2018; Mohammadi, 2022; Mu et al., 2020). Optimal reactor temperature of 35ºC is assumed, entailing pre-heating of the influent flow. Energy obtained from methane helps cover energy requirements for the process. To facilitate decentralization, the operation at short HRT is examined, resulting in a combination of a smaller reactor volume and a high organic loading rate (OLR), which enhances methanogenesis (de Graaff et al., 2010; Gao et al., 2020). An HRT of 2.6 days is established in concordance with Gao et al. (2020). Hydrolysis level of 0.49, and an effluent/influent ratio of 0.95 are assumed, in consistence with common operation parameters. The anaerobic process is simulated using wastewater treatment modelling and simulation software SIMBA[#] to verify biogas production, removal efficiencies, tank volume, and process parameters. Pre-treatment of OFMSW by grinding and mixing with BrW is required to obtain a homogenized influent for the reactor.

After extensive COD elimination in the UASB reactor, nitrogen in the BrW flow is tackled through biological removal in a rotating biological contactor (RBC) performing a one-stage nitritation/anammox process (Windey et al., 2005). An alternative name for the treatment is oxygen-limited nitrification and denitrification (OLAND) process. Annamox-based treatments have been widely used for nitrogen management of wastewater effluents, and they rely on autotrophic anaerobic ammonium oxidation bacteria, the so-called anammox bacteria (Hu et al., 2011; Laureni et al., 2016; Rodriguez-Caballero et al., 2013; Wongkiew et al., 2020). These are able to oxidate ammonium and nitrite intro diatomic nitrogen gas. Water is also produced representing a product with no global-warming potential. Nitrogen recovery is not considered at this stage due to the cost-effectiveness of the annamox process, which additionally avoids larger GHG emissions typical from alternative treatments such as ammonia stripping (Vinardell et al., 2020). The RBC also facilitates the operation and maintenance of the process, which makes this layout the most suitable for a decentralized framework. Biofilm surface of 110 m^2 is calculated according to influent nitrogen concentration leading to a PVC tank volume of less than 0.4 m^3 considering a surface of the carrier material of 350 m^2/m^3. The rotating disks are submerged by 45% and made of PE (Mohammed and Sills, 2022).

After the RBC, a crystallizer following the working principle of a fluidized bed reactor combined with a decanter are proposed as a combination of reactor and accumulation device for the production of struvite ($MgNH_4PO_4 \cdot 6\ H_2O$). In order to avoid the need for air injection, the reactor is proposed to follow the design described by Bhuiyan et al. (2008), which uses a decreasing section to regulate the fluid's velocity inside the unit (Kumar and Pal, 2015). Materials for the reactor and decanter are stainless steel for tank bodies and PVC for junctions and accessories, due to their capacity to resist different pHs and to avoid clogging, respectively (Mbaya et al., 2017; Sena and Hicks, 2018). Reactor and decanter volumes of 0.6 and 0.2 m^3 are found through Strokes' law and assuming retention times of 3.5 and 1 hours for the reactor and decanter respectively. To enable the formation of struvite, magnesium salts need to be dosed in the process unit in a molar ratio of 1.5 Mg^{2+}:PO_4^{3-} to provide optimal struvite production. The obtained product is determined by phosphate concentration as it is the limiting component of the reaction, however the described conditions enable for conversion of 90% of present phosphate to struvite fertilizer (Siciliano et al., 2020). Magnesium is added in the form of $Mg(OH)_2$ according to the presented ratio and influent phosphate concentration. As magnesium dosing can represent up to 75% of the overall operational costs of struvite precipitation, alternative sources of Mg^{2+} ions such as seawater are sometimes considered. However, even if more economical, the use of seawater requires previous filtration and can lower the quality of the produced struvite due to the precipitation of competing compounds such as calcium phosphate, as described in Kumar and Pal, (2015). Therefore, $Mg(OH)_2$ is used considering the relatively small scale of the plant and the optimized design which reduces chemical consumption. Re-circulation of struvite crystal embryos is proposed as a way to enable secondary nucleation in the system.

The process arrangement in the BrW line is as described since even though the OLAND-based process eliminates part of the ammonia present in water apparently reducing struvite-production potential, the limiting component in the struvite reaction is phosphate, which is not targeted for removal in the RBC. Besides, the biological biofilm process requires influent temperature of 30-40ºC (Sobotka et al., 2021), therefore by placing the unit after the UASB reactor the effluent temperature of 35ºC of the anaerobic process can be used to avoid the need for heating the flow twice resulting in relevant energy savings.

2.3.5 Grey water treatment line

Once volumetric losses are accounted for, the treated BrW shows total COD, TN, TP, and TSS concentrations of only 11%, 24%, 21%, and 19% of the influent values, respectively, and negligible BOD. The low volume of BrW compared to GW makes it possible to blend both flows with minor deviations from GW composition. Only slightly increases of TN, TSS, and VSS loads are observed. The mixed stream is the influent of the GW line, consisting of aerobic treatment and sanitization.

A side-stream MBR consisting of an aerated reactor and a crossflow, multitube, hollow-fibre membrane loop is proposed as the biological treatment for GW. Aerobic treatment combined with membrane filtration shows excellent environmental performance in small- to medium- scale systems and provides high pollutant removal efficiencies resulting in potable water effluent qualities. Besides their low footprint and sludge production, especially in comparison with conventional biological treatments, MBRs also have the ability to remove microorganisms, and their modular nature makes them extremely adequate for adaptation to decentralized layouts (Cashman et al., 2018; Cecconet et al., 2019; Kobayashi et al., 2020). As the MBR is closer to the reuse point in the system, it is important to consider its capacity to remove a wider range of contaminants. MBRs show better performance at eliminating active compounds, microcontaminants, androgenic activity, surfactants, and emerging contaminants as a whole when compared with alternative technologies mainly due to the incorporation of membrane filtration and the capacity to hold a richer microbial community (Cecconet et al., 2019). MBR performance at low HRT and SRTs has been proved efficient. The proposed MBR operates at a HRT of 3 days and a volume of 108 m^3. The operational conditions as well as the influent/effluent ratio, volume, and removal efficiencies have been validated using SIMBA[#]. The ultrafiltration (UF) membrane and biological treatment tank's main materials are Polyvinylidene fluoride (PVDF) and stainless steel, respectively.

However, the operational costs of MBR treatment increase those of the complete system substantially, as they can represent 30-40% of total operational costs in decentralized layouts (Garrido-Baserba et al., 2022), and effluents still often require further disinfection, especially when the effluent is destined for reuse, as pathogen presence due to regrowth after the membrane or migration may occur (Cecconet et al., 2019). Membrane fouling also creates the need for regular maintenance. However, this stage ensures adequate pollutant removal, and the flexibility of the treatment helps adapt it to the proposed configuration. RO filtration and UV disinfection are proposed as follow-up treatments to meet desired nutrient and microbial presence regulations in the effluent.

The incorporation of RO in the layout is mainly due to the need for stronger phosphorus removal after all previous treatments in order to meet adequate effluent quality. A flat sheet membrane is proposed. It has been long recognized that membrane technology plays of a key role in decentralized layouts, providing reliable treatment and risk minimization in reuse applications (Fane, 2005). RO is combined with vacuum UV (VUV) disinfection to complete treatment and provide enough pathogen inactivation to meet reuse standards (Hube and Wu, 2021). Even though less relevant, removal rates of COD, TN, TAN, and nitrate (Amanollahi et al., 2021; Piras et al., 2022) considering a treatment time of 20-30 minutes for complete disinfection (Amiri et al., 2020; Khan et al., 2022; Szeto et al., 2020) are included in the study. Employing VUV for disinfection eliminates the need to add chlorine or ozone. Further details on the system's capacity to remove microorganisms are found in the Health and safety section.

2.3.6 Yellow water treatment line

The layout is completed by the YW line, which is especially designed to enable nutrient recovery from urine. Special attention is brought to urea hydrolysis, which can cause undesired odour nuisance, loss of phosphorus and clogging of pipes due to its precipitation with minerals, and ammonia loss due to evaporation (Chen et al., 2017). To avoid uncontrolled hydrolysis and perform stabilization without chemical addition, the process is carried out in an electrochemical cell coupled with a crystalliser (De Paepe et al., 2020a). The process is expected to achieve conversion of TN to TAN with losses of only 3%, high COD removal to create an effluent free of organics, and 30% phosphate recovery in the form of valuable precipitates, leading to a TP recovery rate of 22%. The effluent undergoes full nitrification in an MABR with a membrane module composed of silicone rubber hollow fibres capable of producing 0.15 L of nitrite concentrate per L of influent urine with minimal COD content. Packing density and oxygen demand considered are 500 m^2/m^3 and 3.95 kg/d, respectively, giving a required tank volume of 1 m^3. The combination of an EC and MABR has been proposed as a possible treatment combination to reuse and process urine in space applications, proving to be a compact layout also suitable for extremely decentralized systems (De Paepe et al., 2020a; Zhan et al., 2022). Details on the recovery potential of this configuration can be found in the Resource recovery section.

2.3.7 Rainwater treatment line

Regarding RW collection and treatment, a treatment train consisting of a repetition of the disinfection section of the reuse layout (RO and UV disinfection) is installed with a previous UF filter and a final remineralization unit. Thus, as RO shows removal efficiencies higher than 90% for almost all influent components, a calcite bed contactor is used as a safe and reliable remineralization technique (Garfí et al., 2016; Mohamed Ghali et al., 2017). The combination of RO and UV provides reliable pollutant and microorganism removal efficiencies for potable water applications, and represents more than half of the currently active potable reuse schemes (Jeffrey et al., 2022).

Additionally, a collection tank and a potable water holding tank of 15 m^3 each with polyvinyl chloride (PVC) and flexible polypropylene (FPP) liners are needed to enable proper functioning of the RW treatment system. To provide an impact assessment of the system which is equal among scenarios, tank sizing is done according to potable water demand, even though optimization of this design according to localized rainfall patterns is preferred. The impacts associated with the construction of two tanks are of not much significance to the total results, so the effect of this decision can be considered low.

2.3.8 Additional equipment

Additional equipment necessary for the processing of water and included in the LCA are influent and effluent equalisation tanks, and diverse holding tanks for urine, sludge, process products, and chemicals required for cleaning and operation. Equalisation units are steel tanks of 30 m^3 with a PVC liner and 25 m^3 with a FPP liner for the GW+BrW influent and total effluent, respectively, sized according to Manderso (2018). BrW and OFMSW are mixed and homogenized during pre-treatment in a 3 m^3 stainless steel tank. The urine treatment line requires a 2 m^3 urine and a 5 m^3 treated water holding tank, made of fibreglass and PVC, respectively. Finally, PVC chemical and sludge holding tanks of 10 L and 750 L for monthly chemical recharge and sludge disposal are included.

Details on the dimensions and characteristics of tanks and other supporting equipment (e.g., pumps, valves, holding vessels) can be found in the construction inventory section of the supplementary material.

2.3.9 Health and safety

Even though VUV disinfection weakly reduces oxygen demand and nitrogen concentrations (Piras et al., 2022), its main aim in the layout is to achieve pathogen inactivation to ensure compliance with reuse water standards regarding microbial presence (Adams, 2015).

The combination of MBR+RO+UV provides a highly effective, multi-barrier treatment train for potable water reuse also proved more sustainable than competing alternatives (Akhoundi and Nazif, 2018; Tarpani and Azapagic, 2023). MBR technology allows to achieve RO feedwater quality and provides pathogen removal mainly by size exclusion for protozoa and bacteria and adsorption onto sludge, predation by other organisms, and filtration in the membrane for virus (Katz et al., 2019). RO enables a consistently high quality of effluent water and represents a second barrier against microbial risk in the GW train, with UV being the final disinfection stage. The system is expected to comply not only with standards established for water reuse but also with drinking water quality standards (Ghernaout, 2019; Tang et al., 2018).

However, despite efforts to achieve and maintain a safe water composition comparable to that of tap water, public acceptance of directly reused water and current legislation regarding reuse pose a challenge. While perception of recycled water for non-personal uses (e.g., irrigation, toilet flushing) is positive and approval rates by general population are nearly up 90%, if water is aimed at purposes that involve contact or consumption, acceptance rates plummet to less than 9% (Al-Saidi, 2021; Nkhoma et al., 2021). The main factors triggering public rejection include disgust, distrust, and disinformation. This global reticence to the adoption of recycling measures burdens the deployment of technologies and the development of polices around reuse. Addressing these issues at both an institutional or societal level and at private or personal settings can help increase awareness and adaptation to this new water augmentation technique. Appropriate monitoring of the process, remineralization of the effluent to drinking water mineral contents, and necessary caution with the appearance, taste, and odour of water would be highly relevant if the reused flow were to be employed for potable purposes and would improve consumer perception (WHO, 2022). In the proposed layout, potable water is obtained from RW, and wastewater is reused for non-potable applications.

2.3.10 Resource recovery

A priority of the decentralized system is its adjustment into a circular economy context. As pressure on natural resources increases, recognizing possible strategies to close material loops is of high importance. The recovery of nitrogen and phosphorus (i.e., the two key nutrients present in wastewater and used as fertilizers) could decrease phosphate rock mining and dependence on highly energy-consuming production processes for N- and P-based fertilizers for agriculture by up to 20%. Besides, energy stored in wastewater as COD can be transformed into electrical and thermal energy to damper the energy consumption of the WWTPs themselves or help partially substitute non-renewable sources of energy (Ostermeyer et al., 2022). Other secondary resources that can be recuperated from wastewater include coagulants, metals, and inerts. The presented decentralized configuration focuses on the recovery of energy in the form of biogas and nutrients through source-separation and specific treatment processes aimed at achieving minimal loss of resources.

MFA facilitates the identification of resource loss sites in the treatment trains and has been used to optimize both resource recovery and the approach to sludge management with respect to the original system.

In regards to energy, organic matter is transformed into biogas during anaerobic treatment, which has to be properly managed as the loss of methane in dissolved form in the effluent can be higher than 50% of the total produced gas due to supersaturation (Cookney et al., 2016; Velasco et al., 2018). Gas advection (i.e., transfer) from bed to headspace is limited in UASB treatment as no mixing is involved and fluid velocities are low inside the reactor. When, additionally, a high methane production rate is achieved thanks to high loads of organic matter in the influent as projected in the studied system, supersaturation can be even further increased in UASB reactors than in other anaerobic processes (Crone et al., 2016). Thus, not only the efficiency of the process is reduced, but methane losses also entail a poor performance in terms of resource recovery and can become a source of relevant GHG emissions burdening the sustainability of the system.

Different membrane-based and non-membrane-based (incl. stripping, aeration, and biological oxidation) approaches can be taken to achieve high recovery of dissolved methane in UASB reactors and other anaerobic treatments (Crone et al., 2016). Membrane-based recovery is often recommended as a low-energy consumption approach to methane collection which is also of easy installation as a complement to existing reactors.

Specifically, non-porous membranes are preferred for UASB reactors as they have lower organic material removal efficiencies than alternative anaerobic treatments such as anaerobic membrane bioreactors (AnMBRs), and this could cause pore-wetting in porous membranes. Hollow fibre can be used to fabricate a wide range of types of membranes, which are often used in wastewater treatment due to their versatility and high surface area. Out of two possible configurations (incl. influent flow inside or outside the fibre), effluents with potential to contain suspended solids such as those in the studied system are recommended to be passed through the outside of the fibre to avoid clogging (Crone et al., 2016).

The use of a non-porous hollow fibre membrane contactor (HFMC) made of polydimethylsiloxane (PDMS) operated with vacuum is proposed for the study (PermSelect - MedArray, Inc., MI, USA). The polymer significantly effects the permeability of the membrane and can influence gas collection. With the described equipment, methane recovery of over 92% can be achieved, including both headspace and dissolved gas (Bandara et al., 2011; Cookney et al., 2016; Crone et al., 2016; Gabelman and Hwang, 1999; Velasco et al., 2018). Besides allowing dissolved methane capture, the use of a degassing system in the UASB helps increase the system’s COD removal efficiency due to improved thermodynamic conditions inside the reactor (Bandara et al., 2013; Crone et al., 2020).

The energy requirement of the degassing equipment can be calculated through Eq. 1 where W_{deg} is the power requirement for degassing [Wh], k is the heat capacity ratio, W is the influent molar flow rate [mol*s^{-1}], R is the gas constant, T is the inlet temperature [K], and p_{in} and p_{out} are the inlet and discharge pressures [kPa] (Crone et al., 2016).

$$W_{deg} = \frac{kWRT}{k-1} * \left[\left(\frac{p_{out}}{p_{in}} \right)^{\left(\frac{k-1}{k}\right)} - 1 \right] \quad \text{Eq. 1}$$

Numeric values for the case under study are found in Table 3, resulting in a power demand of 0.052 kWh per m^3 of treated effluent.

Table 3: Parameters regarding energy consumption of dissolved methane recovery. Constants are obtained from Crone et al. (2016).

Parameter	**Value**
Heat capacity ratio (k)	1.295
Influent molar flow rate (mol/s)	0.002
Gas constant R (J/molK)	8.314
Inlet temperature (K)	308.000
Inlet pressure (p_{in})	101.300
Discharge pressure (p_{out})	21.300

Once biogas is captured, direct conversion into heat to cover the reactor and degassing membrane's own heat requirement is an efficient way of satisfying heat requirements and lowering the energy demand of the system. This is commonly done by gas microturbines or with the installation of a combined heat and power generator (Garrido-Baserba et al., 2022). Generators' efficiencies range around 40% for conversion to grid electricity. However, since the UASB operates with high organic matter loads in the influent and therefore has an elevated heat demand, there is no energy surplus from biogas and its energy is used entirely as heat through its combustion after collection.

To obtain the resulting percentage of energy demand of the complete anaerobic system that is covered by biogas, the operation consumption of the reactor and methane production are calculated as follows.

Energy demand in the UASB is required to pre-heat the influent to the operation temperature of 35ºC (T_{op}), required for digestion of high-loaded BrW including food waste with an average specific heat (W) of 4.2 kJ*kg^{-1} (Adami et al., 2020; de Graaff, 2010; Metcalff and Eddy, 2013). A heating efficiency (η) of 80% is assumed in the reactor (Metcalff and Eddy, 2013). The annual energy required for the preparation of the influent is that of 38,200 kWh as calculated from Eq. 2.

$$H = \frac{Q * \rho * W * (T_{op} - T_{in})}{\eta} \quad \text{Eq. 2}$$

A COD-methane conversion rate of 60% is achieved in the UASB operating at 10.55 kg COD*m^{-3}*day^{-1} OLR thanks to the inclusion of OFMSW in the feed mix which is significantly higher than the 40-50% usually achieved by the digestion of BW alone (Gao et al., 2020; Kujawa-Roeleveld et al., 2005; Vinardell et al., 2020; J. Zhang et al., 2021).

As stated, fertilizer-used nutrient recovery is also performed focusing on nitrogen and phosphorus capture. Besides representing a sustainable route towards the obtention of fertilizers, nutrient recovery also eliminates the need for energy-intensive and high sludge production removal processes. Nitrogen and phosphorus removal are tackled by the incorporation of precipitation and membrane filtration units in the layout.

In regard to the BrW line, which treats high loads of both nitrogen and phosphorus, a combination of a crystallizer with $Mg(OH)_2$ addition followed by a decanter is proposed as a way to precipitate struvite ($MgNH_4PO_4 \cdot 6\ H_2O$) through Eq. 3 at pH 8. Precipitation starts when the chemical entities of struvite (i.e., Mg_2^+, NH_4^+, and PO_4^{3-}) are present in the solution and supersaturation by pH increase is reached (Kumar and Pal, 2015). pH higher than 8.5 is avoided to minimize the presence of calcium precipitates and eliminate nitrogen volatilization, which decrease struvite formation. At optimal conditions of pH from 7.5-8.5, more than 90% struvite content is achieved in the solution.

As observed, struvite is both ammonium- and phosphate-rich, while also being able to supply magnesium to plants with the same molecular ratio and therefore being considered a highly valuable by-product of recovery-based wastewater treatment.

$$Mg^{+2} + NH_4^+ + H_nPO_4^{n-3} \rightarrow MgNH_4PO_4 * 6H_2O + nH^+ \qquad \textit{Eq. 3}$$

Struvite is a slow-release, multi-nutrient fertilizer. Unlike other nutrient-rich products of wastewater such as concentrated sewage sludge and even commercial fertilizers, it is free of heavy metals, which avoids accumulation in plats or soil. Moreover, it is only slightly soluble in water making it suitable for fertilizing crops which only receive nutrient supplements once in between considerably long-time intervals. Struvite has been demonstrated to be appropriate for both soil and foliar applications (Kumar and Pal, 2015).

Due to the high concentration of nutrients in urine, recovery is the focus of the YW line. Pre-treatment in the EC helps reduce organic matter content in the fluid with the aim to produce a highly concentrated, nitrate-rich effluent with low COD presence suitable for algae and plant fertilization (Coppens et al., 2016; De Paepe et al., 2020a). EC treatment considers low HRTs of less than a week and achieves COD removals of up to 88% while avoiding nitrogen loss (Walter et al., 2018). A conservative value of 80% removal efficiency is considered in the present study. TN also undergoes conversion to TAN in the cell with nitrogen loss of only 2%. The remaining 98% of nitrogen is found in the form on ammonia and fed into biological treatment focused on nitrification in a MABR which achieves full conversion to nitrate and produces liquid fertilizer (De Paepe et al., 2020a). The pH increase caused by urea hydrolysis causes the precipitation of salts present in urine (Ca, and Mg) in combination with phosphates. Recovery of mainly $Ca_3(PO_4)_2$ is expected as a result which can be reused as fertilizer or in the phosphorus industry. Secondary by-products include struvite and calcite, as well as lower quantities of additional magnesium precipitates (De Paepe et al., 2020b; Randall et al., 2016; Udert et al., 2003).

In order to optimize the recovery potential of the layout, it is considered sludge from biological units is processed with anaerobic digestion enabling for the production of additional biogas and biosolids. These are product of sludge treatment that offers valorisation opportunities and to which conventional treatment often relies on as a way to avoid total nutrient loss. They are also commonly named sewage sludge as they represent the useful fraction of all sludge produced in WWTPs. Existing legislation and guidelines classify biosolids between “Class A” and “Class B” (EPA, 1994), the former meeting stricter requirements regarding pathogen and metals presence, and odour and vector attraction reduction levels. Class A biosolids can be used for all land application purposes, including (but not limited to) agriculture, forestry, and ecosystem reclamation (Collivignarelli et al., 2019; Ostermeyer et al., 2022). Currently, in conventional WWTPs producing biosolids, fractions of 43% and 50% are used in land application in the U.S. and in the EU-28, respectively. An additional 40% of the total are disposed in landfills, and around 15% are incinerated. Other end uses of biosolids include storage and deep-well injection (Collivignarelli et al., 2019; EPA, 2023; Gianico et al., 2021). Developing countries struggle more to close the cycle and reuse sewage sludge, as well as with achieving adequate sanitization before application. Agricultural application is considered as the end use of the produced biosolids.

2.3.11 Sewer system and pumping

The need to separate BrW, GW, and YW and capture RW creates the necessity for additional piping in the building. The already existing sewer network is considered to be suitable for the transport of GW and re-circulation of potable water after treatment. The impacts of extra piping for BrW, YW, and RW are included in the study.

High density polyethylene (HDPE) pipes of standardized diameters of 32 mm and 75 mm are used for the collection of both YW and RW, and BrW, respectively. The length of piping necessary is calculated based on the dimensions of the building described in section 2.3.1. based on the Urban Water Infrastructure Model (UWIM) methodology by Maurer et al. (2013). The UWIM model uses housing density [ρ, dwellings*m^{-2}] and dimensions (A [m^2] and dimensionless model parameter f_2 based on the ratio between length and width of A) to give the meters of private water (L_s [m]) distribution pipeline that adjust to the target settlement through Eq. 4 (Maurer et al., 2010).

$$L_{s,private} = 0.5 * A * \sqrt{f_2 * \rho} \qquad \text{Eq. 4}$$

The housing shape factor (which takes values between 0.2 and 5) is taken as 1.44 (Maurer et al., 2013, 2010).

Pumping is required for the correct collection of BrW and avoidance of pipeline clogging, as well as for the return of treated water to households, both for GW and RW. Sizing of pumps considers a maximum height of 36 m, the flow rates of each type of water, and a water pump efficiency of 70%. In the case of RW, flow rate is sized according to demand. The proposed pumps could deliver adequate water pressure of 50 to 70 psi to showers, sinks, and other house appliances.

2.4 Material Flow Analysis

In industrial ecology, the quantification and representation of materials or energy within a system is commonly achieved through material flow analysis (MFA), a methodology which helps map the movement of the selected flows of interest in their studied context (Graedel, 2019; Laner et al., 2014). This tool is often applied to systems involving resource recycling or reuse to gain insight on their efficiency, therefore being highly adequate for the current assessment, and commonly relies on Sankey diagrams for the clear representation and communication of results. Thus, adhering to common practice, the MFAs for this study present data in a both diagrammatic and numeric way. Representation is carried out in diagram software e!Sankey.

When the analysed system is subject to temporal variance, a dynamic MFA picturing several cycles during a multi-year time period can be used. The MFAs for the present study are static and represent the operation of the decentralized system under normal conditions and in steady state. Variations from the presented flows are to be expected during exceptional events, such as system failure, shutdown, or if pollutant loads deviate from normal. Average values for wastewater originating from households in western countries are shown, and uncertainty is minimized by combining calculations of removal, recovery, and loss of components with simulation results using SIMBA#. A total of 300 days of operation are simulated to ensure that results reflect operation at steady state, and component balances are extracted. Final values for flows included in the MFAs are a result of both methodologies. The treatments represented in the simulation include the UASB reactor, OLAND reactor, and MBR reactor. The struvite reactor, RO membrane, and YW treatment calculations are based on literature and company data (Ostara Nutrient Recovery Technologies Inc., Vancouver, Canada; Applied Membranes Inc., CA, USA; Hydrohm, Ghent, Belgium; Dow, MI, USA; PermSelect - MedArray, Inc., MI, USA; CYPE, Alicante, Spain; Sigmadaf, Girona, Spain; Nagayanagi Co., Tokyo, Japan). Coincidence for both removal efficiencies and treatment parameters (incl. effluent/influent ratio, aerobic and anaerobic reactor volumes and HRTs, biogas production in the UASB reactor, and biofilm surface per volume of RBC) between calculations and simulation are verified. The OLAND process carried out in the RBC is simulated in four separate stages instead of in the reactor with reduced size included in the presented system to enable modelling.

Nutrient (incl. TN and TP), volume, and energy analysis are presented for the decentralized treatment and reuse infrastructure taking the 300 inhabitants building as a basis. Additionally, to represent the flow of organic matter throughout the layout, a COD balance is included. For centralized treatment, nutrient and COD balances are used for comparison with the decentralized layout.

Figures in the Results section and supplementary material are elaborated for pollutant flows including a joint Sankey diagram for COD (dg/d), TN (g/d), and TP (g/d), and a separated MFA only for nutrients with the aim of facilitating visualization. SIMBA# simulations are also found in the supplementary material to showcase conformance between MFA figures and the model of the system. Additionally, a figure representing volume of water (L/d) and energy (kWh/d) consumed and produced by the process in the same diagram is elaborated in accordance with typical practice. Water and energy are often represented together as their relationship is close, considering the obtention of energy through water and the energy consumption originating from water treatment schemes (Sanders and Webber, 2012; United States Department of Energy, 2014). SIMBA# diagram for volume is also provided. Re-circulation of pollutants present in the effluent flow is not illustrated as these flows are not visible when scaled against input pollutant loads. However, values for input concentrations include the re-circulated loads of all chemical entities.

Besides providing a visual and numeric representation of the studied process and the decentralization concept, MFA establishes clear system boundaries, helps define the framework in a consistent way, facilitates the understanding of flow movement within the scope, and ensures consistency with mass balances, not only for the represented flows, but for all relevant to the LCA. As MFA provides a broader-picture overview of the systems that LCA is unable to attain, these two methodologies are often linked and used together for environmental management purposes, with a wide range of studies performing both MFA and LCA to assess environmental performance for the design, assessment, or comparison of alternative systems (Birat, 2020; Padeyanda et al., 2016; Stijn et al., 2020; van Stijn et al., 2022; Withanage and Habib, 2021).

In the performed MFAs, recycled water, nutrients, and energy provide especially relevant data for the LCA.

2.5 Life Cycle Assessment

Life Cycle Assessment (LCA) is a tool capable of quantifying the impacts of a system (e.g., process, activity, or product) by compiling its exchanges with the environment and technosphere. Both inputs and outputs are evaluated, leading to an analysis of the potential sources of harm, the magnitude of their effects, and their overall importance over the life cycle of a product or system.

An attributional approach to modelling is taken, as the aim of the study is to quantify the impacts associated with the treatment of water through the defined system. The alternative would be a consequential LCA, which is used to assess the changes on impacts derived from modifications of the studied activity.

The solid establishment of LCA as one of the main tools for evaluating environmental performance led to the standardization of the methodology and structure that assessment studies follow. This work will adhere to the ISO 14040:2006 and ISO 14044:2006 standards, which were both reviewed and confirmed in 2022 and provide guidelines for the use of LCA. Four interconnected phases consisting of the description of the goal and scope of the study, the building of the inventory, the elaboration of the impact assessment, and the interpretation of results create an iterative process where the decisions taken in each phase are constantly revised and reformulated according to the needs of the study (ISO, 2006). The criteria established for the development of the standardized sections of the LCA in the present study are discussed in the following sections, including the definition of goal and scope, functional unit, and scenarios. Considerations related to the construction of the life cycle inventory and the impact calculations are also detailed.

2.5.1 Goal and scope

2.5.1.1 Goal and approach

LCA has evolved from being a tool used to quantify the life cycle impacts of different activities to a widely used methodology for a broad range of applications. Currently, it is employed in process design, marketing, decision- and policymaking, and facilitates the shift towards sustainable development altogether.

The main goal of this study, which matches that of the LCA, is to evaluate the environmental impacts of water and wastewater decentralization. Therefore, the impact assessment for an on-site recycling treatment layout developed for enhancing the sustainability of water management is carried out. This standardized methodology allows for a transparent analysis which can provide aid in the assessment of the system during its initial design phase and enable comparison with existent water treatment technologies. An attributional, process-based study focused on the designed treatment technology configuration allows for the obtention of quantitative information on the impacts directly caused by the life cycle of the system.

The impact assessment of conventional treatment is carried out to allow for the development of a comparative LCA, with the aim to reveal the differences in performance between the decentralized layout and centralized water treatment.

The scientific community and decision-makers are the two parties at whom the research is principally aimed, as environmental analysis should be in the spotlight when working on the transformation of the water management sector. However, technology manufacturers and all users of the water network can make use of the results of environmental assessments for marketing purposes, or informed decision-making, respectively.

2.5.1.2 Scope and system boundaries

In LCA, boundaries definition involves the identification of the relevant processes and stages of the life cycle of the system, as well as the establishment of a geographical and temporal framework. The selection of the phases included in the analysis must be consistent with the purpose of the study.

Inputs and outputs associated with both the upstream and the downstream of the objective activity should be revised and ideally included in the boundaries if they are necessary for the functioning of the assessed system. In the case of wastewater treatment, upstream activities include the acquisition, transport and processing of materials, energy, and fuel required for the construction and operation of the studied treatment plant, while the main downstream activities involve the management of wastes and recovered resources, as well as the end-of-life of the elements (Corominas et al., 2020; Kobayashi et al., 2020).

Different approaches can be taken according to the extent of the boundaries. While a so-called gate-to-gate scope focuses on the studied system (e.g., the in-plant production of a chemical), if upstream processes are included the LCA takes cradle-to-gate boundaries, and a complete assessment additionally involving downstream exchanges requires a cradle-to-grave view.

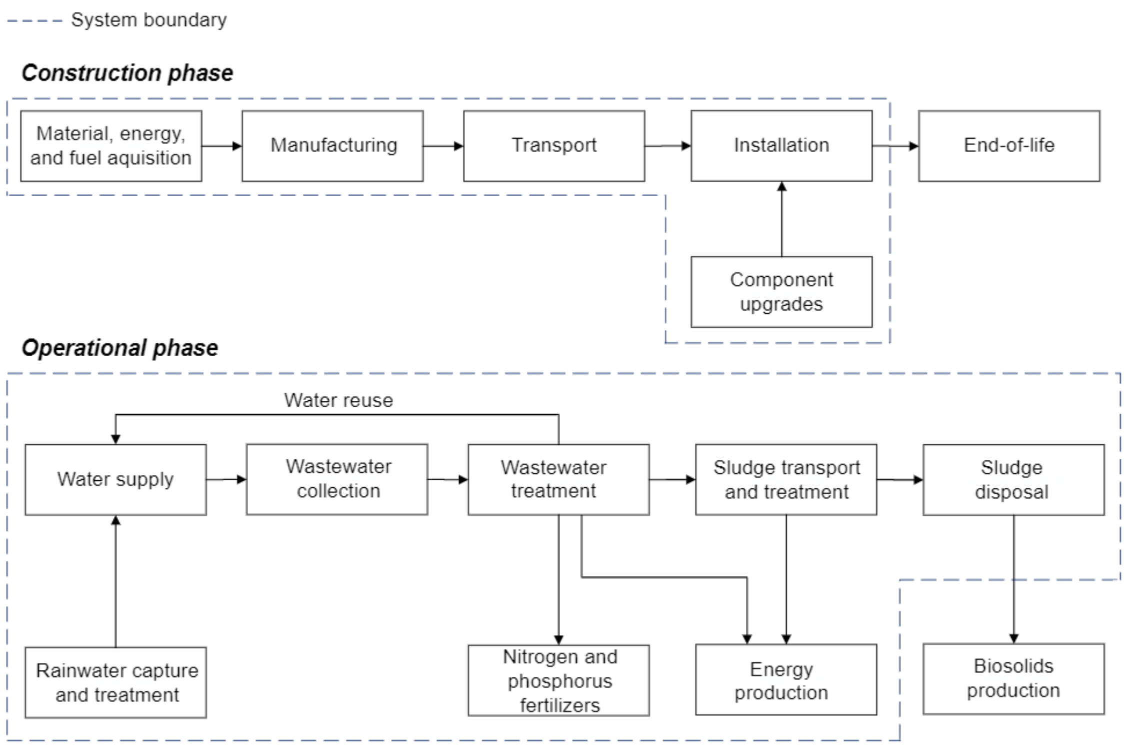

Figure 5: Simplified view of the established boundaries, applicable to both centralized and decentralized treatment.

With the aim to assess the overall environmental performance of decentralized wastewater treatment systems, the boundaries include the construction, operation, maintenance, waste treatment, and management of resources generated by the technology in the European region (see Figure 5). In order to obtain an up-to-date impact assessment, data is collected from reported values, personal calculations, and wastewater treatment simulation software SIMBA#, and combined with database information collected from ecoinvent v3.9.1. which was released in December 2022.

The installation of decentralized systems can lead to significant costs for both existent and new construction buildings, as on-site infrastructure is required for each housing block. Thus, the construction phase has the potential to represent a substantial contribution to total impacts and is therefore included considering the material investment, the production processes of all components, and the installation of the system. The installation of additional piping for BrW, RW, and YW, and the materials and processes required for the manufacturing of the treatment equipment are expected to have the largest impacts in the construction phase. All equipment has a specific lifespan which influences the magnitude of the final construction impacts. Details on materials and lifespans can be found in the construction inventory section of the supplementary material for pipeline, pumps, treatment units, and supporting equipment such as valves, holding tanks, vessels, and filters. The production processes considered for the manufacturing of all components are also listed under the construction inventory section, as well as all considerations taken regarding installation.

Operational impacts are expected to be lower than those of traditional treatment due to optimized processes and smaller treatment volumes made possible by source separation. Additionally, lower energy consumption and chemical additions are required. Emissions to air, water, and land as well as inputs from the technosphere are considered. As a result of on-site reuse, emissions to soil and water originating from the effluent are avoided (Hasik et al., 2017). Micropollutants and microbial pathogens are excluded from the analysis as they fall out of the scope of both current LCA methodologies (Corominas et al., 2020; Harder et al., 2017). Details on exchanges can be found in the operation inventory section of the supplementary material.

The management of sludge is approached through anaerobic digestion with the production of biogas. The sludge anaerobic digester and post-treatment equipment are not assumed to be installed in the building but outsourced. Therefore, transport to the treatment facility is considered.

Resource recovery is a key step of the decentralized process which focuses on minimizing and recapturing waste, as well as working towards a circular economy concept. Recovered resources in the building include energy from biogas in the anaerobic treatment stage, which is employed in the reactor itself to cover part of its energetic demand, and N-based and P-based fertilizers from urine and BrW. The possibilities and implications of substitution are studied and detailed in section 2.5.2.

The total operation time of the decentralized system considered in the study is that of 30 years, and components are assumed to be replaced when required within this period. Membrane upgrades are potentially one of the most material and energy intensive maintenance activity of the process. As suggested by Corominas et al. (2020), secondary maintenance activities with less significance such as the cleaning or oiling of components or those of little environmental impact (e.g., sample extraction or monitoring) are excluded from the study. In concordance with usual LCA practices, end-of-life impacts are considered out of the scope of the thesis as their contribution to final impacts is negligible in comparison with that of operation and installation (Carré et al., 2017; Corominas et al., 2020).

The boundaries for traditional, centralized water treatment are equivalent to those of the decentralized system in the aim to obtain a robust comparison between the two water supply routes. Therefore, the infrastructure and operation of centralized plants is accounted for, as well as the impacts related to the deployment and use of pipeline and pumping stations for water collection, transport, and distribution. Relevant volumetric losses and environmental impacts are associated with sewer networks, therefore their inclusion in the study is crucial for fair comparison. Besides the treatment of wastewater itself, as the decentralized option offers a drinking-water quality effluent with reuse potential, the production of potable water and its distribution is also included in the scope of the centralized treatment scheme (part of water supply in Figure 5). Sludge management, component lifespans, and their respective upgrades are included. Scaling of piping impacts is considered in order to make centralized management comparable to on-site reuse.

2.5.1.3 Functional unit

The functional unit of the study is the need to provide water supply and wastewater treatment to 300 people during a year. The provision of drinking water from rainfall is also included in the site-specific cases.

The total population of 300 inhabitants has total GW and BW demands of 32400 $L*d^{-1}$ and 4500 $L*d^{-1}$ and an annual wastewater production of 13,518 m^3*yr^{-1}. The composition of wastewater providing the pollutant loads entering the treatment lines is specified in Table 1.

2.5.2 Inventory construction

The "*Allocation at the point of substitution*" ecoinvent system model is used, which considers that if waste requiring further treatment is produced within the studied process, the impacts of its treatment are attributed to the main activity. Thus, as waste treatment is included as part of the total activity, in the case that valuable by-products are produced, this rule also applies. Therefore, allocation distributes total impacts among not only those by-products obtained in the main process but also those derived from the treatment of waste. Besides, the "*market for*" datasets of each component are utilized as they include the impacts of the average transport distances within the corresponding area (Wernet et al., 2013). Additional transport needs (e.g., commuting of workers) are added to the inventory independently.

The specific flows, amounts, and activities considered in the inventory are detailed in the supplementary material, as well as the calculations and assumptions made to obtain each exchange. The construction inventory is divided between material requirements, production processes, and installation; while for the operation inventory, special focus is placed on the approach taken regarding recovered resources (incl. water, energy, and nutrients). Table 4 shows the sections in which the inventory is organized.

Within the recovered resources, recycled water and treated RW are considered a replacement for tap water, as they offer the same quality, including all activities that would be required for bringing this resource to households (i.e., extraction, treatment, and transport) through the average regional processes (Corominas et al., 2020; Hasik et al., 2017; Santana et al., 2019). Besides, since the equipment necessary for sludge treatment is not part of the studied decentralized system, its impacts are modelled from ecoinvent. Following the same approach taken in the UASB reactor, sludge valorisation is achieved through its digestion enabling to produce biogas. The “treatment of sewage sludge by anaerobic digestion” dataset from ecoinvent is taken and normalized for the functional unit of this work (Chu et al., 2022).

Table 4: Summary of the sections of the inventory found in the supplementary material for decentralized and centralized water treatment.

	Construction	**Operation**
Decentralized system	Pipeline and pumping materials Treatment unit materials - Brown water line - Grey water line - Yellow water line - Rainwater line - Additional equipment Production processes Installation	Pumping Treatment units - Brown water line - Grey water line - Yellow water line - Rainwater line Resource recovery
Centralized system	Infrastructure - Pipeline and pumping - Potable water production - Wastewater treatment	Pumping Potable water production Wastewater treatment

To maintain consistency with the approach taken for sludge, the fertilizers and energy listed as “recovered” only originate from the decentralized system installed in the buildings (see Figure 4), meaning no biosolids are included, and energy from the UASB is treated independently to biogas from sludge and discounted from the energy consumption of the own anaerobic process.

Crediting of recovered nutrients as fertilizers in the LCA is done through substitution, following common practice (Corominas et al., 2020; Sena et al., 2021; Sena and Hicks, 2018; Temizel-Sekeryan et al., 2021; Wang et al., 2018). Main alternatives are system expansion or allocation of impacts (Ravi et al., 2022). Both phosphorus and nitrogen fertilizer offsets are included (Amann et al., 2018). Not considering magnesium, potassium, and calcium recycling could lead to an underestimation of the substitution potential of the system, however, their significance to the total impacts is assumed negligible due to the low available amounts of these nutrients (Remy and Ruhland, 2006).

Struvite is well established as a replacement for synthetic fertilizers (ISO, 2006; Ravi et al., 2022) with a total bioavailability (Chipako and Randall, 2020). In this study, monoammonium phosphate ($NH_4H_2PO_4$) is elected as the substituted product, as it maintains the N:P ratio of struvite ($(NH_4)MgPO_4 \cdot 6H_2O$)) and is a fertilizer of widespread use (Brye et al., 2022; Desmidt et al., 2015; Ishii and Boyer, 2015; Ye et al., 2021). Ca-P precipitates from urine (mainly calcium phosphate) have a bioavailability rate of 0.7 (Chipako and Randall, 2020) and are treated as inorganic P-based fertilizer. Finally, liquid concentrate from urine is expected to have a 4.5 – 0.29 – 0.8 NPK rating (content of nitrogen, phosphorus, and potassium by weight). It is considered a replacement for N-based and P-based fertilizer (Jurga et al., 2021; Lundin et al., 2000; Ranasinghe et al., 2016) with average bioavailability of 0.9 (Remy and Ruhland, 2006).

The approach to the inclusion of all recovered resources in the LCA is summarized in the operation inventory section in the supplementary material.

For the modelling of the centralized approach to water and wastewater management, the inventory related to potable water production, pipeline deployment, and pumping is built in Brightway2 based on equivalent volumetric requirements to the decentralized system. Four transport distances are considered, as depicted in Figure 6, including the sourcing and distribution of drinking water, and the collection and discharge of wastewater.

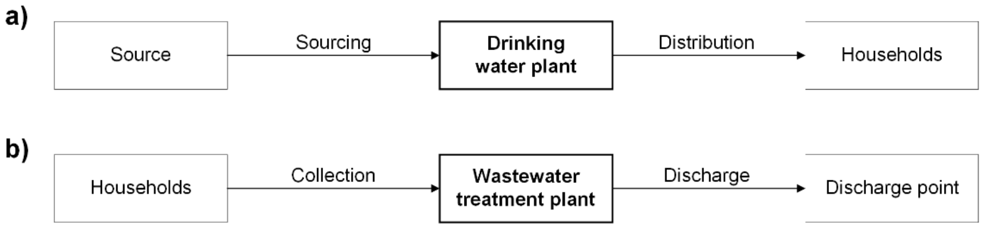

Figure 6: Transport and treatment sections considered in the inventory for centralized potable water and wastewater treatment.

The impacts related to the treatment of wastewater (gate-to-gate impacts for the treatment plant) are based on pre-reported values for conventional WWTPs operating in Europe (Besson et al., 2021; Heimersson et al., 2014; Risch et al., 2014). Additional review of available data regarding the impacts of conventional WWTPs is undertaken and all impacts are compared with those reported by published studies (Gómez-Monsalve et al., 2022; Kraus et al., 2017; Opher and Friedler, 2016; Santana et al., 2019). Review papers by Estévez et al. (2022), Foglia et al. (2021), and Mehmeti and Canaj (2022) are revised for the search of LCA studies which provide the impacts of traditional wastewater treatment and are comparable with this thesis.

Details regarding all considerations and assumptions related to the modelling of the centralized management scenario can be found in the conventional treatment inventory section of the supplementary material.

2.5.3 Impact assessment

ReCiPe (H) v1.03 is used as the impact assessment methodology. Results for the decentralized system are presented for all 18 midpoint indicators except for surplus ore potential (due to lack of available data) and for the 3 endpoint categories (PRé, 2018).

Human toxicity potential category (HTP) is presented as a unified value including both carcinogen and non-carcinogen impacts (HTPc and HTPnc midpoints in ReCiPe v.1.03), and photochemical oxidant formation potential (POFP) is an average between impacts to humans and ecosystems (HOFP and EOFP in ReCiPe v.1.03). Change, eutrophication, and ecotoxicity indicators are especially relevant in the water and wastewater field. The 3 ReCiPe endpoint categories facilitate the summarizing and communication of results (Risch et al., 2014). The ReCiPe indicators, how they relate to each endpoint area of protection, and the units they use are summarized in Figure 7. Boxes in bold mark those indicators included in the present study. Even if the material resources indicator (surplus ore potential or SOP) is not quantified due to lack of midpoint data availability, the endpoint comparison for natural resources does include metal depletion.

Impacts shown in the results section correspond to one year of operation of the plant, meaning all impacts related to the construction phase are annualized in accordance with the expected lifespan of the plant of 30 years.

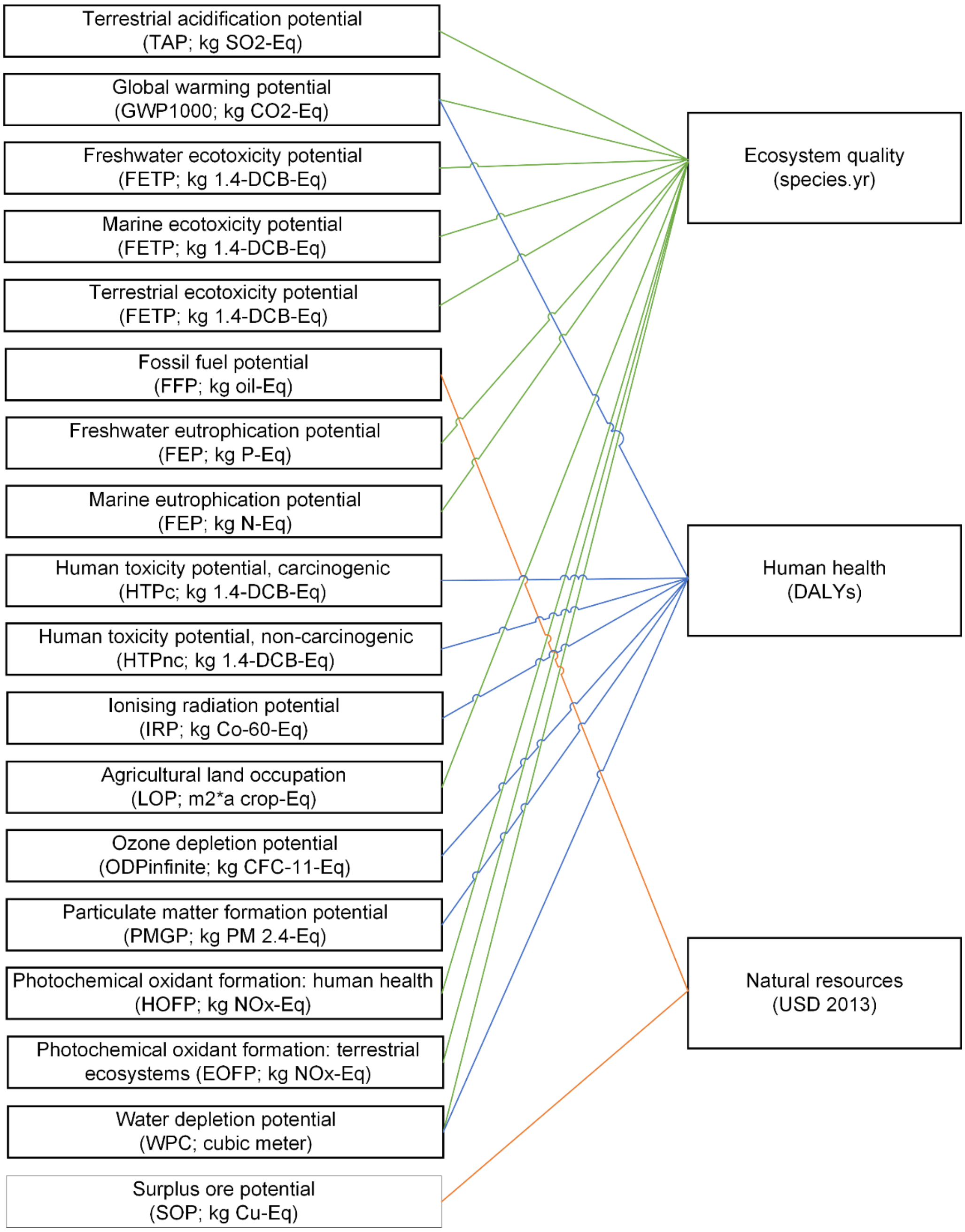

Figure 7: Midpoint indicators and their relation to endpoints.

2.5.4 <u>Interpretation of results</u>

Additionally to the discussion of the LCA results and the drawing of conclusions for the study, the completeness, sensitivity, and consistency of the data is checked, as well as the limitations of the work and the recommendations for further research (ISO, 2006).

2.6 Scenarios

Besides the average scenario for a general European setting, four different geo-specific cases are considered for the cities of Barcelona (Spain), Trondheim (Norway), Vienna (Austria), and Bucharest (Romania). For all scenarios, the inventory is adapted to analyse the influence of local characteristics on the performance of decentralized systems. The main factors that are modified include the electricity mix, the fertilizer obtention technologies, and the climate.

Table 5 showcases the main characteristics of the five proposed scenarios and the abbreviations used to refer to them from this point onward. The selection of these cities is based on the aim to compare different climates (according to the Köppen climate classification) and technological contexts. Climate data is obtained from climatedata.org and distances are calculated in Google Earth v. 9.189.0.0.

Table 5: Summary of the main characteristics for the five scenarios.

	Europe	Barcelona	Trondheim	Vienna	Bucharest
Abbreviation	EU	ES	NO	AT	RO
Köpper climate classification	-	Csa	Dfc	Cfb	Cfa
Climate name	-	Mediterranean	Continental Subarctic	Marine West Coast	Humid Subtropical
Rainfall (mm/yr)	-	614	1123	703	699
Electricity mix (ecoinvent)	RER	ES	NO	AT	RO
Sourcing distance (km)	36.5	52.5	15	165	14.5
Wastewater collection distance (km)	10	6	4	9	11.5

Due to the inadequacy of the use of a European average for rainfall, the EU scenario only considers the wastewater reuse system, and excludes the capture and potabilization of RW. The centralized system modelled for comparison also only includes the pumping and piping necessary for the transport of the volumes related to produced wastewater and recycled water.

In the site-specific scenarios, national ecoinvent datasets allow to study the changes to the impacts caused by the use of regional electricity mixes, fertilizer obtention routes, and waste management technologies. Besides, rainfall patterns help estimate the amount of potable water available in each country and incorporate the RW scheme to the impact assessment. The variations on the final impacts taking the EU average scenario as a reference as studied. Figure 8 showcases the composition of the electricity mixes in the assessed countries.

Out of all scenarios, Barcelona is expected to receive the least rainfall, which influences the decentralized system's capacity to produce potable water. As seen in Figure 8, the country's electricity mix features the highest shares of nuclear and wind power among the studied cases.

Regarding Trondheim, it is expected that its climate can facilitate RW capture, and thus enhance the decentralized system's substitution capacity. Besides, the Norwegian electric mix is strongly based on renewable energies, specifically hydropower (Figure 8) which is potentially highly favourable for the proposed technologies, as they rely mainly on electricity for operation since the use of chemicals is minimized in the system. Austria's mix follows Norway's when it comes to using hydropower, but also includes a fair amount of imported electricity. Finally, Romania's mix is the most partitioned and shows the highest presence of fossil-based electricity.

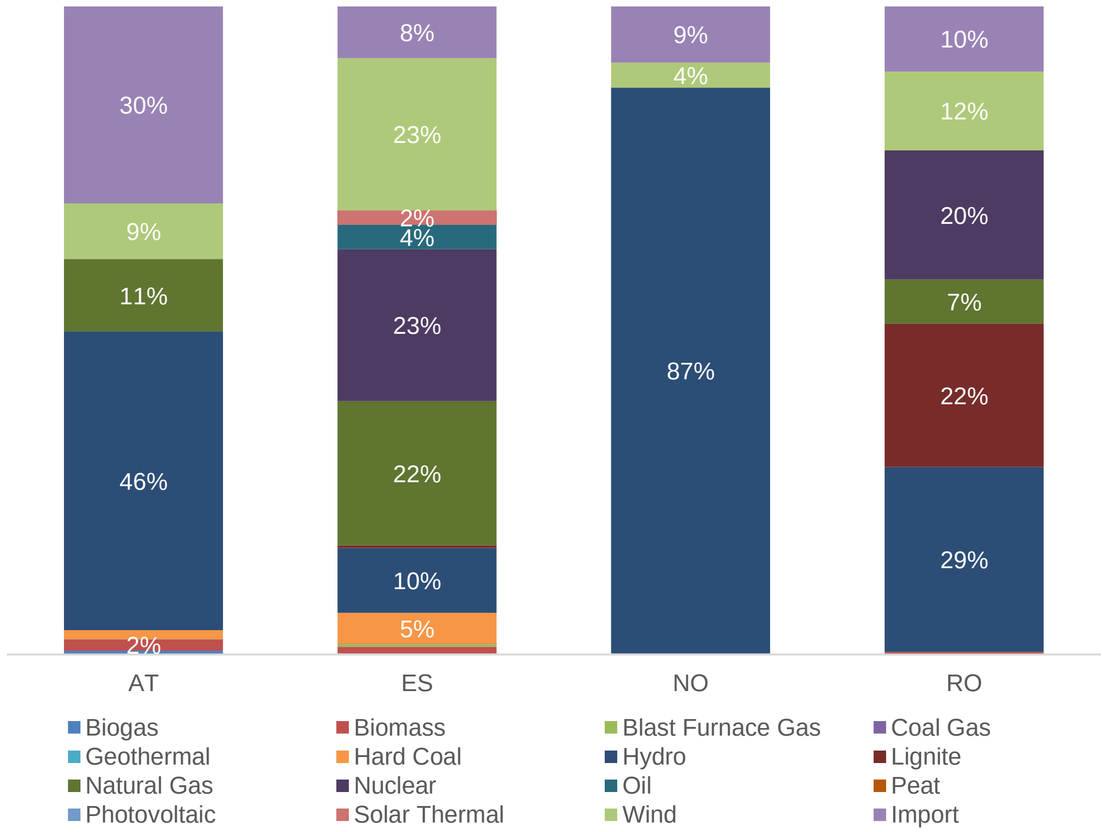

Figure 8: Electric mix used to model the four site-specific scenarios (Wernet et al., 2016). The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

A parallel analysis of the conventional treatment model has been developed to assess the influence on the scores of the characteristics of the water management approach in each city. Water transport distances and characteristics of treatment are modified. Potable water treatment plant and WWTP information, as well as water sources and considerations for each scenario are detailed in the supplementary material.

2.7 Software

The ecoinvent database (v3.9.1), providing data for the modelling of activities or processes is employed for the construction of the inventory. Besides, activities are modelled, and results are processed in the Activity Browser, an LCA open-source software built upon Brightway2. Minitab (v 19.2) statistical software is used during the sensitivity analysis.

Finally, SIMBA# is a simulation platform for treatment plants and networks involving water, wastewater, biogas, and river sections, which is used in this thesis to is used to verify the correctness of the calculations for the design of the UASB, RBC, and MBR units. It considers interconnected modules representing process components such as aeration systems, pumps, biofilm processes, aerobic and anaerobic units, and process water treatments such as anammox. It provides a digital representation of the operation of the selected processes offering information on a wide range of aspects, such as predictions of water composition throughout the system based on pollutant degradation or nitrogen re-solution, gas production and composition, and long-term operation.

3 Results and discussion

3.1 Material flow analysis

MFA assesses the performance of wastewater reuse of the system and provides information on its capacity for resource recovery.

Given the fact that the augmentation system's main aim is to reduce water consumption, a nearly 83% of volumetric recycling is achieved. Besides, only in the principal treatment line and without considering sludge revalorisation, a recovery of 44% of input nitrogen is achieved in the MABR as liquid fertilizer, while an additional 1,4% is extracted in the struvite reactor. In regard to phosphorus, struvite production enables to recapture 26% of the influent mass flow.

Regarding energy, considering operation following the conditions described in the Resource recovery section and a methane content of 78%, 7,720 L of biogas could be produced annually. Simulation results back up these calculations. Considering a 92% methane recovery rate, which is in the lower end of the efficiency range for the selected equipment, over 7,102 L of methane can be converted into energy annually. The remaining 8% of methane is emitted to the atmosphere. From the recovered gas, and assuming a heat of combustion of 38,846 $kJ*m^{-3}$, 19,352 kWh can be produced per year. Thus, methane production and recovery cover up to 51% of the reactor's heat demand. However, as the overall energy demand is higher considering the energy consumption of degassing, pre-treatment, and mixing, the coverage percentage of total energy by biogas is 40.5% if only the main line (i.e., the equipment present in buildings) is considered.

Figure 9 and Figure 10 show the evolution of pollutant flows and volume and energy. Chemical pollutants are represented together with the adequate mass units for the correct visualization of all components, while the diagram for energy and volume uses two different unit types. MFA and SIMBA# diagrams for each component individually, including COD, TN, TP, and water volume can be found in the supplementary material, as well as a Sankey diagram equal to Figure 9 without COD to provide better understanding of nutrient flows in the system.

If the anaerobic digestion of sludge is added, nutrient recovery percentages are increased up to 61% and 89% for nitrogen and phosphorus, respectively. Biogas from sludge could increase energy production by 62%, offering a total of 31,378 kWh per year. This can cover the total demand of the UASB and the additional anaerobic digestion of sludge by 65%. Equal specific heat for sludge and water due to the high water content of the former is assumed to perform the calculations (Kor-Bicakci et al., 2019). The MFA in Figure 11 incorporates sludge treatment for the target processes (i.e., UASB, MBR, and MABR) to showcase the flow of nutrients and organic matter during sludge digestion and dewatering.

If sludge from the biological treatments is processed, resource loss takes place mainly in the biofilm and reverse osmosis processes. The OLAND-based reactor is the only process in the layout involving intentional nutrient removal and could be substituted for nitrogen recovery processes such as ammonia stripping (Kinidi et al., 2018). However, as stated in the Brown water treatment line section, the RBC is considered more suitable for extremely decentralized systems due of its lower maintenance requirements, complexity, and costs. Besides, air pollution derived from stripping can counterbalance the positive environmental effects of recovering ammonia.

Nitrogen and COD loss by gasification in biological treatments as well as conversion and assimilation of bacteria (Bertanza et al., 2017; Paudel et al., 2014; Tawfik et al., 2010) are difficult to avoid in the system, but measures could be incorporated to recover nutrients from the RO unit if necessary. In order to avoid additional use of chemicals, energy, and increased costs, no further treatment is applied to the RO concentrate containing the relatively small fractions of 2.8% and 5.7% of total nitrogen and phosphorus in the system, respectively. However, it could be approached by a combination of struvite and ammonium recovery by electrodialysis and crystallization, or by recirculation to the MBR unit with previous oxidation (Arola et al., 2019).

Finally, nutrient loss in urine treatment is minimal as the combination of EC cell and MABR operating at the described conditions is designed to optimize nitrogen recovery by converting TN to TAN with an efficiency of up to 98% in the electrochemical process enabling full nitrification in the MABR producing a stable nitrate concentrate suitable for cultivation or protein production (De Paepe et al., 2020a). Phosphate loss is reduced by its precipitation in the EC cell. An MFA with the sludge line for nitrogen and phosphorus without COD can be found in the supplementary material and has been used as the principal information for the enhancement of the resource recovery potential of the process.

Material Flow Analysis for extreme decentralization

Nutrient and COD balance

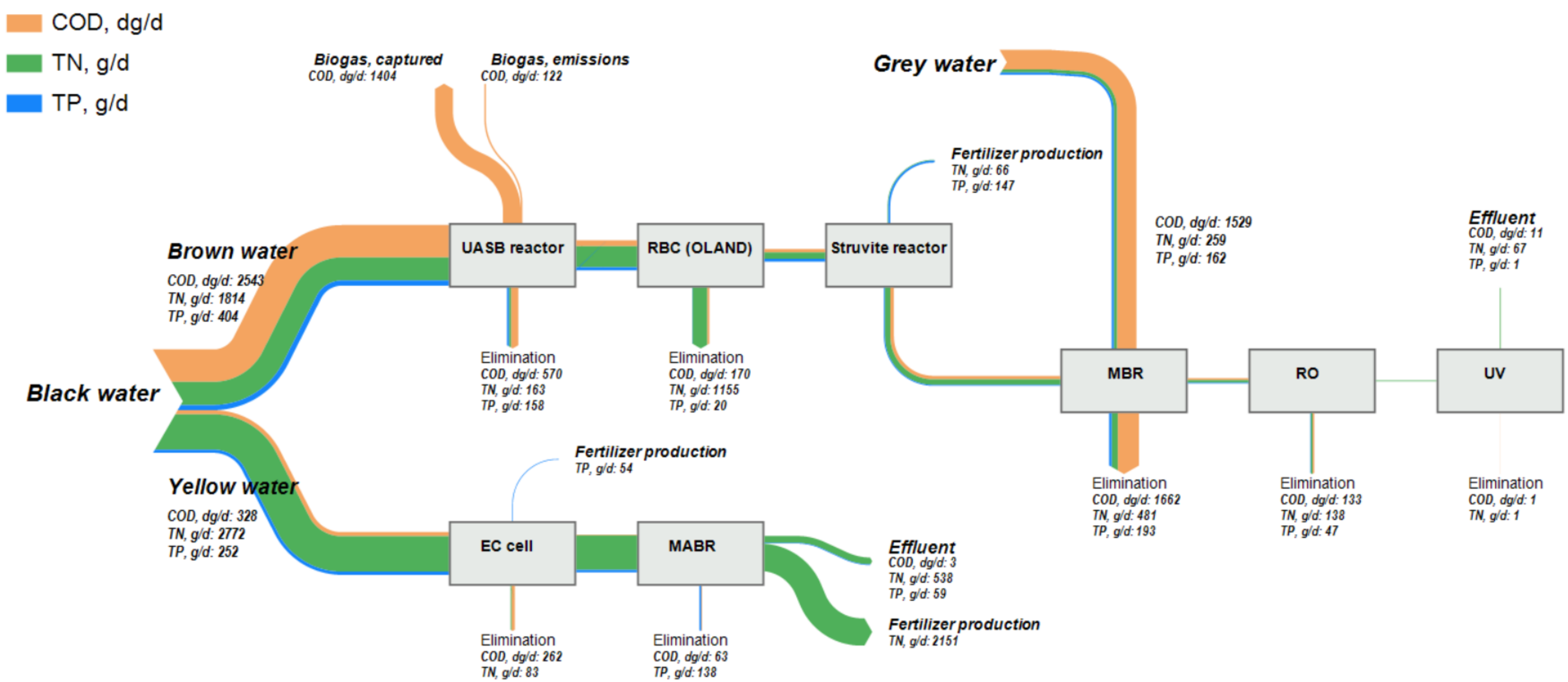

Figure 9: MFA of the decentralized system including mass flows for chemical oxygen demand (COD) in dg/d, total nitrogen (TN) in g/d, and total phosphorus (TP) in g/d, for the reference population of 300 people.

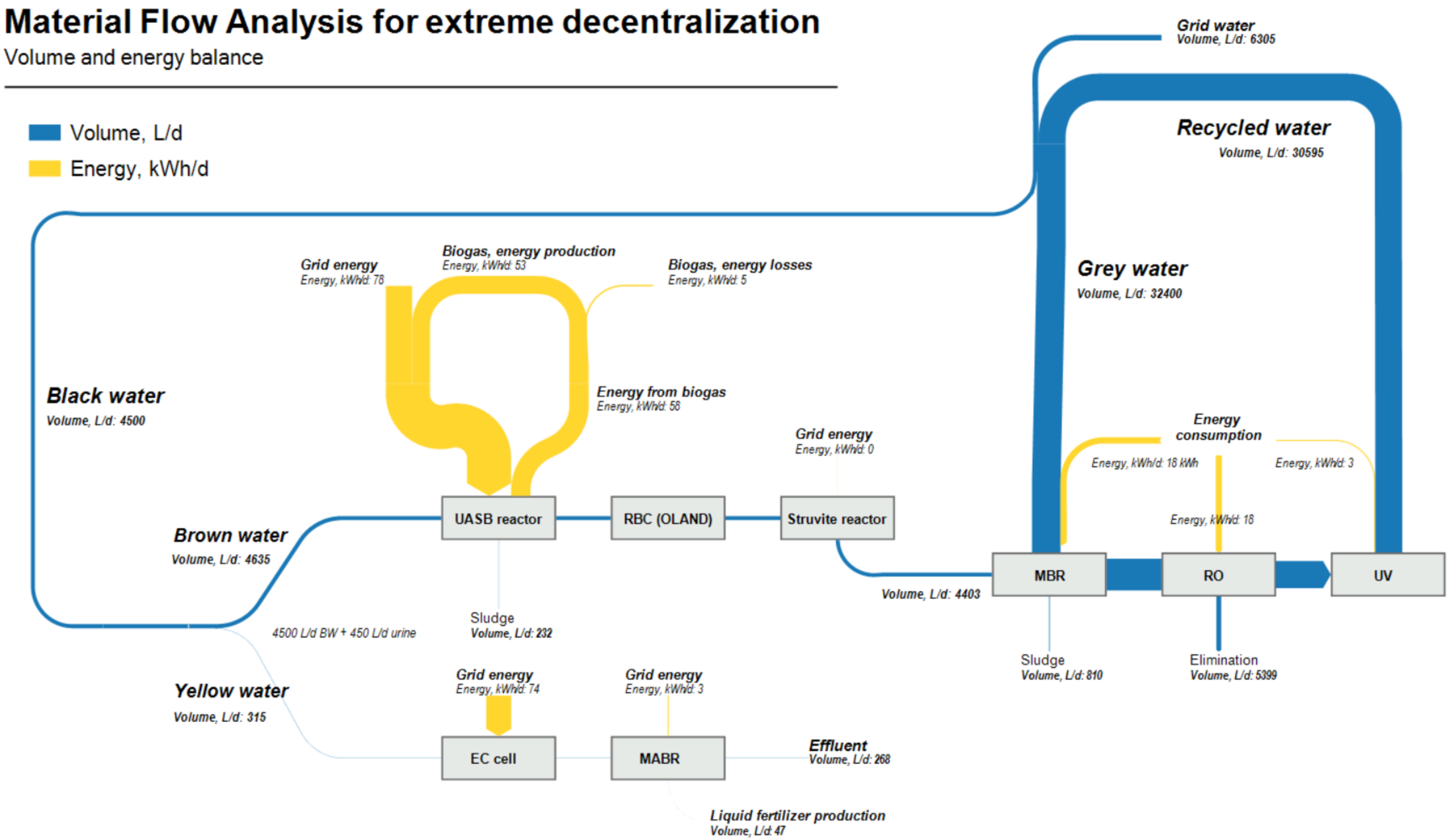

Figure 10: MFA of the decentralized system including water volume in L/d and energy in kWh/d, for the reference population of 300 people.

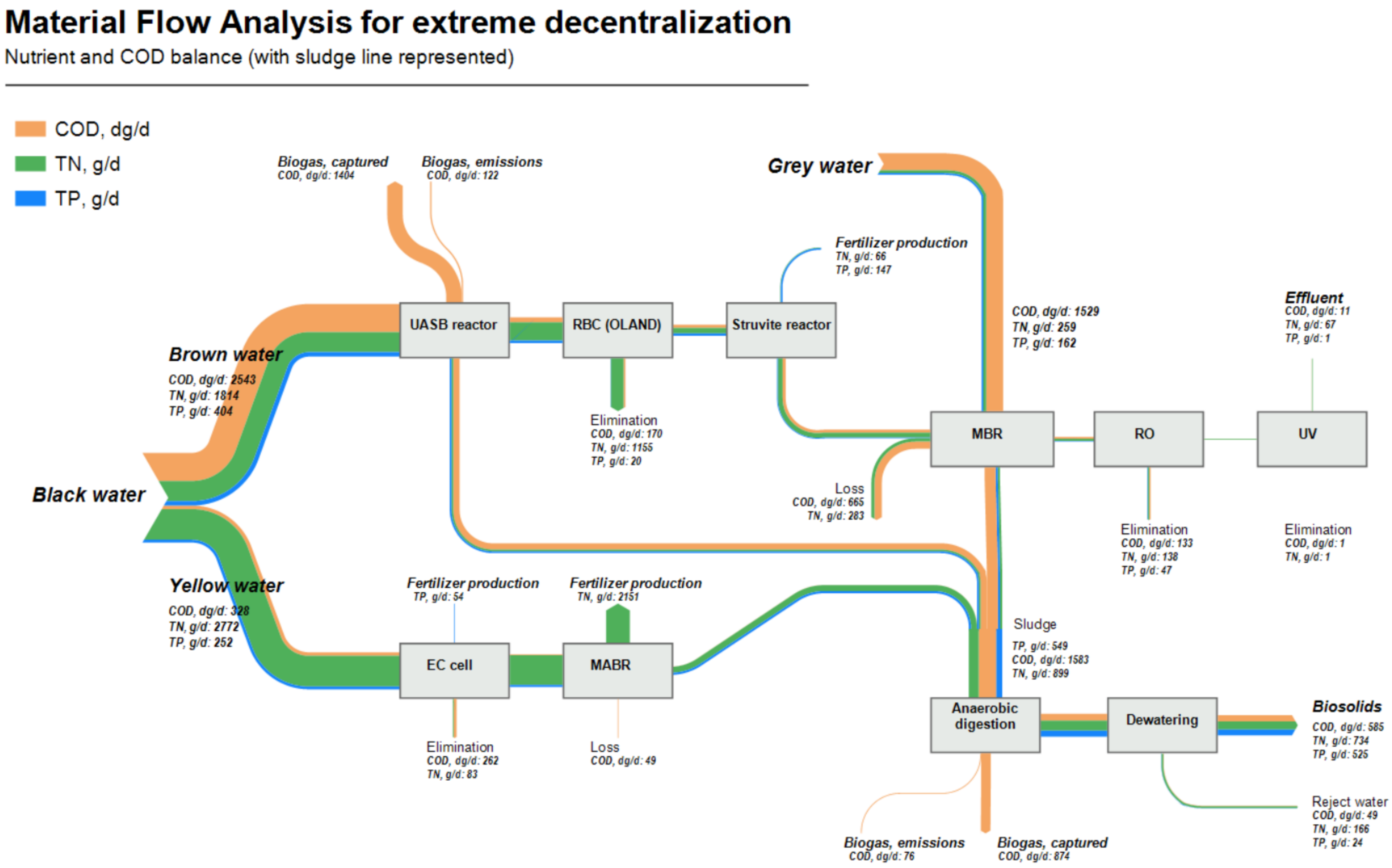

Figure 11: MFA of the decentralized system including mass flows for chemical oxygen demand (COD) in dg/d, total nitrogen (TN) in g/d, and total phosphorus (TP) in g/d also showcasing the sludge treatment line, for the reference population of 300 people.

Material Flow Analysis for conventional treatment

Nutrient and COD balance

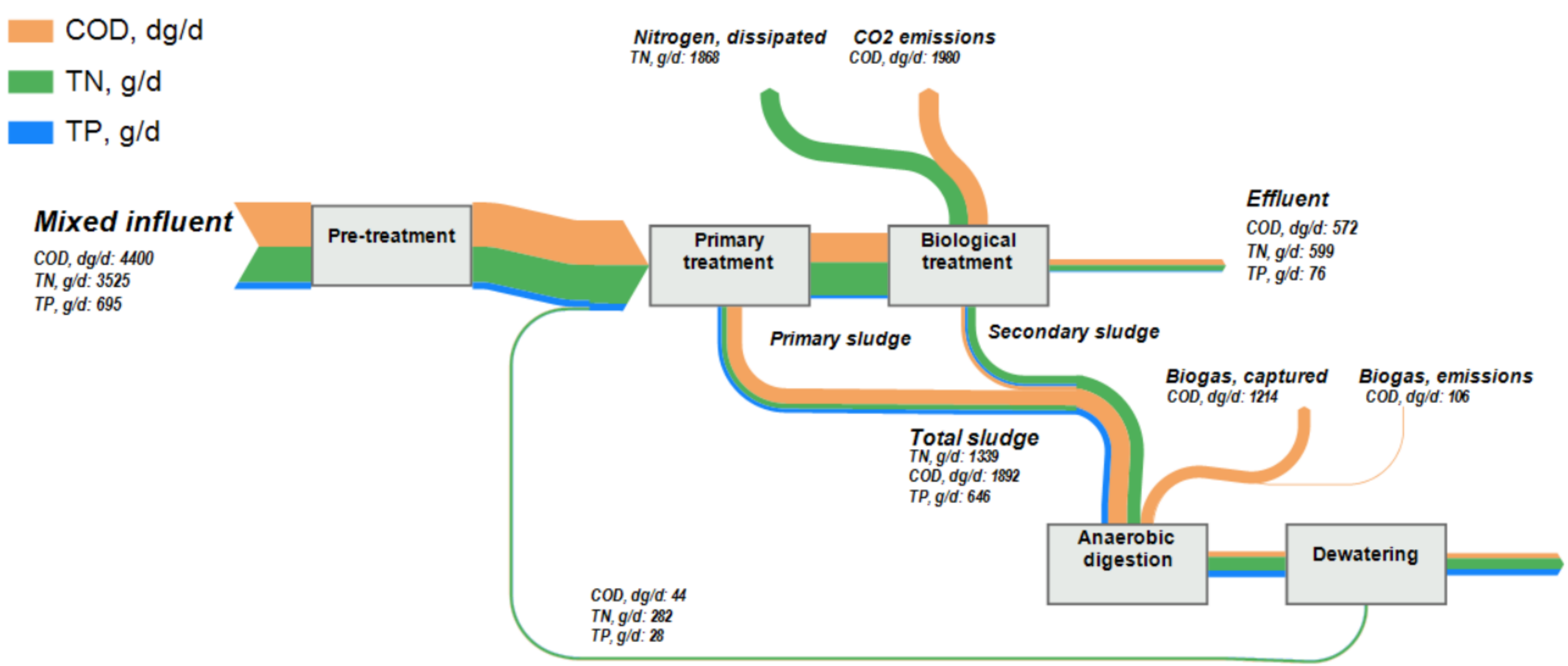

Figure 12: MFA of the conventional system including mass flows for chemical oxygen demand (COD) in dg/d, total nitrogen (TN) in g/d, and total phosphorus (TP) in g/d for the reference population of 300 people.

In order to compare the resource recovery potential offered by the decentralized system with that of conventional wastewater treatment practices, the current state of recovery technology for WWTPs is reviewed.

The simplified MFA of an average conventional WWTP is represented in Figure 12 to showcase the flow of chemical entities within the system. The study focuses on a plant based on the activated sludge process and typical biological nutrient removal, which incorporates anaerobic digestion for the treatment of primary and secondary sludge instead of their more direct disposal. The sludge treatment line produces biosolids which can be valorised through land application. The same figure only for nutrient movement in the treatment line is found in the supplementary material.

As observed in Figure 12, the percentages of recovered nutrients in the product are around 30% for nitrogen and 89% for phosphorus. However, biosolids are not high-value fertilizers or comparable to struvite and urine-diverted fertilizers, which provide high crop yields, good plant uptake, and effectiveness comparable to commercial alternatives. As discussed in Resource recovery, the products obtained though the decentralized treatment also ensure a purer composition and lower levels of metals and other toxic, undesirable components with higher risks associated due to the more specific treatments aimed at source-separated flows. Additionally, it must be noted that, as previously discussed, sludge from the decentralized treatments can also undergo anaerobic digestion and stabilization and produce biosolids increasing the recovery potential of the complete system even further. However, centralized treatment often only relies on sludge processing to achieve some level of recovery. The combination of struvite (or other higher-purity fertilizers) and biosolids is another possible approach to improving the effectiveness of sewage sludge valorisation.

The biogas produced in the centralized treatment for the same influent volume (mixed instead of source-separated) could enable the production of 16,741.5 kWh per year (versus to 31,378 kWh in the decentralized option).

Figure 13 shows a comparison of the resource recovery performance of the decentralized system and conventional treatment, both including sludge processing. Collected TN and TP are classified according to the obtained final product as their value varies significantly. As introduced in Resource recovery, struvite has been proved to be a highly sustainable eco-fertilizer with slow-release of nutrients facilitating uptake from crops and avoiding oversupply. Precipitated struvite has a high concentration of phosphorus, nitrogen, and magnesium, purities of over 85% (versus 4.6% and 2.3% concentrations of nitrogen and phosphorus in biosolids, respectively), and does not require further sanitation or treatment prior to land application as biosolids do (Gianico et al., 2021; González et al., 2021). Urine derived fertilizers, including concentrate and phosphorus precipitates mainly comprised of $Ca_3(PO_4)_2$ are considered adequate for the cultivation of crops (De Paepe et al., 2020a; Goetsch et al., 2020; Hilton et al., 2021; Magwaza, 2020) and also present much purer compositions than biosolids due to their nature as precipitated salts and filtered and concentrated urine.

In Figure 13, graphs for nutrients show how much of TN and TP entering the system is recovered (in mass). As observed, the decentralized system can provide 56% more nitrogen recovery than the conventional system, while also recovering this nutrient as more valuable products. The phosphorus recovery rate is equal between conventional and decentralized treatment, but the decentralized approach allows for the obtention of high-value precipitate fertilizers.

In the graphs regarding energy, the total annual production of both approaches is showcased, taking the decentralized system's energy generation as 100% to showcase its potential for incrementing the conventional treatment's recovery. As observed, the decentralized layout triplicates the energy production of conventional treatment.

Finally, regarding water, consumption from the public network is represented, showing how in the decentralized system water demand is reduced by 83%.

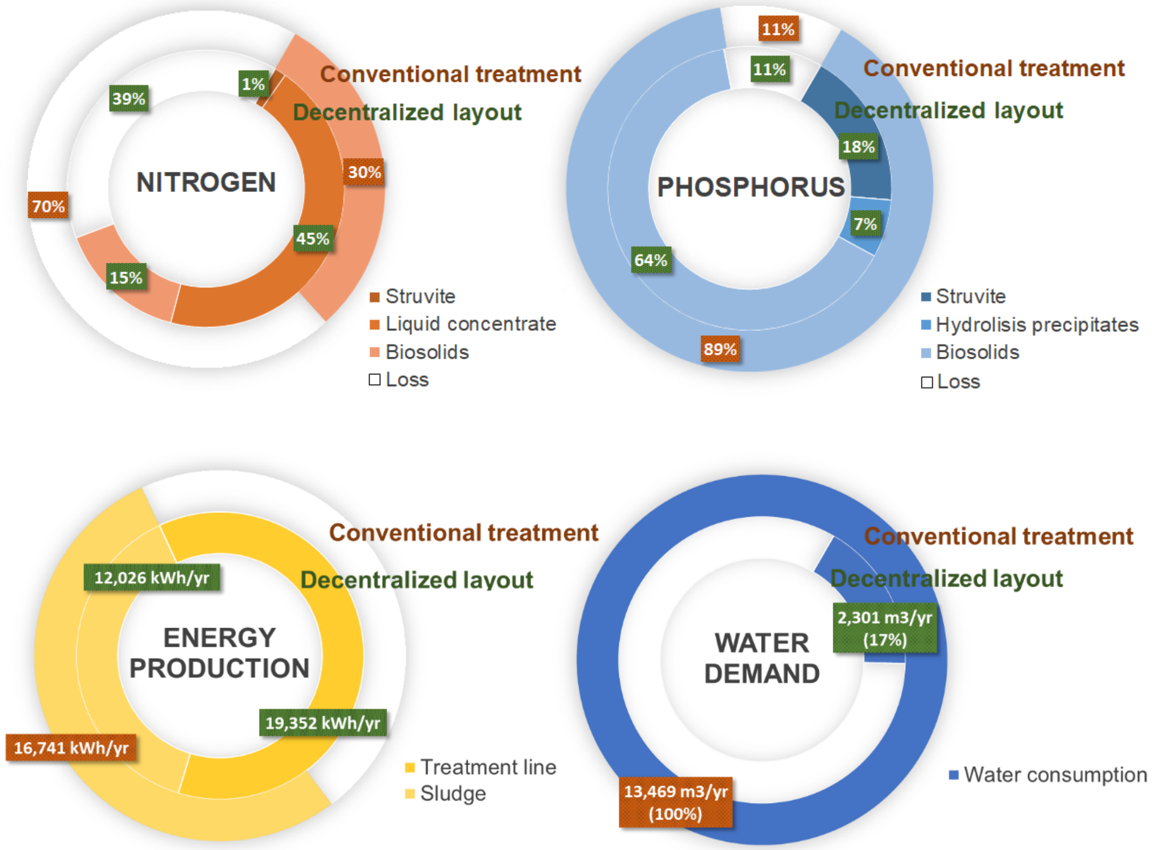

Figure 13: Comparison of the resource recovery potential for the proposed decentralized system and conventional treatment.

3.2 Life cycle assessment

The environmental profile of the decentralized system is compared to that of conventional treatment for a general European setting and four site-specific locations. Total scores for all studied indicators and scenarios can be found in the supplementary material (Tables S21, S22, S23, S24, and S25). Final impacts are broken down to showcase the values for the construction and operation phases separately. Total avoided impacts and values corresponding to water reuse are also included.

The EU scenario evaluates the performance of the decentralized system when only wastewater reuse is considered. The exclusion of RW capture in this scenario potentially hinders the resource recovery capacity of the system, thus the influence of site-specific factors and RW potabilization are studied in the following section covering national scenarios for four cities within Europe. Finally, the adequacy of the data employed for the study and the uncertainty associated with the results is discussed.

3.2.1 *Contribution and comparative analysis for the EU scenario*

For the wastewater reuse case, results indicate that the decentralized system environmentally outperforms conventional treatment in 10/15 of the studied midpoint indicators, including terrestrial acidification potential (TAP), freshwater and marine eutrophication potential (FEP and MEP, respectively), freshwater ecotoxicity potential (FETP), global warming potential (GWP1000), ozone depletion potential (ODPinfinite), fossil fuel potential (FFP), particulate matter formation potential (PMFP), photochemical oxidant formation potential (POFP), and water consumption potential (WCP). It is however not capable to improve the environmental profile of the centralized approach for the remaining three ecotoxicity potential indicators (incl. marine (METP), terrestrial (TETP), and human (HTP) ecotoxicity), and for the ionising radiation potential (IRP), and agricultural land occupation (LOP) midpoints. The scores for all three ReCiPe endpoints are favourable to the decentralization of water treatment.

A breakdown of the impacts between the construction and operational phase of the system to all indicators reveals that the former contributes to a maximum of 10.5% of the total scores (see Table 6). In concordance with Opher and Friedler (2016), the highest influence of the manufacturing and installation of the components is on ecotoxicity. For the proposed system, this is due to the high impact of steel and stainless-steel components, as well as copper production (which is attributed to the manufacturing of motors, batteries, and machinery required for installation) on ecotoxicity indicators. As construction impacts are distributed throughout the lifespan of the technology (Marinoski and Ghisi, 2019), their relevance compared to operation is limited. This is consistent with general indications for LCA in water and wastewater treatment and results reported in existent literature (Corominas et al., 2020; Gómez-Monsalve et al., 2022; Opher and Friedler, 2016).

Table 6: Contributions of the construction and operational phases to total environmental impacts for the EU scenario of the decentralized system.

	Abbreviation	Construction	Operation
Terrestrial acidification	TAP	3,90%	96,10%
Global warming	GWP1000	2,57%	97,43%
Freshwater ecotoxicity	FETP	5,41%	94,59%
Marine ecotoxicity	METP	5,47%	94,53%
Terrestrial ecotoxicity	TETP	10,49%	89,51%
Fossil depletion	FFP	3,85%	96,15%
Freshwater eutrophication	FEP	1,50%	98,50%
Marine eutrophication	MEP	4,73%	95,27%
Human toxicity	HTP	4,83%	95,17%
Ionising radiation	IRP	0,54%	99,46%
Agricultural land occupation	LOP	4,77%	95,23%
Ozone depletion	ODPinfinite	0,63%	99,37%
Particulate matter formation	PMFP	4,49%	95,51%
Photochemical oxidant formation	POFP	3,79%	96,21%
Water consumption	WCP	2,90%	97,10%
Ecosystem quality (endpoint)	-	2,94%	97,06%
Human health (endpoint)	-	6,92%	93,08%
Natural resources (endpoint)	-	9,76%	90,24%

A contribution analysis for all midpoint indicators is presented based on the process and elementary flow contribution data and Sankey diagrams (where avoided impacts are best visualized) obtained from Brightway2's Activity Browser. Total scores for all indicators are broken down for the operation and construction phases to provide insight into the activities causing damage to each midpoint during the lifespan of the plant. Percentages given during the discussion refer to the share of total impacts for either construction, operation, or both phases (i.e., net scores once avoided impacts are discounted). Charts supporting the contribution analysis as well as the impact values for the EU scenario can be found in the supplementary material (process contribution figures section in the supplementary material and table S21).

The **climate change** category reflects the increase in infrared radiative forcing caused by the studied activities and takes special relevance in most LCA studies (Corominas et al., 2020; Huijbregts et al., 2016). It shows a composition of process contributions similar to that for **terrestrial acidification**, which quantifies proton increase in natural soils.

For the decentralized layout, electricity represents 35% and 30% of total impacts for the construction and installation phase and over 74% and 88% for the operational phase, for GWP1000 and TAP, respectively (Figure S9 and S13). The contribution to both midpoints by type of emission is however different, as impacts on global warming are dominated by carbon dioxide (87% during construction and 72% during operation), methane (12% and 23% for construction and operation, respectively), and dinitrogen monoxide (0.84% for construction impacts and 4.4% for the operational phase). While construction emissions causing damage on GWP1000 and TAP are mainly from fossil origin, during operation a total of 13% of CO_2 and 74% of CH_4 impacts are due to non-fossil emissions (Figure S8). Sulfur hexafluoride is the next contributor to GWP1000 during operation. For TAP, sulfur dioxide and nitrogen oxides take up 77% and 21% of construction impacts and 86% and 20% of operational impacts, respectively (Figure S12). During construction, sulfur oxides and nitrate emissions contribute the remaining 3%.

When analysing the results for contributors to impacts further than electricity, activities causing increases to the GWP1000 and TAP indicators differ. For GWP1000, direct emissions from the UASB reactor, the RBC, and MBR contribute 17% of total operational impacts (Figure S8). Nutrient, energy, and water recovery allow a mitigation of 8,890 kg CO_2-Eq*yr^{-1}, with a slightly higher influence from the latter (Table S21). During construction, heat and electricity employed for steel production, and ethylene, hard coal, iron, and clinker obtention are the most relevant activities, contributing 17%, 6.5%, 6%, 4%, and 3% of final impacts.

For TAP, copper production contributes 17% and 14% of total construction and operational impacts, respectively, through direct and indirect emissions. The material's main use is in the electric distribution network, but it also finds applications within the scope of the study including the treatment of solid waste and sewage, and the production of electric motors and equipment needed for installation. An additional 15% of construction impacts derive from heat obtention. Finally, transport of equipment and chemicals and the production of ethylene each add up to 3% of impacts to TAP for both construction and operation (Figure S13). A relevant reduction of operational impacts is achieved by N-based fertilizer and water recovery (15.2 kg and 13.2 kg of avoided SO_2-Eq emissions, respectively), which corresponds to almost 50% of the positive net impacts (Table S21).

Overall, performance is improved by 44% and 52% for the GWP1000 and TAP midpoint categories, respectively, when compared to the conventional approach (Figure 14). For the conventional treatment model, the production of potable water and transport accounts for 23% of total GWP1000 impacts, while wastewater treatment contributes the remaining 77%. The proportion is similar for TAP, where wastewater treatment takes up 73% of impacts (see Figure 18).

The performance improvement can be partly attributed to savings in energy stemming from the elimination of major pumping requirements during water transport. However, the wastewater treatment section in the decentralized system shows higher specific electricity consumption (around 3.5 $kWh*m^{-3}$) than the average conventional process (generally ranging from 0.2 to 2.2 $kWh*m^{-3}$; Sarpong et al., 2020; Walker et al., 2021) mainly due to the inclusion of MBR technology and the UASB reactor treating food waste together with wastewater. Still, the overall benefits of these reactors, including better removal efficiencies yielding higher quality effluents, biogas production, and minimized sludge generation, outweigh the impacts of an increase in operational electricity use and result in an overall enhanced environmental performance (Gao et al., 2021). It must be considered that the wastewater line yields an effluent of drinking water quality, a goal which is not pursued in conventional plants, and starts from a higher organic matter concentration than average. The energy consumption of the decentralized system is in range when compared to centralized forms of reuse, including direct and indirect potable water reuse (DPR and IPR, respectively), ranging from 1.7 to 4.2 $kWh*m^{-3}$. Lower values correspond to IPR treatment based on biological aerated filters and simple DPR, while DPR with advanced drinking water treatment and RO-based IPR show higher consumptions. The energy demand is also lower than averages for alternative water augmentation techniques such as seawater or groundwater desalinisation, which use from 1.9 to 6.7 $kWh*m^{-3}$, with higher consumption required for seawater (Sim and Mauter, 2021; Tow et al., 2021).

Improvement in the GWP1000 and TAP indicators is also linked with reduction of direct GHG emissions (esp. N_2O, as discussed for ODPinfinite, but also CO_2, as the UASB and the MABR minimize emissions), fertilizer recovery, minimization of leakages along the whole treatment system, and water reuse (Santana et al., 2019).

Impacts on **fossil fuel depletion** are mainly linked to coal (66%) and natural gas (28%) use during the operational phase, with a third major contribution from oil (5%). All three are attributed to indirect fossil use from electricity consumption and are therefore highly sensitive to the mix used (Figure S10). With a European average mix, a reduction of only 3% of impacts is achieved with respect to conventional treatment (see Figure 14). Besides electricity, plastic materials cause a remarkable 24% of construction impacts, with the three main contributors being ethylene, polymethyl methacrylate, and propylene, due to the demands of oil and natural gas of their production processes (Figure S11). Yearly, N-based fertilizer recovery and water reuse cut fossil fuel consumption by 2,430 kg oil-Eq, which is around 44% of the plant's operational impacts (Table S21). In the conventional system, impacts to fossil depletion are divided equally between transport and production of drinking water and wastewater treatment (Figure 18).

In concordance with literature, the implementation of decentralized water reuse results in significant improvements to the **aquatic eutrophication indicators**, which show the increase of phosphorus and nitrogen concentration in freshwater and marine water (Kobayashi et al., 2020; Opher and Friedler, 2016; Santana et al., 2019), due to the elimination of nutrient emissions to ground and water bodies originating from pipe leakage and final discharge. The system in the present study achieves reductions of 50%

and almost 100% of impacts for the **freshwater and marine eutrophication** categories, respectively (Figure 14 and Figure 15).

Regarding FEP, the causes of nutrient emissions are phosphate leakages and runoff to surface and ground- water. These can be attributed mainly to the landfilling of spoil from lignite (39% in construction and 20% in operation) and hard coal mining (30% and 69% in construction and operation, respectively). Waste from copper mines takes up the next 21% of construction and 6% of operational impacts (Figure S14). These activities are sources of indirect phosphorus emissions linked with electricity production. The improvement achieved by resource recovery is of around 17% of the total score and is attributed mainly to water reuse, which contributes 71% of avoided impacts (Table S21). For conventional treatment 81% of impacts are linked with wastewater treatment and effluent discharge (Figure 14). While almost the total score for FEP is attributed to phosphates, for MEP, the impacts are shared among nitrate, organic and inorganic nitrogen, ammonium, and additional nitrogen compounds (Figure S15).

As the main contributors to marine eutrophication are leakages to groundwater from sewer networks and, mainly, effluent discharge (Opher and Friedler, 2016), the MEP indicator is the most benefitted from water reuse. In the decentralized system, both sources of emissions to marine water are avoided. The remaining sources of impacts showcase a similar composition than those of FEP, being mainly attributed to spoil treatment, with an additional 14% of operational impacts coming from yellowcake originating from uranium leaching. Besides the elimination of the activities causing the largest share of emissions with marine eutrophication potential, the influence of nitrogen recovery on this indicator is also especially relevant. Avoided emissions are around 73% of the total score and mainly originate from substituted production of ammonium sulphate and NPK (15-15-15) fertilizer (Table S21 and Figure S16).

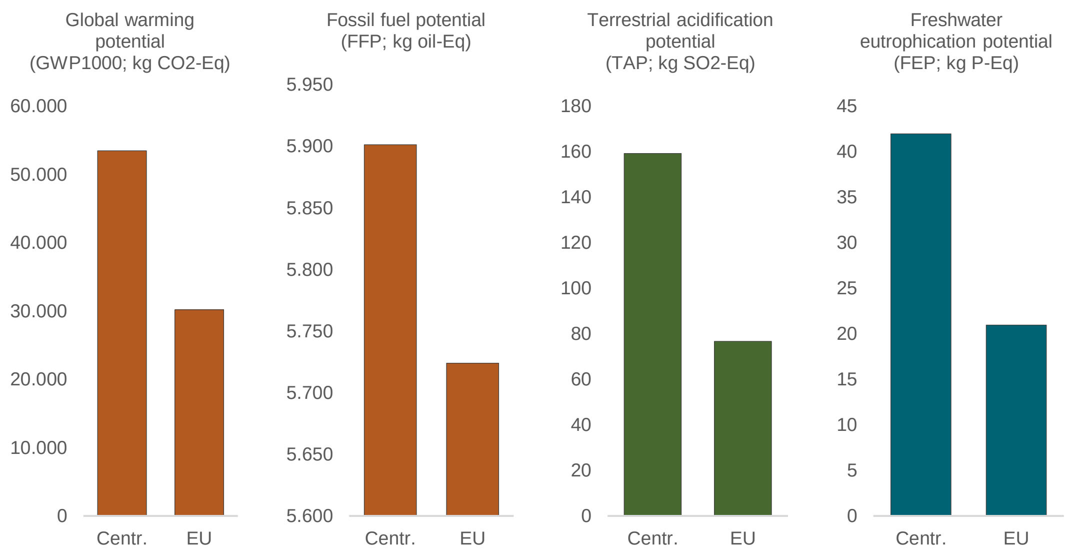

Figure 14: Comparison of total scores for ReCiPe midpoint indicators GWP1000, FFP, TAP, and FEP for conventional centralized treatment (centr.) and the European average scenario of the decentralized system (EU).

The decentralized system outperforms conventional treatment also regarding **water consumption**; however, the impact of water reuse is rather limited for this indicator. Up to 77% of impacts relate to electricity use during operation, and even when the use of 167 m^3*yr^{-1} is avoided thanks to resource recovery, fertilizer production and sludge treatment by anaerobic digestion have a larger impact than water reuse (Table S21). The high contribution of indirect water use by hydroelectric electricity production to the water depletion impact category in decentralized systems is also highlighted by Santana et al. (2019). Regarding construction, 37% of impacts also relate to electricity use, and 12% to the manufacturing of silicone products, which have a water demand of 0.26 kg water*kg^{-1} silicone product (Wernet et al., 2016). The remaining impacts are distributed evenly between the steel and plastic production processes, and the obtention of chemicals including chlorine and oxygen (Figure S17).

The water consumption impacts of centralized treatment are distributed between potabilization/transport and wastewater treatment by 21.5% and 78.5% respectively (Figure 18). An overall 43% reduction of damages is achieved by the decentralized approach (Figure 15).

Land occupation is the one of the impact categories which is deteriorated by the decentralized system's performance (see Figure 15), which is 18% higher than the score of the centralized case. A major part of impacts is linked with forest management activities for electricity production needed for operation (70%), which leaves room for electricity mixes not relying on forestry to score lower than centralized treatment (Figure S18). Avoided impacts mainly linked to phosphorus recovery help the system by shaving the total score by around 27% due to the elimination of the need for phosphate rock mining, an activity of high environmental damage (Table S21). Construction impacts (over 5% of total) are dominated by the installation of the treatments and equipment (30%). For the centralized case, there is a balance where potabilization and transport represent 40% of impacts and the remaining 60% relate to wastewater treatment (Figure 18).

Regarding **ecotoxicity** indicators, the decentralized system is only able to achieve a decrease of 26% of impacts for freshwater ecotoxicity, while marine, terrestrial, and human ecotoxicities are increased by 69%, 121%, and 71%, respectively (Figure 15). Ecotoxicity indicators refer to hazard increase in terrestrial ecosystems, freshwater bodies, and marine water, while human toxicity potential represents the increase of the population's risk to develop cancer and non-cancer disease (Huijbregts et al., 2016).

The household-scale rainwater-greywater recycling system by Marinoski and Ghisi (2019) and the source-separating reuse system for GW and BW by Besson et al. (2021) also failed to reduce the conventional route's impacts for the ecotoxicity categories, including METP, HTP, and FETP. However, even if impact composition for these categories is similar to that obtained by Opher and Friedler (2016), their GW reuse system could achieve better results than the business-as-usual (BAU) route. Their layout, however, did not include BW treatment (which contains the largest pollutant loads). Besides, it treated GW to non-potable reuse quality with a combination of an RBC and a filtration and disinfection stage, therefore avoiding the more energy intensive processes included in this study (i.e., UASB reactor and MBR). Their contribution analysis showed that impacts for the toxicity categories were mainly derived from electricity consumption, especially for the HTP category, but also for both aquatic indicators. TETP was not included in their analysis (Opher and Friedler, 2016).

In this study, the potabilization and transport phase of the conventional route take the most relevance versus wastewater treatment for ecotoxicity out of all indicators (Figure 18) and show similar contributions to impacts than the decentralized system but with lower absolute values. In fact, for all three categories where decentralization performs worse than centralization (METP, TETP, and HTP), the contribution of wastewater treatment to the total impacts of the conventional scenario is especially low (see Figure

18). Besides, impact reductions by resource recovery are rather limited for the ecotoxicity categories, which gives no significant advantage to the decentralized system. Finally, infrastructure takes a more relevant role in the sum of all impacts, especially for **terrestrial ecotoxicity**, which is where decentralization offers the worst performance. For this category, ferronickel, ferrochromium, and copper use to produce steel and stainless-steel components, motors, pumps, and electricity, take up 30%, 26% and 30% of construction and installation impacts (Figure S24). Transport of components contributes an additional 3%. This construction and installation phase accounts for more than 10% of total impacts (Table 6), which are added to the operational impacts, mostly attributed to toxic elements emitted during electricity production (Figure S23), including copper (70% of operation impacts), nickel (8%), zinc (4%), silver (4%), lead (3%), and arsenic (2%). Avoided emissions by resource recovery are around 44,200 kg 1.4-DCB-Eq, cutting the final net score by 24% (Table S21).

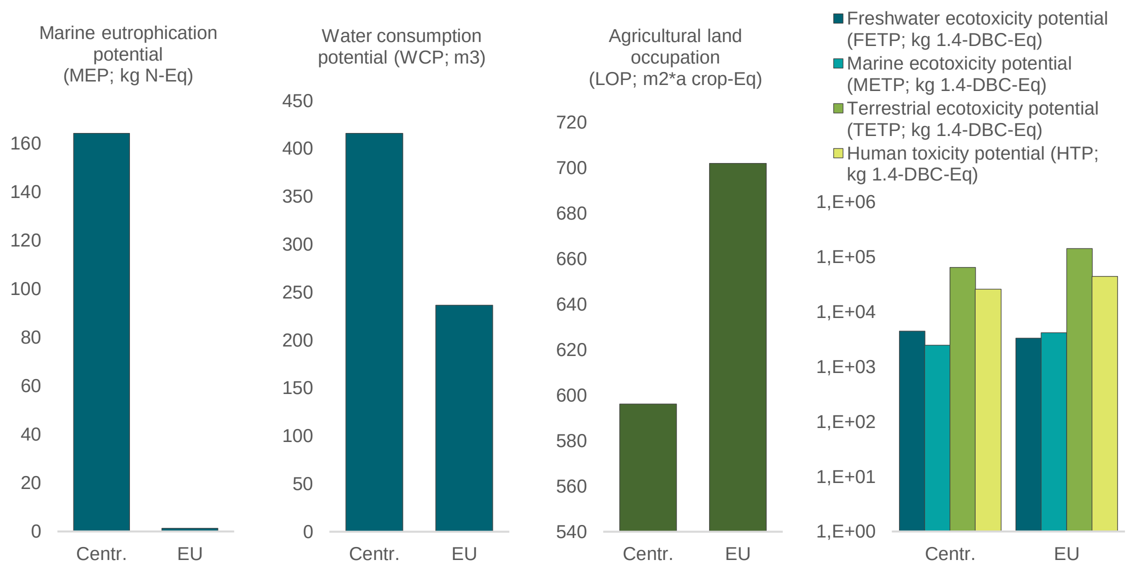

Figure 15: Comparison of total scores for ReCiPe midpoint indicators MEP, WCP, LOP, FETP, METP, TETP, and HTP for conventional centralized treatment (centr.) and the European average scenario of the decentralized system (EU). Note the logarithmic scale for the axis of the ecotoxicity indicators graph.

Process contributions for the **aquatic ecotoxicity** categories (incl. **freshwater** and **marine water**) are almost identical, with the main contributions from heavy metal emissions derived from waste treatment and disposal and electricity production. Copper takes up around 77% of operational impacts for both FETP and METP, followed by the landfilling of spoil from lignite and coal mining, which are responsible for around 11% and 4% of operational impacts for both categories (Figures S19, S20, S21, S22). The construction phase is also dominated by copper emissions (80%) and shows more limited contributions from furnace slag (6%), and nickel (5%). Absolute values for impacts on METP and FETP in 1.4-DBC-Eq are in the same order of magnitude, with avoided impacts of around 16% of these scores (Table S21). However, the centralized approach's impacts are one order of magnitude lower for METP than for FETP due to the contribution of wastewater treatment (Figure 18; Risch et al., 2014).

Finally, **human toxicity** potential is the category which is the most dependant on electricity, in concordance with Opher and Friedler (2016). The presented value is a sum of carcinogenic (HTPc) and non-carcinogenic impacts (HTPnc), which were separated in ReCiPe 2016 (Huijbregts et al., 2016). For the construction phase, copper, steel, and waste landfilling continue to take up all impacts for both the carcinogenic and the non-carcinogenic indicators. Water reuse allows to reduce overall HTP impacts by 24% (Table S21). The main contributors to positive operational impacts which add to total score for HTP are landfilling of spoil from lignite mining (30% of overall score for HTPc and HTPnc), treatment of sulfidic tailings from copper mining (22%), and treatment of spoil from coal mining (10%). All impacts related to mining are attributed to electricity production (Figures S25, S26, S27, S28).

Ionising radiation potential referring to the absorption of radiation from radionuclide emissions is the 5th and last indicator for which decentralization is not favourable. It has the lowest contribution of the construction phase, of below 1% (Table 6). All impacts are related to activities required for electricity production through nuclear processes (Figure S29), including tailing treatment (93%), uranium mining (2%), and treatment of spent nuclear fuel (2%). The scores for this indicator are thus highly sensitive to the electricity mix used. When studied for a European mix, which has around 25% of electricity generated by nuclear power (Eurostat, 2022), IRP is deteriorated by 70% for the decentralized system with respect to conventional treatment (Figure 16) and resource recovery is unable to reduce this difference significantly, being limited to around 14% of the total net score (Table S21).

The impacts to the **ozone depletion** potential category assessing decrease in stratospheric ozone are dominated by the operational phase and derive mainly from direct nitrous oxide emissions originating at the MBR, MABR, and RBC (Figure S31). Thus, the nitrification and denitrification processes for nitrogen treatment in each of these reactors emit approximately 56%, 2% and 42% of total N_2O, respectively. The best performance is achieved by the MABR as it is designed to minimize nitrous oxide emissions (Kinh et al., 2017). The overall emission factor of the plant is that of 0.26% of influent TN, which is well below the average of 1.1%-1.6% associated with conventional plants (de Haas and Andrews, 2022; Zhongming et al., 2019). The next major contributor to the ODPinfinite midpoint after N_2O is electricity use (Figure S30).

Resource recovery, and mainly the substitution of the production of phosphorus fertilizer which is a large source of ozone-depleting substances (ODSs), allows to cut 0,011 kg CFC-11-Eq*yr^{-1}, when the total net score is that of 0,05 011 kg CFC-11-Eq*yr^{-1} (Table S21). This reduction combined with the lower N_2O emission factor enable the decentralized system to narrow down the impacts of the BAU scenario by 71% (see Figure 16).

Infrastructure is responsible for 4.5% of **particulate matter formation**, calculated as particulate matter intake by population, with the main contributors to construction impacts being electricity production for the manufacturing and installation of equipment (28%), ferrochromium for steel tanks and components (22%), and heat use (8%). Regarding operation, 86% of the impacts refer to electricity consumption, while magnesium dosing for struvite precipitation represents another 4%. Annually, avoided emissions add up to 12.9 kg PM2.5-Eq thanks to resource recovery (Table S21). The overall performance of the decentralized system is 26% better than that of BAU practices (Figure 16).

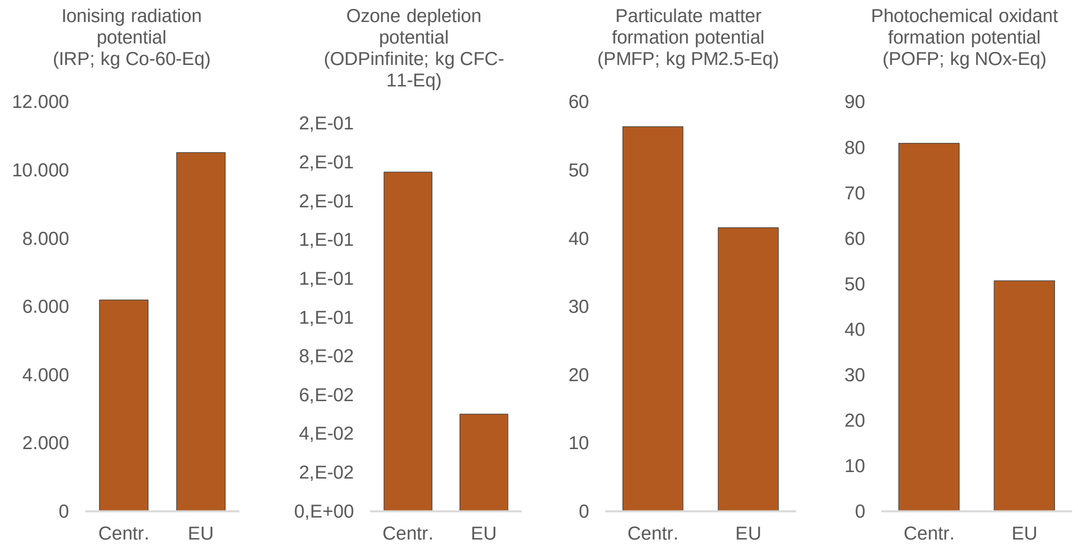

Figure 16: Comparison of total scores for ReCiPe midpoint indicators IRP, ODPinfinite, PMFP, and POFP for conventional centralized treatment (centr.) and the European average scenario of the decentralized system (EU).

The **photochemical oxidant formation** score is an average between the impact on this category on human health (photochemical oxidant formation potential: human health; HOFP) and on terrestrial ecosystems (photochemical oxidant formation potential: human health; EOFP). The former calculates the increase in tropospheric ozone concentration, while the latter refers to the increase which is potentially intaken by population (Huijbregts et al., 2016). Damage of ozone formation to ecosystems was added in ReCiPe 2016 to complement the previous considerations regarding its effect on human health. For this study, values obtained for EOFP are higher than those obtained for HOFP and thus an average is used. Results for HOFP indicate the decentralized system emits a total of 48.8 kg NOx-Eq yearly, while for EOFP, the total is that of 52.6 kg NOx-Eq. Both results are below the value for the conventional treatment of 80.92 kg NOx-Eq*yr^{-1}. If the average is used, a decrease of 37% of emissions is achieved (Figure 16) and 17.3 NOx-Eq*yr^{-1} are avoided through resource recovery (Table S21).

Emissions of NOx and non-methane volatile organic compounds (NMVOCs) causing ozone formation happen mainly during operation in the decentralized plant, with electricity being the main contributor, responsible for around 67% of operational impacts, followed by emissions from the MBR (10%), transport (7%), and magnesium production for dosing in the struvite reactor (4%). Construction impacts are much lower and attributed to electricity production (29%), diesel combustion in building machinery (6.5%) the manufacturing of plastics (5%), and the obtention of natural gas (3%) (Figures S34, S35, S3, and S37).

While the analysis of midpoint indicators allows for a transparent and effective assessment of the origin and composition of impacts, communication of results to a general public is generally facilitated by the use of endpoint indicators. ReCiPe (H) v1.03 employs three higher impact aggregation levels which showcase damages to ecosystem quality, human health, and natural resources. These integrate the impacts on the midpoints relevant to each area of protection and provide a comprehensive and compact result of the LCA. However, endpoint indicators weight the relevance of each damage according to additional assumptions, thus incorporating higher levels of uncertainty and judgment. As the results of the present study are suitable for use in the aid of policymaking and for the comparison of different technological alternatives for water and wastewater management, the midpoint analysis is complemented with the presentation of results in a summarised manner using the three ReCiPe endpoints.

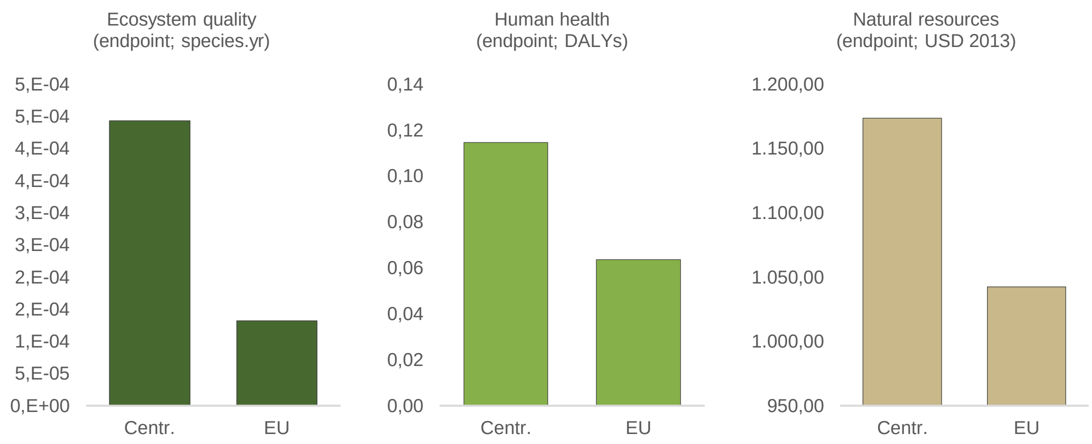

Figure 17: Comparison of total scores for ReCiPe endpoint indicators for conventional centralized treatment (centr.) and the European average scenario of the decentralized system (EU).

As observed in Figure 17, the use of endpoint indicators is favourable to decentralization, as impacts on ecosystem quality, human health, and natural resource use are cut by 70%, 44%, and 11% with respect to conventional treatment. The indicators contributing to each score are listed in Figure 7. The best performance is achieved for ecosystem quality, which is also the endpoint that encompasses more midpoint categories, followed by human health. Besides, the positive results for natural resource depletion provide information about the non-quantified ReCiPe midpoint (surplus ore potential), which is mainly impacted by the mining of rare earth oxides in the decentralized system. Still, as observed in (Figure S40) and considering the importance of the operational phase presented in Table 6, the use of fossil resources has a higher weight than the metal depletion potential on the natural resources indicator.

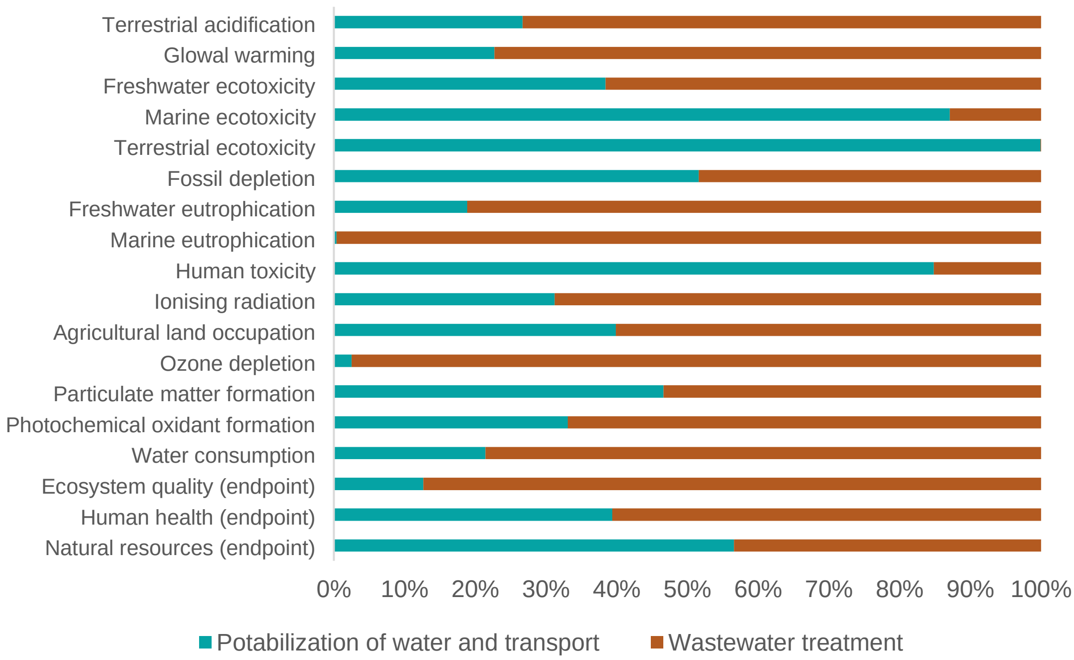

Figure 18: Distribution of impacts for the centralized system in the EU scenario.

3.2.2 Site-specific scenarios

As seen, the results of the contribution analysis highlight the dependence of the decentralized system on electricity, which positions the energy sector as a relevant source of impacts beyond the water management scene. The electric mix is in fact known as a main influencer of the damage caused on the environment by water treatment facilities as a whole (Jeong et al., 2018; Kobayashi et al., 2020). Site-specific scenarios for four cities (incl. Barcelona, Trondheim, Vienna, and Bucharest) are modelled to assess the variations in the results arising from a change in the sources of electricity powering the decentralized layout. All other activities in the inventory which can be adapted to rely on national datasets are modified, and RW capture according to each country's rainfall patterns is included.

Results are presented for all midpoint and endpoint indicators, taking the EU scenario as a reference, and a summary of all variations with respect to the EU case is provided in Table 7. Tables S22, S23, S24, and S25 in the supplementary material showcase the total damages and avoided impacts for all indicators for the site-specific scenarios.

The inclusion of site-specific considerations regarding water transport distances and water treatment techniques in the modelling of the conventional treatment scenario does not modify the impact scores significantly (see section 4.1 and table 20 of SM). Still, electricity consumption has a large effect also on conventional treatment and thus, the impacts for the four country-specific scenarios are not directly compared to the model for conventional treatment based on an average European electricity mix. Figures presented in this section include a line representing the impact of conventional treatment in the average European setting; however, this is only for reference and to help visualize the potential change that can be achieved as a result of an electricity mix change.

As analysed in section 3.2.1, impacts on GWP1000 are dominated by indirect CO_2, CH_4 and N_2O emissions from the use of electricity from fossil origin (73% of total impacts, including both the construction and operation of the decentralized system). The contribution assessment showcases that electricity obtained from lignite, hard coal, and natural gas is the most carbon intensive and therefore causes the highest damages to the climate change indicator. Thus, the relevant weight of electricity from lignite in the RO mix (Figure 8) causes an increase in impacts, while the rest of scenarios achieve moderate to high improvements. NO leads the classification as it cancels almost the totality of electricity-bound impacts, with a global improvement of 71% (Table 7). For the NO scenario, direct emissions from the treatment trains take dominance over impacts. In the ES scenario, the main impacts on GWP1000 are caused by electricity production from natural gas (which is consistent with the mix of the country, see Figure 8), while for AT, damages are mainly due to imported electricity from Germany and the Czech Republic.

Coal burning (incl. lignite, and hard coal) is also a principal source of sulfur dioxide and nitrogen oxides, which contribute to the acidification of soils through nutrient deposition. The TAP indicator is affected by these emissions and thus the changes in impacts are directly related to the share of fossil fuel energy in each country's mix. While for TAP, differences from the changes observed for GWP1000 exist, NO continues to perform better than any other scenario, and AT and ES also achieve impact reductions with respect to the EU average. The RO scenario sees a deterioration of over 50%, which is majorly above the 3% increase for GWP1000, as the importance of electricity on the former is higher. Thus, RO's high share of electricity from lignite has more weight on TAP than on GWP1000, as climate change is affected by a wider range of activities, an important share of which corresponds to direct emissions occurring independently of the used electric mix. For TAP, AT is beneficiated by a higher reduction in impacts than ES due to N-based fertilizer recovery (Figure 19). This allows the scenario to score better than ES, something that is not possible for GWP1000, as resource recovery has a higher weight on TAP than on climate change (Tables S22 and S24 and Figure S41). The same situation is encountered for the FFP indicator, for which impacts are mostly due to imported electricity (Figure 19). Besides, the ES mix relies on natural gas, which has a higher fossil fuel potential than the rest of fossil resources (Huijbregts et al., 2017).

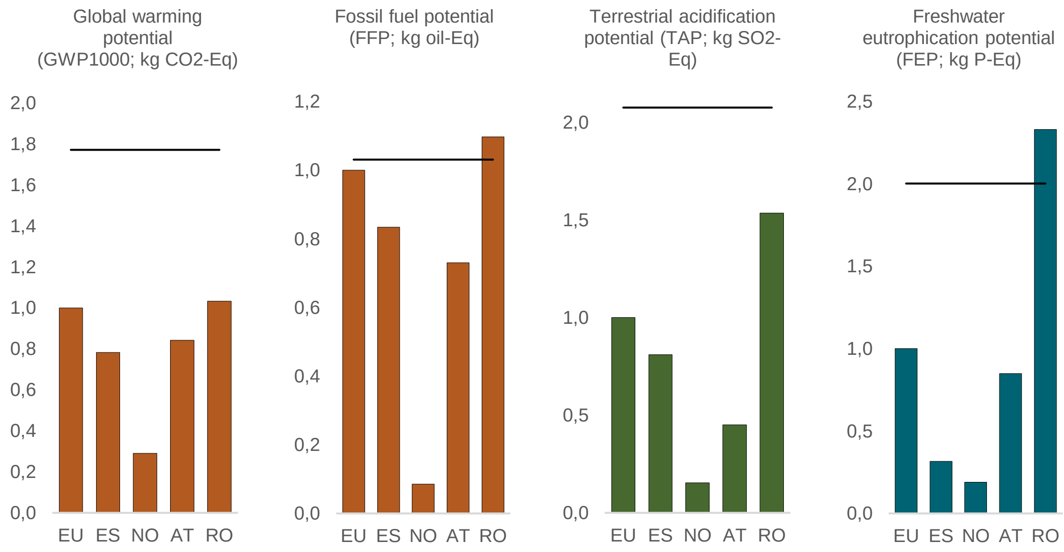

Figure 19: Comparison of total scores for ReCiPe midpoint indicators GWP1000, FFP, TAP, and FEP for the four site-specific scenarios taking the EU case as a reference. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

As observed in the contribution analysis, the disposal of waste from lignite and hard coal mining are the main causes of nutrient emissions to freshwater, which strongly impair the performance of the RO scenario and cause 90% of damages to the FEP indicator (Figure 8). Impacts on FEP for Trondheim are minimal due to the renewable nature of Norway's electricity mix and are even further reduced by water reuse, which cuts down the final score by another 40%. For marine waters, the combination of barely any nutrient emissions linked with electricity production and the higher RW capture shaving the already minimal impact value achieve a negative score on MEP for the NO scenario (Figure 20). Avoided emissions due to substituted nitric acid and ammonium nitrate production take a relevant role, causing liquid fertilizer from urine and struvite to be the largest contributors to avoided impacts on marine eutrophication not only in Norway but in all studied countries (Tables S22, S23, S24, and S25). For MEP, the score of conventional treatment is greatly over that for all scenarios and is thus not represented to maintain the scale of the graph.

Despite the highly favourable performance of the NO scenario, ReCiPe's methodology causes the high share of hydroelectric power in Norway's electric mix to result in an impact increase of over 530% with respect to the EU scenario (Figure 8 and Figure 20). Pressurized nuclear power reactors also require large volumes of water, which translates into higher WCP. The ES and RO mixes have the largest shares of nuclear power; however, the former is the best performing scenario in terms of WCP as the light-water moderated reactors in Spain's mix consume 0.076 m^3*kWh, and the share of hydroelectric power is the lowest out of all scenarios. For RO, the production of electricity using heavy-water reactors has a consumption of 0.37 m^3*kWh (Wernet et al., 2016).

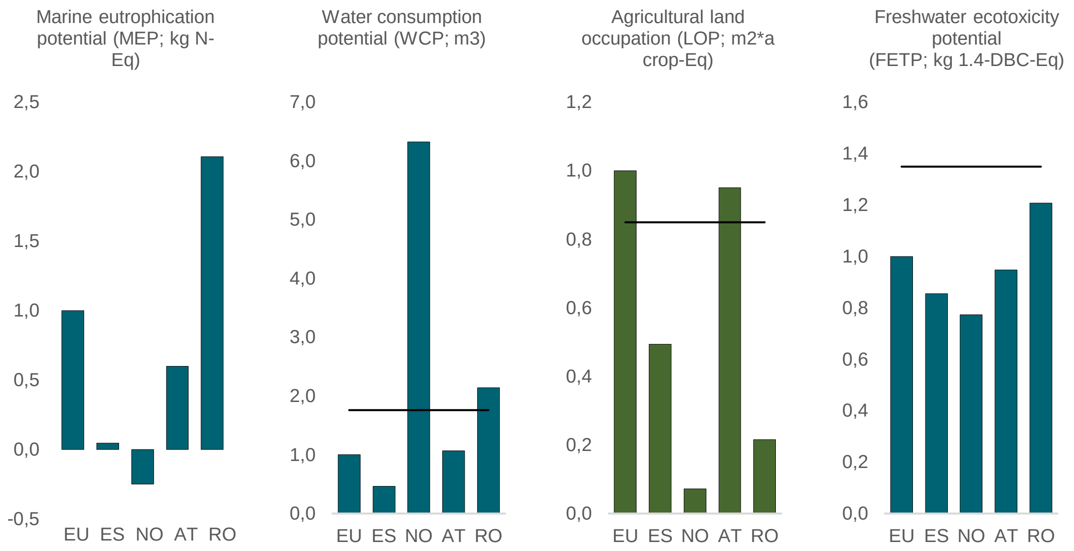

Figure 20: Comparison of total scores for ReCiPe midpoint indicators MEP, WCP, LOP, FETP for the four site-specific scenarios taking the EU case as a reference. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

Similarly to the case of hydroelectric power, even if it yields substantial environmental benefits, the production of energy from biomass causes the impacts on the use of land quantified through the ReCiPe methodology to be incremented. The use of heat and power co-generation in Austria results in impacts to the LOP category to be significantly higher than those of any other scenario (Figure 20).

Variations between scenarios are limited for all three ecotoxicity indicators (Figure 20 and Figure 21). Even if a large set of emissions of toxic elements are indirect and originate from electricity consumption (Contribution and comparative analysis for the EU scenario), they are linked with the use of copper in distribution networks, making them independent of the source of electricity being transported. The changes observed for freshwater and marine ecotoxicity are almost identical (Table 7), as the composition of impacts on FETP and METP also are (Contribution and comparative analysis for the EU scenario section). For terrestrial ecotoxicity, the ES scenario is relevantly deviating from the pattern observed for FETP and METP, showing impacts higher than AT. The contributions of both electricity distribution and transformation are observed to be higher for Barcelona, and the potential of impact reduction by resource recovery for this scenario is lower than for Vienna due to inferior weight of avoided phosphorus fertilizer production.

In human toxicity, the construction of the water distribution network has a principal contribution to total carcinogenic impacts, and thus the effect of water reuse is of special importance. The point to which resource recovery can achieve a reduction of impacts is directly related to the amount of rainfall of each country. Even if carcinogenic impacts are countered by the elimination of water transport pipeline, electricity consumption increases the scores for the non-carcinogenic category. Treatment of waste from lignite mining plays a main role in the emission of toxic elements, causing the RO scenario to lead the classification for the most unfavourable case also for HTP (Figure 21). As in

GWP1000, imports of electricity from lignite cause the AT scenario to score higher than the ES case.

As the decentralized system itself is not responsible for the emission of radioactive elements, the performance on the IRP indicator is strongly tied to the level of nuclear power in each country. This positions the ES and RO scenarios as the principal uranium users (Figure 8 and Figure 21). Treatment of tailing from uranium milling is the principal source of emissions for all cases, and in the RO case the managing of spent nuclear fuel also takes relevance. Water reuse helps palliate these impacts, and for the cases where barely any nuclear power is used (i.e., NO and AT), the effect of resource recovery reduces the impacts to only 17%, and 60% of what they would be if no water was recycled (Tables S23 and S24).

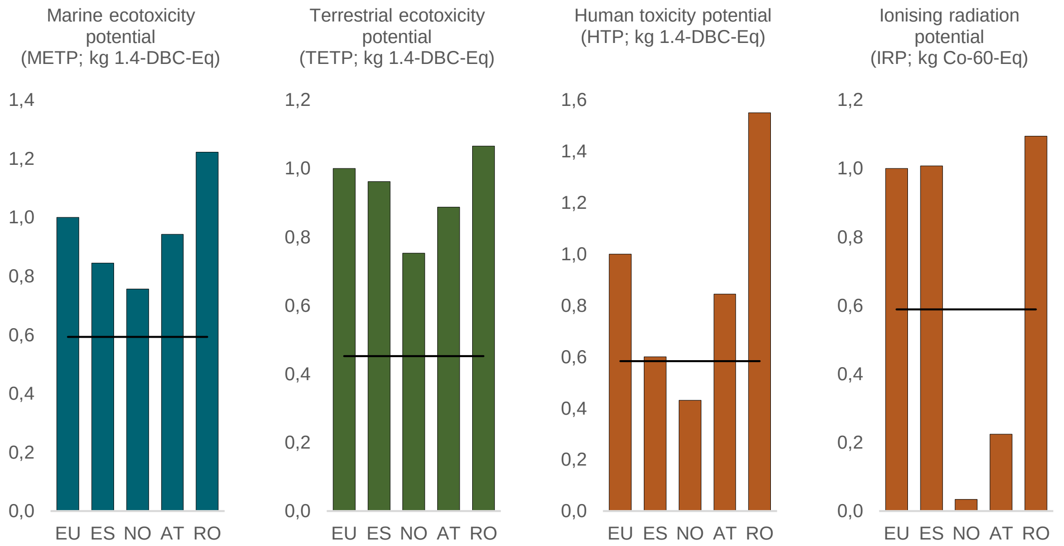

Figure 21: Comparison of total scores for ReCiPe midpoint indicators METP, TETP, HTP, and IRP for the four site-specific scenarios taking the EU case as a reference. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

For ozone depletion, over 85% of impacts correspond to direct emissions from the treatment trains (Contribution and comparative analysis for the EU scenario section and Figure S31), and thus variation between countries is low (Table 7). For this indicator, NO achieves the best performance, followed by RO. Scores for ES and AT are similar to that of the European average scenario. Electricity generation from natural gas appears to be the main cause of differences between scenarios, as impacts from other relevant factors (incl. sodium hydroxide production, resource recovery, and water pump operation) are similar for all countries, which is in concordance with references (Atilgan and Azapagic, 2016). Thus, the higher share of natural gas (Figure 8) for ES and AT gives RO an environmental advantage in this category (Figure 22).

Electricity contributes to over 80% of the formation of particulate matter, with lignite having the highest impact once again (Figure 22). Water reuse is favourable for NO and AT and gives ES a disadvantage in terms of avoided impacts through resource recovery. For the photochemical oxidant formation potential, around 35% of emissions occur at the MBR, during the manufacturing of chemicals required for plant operation, and transport of components (Contribution and comparative analysis for the EU scenario), which are all independent from electricity use. The remaining 65% is highly dependent on the amount of electricity from fossil fuels used in the region's mix, especially hard coal and lignite, which makes the ES and RO scenarios to be the most unsustainable.

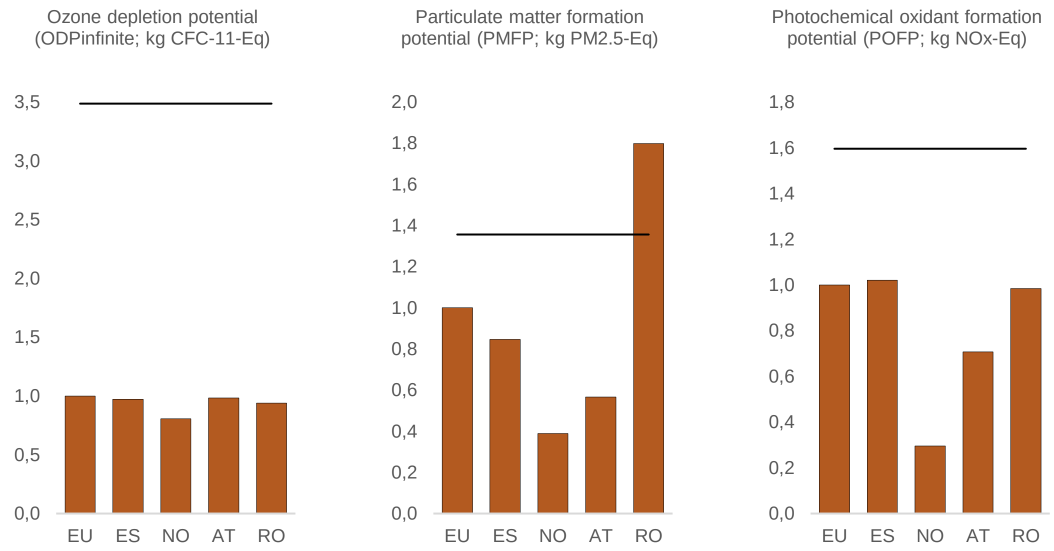

Figure 22: Comparison of total scores for ReCiPe midpoint indicators ODPinfinite, PMFP, and POFP for the four site-specific scenarios taking the EU case as a reference. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

While for most indicators, the switch of country does not affect the environmental advantage or disadvantage in net score of the decentralized system with respect to conventional treatment, some changes do occur. On the one hand, for FFP, FEP, WCP, and PMFP, the RO case's impact surpasses that of centralized treatment. Besides, Norway's high use of hydroelectric power causes impacts on the WCP indicator to also surpass those of conventional treatment in an average European setting. On the other hand, the implementation of higher fractions of electricity from renewable energies, lower shares of nuclear energy, and lower contributions of biomass also help achieve better performances for HTP, IRP, and LOP, respectively, which position the decentralized system's impacts below those of the centralized case. However, comparison should be made against centralized treatment models that consider that conventional treatment is also powered by the same regional mixes in every case.

Still, the high dependence of the water reuse system on electricity indicates that its performance could be significantly enhanced by its coupling with, for instance, solar photovoltaic energy. Besides, and due to their high dependence on electricity, the implementation of reuse systems will unavoidably influence the urban energy consumption patterns of the future. Thus, understanding of the complex link between water and energy is a key step when designing sustainable water treatment schemes, a process that calls for a holistic approach if low global environmental impacts are desired. (Hamiche et al., 2016; Khalkhali et al., 2021; Vakilifard et al., 2018).

The changes in midpoint scores cause an impact also on the endpoint categories, where ecosystem quality and human health show a similar variation pattern between countries (Table 7) with the exception of the RO scenario, that is only 19% over the EU score for ecosystem quality but 50% higher for human health. It is observed that the HTP, PMFP, and WCP midpoints take up over 98% of impacts on the human health endpoint for RO due to their higher values already as midpoints and the weight attributed to them by midpoint to endpoint characterization factors.

The score for natural resources is solely dependent on the use of fossil and mineral resources, and the endpoint on which infrastructure has the highest influence. Construction impacts come from the mining of rare earth elements for the production of motors for mixing, reactor operation, and grinding, and showcase no relevant variations between scenarios. Operational impacts are however dominated by petroleum and natural gas production, two fossil resources for which endpoint characterization factors are the highest (Huijbregts et al., 2017). This causes the ES case to particularly underperform when compared to the rest of scenarios. Meanwhile, Trondheim's case is able to compensate the totality of impacts with resource recovery.

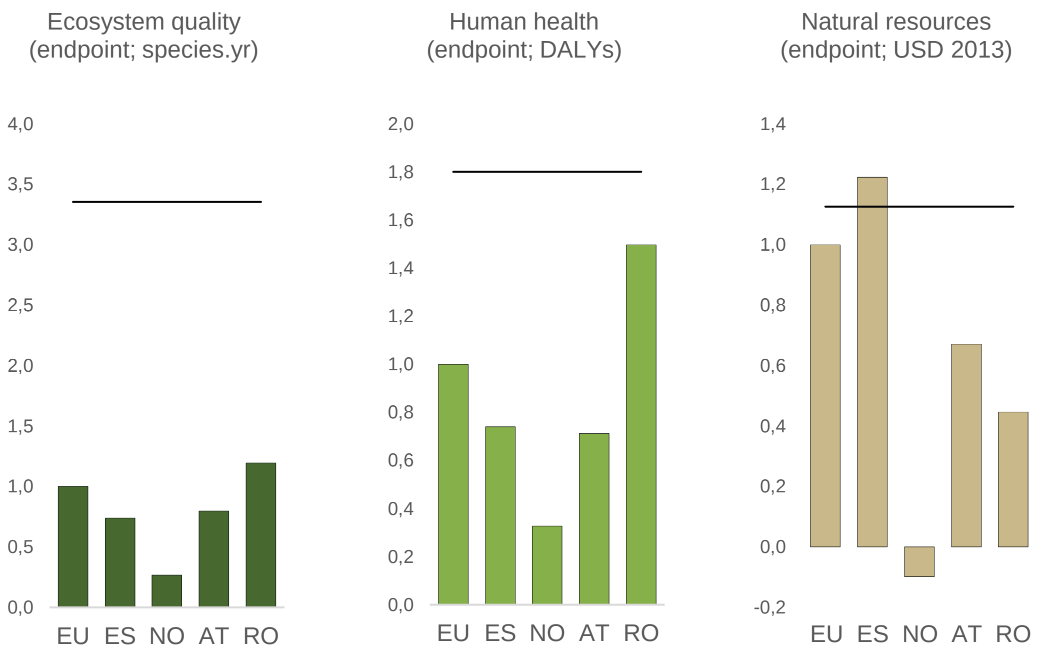

Figure 23: Comparison of total scores for ReCiPe endpoint indicators for the four site-specific scenarios taking the EU case as a reference. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

Water reuse including RW capture allows for an average of 43% of avoided impacts, with the rest corresponding to fertilizer and energy recovery. The highest shares of avoided impacts linked to water reuse are observed for the FEP, IRP, and HTP midpoints, while for ODPinfinite the effect of water use is minimal (Tables S22, S23, S24, and S25). Figure S41 shows a summarized view of the influence of resource recovery on the scores for each studied ReCiPe indicator (incl. midpoints and endpoints).

When focusing on drinking water reuse, and as observed in (Table S26), the implementation of RW capture is favourable for the decentralized layout as impacts caused by the installation and operation of RW treatment equipment are compensated by the benefits of potable reuse. As expected, the NO scenario shows the best results, correspondingly to the amount of rainfall per year of the country. However, in some categories, the avoided impacts are unable to counter the damages caused by treatment. In the ES scenario, the lower amount of rainfall causes the implementation of RW reuse in the system to deteriorate the state of the ecotoxicity indicators, as well as the fossil potential category (and consequently the natural resources endpoint). Thus, in countries with low amounts of rainfall, the inclusion of the RW line in the decentralized layout must be preceded by detailed analysis.

Table 7: Summary of the variations on total net impacts caused by the inclusion of site-specific considerations. Percentages show change with respect to the EU scenario. All scores refer to decentralized system. The abbreviations used are the ones assigned to the site-specific scenarios and refer to the cases of Vienna (AT), Barcelona (ES), Trondheim (NO), and Bucharest (RO).

Midpoints	ES	NO	AT	RO
FEP (kg P-Eq)	-68%	-81%	-15%	133%
FETP (kg 1.4-DBC-Eq)	-14%	-23%	-5%	21%
FFP (kg oil-Eq)	-17%	-91%	-27%	10%
GWP1000 (kg CO2-Eq)	-22%	-71%	-16%	3%
HTP (kg 1.4-DBC-Eq)	-40%	-57%	-16%	55%
IRP (kg Co-60-Eq)	1%	-97%	-78%	9%
LOP (m2*a crop-Eq)	-51%	-93%	-5%	-78%
MEP (kg N-Eq)	-95%	-125%	-40%	111%
METP (kg 1.4-DBC-Eq)	-16%	-24%	-6%	22%
ODPinfinite (kg CFC-11-Eq)	-3%	-20%	-2%	-6%
PMFP (kg PM2.5-Eq)	-15%	-61%	-43%	80%
POFP (kg NOx-Eq)	2%	-70%	-29%	-2%
TAP (kg SO2-Eq)	-19%	-85%	-55%	54%
TETP (kg 1.4-DBC-Eq)	-4%	-25%	-11%	7%
WCP (m3)	-54%	532%	7%	114%

Endpoints	ES	NO	AT	RO
Ecosystem quality (species.yr)	-26%	-73%	-20%	19%
Human Health (DALYs)	-26%	-67%	-29%	50%
Natural resources (USD 2013)	22%	-110%	-33%	-55%

Improved performance Worsened performance

4 Sensitivity analysis

Impact assessment studies are data-intensive and thus the quality and reliability of the information employed is critical. A data analysis to assess the level of uncertainty associated with the LCIs used from ecoinvent is performed. A two-step process is carried out involving the generation of 100 scenarios through Monte Carlo sampling and the statistical analysis of the obtained data.

The Monte Carlo simulation is performed in the Activity Browser considering 100 iterations of each LCI, including uncertainties associated with all the technosphere flows, exchanges with the biosphere, characterization factors, and parameters for all midpoint and endpoint indicators. The simulation is firstly run for the five decentralization scenarios (incl. EU, ES, NO, AT, and RO). The obtained impact data is then transferred to Minitab (v 19.2) statistical software to verify the significance of the differences between countries. Samples are tested using the Games-Howell method for comparison of groups of data with varying variances with a confidence interval of 95%. Box plots and Games-Howell test results can be found in the sensitivity analysis section of the supplementary material.

The Games-Howell test helps determine which differences between site-specific scenarios are significant, while box plots help visualize the uncertainty to which the results are subject for each environmental category. Uncertainty linked to LCI data is the highest for the FEP, IRP, TETP, and HTP categories. The EU, RO and ES scenarios show non-significant differences in the case of IRP and TETP, and for HTP the influence of site-specific factors takes even less relevance, especially in the case of non-carcinogen impacts. It is confirmed that regional data does not influence the performance of the system relevantly to the ODPinfinite category, as results appear similar even when uncertainties for this indicator are low. The only city being clearly separated from the rest in terms of ozone depletion is Trondheim, with a much lower impact.

The test also confirms that even when uncertainty is considered, the indicators which are more sensitive to location changes are the TAP, MEP, LOP, and PMFP midpoints. Total scores for all countries and the European average are also significantly different between each other for the ecosystem quality and natural resources endpoints.

The rest of midpoints and the human health endpoint show some overlapping of final scores for the countries which appear the closest in the graphs presented in the Contribution and comparative analysis for the EU scenario. Differences between the EU and RO cases for GWP100 and FFP appear non-significant, as well as those between EU and AT for FEP and WCP. For the aquatic ecotoxicity indicators, close scores of EU and AT and ES and NO also position these countries at the same level. Finally, ES is too close to EU to show significant changes in the POFP indicator. The category where conclusions are altered by the sensitivity analysis is the human health endpoint. While for this category, the RO case is distinctly higher than all other location-specific cases, the differences between the different cities and the European average prove non-significant.

In view of the obtained results, the European average scenario is further studied to assess the influence of uncertainty associated with ecoinvent data on the results of the comparative LCA. While Besson et al., 2021 and Risch et al., 2014 carry out an uncertainty study of the impacts of centralized WWTPs by changing parameters which influence the results such as electricity consumption and location, the uncertainty tied to the data collected from ecoinvent is not reported. When reviewing the results of the sensitivity analysis for the European average case, it is observed that for the FFP midpoint and the human health endpoint the decentralized system performs worse than the centralized approach in 74% and 59% of the modelled scenarios, respectively. The natural resources endpoint is also under discussion as around 45% of scenarios are above the value for the centralized treatment. For the HTP, FEP, FETP, LOP, METP, and PMFP indicators, the conclusions of the comparison are reversed in minor shares of 31%, 20%, 16%, 4%, 3%, and 2% of scenarios, respectively.

All these indicators where at least one scenario changes the decision of the contribution analysis are further studied in Minitab. The Monte Carlo simulation is run for the part of conventional treatment modelled in the thesis (incl. potabilization and transport of water), and Games-Howell tests are performed comparing centralized treatment against decentralized treatment (EU scenario). Results for all the aforementioned indicators prove significantly different and thus the conclusions of the comparative analysis remain solid (incl. the natural resources endpoint, and FFP, FETP, METP, FEP, LOP, and PMFP midpoints), with the exception of the HTP midpoint and the human health endpoint (see the Sensitivity analysis section in the supplementary material). It is acknowledged that even higher uncertainty can potentially be attributed to centralized treatment as wastewater treatment has a high weight on the total impacts (Figure 18). Thus, the HTP midpoint and the human health endpoint could benefit from further study.

5 Limitations and recommendations for further research

The present thesis assesses the environmental performance of a decentralized layout composed of a specific selection of technologies designed for resource recovery and more sustainable water treatment.

The research aims to contribute to the path towards more sustainable water management systems and extend the current knowledge on the environmental performance of decentralized approaches. The findings are especially relevant to the scientific community and decision-makers to both aid the study of alternatives to conventional water treatment and to support the implementation of new technologies. In this section, the limitations of the study are visited, and further research is proposed with respect to the design of the decentralized treatment, the results of MFA and LCA, and the comparison between conventional and decentralized treatment.

The choice of treatments which constitute the studied decentralized layout in this thesis is backed by previous research (Garrido-Baserba et al., 2022). The aim is to continue analysing the performance of technologies for which the techno-economic feasibility has already been confirmed. However, the presented case is not a conclusive take on decentralized technological schemes. Further study on the environmental impacts of other decentralized layouts or a combination of centralized and decentralized technologies would also be key when designing water treatment systems with a focus on sustainable development. Hybrid solutions are advocated as they could potentially offer the possibility to combine the benefits of both approaches (Keller, 2023).

The study's results are favourable to the decentralized system and indicate it efficiently minimizes resource loss and outperforms average European centralized WWTPs in the impact assessment. However, both MFA and LCA can be used to further improve the design of the decentralized treatment trains.

On the one hand, MFA highlights resource loss in the RBC and RO processes, where resource recovery could be further strengthened by the implementation of measures for waste management or treatment substitution (see the Material flow analysis section). The adoption of these approaches would however change the impacts of the system, which would need to be revised.

On the other hand, LCA results indicate that the substitution of the MBR with an alternative treatment could decrease direct impacts on the GWP1000, ODPinfinite, and POFP indicators and indirect emissions due to electricity consumption, which would affect all impact categories. Besides, construction impacts could also be greatly reduced by the change. Trickling filters are energy-efficient processes with high COD removal which are often included in water reuse schemes (Arous et al., 2023; Henrich and Marggraff, 2013). This fixed film process could potentially be incorporated into the layout to take the position of the MBR. However, the impacts of this change on final water quality would need to be assessed, as it is highly likely that a further separation stage would be required after the filter. Conversely, the current MBR ensures adequate treatment to reuse standards. Thus, if further treatment would be needed to meet adequate effluent quality, the global impacts of this change should again be compared with the current damage caused by the use of an MBR.

Regarding completeness, industrial ecology studies depend on the availability of reliable data and are often restricted by the lack thereof. In this case, the additional challenge to compare an existing activity (i.e., centralized treatment) with a system under the design phase causes the inventory construction to be limited by the lack of data for specific concepts, such as heavy metals, micropollutants, and microbial pathogens. While these flows are often out of the scope of LCA studies for WWTPs (Corominas et al., 2020), their inclusion could be favourable for increasing the level of detail of the assessment. Further knowledge of the decentralized system would also help provide insight for the site-specific cases through the inclusion of data regarding local pollutants, or the effect of temperature. Finally, additional information related to secondary processes, such as the consumption of cleaning chemicals, could also provide more insight into the damages caused by the system and decrease the uncertainty tied to the impact scores. Real-life implementation and monitoring of the layout could allow to obtain missing information and enrich the inventory. Based on the results, the MEP category could beneficiate the most from further study, since while the conclusion that there is a relevant performance improvement with respect to conventional practices for marine eutrophication is solid, the possibility that an underestimation of impacts to this indicator has occurred is acknowledged. It must be considered that if further information was added to the inventory for decentralized treatment, the same would be required for the traditional approach, to maintain the level of detail and boundaries for fair comparison. The analysis of the influence of seasonality could also help further expand the impact assessment, as stated by Rashid et al. (2023).

A lack of research covering the totality of activities involved in water management hinders the comparison of the decentralized system with conventional treatment. Many studies are limited to a specific treatment unit or step of the process (Rashid et al., 2023). The analysis of review papers focusing on LCA applied to wastewater treatment (Estévez et al., 2022; Foglia et al., 2021; Mehmeti and Canaj, 2022) combined with the general search of related studies yielded three main articles reporting the impacts of conventional WWTPs for a comparable setting, methodology, and level of detail (Besson et al., 2021; Heimersson et al., 2014; Risch et al., 2014). Four additional papers were used to compare the results of the LCA for conventional treatment to pre-reported impacts (Gómez-Monsalve et al., 2022; Kraus et al., 2017; Opher and Friedler, 2016; Santana et al., 2019). Even if differences especially in the geographic location of these studies difficulted comparison, they also provide impacts for centralized treatment following a conventional treatment scheme and were consulted. An even more reduced number of studies assess the impacts of the water management system consisting of not only wastewater treatment plants, but also pipeline networks, pumping stations, and drinking water plants. A good example is the thesis by Venkatesh (2011), which acknowledges the complex interdependencies between the sub-parts of the system. However, current research on the impacts associated to the anthropogenic components of the water cycle as a whole is insufficient, and the sector could greatly beneficiate from this information.

Thus, concerning the comparative LCA, further research on the impacts of water management as a system would allow for more robust comparisons of the performance of new technologies with the conventional approach. The modelling of centralized WWTPs with site-specific data could allow to judge whether the ratios of midpoints and endpoints in which decentralization's impacts are lower is maintained when the location of the study changes. Finally, data-related uncertainty information for the centralized case of wastewater treatment would help further explore impacts, especially on the HTP midpoint and human health endpoint.

6 Conclusions

The present thesis aims to provide insight into the environmental profile of a decentralized water treatment system composed of technologies for which the techno-economic feasibility has been previously assessed with positive results (Garrido-Baserba et al., 2022), in order to further expand the knowledge on alternative approaches to water treatment.

The assessment of the environmental implications of the installation and operation of the totality of infrastructure involved in water management through different approaches is a challenging but necessary task in the path towards the design of sustainable water systems. The performance assessment of a promising decentralized approach and its comparison with conventional treatment in this thesis include all utilities involved for both cases. Despite unavoidable limitations linked to data availability and to the need to define a realistic scope for the thesis, the favourable results for decentralization provide a reason and a knowledge basis for the further expansion of this research.

The study satisfactorily fulfils the proposed aims in section 1.2. The conclusions are presented in a numbered list in accordance with the research question they answer (see section 1.2).

1. The implementation of source-separation allows for more effective treatment adapted to each wastewater flow (including urine, brown water, and grey water), which facilitates focusing on resource recovery.

   The studied treatment train achieves recovery rates of 83%, 46%, and 26% of input water, nitrogen, and phosphorus, respectively. Biogas produced in the up-flow anaerobic sludge bed reactor treating the organic fraction of the building's waste can cover up to 40.5% of the energy demand for brown water and waste treatment. Resource loss can be minimized by the anaerobic digestion of sludge from the biological treatments. If optimized management of waste is carried out, recovery percentages can be increased to 61% for nitrogen, 89% for phosphorus and 65% for energy.

2. The damages to the environment caused by the implementation of the decentralized layout are mainly due to the operational phase, which contributes over 90% of the impacts to all studied indicators (Table 6). Infrastructure takes the highest relevance for the ecotoxicity midpoints and the natural resources endpoint, with a maximum impact share of 10.5% for terrestrial ecotoxicity.

   The contribution analysis highlights the role of resource recovery in damage mitigation, which is especially relevant for the water consumption, marine eutrophication, terrestrial acidification, and freshwater eutrophication potential categories. For these indicators, avoided impacts represent 71%, 70%, 48%, and 42% of net damages. The average impact reduction for midpoints is that of 34%.

   The activities and factors with higher direct contribution to impacts during the operational phase include emissions of greenhouse gases and ozone-depleting substances from the treatment trains and chemical production for dosing in the reactors. Regarding the construction phase, steel, stainless-steel, and silicone components and the manufacturing and installation of equipment are the main sources of damage.

   Indirect impacts arising from beyond the water sector take great relevance in the environmental profile of the decentralized layout. As the minimization of ancillary chemical additions is prioritized, most treatment units rely on electricity to operate. Its production and the generated waste, as well as its distribution, greatly affect the performance of the reuse layout. Thus, on the path towards optimized resource management, the decentralization of water treatment and energy production are intrinsically linked together. Photovoltaic-powered water treatment technologies could provide an integrated solution with low environmental impacts.

3. Through the implementation of wastewater reuse, water consumption in the building is reduced to only 17% of the original demand and nitrogen and energy recovery are doubled. Figure 13 summarises the comparative assessment between conventional and decentralized wastewater treatment in terms of resource recovery.

   The results of the impact assessment showcase the potential of the proposed decentralized system to reduce the environmental impacts of current practices in water treatment to 10/15 of the studied midpoint indicators. The global warming, terrestrial acidification, marine eutrophication, ozone depletion, and photochemical oxidant formation environmental categories are the most beneficiated from the use of decentralized technology, which reduces impacts by 44%, 52%, 99%, 71%, 37%, respectively. Meanwhile, conventional treatment has lower impacts for 5 midpoint indicators including land occupation, marine and terrestrial ecotoxicity, human toxicity, and ionizing radiation potential, for which the decentralized layout shows impacts 18%, 69%, 121%, 71%, 70% larger, respectively (Figure 14, Figure 15, and Figure 16).

   All three endpoint categories are favourable to decentralization in an average European setting, with observed impact reductions of 70% for ecosystem quality, 44% for human health, and 11% for natural resources (Figure 17).

   A sensitivity analysis relying on Monte Carlo simulation reveals that in 59% and 31% of the modelled cases conclusions could be reversed for the HTP midpoint and the human health endpoint, respectively. Results of the comparison between centralized and decentralized treatment are solid in terms of data uncertainty for the rest of indicators.

   For the centralized approach, wastewater treatment takes the highest weight on total impacts; however, for marine and terrestrial ecotoxicities, human toxicity, and fossil depletion, water transport and freshwater potabilization showcase higher damage potential than WWTPs (Figure 18).

4. The variations in impacts observed between site-specific cases follow an expected pattern as they are directly linked to the weight of electricity in each indicator. Avoided impacts through resource recovery are also affected by the types of fertilizer used in each country and their respective obtention routes, as well as by the amount of available rainwater (discussed for RQ5). The impact of location-specific factors has a limited influence on the construction phase.

   Table 7 summarizes the increases or decreases in impacts for the four studied cities with respect to an average European setting. The AT scenario shows the least variation while the high presence of fossil fuels in the mix for the RO case causes an average 35% deterioration to midpoint indicators. For the NO scenario, the effect of burden-shifting is clear as a mix heavily based on hydroelectric energy allows for relevant improvements in all midpoints but strongly deteriorates the water consumption potential category. The ES scenario achieves moderate impact reductions for all midpoints except for ionising radiation potential and photochemical oxidant formation potential.

5. The inclusion of rainwater capture and reuse yields positive results for the environmental analysis in all scenarios, as impacts caused by the rainwater treatment equipment during its life cycle are globally lower than the benefits obtained from potable reuse. However, in the scenario with lower rainfall (ES), the incorporation of rainwater reuse increases impacts to four midpoint indicators (incl. all three ecotoxicity potentials and fossil fuel depletion) and to the natural resources endpoint. For the AT and RO cases, only midpoints regarding ecotoxicity are deteriorated. The scenario with the highest amount of rainfall (NO) expectedly offers the best performance (Table S26).

   While water and nutrient recovery allow for relevant avoided impacts in all locations, rainwater capture capacity highly influences the potential of the rainwater line to be favourable for the decentralized system. Thus, the decision to include rainwater reuse in the layout should be accompanied by a site-specific assessment of the damages and benefits caused by the installation of the necessary equipment.

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