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How decentralized treatment can contribute to the symbiosis between environmental protection and resource recovery

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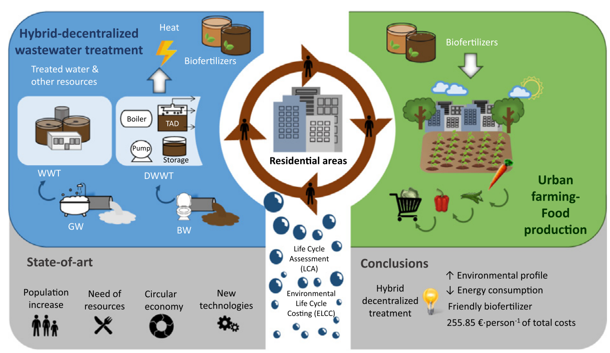
HIGHLIGHTS

- Environmental-economic analysis for wastewater treatment and urban farming
- Life cycle assessment and environmental life cycle costing were the methodologies used.
- The best environmental profile corresponds to a hybrid-decentralized system.
- Internalisation of some external costs is relevant to the polluter pays principle.

GRAPHICAL ABSTRACT



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ABSTRACT

Challenges associated with the sustainability of the water cycle pose new opportunities for resource recovery and greater environmental protection. While centralized wastewater treatment plants must evolve in their design and operation to adapt to a scenario of increasing demand for water, resources and energy, the decentralized approach emerges as an option to be considered in small communities or developing residential areas where bioenergy production can be improved through the recovery of organic matter in segregated streams or where the investment in the sewer network for connection to a centralized facility may be technologically or economically unfeasible. The main objective of this work is to evaluate the environmental and economic profile of a hybrid-decentralized configuration for the purpose of efficient wastewater management and resource recovery and its comparative evaluation with the centralized treatment scenario. Beyond water reclamation, decentralized treatment offers the possibility of valorization of digestate streams as nutrient sources for horticultural or ornamental crops in the vicinity of the plant. Based on the results of the environmental profile, this manuscript shows that the decentralized treatment approach is in line with the philosophy and guidelines of the circular economy, as it allows the use of reclaimed water and biofertilizers under safe and environmental-friendly conditions.

1. Introduction

The need for water, energy and food are key elements inherent to a world population with a growing demand for increasingly scarce exploitable resources (Sheikh et al., 2019). Agriculture is the main water user and accounts for up to 80% of freshwater withdrawals, with irrigation of food crops being the dominant use (Velasco-Muñoz et al., 2018). The use of reclaimed wastewater can represent a resource recovery option for small and medium-sized agricultural areas as it not only supplies irrigation needs, but also reclaimed water can provide a potential source of nutrients for crops (Poustie et al., 2020). The use of

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reclaimed water has become a reality in different regions of countries in the Mediterranean area such as Murcia (Spain), where more than 100 $Mm^3 \cdot year^{-1}$ are reused for agriculture or the rural area surrounding the city of Milan (Italy) that uses high quality treated wastewater for agricultural irrigation (SUWANU Europe, 2019). Although the EU policy in water issues has been especially intense in recent times, the extensive wastewater recycling has also been supported by other countries. For example, Israel treats and reuses around a 90% of the municipal wastewater (Craddock et al., 2021). Zhu and Dou (2018) have reported for 2015 a utilization ratio of 30% of the reclaimed water in Beijing (China). California and Florida (United States) have been irrigating agricultural crops with recycled water for decades (more than 50 years for Florida) (Parsons, 2018; Sheikh et al., 2018).

Considering wastewater treatment alternatives, centralized systems are associated with high energy consumption and extensive sewerage infrastructure, although it presents as main strength the proven technological robustness linked to widely developed treatment strategies (Ashok et al., 2018; Jung et al., 2018). However, when it comes to reusing reclaimed water from centralized wastewater treatment systems, the distance between facilities and agricultural areas can become a major drawback, as a reclaimed water distribution network has to be enabled that would duplicate treatment, reclamation and discharge infrastructures (Qureshi, 2020). On the other hand, the need for agricultural water may be seasonal, which implies the storage of large volumes of reclaimed water (Cirelli et al., 2008). In this framework, decentralized wastewater plants relieve pressure on central plants that may be at their capacity limit, while reducing wastewater pumping costs (Capodaglio, 2017). Moreover, the decentralized treatment approach can address the challenges of reuse of reclaimed water and biofertilizers by considering their application in green areas and agricultural lands in the surrounding areas of the facilities (Bisschops et al., 2019; Malila et al., 2019).

A classic approach to decentralized systems is based on the segregation of three waste streams: black water (BW) generated in the toilets, grey water (GW) from laundry, showers or dishwashers and kitchen waste (KW) (Wielemaker et al., 2018). Segregation of BW streams avoids their dilution and improves the resource recovery performance in the form of biogas and nutrients (Zeeman and Kujawa-Roeleveld, 2011). Up-flow anaerobic sludge blanket (UASB), anaerobic membrane bioreactors (anMBR) and thermophilic anaerobic digestion (TAD) are examples of anaerobic technologies for BW treatment (Kujawa-Roeleveld et al., 2006; Pretel et al., 2016). On the other hand, the management of GW after treatment allows its reuse for refilling toilet flushing as well as irrigation water (Skambraks et al., 2017).

A holistic and comparative assessment between both treatment options as well as activities related to the reuse of recovered flows should be carried out in the context of an environmental and economic assessment (Garrido-Baserba et al., 2018; Kobayashi et al., 2020). When considering the introduction of a water reuse scheme, it is important to examine the benefits and drawbacks that could be considered in the decision-making process to provide a clear conclusion (Ofori et al., 2021). Environmental and Economic life cycle methodologies not only consider the impacts and costs associated with the treatment stage, but also the implications derived from the sanitation network, the recovery of resources such as energy and biofertilizers, and the impact of fertilization activities (Resende et al., 2019; Morelli et al., 2018).

This manuscript comprises an environmental assessment from a life cycle perspective following the LCA methodology that integrates the water reclamation and the production of biofertilizers in wastewater treatment systems with their use in crops in residential areas for food production. A comparison of various scenarios has been proposed considering technological and scale differences in the wastewater treatment facility and the electrical energy source and requirements of the vacuum toilets of an innovative system for black water treatment.

The economic evaluation will be only performed for this decentralized BW treatment using an Environmental Life Cycle Costing (ELCC) with the estimation of internal costs (CAPEX and OPEX) and external costs (monetization of environmental impacts) (Roh et al., 2018).

2. Materials and methods

2.1. Description of scenarios under comparison

2.1.1. Overall description

This study comprises an environmental assessment for a baseline scenario from which the nutrients recovered from the wastewater of a decentralized/hybrid treatment facility were used for the fertilization of the crops of in an urban area. The scenario has been divided into two different subsystems: the first (*SS1 - wastewater management and fertilizer production*) considers the management of segregated wastewater flows in a decentralized facility (for black water) and in a centralized facility (for grey water) and the manufacture of mineral fertilizers. The second (*SS2 - gardening activities*) includes agricultural production as well as emissions from cultivation and fertilization. This decentralized baseline scenario (DBS) was compared to a centralized scenario (CS) where the resources for land fertilization were recovered from the combined wastewater treatment of the black water (BW) and grey water (GW) in a centralized facility instead of a hybrid treatment. Fig. 1 depicts the flowchart of the configuration of both scenarios.

2.1.2. Wastewater management in subsystem SS1 for the decentralized baseline scenario (DBS)

The decentralized BW management has been proposed in the framework of Run4Life (Recovery and Utilization of Nutrients 4 Low Impact Fertilizer) project as a decentralized treatment for a residential or urban development area of 80–90 inhabitants where ultra-vacuum toilets have been installed. Interest in these low-flow toilets has increased in recent years due to the minimal water consumption, which means that the BW stream has a high concentration of organic matter, ensuring a higher yield of biogas production in the anaerobic digestion treatment. It consists of a compact unit that incorporates an intermediate discharge tank, where the vacuum is maintained and is connected to the toilet through an inlet valve. In quantitative terms, an average consumption of approximately 0.5–0.9 L of water per flush was considered, corresponding to an average BW production flow of 0.3 $m^3 \cdot day^{-1}$ and an average electricity consumption of 13.33 $kWh \cdot m^{-3}$.

In this neighbourhood, BW is collected separately and sent to a decentralized system consisting of a thermophilic anaerobic digestion unit (TAD) operating at 55 °C. The biogas produced is sent to a boiler to produce heat required for the operation of the TAD and the excess can be supplied to the neighbourhood. The composition of the biogas is 72.6% methane, 27.3% carbon dioxide and 0.1% other gases. The pathogen -free liquid and solid biofertilizers produced are used for fertilization activities in urban gardens, where residents have a small plot of land for horticultural crops (Zhang et al., 2020). The GW, on the other hand, was managed in a centralized facility. The sludge produced in the GW treatment is not relevant and it is not feasible to use it for crop fertilization due to the small quantity of sludge produced (Tervahauta et al., 2014).

2.1.3. Wastewater management in subsystem SS1 for the centralized scenario (CS)

This alternative scenario considers the same population where conventional toilets are installed so that GW and BW streams are not segregated and are treated in a conventional centralized facility for a total capacity of 55,000 $m^3 \cdot d^{-1}$ (Morera et al., 2017). Conventional toilets use more water per flush (7 L/flush), which implies the production of 4.17 m^3 BW per day. The treatment plant is divided into five sections: pumping and pre-treatment, primary treatment, activated sludge, anaerobic digestion and composting. As main outputs of the system, electricity, compost and treated water are considered.

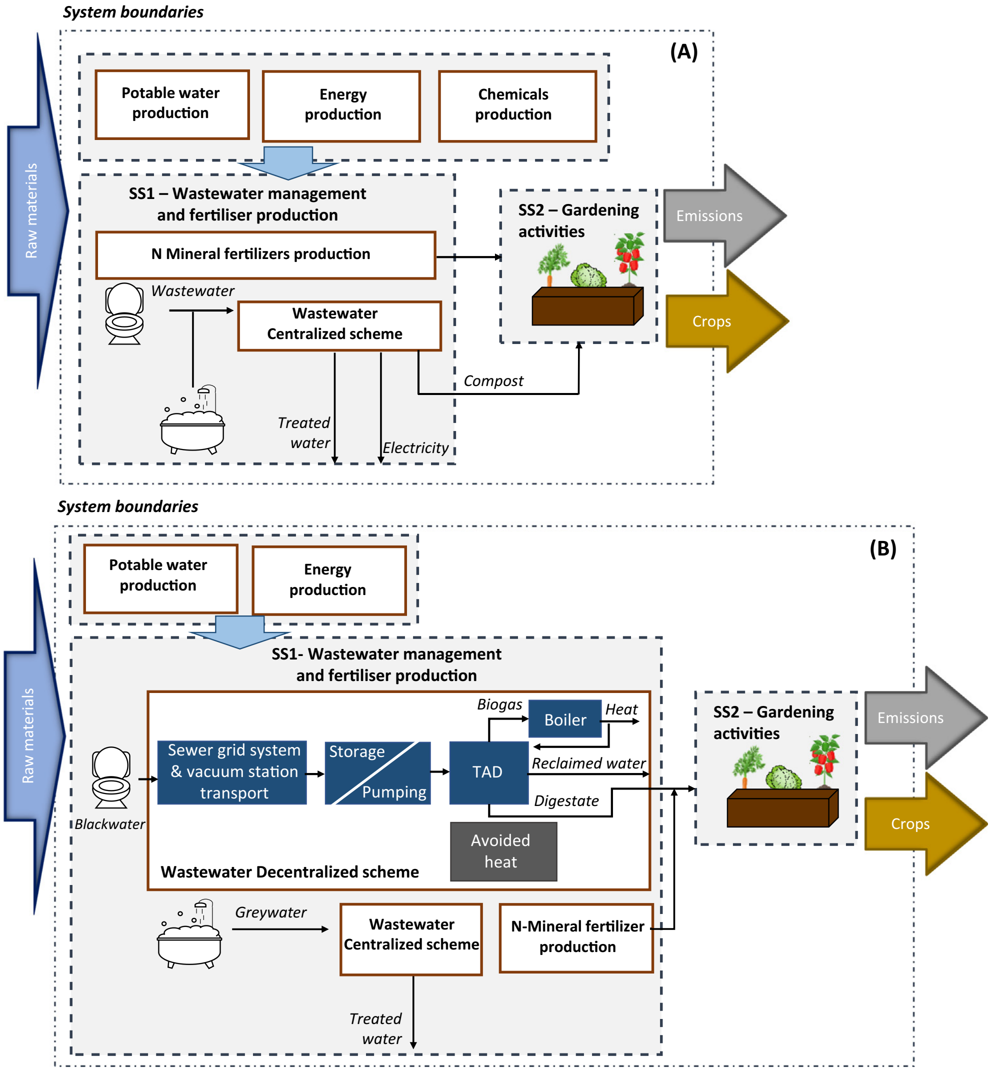

Fig. 1. Flow chart and subsystems of the CS (Centralized Scenario) (a) and DBS (Decentralized Baseline Scenario).

2.1.4. Fertilizer production in subsystem SS1 and subsystem SS2

The amount of nutrients recovered from the wastewater management in CS or in the decentralized scenario DBS where used to grow vegetables (peppers, bush beans, carrots and cabbages) in the land plots of a residential area. In this sense, biofertilizers could be applied to agricultural land embedded in urban soil through two approaches: urban farming and urban gardening (Pölling et al., 2017). Both alternatives, despite their similarities, are not semantically the same. Urban farming takes advantage of the proximity of densely populated areas by harvesting products that can provide economic benefits. Gardening, however, is more related to the production of food and non-food systems that fulfil the social benefits of agricultural activity itself being independent of economic gains (Lohrberg et al., 2016). Thus, the concept of urban farming was integrated with the wastewater treatment in this manuscript. The resources obtained from wastewater treatment systems may not be sufficient to ensure the required N and P fertilization levels for the crop, implying the need for extra input of mineral fertilizers. A minor amount of N-based mineral fertilizer was required to meet nutrient needs. In contrast, P and K demands were fully met by the biofertilizer.

2.1.5. Alternative scenarios to decentralized baseline scenario (DBS)

Regardless of the treatment scheme, differences in energy can be detected between centralized and decentralized systems due to the pumping distance and the flows of tap water and wastewater. Energy consumption has been reported as a remarkable parameter in the environmental profiles shown in many LCA studies conducted for wastewater treatment plants (Takeshita et al., 2020; Rashid and Liu, 2020; Wang et al., 2019). Therefore, the energy demand of both scenarios was examined and compared to effectively reduce the environmental impacts.

Based on the results obtained from the decentralized baseline scenario (DBS) and with the aim of reducing the energy demand, an alternative decentralized scenario with energy reduction in the vacuum toilets (DRS) was proposed. The electricity demand considered for the toilets was 7.76 $kWh \cdot m^{-3}$ which is a data provided by commercial suppliers (a variation of about 50% was identified) (SwedEnviro, 2001). Two more scenarios were proposed. In one of them the electricity is supplied by photovoltaic cells (DSS) while in the other the analysis focuses on the joint implementation (DJS) of the measures taken in the DRS and DSS scenarios. These new scenarios have the same scheme shown in Fig. 1B for the DBS scenario, differing only in the energy considerations taken for the vacuum system.

Further research is needed in the development of vacuum toilets to make them more environmentally competitive, since a large difference has been identified in the literature/market not only in the electricity demand for the vacuum toilets but also in the water consumption per flush. Conventional toilets demand large amounts of water (approx. 6–12 L/flush) and novel technologies are being developed to reduce this consumption by promoting that water-saving wastewater management, which has potential environmental and economic benefits. Low-vacuum and vacuum toilets reduce drastically the amount of wastewater, which is more concentrated in organic matter. Therefore, it would be necessary to perform a sensitivity analysis to identify the influence of these parameters (electricity and water).

The ultra-low vacuum toilets considered in the hybrid scenarios have a lower electricity water demands than other commercially available vacuum toilets. The energy consumption for this kind of toilets can be of up to 34 kWh·person^{-1}·year^{-1} (Mohr et al., 2018) although numerous studies reported that this energy is in the range of 15–30 kWh·person^{-1}·year^{-1} (Münch and Winker, 2009; Remy and Jekel, 2012; Herrmann and Hesse, 2002). Concerning to the water consumption, a normal range is between 0.70 and 1.50 L·flush^{-1} (Remy and Jekel, 2012; Mohr et al., 2018; SwedEnviro, 2001). However, some technical reports have identified improvements in energy and electricity consumption that make the well-known vacuum toilet competitive enough with other emerging technologies (such as ultra-low vacuum toilets). The electricity and water demand of vacuum toilets has been found to be 4 kWh·person^{-1}·year^{-1} and 0.50–1 L·flush^{-1} (Otter Vacuum, 2021).

Considering these references and the effect on the environmental profile of vacuum systems from the electricity demand, different alternative scenarios have been considered for discussion to compare our results with those of the literature and the conventional system. Hence, four scenarios have been proposed for comparison considering the same neighbourhood.

The water consumption is identical in three of them. Thus, they have been named according to their energy consumption. The scenario with the highest demand (H) was designed using Mohr et al. (2018) data. The remaining scenarios were designated as M (medium energy demand), L (Lower energy demand), and W (different water consumption). Data from Münch and Winker (2009) were considered for scenario M, data from Remy and Jekel (2012) for scenario L and data from Otter Vacuum (2021) for scenario W. The energy and water demand for each of them is: 25 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1} (scenario H), 22.5 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1} (Scenario M), 15 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1} (Scenario L) and 4 kWh·person^{-1}·year^{-1} and 1 L·flush^{-1} (Scenario W).

2.2. LCA framework

2.2.1. Goal and scope definition

The environmental analysis has been performed following the Life Cycle Assessment (LCA) methodology under ISO standards 14040:2006 and 14044:2006. The objective is to evaluate the environmental profile of systems with integrated wastewater management and agricultural activities. The impacts are compared in relation to the origin (from centralized or hybrid configurations) of the biofertilizer obtained from wastewater treatment to fertilize the crops of a residential area. In this analysis, attention has been paid to the operation stage and, therefore, the construction of the different infrastructures corresponding to both scenarios were excluded, since their contribution is negligible (Arias et al., 2020; Santana et al., 2019).

2.2.2. LCA data collection

The functional unit (FU) was defined as 1 m^2 of urban garden as integrates the valorization of the biofertilizer from the wastewater treatment process. This unit was considered as a reference for reporting all inventory data based on mass and energy flows. Many wastewater treatment facilities have been analysed using a process perspective that studies one of the most important functions of the facility: wastewater treatment (Corominas et al., 2013). However, the multifunctionality of these facilities must also be considered and a product perspective should also be used to reflect the resource recovery function of the new treatment plants (Moretti et al., 2020). The most typical functional unit for this type of perspective is related to the valorization of sludge (Leung Lam et al., 2020). Broader limits of the system have been considered in this study. Thus, the defined functional unit has also incorporated activities related to food production in urban gardens under the concept of urban farming (Dorr et al., 2021).

Primary data were handled for water consumption in both conventional and vacuum toilets, as well as for the electricity requirement in the latter. As for the BW decentralized wastewater treatment system, the information used to carry out the analysis was based on direct data provided by the Run4Life pilot plant located in Sneek (The Netherlands). A summary of the main inputs and outputs for the BW decentralized scheme is detailed in Table 1.

The environmental impacts associated with the background processes of the subsystems SS1 and SS2 were estimated based on data available in the Ecoinvent v3.0.0.1 database. The following processes were considered: tap water consumption, electricity production, small scale-heat production (avoided burden), reclaimed water (avoided burden) and mineral fertilizer production. Since there is a surplus heat flow in the BW decentralized treatment strategy, it was assumed that the recovered heat avoids the production of an equivalent amount of heat from natural gas (Leung Lam et al., 2020). Thus, the impacts associated with the production of this fossil heat were deducted from those derived from SS1 (see Fig. 1B). Similarly, an avoided allocation was also assumed for the reclaimed water generated in the decentralized BW treatment, that is the water content of the liquid stream will save a comparable amount of irrigation water. The distance between the centralized facility and the urban plots becomes a drawback for pumping water. Therefore, the treated water is discharged directly into the environment instead of being reused as reclaimed water.

The biofertilizers from wastewater treatment facilities has been applied in the urban areas to supply the exact demand of nutrients to the crops when the phosphorus is the limiting element.

The nutrient requirements (as mineral fertilizer) in the subsystem SS2 for the cultivation of the same tract of land destined to each of the selected crops (carrots, bush beans, cabbage and peppers) was

Table 1
Main inventory data for the BW decentralized scheme, step common in Scenarios DBS, DRS, DSS and DJS. Data are reported per functional unit that is, 1 m^2.

	DBS	DRS	DSS	DJS
Inputs from the technosphere				
Materials				
COD (kg)	0.74			
TN (kg)	0.01			
TP (kg)	$3.00 \cdot 10^{-3}$			
K (kg)	$1.50 \cdot 10^{-2}$			
Tap water (kg)	20.41			
Energy[a]				
Electricity (kWh)- tap water pumping	$4.08 \cdot 10^{-3}$			
Electricity (kWh)- toilets and sewer	0.27	0.16	0.27	0.16
Electricity (kWh)- pumping to TAD	0.06			
Outputs to the environment				
CO_2 (g)	183.45			
S (mg)	133.80			
Outputs to the technosphere				
Liquid digestate to SS2				
N (g)	8.92			
P_2O_5 (g)	6.61			
K_2O (g)	17.61			
Solid digestate to SS2				
N (g)	0.57			
P_2O_5 (g)	0.27			
K_2O (g)	0.51			
Avoided flows				
Heat (kWh)	0.77			
Irrigation water (L)	20.41			

[a] Electricity demand in scenarios DSS and DJS is solar power.

14.5 g N·m^{-2}, 4.12 g P·m^{-2} and 1.34 g K·m^{-2} (Shresthal et al., 2020). It should be noted that the nutrient assimilation efficiency of a biofertilizer is lower than that of a mineral fertilizer (Ashekuzzaman et al., 2021). Moreover, the liquid and solid digestates of the BW treatment in scenario DBS (and DRS, DSS, DJS by extension) and the compost from CS have different fractions of N and P available for the crops and therefore additional nitrogen-based mineral fertilizer is required to meet crop needs in SS2.

An average mineral replacement ratio was 52% for solid fertilizer and compost and 72% for liquid digestate for nitrogen and 60% for phosphorus (Ashekuzzaman et al., 2021). Nitrous oxides (N_2O) emissions were estimated according to the Intergovernmental Panel on Climate Change (IPCC, 2019). Tier 1 nitrogen dioxides (NO_x) and Tier 2 ammonia (NH_3) emissions were calculated according to the methodology of the European Environment Agency and the European Monitoring and Evaluation Program (EMEP/EEA, 2019). Nitrate (NO_3^-) leaching (Faist Emmenegger et al., 2009), and phosphorus (PO_4^{-3}) leaching and runoff (Shresthal et al., 2020) were also taken into account. CO_2 emissions resulting from land use change were not included as it has been assumed that the garden area has been devoted to agriculture for the last 20 years. The main inventory data corresponding to the scenarios under study are shown in Table 2.

2.2.3. Life Cycle Environmental assessment method

Two impact assessment methods have been considered in the analysis. Firstly, the ReCiPe 2016 Hierarchist Midpoint method V1.03 World (2010) (Huijbregts et al., 2017) has been used for the selection of characterization factors required to estimate the environmental burdens, which were reported according to a set of impact categories at midpoint level: global warming (GW), stratospheric ozone depletion (SOD), terrestrial acidification (TA), freshwater eutrophication (FE), marine eutrophication (ME), terrestrial ecotoxicity (TET), freshwater ecotoxicity (FET), marine ecotoxicity (MET) and fossil resource scarcity (FRS). Secondly, the ReCiPe 2016 Hierarchist Endpoint method V1.03 World (2010) H/H (Huijbregts et al., 2017) has been considered to benchmark the profiles of the four decentralized scenarios and the centralized one. The choice of this endpoint approach allows reporting a single score that facilitates the communication of the environmental results. For this purpose, the normalization and weighting factors taken from ReCiPe 2016 Endpoint method have been considered, reporting the environmental results in terms of three indicators to express the relative severity of damage according to the Human Health (HH), Ecosystem Quality (EQ) and Resource Scarcity (RS). Those three endpoint indicators are the result of an aggregation process from specific midpoint categories using endpoint characterization factors. In this analysis, only the impact categories previously selected for environmental assessment at midpoint level have been considered for the estimation of the single environmental score, considering the corresponding normalization and weighting factors established by the method. The SimaPro software v9.0.0.29 (PRé Consultants, 2021) has been used for the computational implementation of the life cycle inventories.

Table 2
Main inventory data (per m^2) for the Subsystem SS2 corresponding to gardening activities.

	DBS/DRS/DSS/DJS[a]	CS
SS1- production and treatment activities		
Outputs to technosphere		
N-Mineral fertilizer to SS2 (g)	7.78	7.37
N-biofertilizer to SS2 (g)	9.50	11.88
P-biofertilizer to SS2 (g)	6.87	6.87
K-biofertilizer to SS2 (g)	18.11	2.61
Outputs to the environment		
Treated water (L)	–	4508.11
SS2- gardening activities		
Inputs from technosphere		
N-Mineral fertilizer from SS1 (g)	7.78	7.37
N-biofertilizer to SS1 (g)	9.50	11.88
P-biofertilizer to SS1 (g)	6.87	6.87
K-biofertilizer to SS1 (g)	18.11	2.61
Outputs to technosphere		
Bell peppers (kg)	1.05	1.05
Carrots (kg)	0.89	0.89
Cabbages (kg)	0.83	0.83
Bush Beans (kg)	0.89	0.89
Outputs to the environment		
Emissions into air		
N_2O- Direct (g)	0.27	0.30
N_2O- Indirect (g)	0.10	0.12
NH_3 (g)	3.26	3.79
NO (g)	2.98	2.87
Emissions into water		
PO_4^{-3} (g)	0.18	0.18
NO_3^- (g)	18.86	20.47

[a] This column includes all the inventory of the hybrid scenarios. Same inventory for all of them in subsystem SS2.

2.3. ELCC framework

The economic analysis performed within this manuscript studies the construction and operational phases of the black water decentralized treatment included in the subsystem SS1 of the DBS scenario to meet the requirements of the Directive 91/271/EEC195. The Environmental Life Cycle Costing (ELCC) is a methodology that estimates the monetary value of both internalities and externalities (at least one). The internal costs can be classified into two main categories: CAPEX (capital expenditure) and OPEX (operating expenditure).

The capital costs are related to the investments required to purchase and install the assets of the treatment process. The initial investment comprises fixed assets and working capital. Fixed costs have been estimated as an acquisition cost through purchase. The storage tank for black water, the reactor feed pump, the TAD reactor and the boiler are the main units of the decentralized wastewater treatment system. The storage tank costs can be determined from The National Tank Outlet (2021). The thermophilic anaerobic reactor of the decentralized scenarios has the same configuration of a typical UASB reactor. Libhaber and Orozco-Jaramillo (2012), Sato et al. (2007) and Puchongkawarin (2015) have provided estimations for UASB (Upflow Anaerobic Sludge Blanket) reactors and Soltero et al. (2018), Picardo et al. (2019), Sandvall et al. (2017) and Gowreesunker and Tassou (2016) for boilers. Apart from the purchased costs of the different equipment, other items have been considered: costs of the hydraulic and electrical connections (as 15% of the equipment costs). The indirect costs of execution were related to the general costs of the construction company (15% of the equipment and connection costs) and profits of the construction company (10% of the equipment and connection costs) (Acampa et al., 2019). A lifetime of 30 years and a discount rate of 5% were assumed (Karczmarczyk et al., 2021). Thus, all assets have been amortized annually at a constant depreciation rate (12%) and the residual value has been assumed zero (Ernst and Young, 2018). Because of the long-time horizon selected for the economic analysis of BW treatment facility, new investments for machinery replacement have been considered.

The OPEX includes all costs (variable and fixed) of goods and services for the daily operation and management of the facilities as well as the benefits from the sale of the products. The cost of the water consumed in households varies from one European country to another with prices ranging from 1.07 to 9.32 €/m^3 for 2020 (prices for Bulgaria and Denmark, respectively) (EurEau, 2020). However, these costs are already undertaken directly by the users and, thus, the costs associated with water demand are not covered by the overall budget of the decentralized wastewater treatment plant and they were left out of the ELCC boundaries. For similar reasons, the ultra-low vacuum toilets, and sewer system (collection and conveyance) costs were not included. The total variable cost of energy, such as heat or electricity, is estimated from the Energy Price Data Visualization Tool of Eurostat (Canaj et al., 2021). The average European price of electricity for a non-domestic consumer with a demand lower than 20 MWh is 0.26 €·kWh^{-1} and that of natural gas for heat production is 0.06 €·kWh^{-1} (Eurostat, 2020). As there is not consumption of chemicals in the BW decentralized system analysed, the operational costs are associated only with the electricity demand. In this sense, electricity consumption is a drawback from an economic point of view. The BW pumping from the storage tank to the thermophilic anaerobic reactor has been assumed as the only main consumer device because the energy required by the reactor is supplied by heat from the revalorization of the produced biogas.

Fertilizer transportation costs were considered insignificant because they were applied in the residential area under the concept of urban farming. Part-time staff labour costs were included in the estimates for occasional maintenance (2 h per week for the normal operation, 4 h extra for the monthly maintenance and 8 h twice every year for general maintenance) (Jung et al., 2018). Other costs included were insurance and taxes, supervision and general services.

Revenues can be expected from the operation of the BW decentralized treatment facility. Prices have not been assumed constant for these estimates and the Harmonized Consumer Prices Index (HCPI) has been considered to be 2.20% (average European value of the last 20 years) (European Central Bank, 2021). Internal benefits come from the excess heat that is exported to nearby households as well as from the valorization of biofertilizers in crop cultivation and fees from the wastewater service imposed to users. The prices of nutrients such as nitrogen, phosphorus and potassium (as fertilizer components) can be estimated from the reported values of mineral fertilizers (nitrogen, DAP or diammonium phosphate and potassium chloride) (World Bank Group, 2020).

Operational external costs are calculated as a way to compensate for environmental pollution from the monetization of LCA results, which reports the conversion of the physical environmental impacts into financial ones according to the set of environmental prices provided by De Bruyn et al. (2018) for the ReCiPe midpoint impact categories (Hunkeler et al., 2008). Thus, the environmental costs were estimated translating the LCA results of a gate-to-gate assessment for the BW decentralized treatment for the impact categories analysed. Since the environmental impact contribution of the construction phase has been assumed negligible in the LCA assessment, the externality costs of the construction stage were also disregarded.

Finally, the total costs are represented by the Net Present Value (NPV), as shown in Eq. (1) where n and i are the life span and the discount rate, respectively (Hermelink and Jarger de, 2015).

$$NPV = CAPEX + \sum_{n} \frac{OPEX}{(1+i)^{n}} \qquad (1)$$

3. Results and discussion

3.1. Environmental outcomes

3.1.1. Comparison between the decentralized baseline scenario (DBS) and the centralized scenario (CS)

The comparative results of the life cycle impact assessment of the DBS and CS scenarios are shown in Fig. 2. Based on the outcomes of the analysis, there is a notable difference in the total relative contribution profile between the baseline scenario DBS considering decentralized BW treatment and the one corresponding to a total (BW and GW) centralized treatment. For all categories, the total loads of the hybrid scenario are minor than those of the centralized scenario. The mayor difference in the total environmental profile between both scenarios is given by the FRS category (74.8% of reduction) followed by TET (70.1%), MET (66.8%) and FET (66.7%).

The production of fertilizers to supply the necessary nutrients to energy and food plantations is, together with the emissions derived from their application and the use of machinery (in large-scale crops), one of the greatest environmental impacts of agricultural systems. Parajuli et al. (2017) have demonstrated the large impact on the environment of the use of synthetic fertilizers to produce raw materials that can be used in biorefineries. Ghasempour and Ahmadi (2018) and Romeiko (2019) have also reached the same conclusion for food crops of corn, soybean and wheat. Unlike these studies, in this manuscript the nutrients obtained for a small-scale agricultural system located in an urban area come mainly from the valorization of wastewater resources, although in both cases the indirect impacts due to the production of fertilizers and transport and direct emissions derivative from their application can be clearly differentiated. Thus, the results of the centralized and hybrid scenario were disaggregated in Fig. 2. The following four processes stand out in the environmental profile: production of N-based mineral fertilizer, the wastewater treatment, diffuse emissions from the application of fertilizers in the garden and the transportation of the compost produced in the centralized facility.

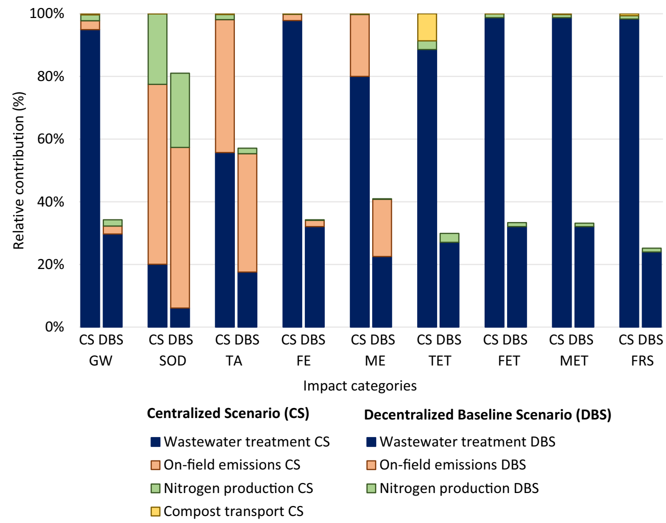

Fig. 2. Distribution of burdens between the contributing processes involved and comparison of the DBS (Decentralized Baseline Scenario) and CS (Centralized Scenario) scenarios. GW: Global Warming; SOD: Stratospheric Ozone Depletion; TA: Terrestrial acidification; FE: Freshwater Eutrophication; ME: Marine Eutrophication; TET: Terrestrial Ecotoxicity; MET: Marine Ecotoxicity; FRS: Fossil Resource Scarcity.

The main difference on the environmental profile of the scenarios is given by the environmental impacts of the biofertilizer production process (wastewater treatment facility). This process represents the highest environmental impact in most categories (80–98%), except for SOD (20.1%) and TA (55.7%). For this reason, the comparison between the DBS and CS scenarios is mainly due to the different characteristics of the wastewater treatment systems. Various studies have demonstrated the reduction of the environmental impact of decentralized wastewater systems when they are compared to centralized facilities (Ishii and Boyer, 2015; Opher and Friedler, 2016; Romeiko, 2020; Xue et al., 2016). Even though, there are others more sceptical about these environmental benefits. Rish et al. (2021) have studied and compared six wastewater management scenarios. They have concluded that centralized systems better manage their emissions (less impact on the ecosystem category) while decentralized ones are best on the category of resources. Besides, the environmental improvement is also dependant on the length of the drain connections. Igos et al. (2012) obtained inconclusive results when both types of facilities were compared considering the removal of pharmaceutical compounds. The comparison between both types of systems is complex since it depends on the technologies used, the segregation at source of the wastewater, the specificity of the local conditions and the objective of the study. Differences on the environmental impact have also been detected between different types of small-scale facilities (Lourenço and Nunes, 2021; Machado et al., 2007; Lopsik, 2013; Garfí et al., 2017; Gallagher and Gill, 2021).

The production of mineral nitrogen fertilizer in DBS is an environmental hotspot in the category of SOD (29%) although also affects to other categories such as GW (5.8%), TET (9.6%), FET (3.8%), MET (3.4%) and FRS (4.7%) due to the background processes involved in its production which entail emissions of N_2O, contributing to GW and SOD and metals (Cu and V among others) accounting for ecotoxicity. The transportation of the compost produced in the centralized facility is only relevant in the TET (8.7%, heavy metal emissions from fuel) in the CS scenario being below 0.6% in the remaining impact categories.

On-field emissions from the application of fertilizers play a key role in SOD (63%, dinitrogen monoxide), TA (66%, ammonia) and ME (45%, nitrate) for DBS. It should be noted that the estimation of these emissions considers the emission factors established by EMEP/EEA (2019), where the factor applied to nitrogen emission in the form of ammonia is double in the case of biofertilizer application instead of mineral-based fertilizers. However, similar mass quantity of mineral fertilizers was used in both scenarios (7.78 g $N \cdot m^{-2}$ for DBS and 7.37 g $N \cdot m^{-2}$ for CS) and thus differences between scenarios on the environmental profile related to on-field emissions (also to mineral nitrogen production) were not significant.

3.1.2. Environmental performance of the wastewater treatment in subsystem DBS-SS1

The wastewater treatment of the hybrid scheme (DBS-SS1), where the biofertilizers are produced, can be designated as the stage with the highest contribution in seven impact categories: GW, FE, ME, TET, FET, MET and FRS. The relative characterization results corresponding to the subsystem SS1 (focusing on wastewater management) of scenario DBS are shown in Fig. 3. Considering the profile shown in this figure, three processes involved in the system play key roles according to the impact categories analysed: the GW treatment, the tap water supply, the energy consumed in the vacuum toilets. The grey water treatment in a centralized plant is behind the worst results, which is attributed to the energy demand for aeration and the consumption of sodium aluminate in the secondary treatment. The impact is as low as 43% (SOD) and as high as 97% (ME). The tap water supply to the households is also relevant with shares from 2% (ME) to 48% (SOD). Thus, the implementation of ultra-low vacuum toilets for the collection of faces and urines reduces the environmental impact related to water consumption. However, the ultra-low vacuum toilets and sewer system and the pump

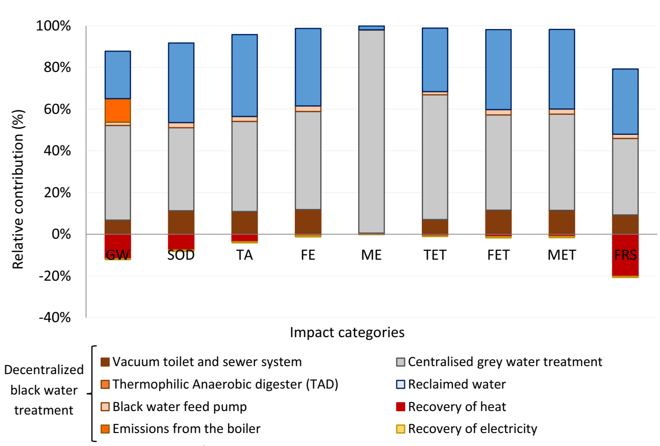

Fig. 3. Environmental profile of the hybrid wastewater treatment process included in SS1 for the DBS (Decentralized Baseline Scenario) scenario. GW: Global Warming; SOD: Stratospheric Ozone Depletion; TA: Terrestrial acidification; FE: Freshwater Eutrophication; ME: Marine Eutrophication; TET: Terrestrial Ecotoxicity; MET: Marine Ecotoxicity; FRS: Fossil Resource Scarcity.

located in the influent in the TAD are the only equipment with environmental repercussions in the decentralized BW treatment. The water reduction in the toilets should be accompanied by an electricity reduction. As illustrated in Fig. 3, this energy consumption is relevant in all impact categories but marine eutrophication with an upper and lower range values of 8–12% for the remaining categories. Moreover, the CO_2 gaseous emissions from burning the biogas in the boiler are also significant in GW (13%).

3.1.3. Energy for wastewater treatment in subsystems SS1- DBS and SS1-CS

Table 3 disaggregates the energy consumption from the cradle and per functional unit of the subsystem SS1 (production of mineral nitrogen fertilizer not included). Extraction and treatment of the water, the tap water and wastewater pumping, and the wastewater treatment were the stages analysed. Considering the drinking water supply, the energy consumption associated with drinking water purification is estimated to be around 0.42 $kWh \cdot m^{-3}$, a value in line with the results reported by Wakeel et al. (2016) (0.01–0.44 $kWh \cdot m^{-3}$), Majid et al. (2020) (0.46–0.92 $kWh \cdot m^{-3}$), Porse et al. (2020) (0.32–0.65 $kWh \cdot m^{-3}$), Yoon et al. (2018) (0.17–0.96 $kWh \cdot m^{-3}$) and Smith and Liu (2017) (0.29 $kWh \cdot m^{-3}$) and Macharia et al. (2021) (0.2–4.07 $kWh \cdot m^{-3}$).

Tap water is transported through a 40 km network pipeline to the dwellings. The energy consumption for this activity (independent of wastewater treatment) is $5 \cdot 10^{-3}$ $kWh \cdot m^{-3} \cdot km^{-1}$ (Plappally and Lienhard, 2012). The differences in water demand for the ultra-low vacuum and conventional toilets: 0.3 $m^3 \cdot d^{-1}$ and 4.17 $m^3 \cdot d^{-1}$, imply different electricity consumption for the BW pumping for the same specific energy and distance ($5 \cdot 10^{-3}$ $kWh \cdot m^{-3} \cdot km^{-1}$ and 40 km). Gu et al. (2017) reported values of 0.02–0.37 $kWh \cdot m^{-3}$ for wastewater collection and pumping. The consumption of energy to pump GW can be similarly estimated.

Despite the high energy consumption of the vacuum toilets in the BW treatment of DBS, the net balance is negative (self-sustainable) which reduces considerably the global energy balance of the hybrid system. The use of vacuum toilets with a stream segregation at source and decentralization prevents the phenomena of dilution, which translates into better energy recovery (Capodaglio, 2017). Then, this wastewater system can be classified as high-energy production process due to the generation of thermal energy from biogas during the treatment of concentrated black water by anaerobic digestion. In a large-scale digestion process at room temperature, energy requirements are in the range of 0.05–0.1 $kWh \cdot m^{-3}$ due to pumping needs (de Mes et al., 2003). However, small facilities are affected by the diseconomies of scale in the energy consumption and production. Thus, in the BW treatment of the DBS scenario the energy demand per pumping reaches 2.96 $kWh \cdot m^{-3}$ per BW influent. The energetic effects according to the size of the facilities have been addressed in multiple studies. For example, Vaccari et al. (2018) have investigated four types of Italian facilities according to their size to identify their potential for energy savings. The energy needs of small plants (<2000 population equivalent) was around of 0.21–1.77 $kWh \cdot m^{-3}$ while in larger installations (>100,000 population equivalent) this demand reduced to 0.14–0.71 $kWh \cdot m^{-3}$. Christoforidou et al. (2020) have reported similar results with values above 245 $kWh \cdot person^{-1} \cdot year^{-1}$ for small plants and values between 22 and 95 kWh $kWh \cdot person^{-1} \cdot year^{-1}$ for large facilities. Many other studies have shown this increase in the ratio between the size of water treatment plants and energy consumption (Trapote et al., 2014; Gu et al., 2017).

The consumption of the thermal energy necessary to increase the temperature of the influent up to 55 °C in the anaerobic digestion is offset by the energy produced from the revalorization of the biogas. Despite the energy disadvantage, the increase in temperature contributes to the removal of pathogens. The concern about the risks of applying biofertilizers in agricultural systems (not only environmental but also

Table 3
Summary of the energy balance per type of facility express as $kWh/m^2 \cdot y$.

		CS[a,d]	DBS (BW-GW)[d]	DBS (BW)[c]
Energy consumption	Extraction and treatment of water[e]	1.88	0.53	0.01
	Pumping of tap water	0.90	0.26	$4.08 \cdot 10^{-3}$
	Wastewater transportation to treatment	0.90	0.41	0.27
	Wastewater treatment electricity demand	1.66	0.53	0.06
	Wastewater treatment heat demand	–	–	–
Net energy production	Electricity production (WWTP/DWTP)	−0.06[b]	–	–
	Heat production (WWTP/DWTP)	–	−0.77	−0.77
	Net balance	5.29	0.96	−0.42

[a] The biological treatment of the facilities is the typical aerobic activated sludge.
[b] Energy recovered through the anaerobic digestion of the sludge.
[c] Only black water is treated.
[d] Treatment with black water and grey water.
[e] Energy from extraction and treatment only includes pumping and direct energy in the water treatment. Energy from chemical production is excluded.

human health) including micro-pollutants (both microbial and heavy metals and antibiotic compounds) has led to the appearance of new standards for their regulation. Several studies have analysed in depth the new legislative aspects that affect both solid and liquid organic fertilizers. Some of them are Collivignarelli et al. (2019), Reuland et al. (2021), Rosemarin et al. (2020) and Lavnić et al. (2017).

The wastewater pumping has higher energy demand for the hybrid system than the centralized system per habitant equivalent (25.88 and 19.47 $kWh \cdot person^{-1} \cdot year^{-1}$) for a distance of 40 km of centralized sewer network. However, the installation of the vacuum toilets has proven to reduce this energy demand for higher length of the sewerage infrastructure. Jóźwiakowski et al. (2018) has reported a 71.62 km sewage system and 114.6 km water supply system for Zamość (Poland).

Table 3 shows how per square meter of land fertigated with biofertilizers this energy consumption is lower for the hybrid system. If expressed per habitant equivalent, the hybrid (from cradle to gate of the facility) also has a better environmental profile. Around 79.57 and 75.75 $kWh \cdot person^{-1} \cdot year^{-1}$ are needed for the CS and DBS systems, respectively. The segregation of the wastewater from source to obtain a concentrated BW stream to recover energy is a good strategy to decrease the negative effects on the environment related to the energy consumption.

As shown in Table 3, the hybrid system has lower energy requirements for pumping and treatment of the tap water. The energy consumed was 60.11 and 49.86 $kWh \cdot person^{-1} \cdot year^{-1}$ for each of them (CS and DBS, respectively). The reason is related to the lower tap water requirements of the toilet which implies a lower energy demand for treatment and pumping to the houses. In both cases the difference in the energy demand is around 17%. This deviation is higher (72%) when the results are expressed per functional unit (square meter of land fertigated). Considering only extraction, treatment and pumping of the tap water the energy consumed was 3.69 and 1.20 $kWh \cdot m^{-2} \cdot year^{-1}$ for scenarios CS and DBS. The disparity in results is related to the amount of nutrients recovered from the treatment processes and used in the land. The hybrid system is more efficient recovering nutrients from the wastewater and thus more land can be fertilized.

3.1.4. Environmental performance of scenarios DRS, DSS and DJS

The reduction of the environmental impact goes through a sustainable design of the treatment facilities with a decrease on the consumption of materials and energy or with an exchange of the resources for other more environmentally friendly (Machado et al., 2007). LCA studies have revealed that the fossil based-energy consumption is the main contributor to environmental pollution during wastewater treatment and not only for the decentralized processes (Takeshita et al., 2020; Szulc et al., 2021). A reduction in energy consumption can be achieved using internal resources (energy from biogas), using more efficient equipment or through proper system operation.

The difference between scenarios DBS and DRS is the electricity consumption data of vacuum toilets. The possibility of changing the toilet for a more energy efficient one is being evaluated. A reduction close to 42% of electricity consumption has been assumed in Scenario DRS considering a bibliographic data as detailed in Section 2.1. A European average profile has been considered for the estimation of the environmental loads associated with the production of the required electricity. This assumption implies a noticeable improvement in categories such as GW (−3.6%), TA (−1.7%), FE (−5.2%), TET (−3.0%), FET (−5.3%), MET (−5.3%) and FRS (−7.0%). The relative profile associated with this scenario is depicted in Fig. 4. The burdens of the hybrid treatment scheme integrated with the urban farming concept are lower due to the reduction in electricity demand in the toilets. The GW impact category has experienced a 4.15% of reduction if related only to the wastewater treatment and not to the recovery of nutrients in the crops. Zawartka et al. (2020) have obtained similar lowering of GW emissions (3.65%)

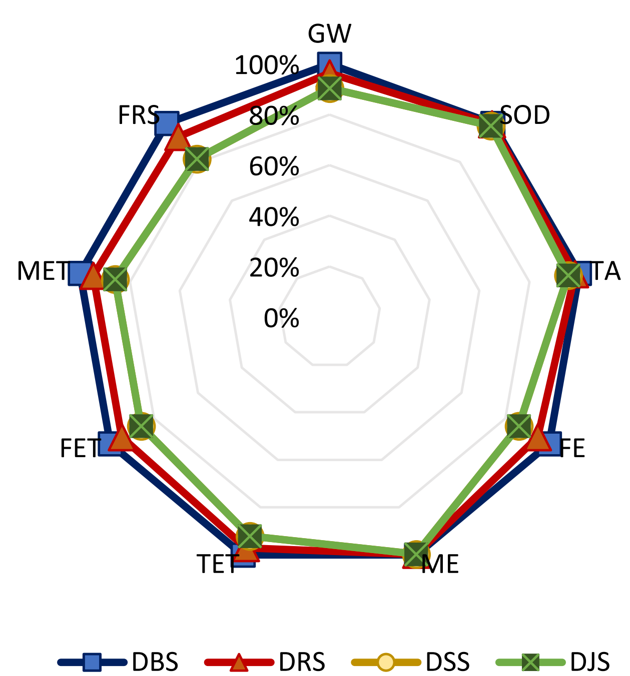

Fig. 4. Comparative relative profile of the scenarios with reduction in energy demand (DRS), with renewable energy (DSS) or with joint implementation of energy reduction and renewable source (DJS) with the baseline scenario (DBS). GW: Global Warming; SOD: Stratospheric Ozone Depletion; TA: Terrestrial acidification; FE: Freshwater Eutrophication; ME: Marine Eutrophication; TET: Terrestrial Ecotoxicity; MET: Marine Ecotoxicity; FRS: Fossil Resource Scarcity.

when they have increase (around a 10%) the use of biogas to produce electricity in a centralized facility.

On the other side, renewable energy can be used instead of fossil fuels, contributing to the decarbonization of treatment plants. Some possibilities are hydroelectric energy, photovoltaics, or the generation of electricity from the incineration of sludge. The energy source to supply the electricity required by the vacuum toilets and feed pump to the TAD in the scenario DSS is solar energy. Muñoz et al. (2006), Guo et al. (2019) and Drouiche et al. (2013) are some examples of solar energy integration in wastewater treatment plants. The change in supply proposed for the DSS scenario leads to remarkable environmental benefits in all categories (except in ME and SOD). The largest environmental improvement in DSS occurs in FRS (around 18.5%) due to the non-dependence on fossil resources to produce the electricity required by the toilets (to supply tap water and vacuum BW). The quantification of the environmental impact decrease by taking both measures has been carried out with the DJS scenario. The improvement for each category was: GW (−9.5%), TA (−4.5%), FE (−13.9%), TET (−7.9%), FET (−14.1%), MET (−14.0%) and FRS (−18.5%). The impacts of the scenario DJS are quite like to those of scenario DSS. Once the impact has been reduced considerably with the measures taken in scenarios DRS and DSS, changes in the energy in the vacuum toilets result in negligible impacts compared to the other processes of the system.

3.1.5. Comparison between scenarios and sensitivity analysis

The scenarios DBS, DRS, DSS and DJS proposed for analysis incorporate the use of low-vacuum toilets in the neighbourhood, while managing the wastewater in a hybrid system to satisfy both wastewater treatment and the production of the necessary nutrients.

In this sense, an environmental comparison is made between the centralized system CS and the novel ones based on an endpoint method to have a single environmental score per system (in millipoints -mPt) and facilitate the message to stakeholders. The result of the comparison is depicted in Fig. 5, which is intended to represent the magnitude of the overall impact. The corresponding shares of each endpoint category and each process involved are also represented on Fig. 5a and b per scenario. The single score for the centralized scenario (79.75 mPt) is considerably higher than that of the innovative systems. The estimated score for scenario DBS, the novel scenario that has just been established in a neighbourhood, is 22.26 mPt. This score is 1.03, 1.10 and 1.10 times higher than those of scenarios DRS, DSS and DJS respectively, which have been designed on the basis of scenario DBS but reducing electricity demand by vacuum toilets, considering solar panels to meet electricity needs or taken into account both measures. For this reason, this strategy could support the installation of solar panels on building terraces and vacuum toilets with lower energy demand. Thus, the four proposed innovative scenarios can be considered potential substitutes for the centralized system according to the environmental profile.

Based on the results of the DRS scenario and due to the large difference of electricity demand and water consumption per flush identified in the literature for the vacuum toilets, a sensitivity analysis has been performed to identify the influence of these parameters. For comparison, the electricity demand was taken directly from the European grid in all scenarios, which is the current situation in the pilot neighbourhood. Fig. 6 shows the comparative profiles between the four alternative scenarios proposed for this sensitivity analysis as described in Section 2.1.

Considering these profiles, only scenarios L (Lower energy demand) and W (different water consumption) are competitive with the established novel scenario (scenario DBS, Fig. 5) although only scenario W enhances the environmental profile when compared to DRS. This scenario results in 5.6% and 2.2% reduction in the environmental score in relationship to DBS and DRS. Although this scenario requires twice the water consumption, it is energy optimized implying an 88% reduction in the electrical demand in the toilets. Scenarios DSS and DJS should be the best option. Therefore, Scenario DBS should be further investigated to make it more environmentally friendly, focusing on optimizing the electricity consumption of the pneumatic system.

On the contrary, the other alternative scenarios (H, M and L) derive higher environmental scores compared to scenario DRS. The demand for more water and electricity is behind these worse profiles. Nevertheless, in all scenarios, improvements are achieved compared to the

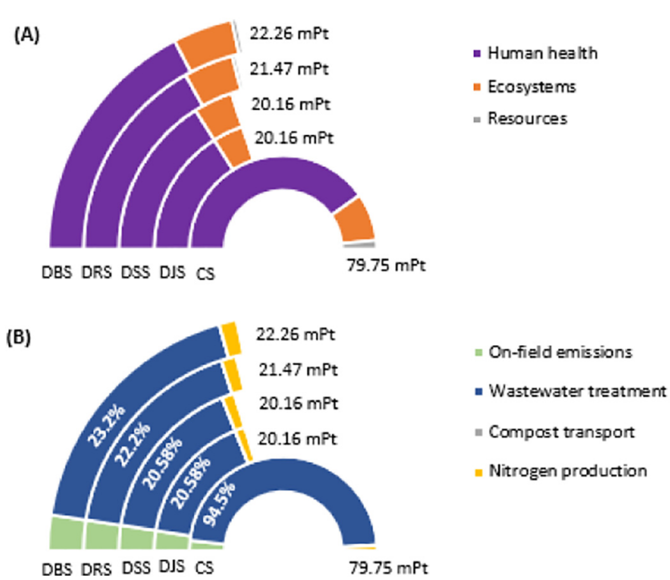

Fig. 5. Comparison of the magnitude of the overall impact (single score) of the different hybrid scenarios and the centralized scenario. (a) Relative contribution per endpoint impact category and (b) relative contribution per process involved. CS: Centralized Scenario; DBS: Decentralized Baseline Scenario; DRS: Decentralized with energy Reduction Scenario; DSS: Decentralized with Solar energy Supply Scenario.

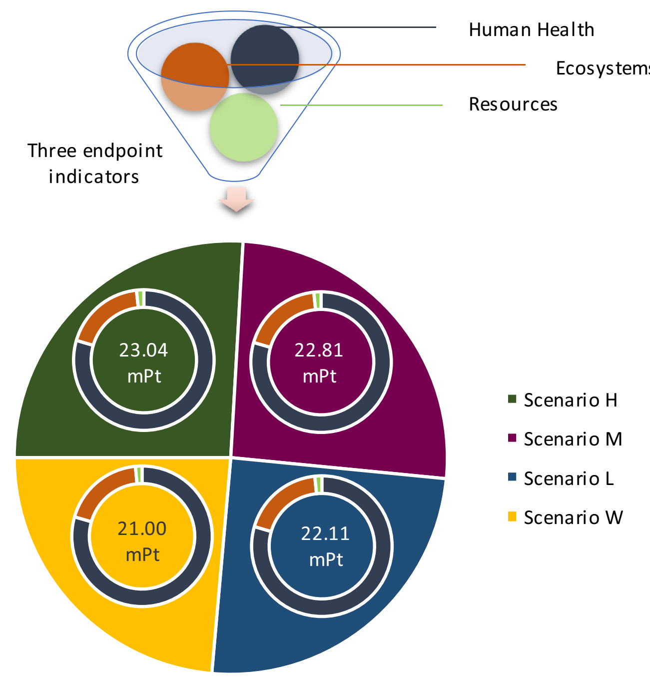

Fig. 6. Comparative single score profiles for the DBS (Decentralized Baseline Scenario) scenario with the implementation of alternative vacuum toilets (data from literature) in four differentiated scenarios (H, M, L and W). Scenario H: 25 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1}; Scenario M: 22.5 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1}; Scenario L: 15 kWh·person^{-1}·year^{-1} and 1.40 L·flush^{-1}; Scenario W: 4 kWh·person^{-1}·year^{-1} and 1 L·flush^{-1}.

conventional system, which supports the interest in promoting the installation of this type of toilet combined with a decentralized system and BW segregation to simultaneously treat biowaste and produce biofertilizers.

3.2. Economic results

The results obtained by the NPV (Net Present Value) for the BW decentralized facility included in the DBS scenario shows a lack of financial self-sufficiency in wastewater management. The total NPV for a discount rate of 5% is 334,304.35€. If expressed per population equivalent the annual average price result is 255.85 €·person^{-1} (replacement costs included). The investment for the operation and construction cannot be recovered when considering the current tariffs imposed on habitants for the service of wastewater treatment (without considering that the imbalance is covered by public subsidies) as an income. However, Moral Pajares et al. (2019), Reynaud (2016), Valero et al. (2018) and other studies conducted for wastewater treatment plants have demonstrated that the treatment fees currently applied do not guarantee sufficient benefits to support the investment in this type of facilities. For example, Acampa et al. (2019) have calculated for different low-medium size wastewater treatment plants the total parametric capital cost, which is in the range of 56.26–95.73 €·person^{-1} for 5000–4500 inhabitants. Zessner et al. (2010) concluded that the annual average price in terms of wastewater treatment was at least 90 €·person^{-1}. Maurer et al. (2005) have estimated and equivalent investment in the range of 272–2179 US$·person^{-1} for decentralized wastewater treatment plants which mainly depend on the sewer system solutions. The equation of the parametric cost curve for wastewater treatment plant of Acampa et al. (2019) would provide a construction cost of 246.10 €·person^{-1} for the same number of inhabitants of the BW facility. This cost is higher than the one estimated for the decentralized facility: 196.88 €·population^{-1} (construction and operational costs) or 36.54 €·population^{-1} (only construction costs). The assumptions made for the first year of operation regarding the total construction, fixed and variable costs were summarized in Table 4.

The operational variable costs are related to the pumping of the BW to the TAD. Approximately 0.37 €·m^{-3} of BW will be needed for such purpose. On the other hand, the fixed costs are 124.09 €·m^{-3} and 55.52 €·m^{-3} of BW (45%) are related to labour costs. Although a decentralized wastewater treatment plant requires part-time staff for occasional maintenance, the labour costs cannot be considered negligible for this scenario (Jung et al., 2018). Molinos-Senante et al. (2010) have estimated a cost of 0.07€·m^{-3} for staff in a centralized wastewater treatment plant, which would give a total cost of 7.80 €·year^{-1} this facility instead of the 6080 €·year^{-1} estimated. The solid and liquid fertilizers, biogas heat yield and wastewater treatment service would provide a profit of 6.08 €·m^{-3} of BW influent in the first year of operation but due to inflation (a rate of 2.20%) this benefit could reach 11.67 €·m^{-3} after 30 years. On the other hand, it is assumed that the costs for a user of an urban garden are those of fertilizers: 0.012 €·m^{-2} for biofertilizers and mineral fertilizers in DBS (average cost).

Table 4
Inventory data for the construction, maintenance and operational costs.

Economic item	Unit	Value	Source
Black water storage tank	€	1000.0	The National Tank Outlet (2021)
Pump	€	5000.0	Run4Life project
TAD	€	28,900.0	Run4Life project
Boiler	€	1576.9	Sandvall et al. (2017)
Costs of the hydraulic and electrical connections	€	5471.5	Acampa et al. (2019)
General costs of the construction company	€	6921.5	Acampa et al. (2019)
Profits of the construction company	€	4614.3	Acampa et al. (2019)
Construction costs	€	57,679.1	–
Working capital	€	2342.3	–
Staff	€·year^{-1}	6080.0	Run4Life project
Maintenance	€·year^{-1}	2097.4	Humphreys (2005)
Insurances and taxes	€·year^{-1}	419.5	Humphreys (2005)
Supervision	€·year^{-1}	912.0	Humphreys (2005)
General services	€·year^{-1}	4544.7	Humphreys (2005)
Fixed costs	€·year^{-1}	13,588.5	–
TAD	€·year^{-1}	0.0	–
Pump	€·year^{-1}	40.6	Eurostat (2021)
CO_2 emissions	€·year^{-1}	59.6	Delf (2010)
Variable costs	€·year^{-1}	100.1	–
Benefits	€·year^{-1}	665.4	–
NPV (Net Present Value)	€	334,304.4	–

In agreement to the Environmental Life Cycle Costing (ELCC) methodology, environmental costs were also included for the estimation of the NPV although only direct external costs (currently considered within the polluter pays principle) were internalized. Thus, only the impact of the CO_2 emissions was included in the budget, as already done by Roh et al. (2018). In the BW decentralized treatment, the direct external costs are associated to the carbon dioxide emissions in the boiler (0.54 €·m^{-3}) while indirect external costs are from the impacts of the production of electricity for pumping and avoided flows. Regarding to the other external costs, the most important category is TET with a relative contribution of 98.5% compared to other categories. The final environmental costs are $8.2 \cdot 10^{-2}$ €·m^{-2} of land fertigated.

4. Conclusion

This study analyses the environmental profile of an emerging hybrid wastewater treatment scenario that is integrated within the concept of urban farming for resource recovery. The BW decentralized system included in the hybrid scenario will be study following the LCA methodology. Different farming-wastewater treatment scenarios have been designed and compared with another scenario with the same characteristics of the crop land but with a centralized system for wastewater treatment. The environmental results show that the emerging scenarios in which energy and nutrients are recovered from black water, despite demanding energy in the toilets, result into an improvement for the environment. The incorporation of the avoided heat within the system boundaries due to the extra heat produced using biogas in a boiler implies environmental credits that offset the environmental burdens derived from other steps included in the decentralized system. As for the economic evaluation, the NPV of the BW decentralized treatment facility was 255.85 €·person^{-1}, which is in line with the results obtained by other studies but also reflects how the treatment fees currently applied do not guarantee sufficient benefits to support the investment. The proposed configurations are appropriate for including wastewater treatment schemes within the circular economy, as they are flexible systems that allow resource recovery close to the point of generation. However, further research is needed, mainly concerning the electricity demand by vacuum toilets and in the centralized treatment technologies, as it is the bottleneck of the environmental profile.

CRediT authorship contribution statement

Sofía Estévez: Methodology, Formal analysis, Investigation, Writing – original draft, Visualization. **Sara González-García:** Methodology, Formal analysis, Writing – original draft. **Gumersindo Feijoo:** Validation, Writing – review & editing, Supervision. **María Teresa Moreira:** Conceptualization, Validation, Writing – review & editing, Supervision.

Declaration of competing interest

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

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