# A novel water-food-energy framework for a comprehensive assessment of resource recovery from wastewater treatment plants

Bhambhani, Anurag; Jovanovic, Oriana; Kjerstadius, Hamse; Trapani, Daniele Di; Mannina, Giorgio; Peter van der Hoek, Jan; Kapelan, Zoran

**DOI**
10.1016/j.jclepro.2025.144716

**Publication date**
2025

**Document Version**
Final published version

**Published in**
Journal of Cleaner Production

Journal of Cleaner Production 489 (2025) 144716

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

# A novel water-food-energy framework for a comprehensive assessment of resource recovery from wastewater treatment plants

Anurag Bhambhani [a,*], Oriana Jovanovic [a], Hamse Kjerstadius [c], Daniele Di Trapani [d], Giorgio Mannina [d], Jan Peter van der Hoek [a,b], Zoran Kapelan [a]

[a] *Delft University of Technology, Faculty of Civil Engineering and Geosciences, Stevinweg 1, 2628 CN, Delft, the Netherlands*
[b] *Waternet, Research & Innovation, Korte Ouderkerkerdijk 7, 1096, AC, Amsterdam, the Netherlands*
[c] *Nordvästra Skånes Vatten och Avlopp AB (NSVA), Rönnowsgatan 12, 252 25, Helsingborg, Sweden*
[d] *Engineering Department, Palermo University, Viale delle Scienze, Ed. 8, 90128, Palermo, Italy*

ARTICLE INFO

Handling Editor: Mingzhou Jin

*Keywords:*
Water-food-energy nexus
Resource recovery
source separation
Decentralization
wastewater treatment

ABSTRACT

There is a trend towards decentralized source separation (DSS) for wastewater treatment and resource recovery. An assessment framework is required to assess whether implementing a DSS treatment over a conventional centralized one is advantageous. This framework needs to account for the performance of the wastewater treatment plant (WWTP) and the effect that resource recovery has on closely-linked sectors such as food and energy production. A framework is lacking that covers the economic dimension, the circularity, the nature reciprocity of resource recovery and that can be applied to real-life cases. A novel WFE framework has been developed here to compare a conventional centralized and a DSS-based WWTP. This novel WFE framework contains assessment methods that are reproducible, and applicable to real-life cases. It also accounts for the local climatic conditions that determine irrigation water requirements. The comparison results revealed that the need to construct new DSS infrastructure leads to a lower economic efficiency of water treatment. Further, chemical-intensive treatment reduces the DSS's material resource circularity and efficiency. Using heat pumps increases the energy use of the DSS WWTP, causing a reduction in water treatment energy efficiency. However, the advantages of DSS show up in the freshwater and nutrient efficiency of food production as well as in the energy self-sufficiency of the WWTP. The novel WFE framework contains indicators specific to water treatment and the food production sectors to improve inter-sectoral communication. Also, including the nature reciprocity assessment can help demonstrate the issue with treated wastewater discharge, especially in arid regions with low stream flows. It can potentially help improve the acceptance of treated wastewater-based reuse. To conclude, the novel framework helps to assess real-life case studies in a more integrated and holistic way. It can help make decisions related to decentralization and source separation by simultaneously considering the water treatment, energy production, and food production sectors.

## 1. Introduction

Resource recovery from wastewater treatment plants (WWTPs) is essential to the circular economy. The recovered resources, such as treated wastewater, nutrients (N and P), and organic matter have to be recycled to close the material loops (Capodaglio, 2017; Mannina et al., 2021). These resources often are to be used in other sectors including food production (FP), energy production (EP), and manufacturing industries (Zarei, 2020). These sectors have their functions and sustainability goals associated with that particular function, which can differ from those of the wastewater treatment (WT) sector. For example, the main function of the FP sector is the production of food crops and its sustainability goals could be minimizing the use of freshwater and industrial fertilizers.

The water treatment sector is closely linked to food and energy production, engendering the concept of the water-food-energy (WFE) nexus (El-Gafy, 2017; Molajou et al., 2021). Since these three sectors have their functions and associated sustainability goals and indicators, assessing the sustainability performance of the entire nexus is a complex undertaking (Albrecht et al., 2018; Dargin et al., 2019). While

* Corresponding author.
*E-mail address:* a.bhambhani@tudelft.nl (A. Bhambhani).

https://doi.org/10.1016/j.jclepro.2025.144716
Received 15 August 2024; Received in revised form 23 December 2024; Accepted 6 January 2025
Available online 8 January 2025

significant work has been done to develop WFE nexus frameworks, a lack of specific and reproducible assessment methods has been pointed out by several authors (Albrecht et al., 2018; Cairns and Krzywoszynska, 2016; Nhamo et al., 2020; Shannak et al., 2018). Furthermore, WFE nexus frameworks lack the consideration of the local geography, climate, and other factors which are consequential for the assessment (Shannak et al., 2018). Therefore, a novel framework is needed to assess the sustainability performance of the WFE nexus.

Wastewater treatment and resource recovery can be achieved through conventional centralized treatment, decentralized source separation (DSS) or their hybrid (Poustie et al., 2015). Conventional centralized treatment enjoys economies of scale because it serves a larger population and usually has a lower net energy consumption, and land use (Besson et al., 2021; Firmansyah et al., 2021; Roefs et al., 2017). DSS refers to a combination of decentralization and source separation implying that the wastewater is separated into different streams at the source and the treatment, reuse, or disposal of all or some of the streams are achieved very close to the point of generation. DSS can offer some advantages from the resource recovery perspective. Due to the lower dilution of organic matter and nutrients, resource recovery could be more efficient from source-separated streams (Pasciucco et al., 2022).

The comparison between the conventional centralized WWTPs and DSS ones has yielded mixed results depending on the sustainability indicators used (Cardoso et al., 2021; Firmansyah et al., 2021; McConville et al., 2017) and little work has been done to compare them from the WFE perspective. Given the above knowledge gaps, this paper aims to develop a novel, WFE-based framework that allows for a more integrated and holistic comparison of conventional centralized and DSS-based solutions by accounting for relevant climatic, geographical and economic factors. The novel framework is meant for the decision-makers in the WT sector but can serve to communicate the potential benefits to the FP sector as well. The novelty of this framework is that it offers an integrated and holistic way to account for the links between the water, food, and energy sectors, allowing for a better comparison between the two approaches.

The paper is organised as follows. The background of the WFE nexus is presented in Section 2, along with an introduction to the concepts of decentralization, source separation (SS), efficiency, circularity, energy self-sufficiency, and nature reciprocity. In Section 3, the novel WFE framework is presented along with the assessment methods. Section 4 contains the description of the two case studies used in this research, and Section 5 contains the results of the comparison between them. A discussion about the WFE framework's application to compare the two cases follows in Section 6. In Section 7, conclusions about the novel WFE framework, the comparison between DSS and conventional treatment approaches, and recommendations are presented.

## 2. Background

### 2.1. Decentralized source separation versus conventional centralized treatment

Although decentralization and source separation are two independent concepts, they are often discussed together as some authors believe that their combination is key to capturing the benefits of both (Guest et al., 2009; Opher and Friedler, 2016; Roefs et al., 2017). In this study, the two concepts are considered together in one system and thus, a conventional centralized WWTP is compared to one with decentralized source separation. The two systems considered in this study are defined as follows. Conventional centralized wastewater treatment collects and treats domestic wastewater and sometimes stormwater from large urban or peri-urban regions utilizing extensive networks of pipes and pumps at a central location and discharge the treated wastewater at a nearby-located discharge point (Larsen, 2019). Decentralized source separation is an approach wherein wastewater is separated into different streams at the source and the treatment, reuse, or disposal of all or some of the streams are achieved very close to the point of generation. This type of treatment is used most commonly for a cluster of homes or isolated settlements (Larsen, 2019).

Domestic wastewater can be separated into black water (BW) and grey water (GW). BW carries roughly 90 % of the N, 77 % of the P and 55 % of the organic matter measured by COD (Roefs et al., 2017). DSS could refer to the separate collection of urine, or to the collection of the BW and GW using different pipes or even the separate collection of urine, BW, and GW, the latter two options being more promising because of more resource recovery opportunities (Besson et al., 2021). Due to the differences in their compositions, while the GW is more suited for treated wastewater reuse because of low pathogen concentrations (Paulo et al., 2013), the BW can be targeted for energy and nutrient recovery (Pasciucco et al., 2022). Fig. 1 demonstrates the basic idea behind conventional centralized and decentralized source separation approaches to wastewater treatment and resource recovery.

DSS can offer some advantages from the resource recovery perspective. Due to the lower dilution of organic matter and nutrients, resource recovery is more efficient from source-separated streams (Pasciucco et al., 2022). For example, Kjerstadius et al. (2017) found a higher nutrient recovery efficiency with a lower carbon footprint resulting from the separate GW and BW collection when compared to a mixed stream collection.

On the other hand, DSS treatment often lacks adequate financial and technical support and suffers from diseconomies of scale, especially for energy use efficiency, and technological and political lock-in (McConville et al., 2017). The lock-in can be attributed to the narrative that a DSS approach needs to be profitable from the start and thereby can preclude the government from funding such a transition (Ampe et al., 2020). Additionally, SS is often considered immature and risky by wastewater experts (Guest et al., 2009), and expensive to monitor (Diaz-Elsayed et al., 2019). Furthermore, high upfront capital costs for retrofitting toilets and installing extra pipelines are barriers to the more widespread adoption of DSS (Diaz-Elsayed et al., 2019). Additionally, due to a lower dilution, the higher concentration of contaminants in the source-separated streams are a risk for sudden point source pollution and a threat to public health (Schoen and Garland, 2017).

Therefore, a DSS-based treatment has both advantages and disadvantages depending on factors such as energy consumption, freshwater sources, and nutrient emissions (Opher and Friedler, 2016). These pros and cons need further study subject to different contexts and system boundaries (Lam et al., 2015; McConville et al., 2017). Further, the advantages of DSS may be more prominent when considering the other sectors where the resources recovered from a WWTP are used. Therefore, a WFE framework can clearly demonstrate and quantify the advantages of a DSS WWTP. Yet, to the extent of the author's knowledge, no WFE framework has been used to compare a conventional centralized and a DSS treatment approach. This knowledge gap will be covered in this paper.

### 2.2. The water-food-energy nexus

The water-food-energy (WFE) nexus refers to the complex links and dependencies between the three resources and the need for cross-sectoral coordination for their utilisation (Smajgl et al., 2016). The need for a cross-sector perspective was felt as numerous interactions exist between the three sectors. Water is required to produce food (e.g., irrigation) and for energy production (e.g., for cooling of power plants, to produce hydroelectricity). Energy is used throughout food production and transportation and for water production and treatment (El-Gafy, 2017).

Most of the discussion around the WFE nexus remains theoretical (Albrecht et al., 2018). Although there is a limited number of frameworks addressing all three sectors (Shannak et al., 2018), an integrated approach to facilitate cross-sectoral coordination is needed (Nhamo

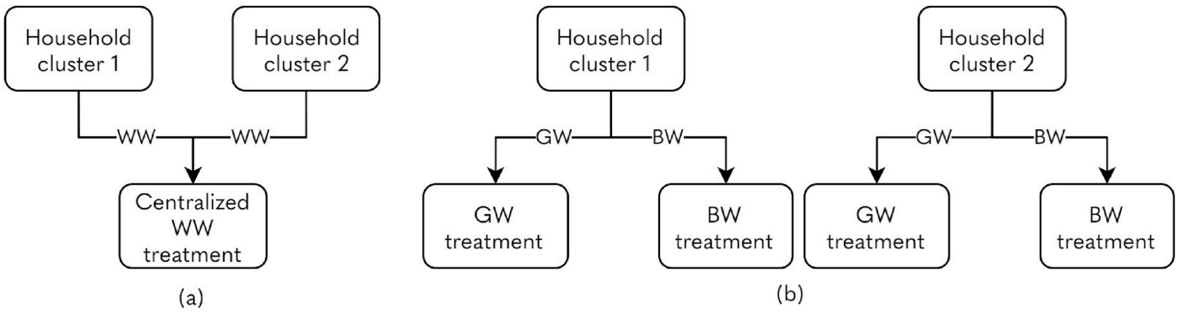


**Fig. 1.** General representations of the (a) conventional centralized and (b) decentralized source separation approaches to wastewater treatment and resource recovery. Whereas, in the conventional centralized approach, the wastewater (WW) from multiple household clusters are treated at a centralized WWTP, in a decentralized source separation approach, grey water (GW) and black water (BW) are separately treated at multiple decentralized WWTPs.

et al., 2020). Fetanat et al., (2021) developed a decision-making WFE framework for energy recovery from WWTPs, but other resources, such as treated wastewater and nutrients, were not covered.

Yi et al. (2020) and Simpson et al. (2022) developed composite indicators to measure and monitor the individual performances and the linkages between the three sectors at a national level and the provincial scales. However, this may not serve the decision-makers to design and monitor the resource recovery solutions at a WWTP scale. Furthermore, these frameworks lack the consideration of circularity indicators. The WFE nexus index method developed by El-Gafy (2017) included indicators to calculate the total water and energy consumption for food production, but it does not account for the circularity and is focused on the FP sector. Thus, the few existing frameworks lack in some aspects.

Moreover, Shannak et al. (2018) pointed to the fact that the economic dimension is not sufficiently included in most WFE frameworks. Also, these lack factors such as the local climate which will dictate the water requirement (e.g., for irrigation) (Shannak et al., 2018). The frameworks have also been criticised for lack of practical applicability (Cairns and Krzywoszynska, 2016), and a lack of analytical tools and reproducible methods to evaluate real-world cases (Albrecht et al., 2018; Nhamo et al., 2020).

Therefore, a new WFE framework with reproducible methods is presented in this paper to compare a conventional centralized and a DSS-based approach to wastewater treatment and resource recovery. The novelty of this framework is that it offers an integrated and a more holistic way to account for the links between the water, food, and energy sectors, allowing for a better comparison. The assessment is centred around a WWTP and is meant for the decision-makers to assess the resource recovery solutions at the WWTP scale. The methods of this framework will allow the decision-makers to evaluate how a particular resource recovery solution can contribute positively to the FP sector subject to the specific water requirements of a particular region. This may also serve to improve the inter-sectoral communication which is often found wanting in the WFE nexuses (Greer et al., 2020). Further, a circularity assessment method will also be used as part of this framework as the current WFE frameworks usually lack circularity indicators.

## 3. Material and methods

### 3.1. WFE assessment framework

The WFE assessment framework developed in this paper is presented in Fig. 2. This framework centres on the water treatment sector while integrating the food and energy production sectors. It uses indicators from all three sectors, with functional units assigned to water treatment and food production. For water treatment, the functional unit is the

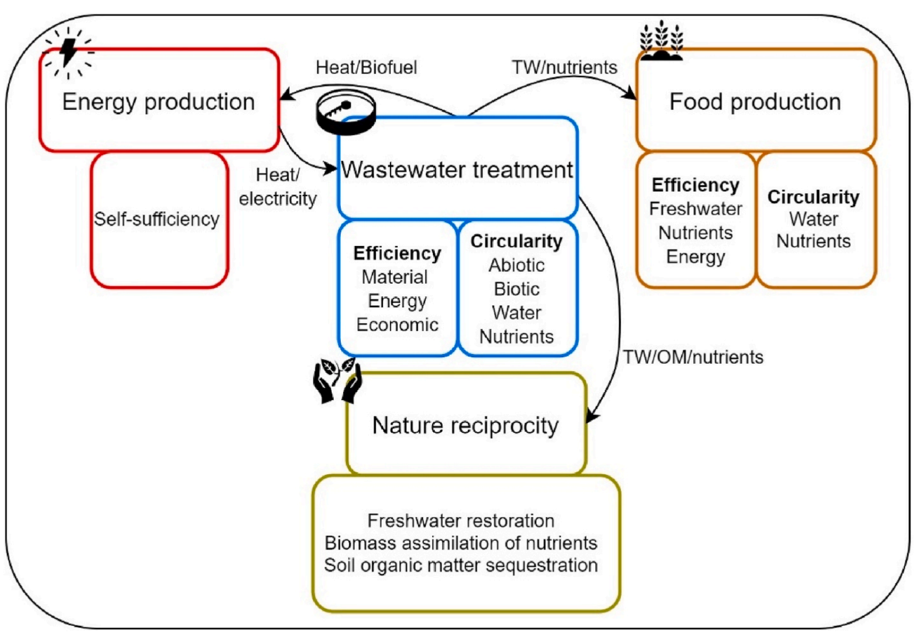


**Fig. 2.** The novel WFE framework showing some common resource exchanges. The figure shows the three sectors i.e., the water treatment, energy production, and food production, along with the their indicators.

treatment of a specific volume of wastewater to meet effluent standards. In food production, it is the irrigation and fertilization of agricultural land. From the energy perspective, the goal is wastewater treatment plant (WWTP) self-sufficiency. Below, the framework's indicators are briefly discussed, as some were detailed in previous publications.

### 3.1.1. Efficiency assessment

Efficiency can be broadly defined as the ratio between useful outputs (benefits) and the inventoried flows (of resources, energy, money, etc.) or environmental impacts. Huysman et al. (2015) defined two levels of efficiency: Level 1 refers to the ratio between benefits and inventoried flows and level 2 is the ratio between the intended effects or benefits and the environmental impact (eco-efficiency). In this paper, level 1 efficiency indicators will be used. This is because this framework is meant for decision-makers who may not have the necessary skills and resources to conduct an LCA. If necessary and in the case that LCA results are available, they can be easily included as denominators in the efficiency formulae. The efficiency indicators will be expressed as the ratio between the functional unit of a process (the useful output) and the inventoried flows (material, energy, and economic investment). To illustrate, the energy efficiency of the water treatment sector will be defined as the ratio between the volume of wastewater treated ($m^3$/y) to the energy consumed (kWh/y). A higher efficiency value is naturally more desirable. For the material efficiency indicators, the ratio between the functional unit of a process and the linear flowing material flows (kg-linear/year) of the process is used. The linear flow is obtained as part of the material circularity indicator calculation, which is explained in depth in Bhambhani et al. (2023).

Efficiency is calculated separately for the water treatment and food production sectors, using their respective functional units as numerators.

**Water Treatment**: The functional unit is the annual volume of wastewater treated ($m^3$/year) per person equivalent (p.e.) to meet local effluent standards.

**Food Production**: The functional unit is defined as the irrigation and fertilization of arable land required to support the same p.e. Here, the arable land area per person, based on The world bank (2024), and the most cultivated EU crop, common wheat (Eurostat, 2024), are used to estimate water and nutrient needs. These requirements can also be calculated for other crops if necessary.

The authors focus on the inventoried flows most relevant to the interaction between the water treatment and food production sectors. For the water treatment sector, these include material, energy, and economic costs (Capex and Opex). For the food production sector, freshwater, nutrients, and energy are considered. Although the centralized WWTP is already operational, the Capex pertains to upgrades aimed at improving resource recovery, detailed in the case study section. Based on the above-mentioned functional units and inventoried flows, the following indicators are used:

**Material Efficiency ($E_{WT\text{-}Mat}$ in $m^3$/kg-linear):** The ratio of the annual wastewater volume treated ($V_{WW}$, $m^3$/y) to the linear mass flow of resources through the WWTP. The linear mass flow is calculated as the product of the mass of resources used ($M_{in}$, kg/y) and a linear flow indicator (1 - $MCI_{mixed}$).

**Energy Efficiency**: The volume of wastewater treated annually per unit of energy input to the WWTP ($E_{in}$, kWh/y).

**Economic Efficiency**: The annual wastewater volume treated divided by the total annualized costs (€/y), including capital and operational expenditures of the WWTP.

For the food production process, efficiencies are calculated as follows:

**Freshwater Efficiency ($E_{FP\text{-}FW}$, $m^2/m^3$-linear)**: The annual irrigated land area ($A_{agri}$, $m^2$/y) divided by the linear flow of irrigation water, expressed as the product of the irrigation water volume ($V_{irr}$, $m^3$/y) and the linear flow indicator (1 - $MCI_{FW}$).

**Nutrient Efficiency ($E_{FP\text{-}Nut}$, $m^2$/kg-linear)**: The annual irrigated land area divided by the linear flow of nutrients, calculated as the product of the nutrient mass used ($M_{Nut}$, kg/y) and the linear flow indicator (1 - $MCI_{Nut}$).

**Energy Efficiency ($E_{FP\text{-}En}$, $m^2$/kWh)**: The annual irrigated land area divided by the energy consumed for irrigation ($E_{irr}$, kWh/y).

These efficiency equations, along with those for circularity, nature reciprocity, and energy self-sufficiency, are summarized in Table 1.

### 3.1.2. Circularity assessment

The circular economy (CE) is an alternative to the current linear economy that aims to recirculate resources within the economic production system to maximise the recovery of value (Corona et al., 2019). Circularity is a measure of the extent to which the CE has been implemented, in other words, the extent to which virgin resource extraction and unrecovered waste generation are avoided (Ellen MacArthur Foundation, 2019). Circularity may be assessed at the level of the economy but here the focus is on the water treatment and food production sectors. In this paper, the modified material circularity indicator (MCI) from Bhambhani et al. (2023) will be used.

Here, the circularity assessment measures the percentage of the total resource flows through the water treatment and the food production processes that are circular and the focus is on resources that are relevant to the WWTP and/or are exchanged between the two sectors. These include mixed resources which refer to any material resource (precipitation chemicals, acids, bases, biochar, etc.) being used in the water treatment process. Further, the circularity values are calculated for

**Table 1**
Indicators used in the novel WFE Framework.

| Water treatment (WT) | Formulae |
|---|---|
| Circularity (mixed, biotic, nutrients, and water) | $V = M(1 - RSIF)$<br>$W = M(1 - RSOF - RGOF)$<br>$LFI = \frac{V+W}{2M}$<br>$MCI = 1 - LFI$ |
| Material efficiency | $E_{WT-Mat}\ (m^3/kg\ linear) = \frac{V_{WW}\ (m^3/y)}{M_{in}(kg) \times (1 - MCI_{mixed})}$ |
| Energy efficiency | $E_{WT-En}(m^3/kWh) = \frac{V_{WW}\ (m^3/y)}{E_{in}\ (kWh/y)}$ |
| Economic efficiency | $E_{WT-Econ}\ (m^3/€) = \frac{V_{WW}\ (m^3/y)}{Capex + Opex\ (€/y)}$ |
| **Food production (FP)** | |
| Circularity (water and nutrients) | $V = M(1 - RSIF)$<br>$W = M(1 - RSOF - RGOF)$<br>$LFI = \frac{V+W}{2M}$<br>$MCI = 1 - LFI$ |
| Freshwater efficiency | $E_{FP-FW}\ (m^2/m^3\ linear) = \frac{A_{agri}\ (m^2/y)}{V_{irr}(m^3/y) \times (1 - MCI_{FW})}$ |
| Nutrient efficiency | $E_{FP-Nut}\ (m^2/kg\ linear) = \frac{A_{agri}\ (m^2/y)}{M_{Nut}(kg/y) \times (1 - MCI_{Nut})}$ |
| Energy efficiency | $E_{FP-En}\ (m^2/kWh) = \frac{A_{agri}\ (m^2/y)}{E_{irr}\ (kWh/y)}$ |
| **Nature reciprocity** | |
| Freshwater restoration | $FR = \sum_{i=1}^{12} (Q_{dis\,i} \times (1 - WPL_i))$ |
| Biomass assimilation of nutrients | $BA = M_{inf} \times NRE \times NUE$ |
| Soil organic matter sequestration | $SS = (1 - VS/100) \times OM_{soil}$ |
| **Energy production (EP)** | |
| Energy self-sufficiency | $ESS = \frac{E_{rec}\ (kWh/y)}{E_{in}(kWh/t)}$ |

**Acronyms**
V = virgin mass; W = unrecovered mass; RSIF = restorative input fraction; RSOF = restorative output fraction; RGOF = regenerative output fraction; M = mass of resources; MCI = material circularity indicator; LFI = linear flow indicator; $Q_{dis\ i}$ = treated wastewater discharge in month i; $WPL_i$ = water pollution level in month i; NRE = nutrient recovery efficiency; NUE = nutrient uptake efficiency; VS = volatile solid component; $OM_{soil}$ = organic matter added to the soil; $E_{rec}$ = Recovered energy; $E_{in}$ = Input energy.

biotic resources, nutrients, and water for the water treatment. For food production, the circularities of freshwater and nutrients are calculated. The MCIs are calculated as follows:

**Virgin input (V in kg/y):** The fraction of resource input that is restorative in nature (RSIF) is subtracted from 1 and multiplied by the total resource input (M in kg/y).

**Unrecovered waste (W in kg/y)**: The fraction of restorative and regenerative output flows (RSOF and RGOF) is subtracted from 1 and multiplied by the total resource input (M in kg/y).

**Linear flow indicator (LFI)**: The sum of W (kg/y) and V (kg/y) is divided by two times the total resource input (M in kg/y).

**Material circularity indicator (MCI)**: The LFI is subtracted from 1 to give the MCI.

The relevant equations are presented in Table 1. To understand the background of the equations, the reader is directed to Bhambhani et al. (2023).

### 3.1.3. Nature reciprocity assessment

Nature reciprocity can be defined as the positive effects on the natural environment through a re-balancing of resource stocks (Bhambhani et al., 2024). The potential as well as duty of human society to actively benefit the natural environment has been ignored in conventional sustainability discourses (Bhambhani et al., 2024). In this framework, this potential will be considered and assessed using the nature reciprocity indicators. Nature reciprocity is here defined as the re-balancing of the stocks of freshwater, N, P, and organic matter that can be achieved using the resources recovered from WWTPs.

The three nature benefits considered here are related to the water, nutrient, and carbon cycles and the three indicators used are freshwater restoration (FR), biomass assimilation of nutrients (BMA), and soil organic matter sequestration (SS). The first indicator represents the quantity of freshwater that a WWTP restores to the natural environment through treated wastewater discharge, considering the quality of the treated wastewater. This is calculated by subtracting the fraction of the stream flow (measured using the water pollution level or WPL) required to dilute the WWTP effluent from the monthly discharge of the WWTP ($Q_{dis}$ in $m^3$/month).

The BMA quantifies the N and P that are cycled back into the natural environment through biomass assimilation. This indicator accounts for the mass of nutrients flowing into a WWTP ($M_{inf}$ in kg/y), the WWTP's nutrient recovery efficiency (NRE in fractions), and the recovered nutrient uptake efficiency (NUE in fractions) of the recovered nutrient products applied to agricultural fields.

Lastly, the SS indicator measures the quantity of organic matter sequestered in the soil. The indicator considers the total organic matter applied to the soil ($OM_{soil}$ in kg/y) and the stability of the organic matter measured using the volatile solids component (VS in %).

The three nature benefits assessed are linked to the water, nutrient, and carbon cycles, measured using freshwater restoration (FR), biomass assimilation of nutrients (BMA), and soil organic matter sequestration (SS).

**Freshwater Restoration (FR):** This measures the quantity of freshwater a WWTP returns to the environment via treated wastewater discharge, adjusted for effluent quality. It is calculated as the WWTP's monthly discharge volume ($Q_{dis}$ in $m^3$/month) minus the stream flow fraction needed to dilute the effluent, determined by its water pollution level (WPL).

**Biomass Assimilation of Nutrients (BMA):** This indicator quantifies the nitrogen (N) and phosphorus (P) cycled back into the environment via biomass. It factors in the WWTP's nutrient recovery efficiency (NRE), the nutrient uptake efficiency (NUE) of recovered products, and the nutrient inflow to the WWTP ($M_{inf}$ in kg/y).

**Soil Organic Matter Sequestration (SS):** This measures the organic matter sequestered in soil, based on the total organic matter applied ($OM_{soil}$ in kg/y) and its stability, evaluated by the volatile solids content (VS%).

These equations are presented in Table 1, and for a detailed explanation of these indicators, the readers are referred to Bhambhani et al. (2024).

### 3.1.4. Energy self-sufficiency assessment

Energy self-sufficiency is here defined as the quantity of energy recovered (kWh/y) from a WWTP (heat + electricity) expressed as a percentage of the energy used (kWh/y) by the WWTP. This is the definition used by several authors when discussing the concept of energy self-sufficiency including Maktabifard et al. (2018), Wang et al. (2016), Yan et al. (2017), among others. This is one way to measure the progress of a WWTP towards sustainability concerning energy use and will be used in the novel framework.

Water treatment and agriculture account for up to 5 % of the total electrical energy demand of some countries (Longo et al., 2016). WWTPs require energy to run several treatment processes and the requirement varies according to factors such as the processes, the location of the WWTP, pollutant loads, environmental standards, and the infrastructure age (Maktabifard et al., 2018). The energy use is expected to grow significantly in the coming years (Yan et al., 2017). However, energy can also be recovered from WWTPs in various ways. Whereas, anaerobic digestion of excess sludge can yield biogas that can be used to produce electricity, heat exchangers and heat pumps may be employed to recover thermal energy. Further, microbial fuel cells can convert the organic energy present in the wastewater directly to electricity (Wang et al., 2016). This has led to the discussion of energy-self-sufficient or energy-neutral WWTPs. This indicator is calculated as follows:

**Energy self-sufficiency (%)**: This is the ratio between the energy recovered (kWh/y) and the energy used by a WWTP (kWh/y) multiplied by 100.

Therefore, 100 % energy self-sufficiency implies that the WWTP can theoretically supply all of its energy requirements. If the WWTP can produce more energy than it requires for its functioning, then the indicator value will be more than 100 % and if no energy is produced by the WWTP, the value will be 0 %. In the latter case, all of the energy requirement of the WWTP has to be externally produced. It is important to note that even if the energy produced at the WWTP is used for a purpose unrelated to the WWTP (e.g., transportation fuel), it is still counted. The indicators for the efficiency, circularity, nature reciprocity and energy self-sufficiency assessment can be found in Table 1.

## 4. Case studies

The semi-hypothetical case studies described here are based on two real-life cases but contain a few assumptions and simplifications which will be pointed out in their descriptions. These have to do with the fact that some features of the real cases are still under planning and may be implemented in the future but have been included in this paper.

### 4.1. Corleone: a centralized conventional WWTP

Corleone is an activated sludge (AS) WWTP treating 3700 $m^3$/d of domestic wastewater and is designed for 12000 p.e. (Mannina et al., 2022). The AS reactor is supplied with an intermittent aeration (IA) system to decrease energy use. Furthermore, an oxic settling anaerobic (OSA) reactor is present to reduce the excess sludge quantity. Some of the wastewater will pass through an ultrafiltration (UF) unit to produce irrigation water (Mannina et al., 2022). A single UF module is currently operational and has a capacity of 25 $m^3$/h thus limiting the quantity of irrigation water that can be supplied. Since this WWTP is a conventional one, the authors here have assumed that its distance from the agricultural fields is 2 km on average. This has been chosen arbitrarily since no data was available and a sensitivity analysis is conducted. It must be noted that the excess sludge is landfilled. In the future, it is expected to be composted and the compost to be used in agriculture. The composting process has been included in this case study. A flowchart of the Corleone case study is shown in Figs. 3 and 4 shows a map and some elements of the Corleone WWTP.

### 4.2. Helsingborg: a DSS WWTP

The Helsingborg case study consists of a DSS WWTP with a capacity of 12000 p.e. Each house has three sewer pipes: one for BW, another for GW, and the last one for food waste collection. For the case study, food waste collection and treatment have been excluded from the system boundary, and only the wastewater flows are considered for a fair comparison with Corleone.

The BW is collected from the vacuum toilets using vacuum sewers and transported directly to an up-flow anaerobic sludge blanket septic tank (UASB-ST) for biogas production. The digestate is subject to struvite precipitation for P recovery and ammonia stripping for the recovery of ammonium sulphate for agriculture. The digested sludge is composted to produce soil (75 %) or applied to fields directly (25 %) (Kjerstadius et al., 2017). The GW is separately collected through a low-pressure sewer and treated in an activated sludge (AS) unit. The excess sludge is directed to the UASB-ST along with the BW. Furthermore, thermal energy is recovered using heat pumps from the GW effluent from the AS unit.

The effluent of the AS unit is treated in a post-precipitation unit before being discharged into the ocean (Kjerstadius et al., 2017). According to the case study owners, the treated wastewater may be used for irrigation of urban farms for food production in the future. Therefore, it has been assumed here that the same quantity of treated wastewater as in the Corleone case study is used for irrigation of these farms. The case study flow chart, a map and a photo are presented in Figs. 5 and 6.

## 5. Results

The detailed calculations of all the indicators are shown in the Supplementary material. The water treatment material efficiency of Corleone (735.8 $m^3$/kg-linear) is much higher compared to that of Helsingborg (1.4 $m^3$/kg-linear). Helsingborg uses resource-intensive treatment processes such as struvite precipitation requiring magnesium, and citric acid and ammonia stripping requiring sodium hydroxide, sulphuric acid, and citric acid. These processes require industrial chemicals that are difficult to recycle leading to a largely linear flow (i.e., a low $MCI_{mixed}$). Thus, the material efficiency equation discussed in Table 1 penalizes for a low circularity of these resources. Corleone, in contrast uses lower quantities of chemicals relying more on producing treated wastewater for irrigation and not on nutrient recovery processes.

The precipitation chemicals such as magnesium chloride (for struvite precipitation) in Helsingborg add to the material use of the WWTP. But, this chemical gets recovered as part of the struvite crystals and can be recycled in agriculture. However, the bigger problems are the chemicals used for pH control, such as NaOH, which are manufactured in industries and are difficult to recycle. NaOH is used here to control the pH of the ammonia stripper and citric acid for cleaning the precipitation equipment. Ye et al. (2020) and Sakthivel et al. (2012) mention a high chemical use intensity of precipitation-based nutrient recovery technology. The industrial chemicals used for pH control and/or cleaning are difficult to recycle and cause a low circularity and consequently, a low material efficiency.

The energy efficiency of the Corleone WWTP (4.6 $m^3$/kWh) is also significantly higher compared to Helsingborg (0.2 $m^3$/kWh). This is because the energy efficiency is calculated by dividing the volume of wastewater treated ($V_{WW}$) by the input energy ($E_{in}$) as explained in the equation of energy efficiency in Table 1. The main reasons for the low energy efficiency of Helsingborg are the energy-intensive heat pumps that recover thermal energy, responsible for 75 % of the total energy use ($E_{in}$). However, they pay off in a high energy self-sufficiency, discussed later. Heat pumps transfer thermal energy from a low-grade source (such as wastewater) to a working fluid and then raise its thermal energy content using mechanical energy (Culha et al., 2015). More energy-efficient designs for heat pumps need to be researched (Chae and Ren, 2016) but the inclusion of heat energy recovery using heat pumps can support energy-positive WWTPs as noted by Barroso Soares (2017) and confirmed by this study.

In terms of economic efficiency, a similar relationship is found where Corleone (6.5 $m^3$/€) performs substantially better than Helsingborg (0.2 $m^3$/€) because of a higher capex for Helsingborg in the economic efficiency formula discussed in Table 1. All costs were normalized using purchasing power parity (PPP). This result was expected as Corleone is an existing WWTP and the only capital costs it incurs are for the repair of the UF unit and the infrastructure required to connect the WWTP to a storage tank for irrigation. Helsingborg was constructed recently and thus involves green-field costs. Yet, this comparison is important for the consideration of constructing decentralized/source separation systems to replace old conventional WWTPs. Since most existing WWTPs are of the conventional centralized type, their associated capital costs only pertain to the repair of the existing infrastructure. In contrast, to construct DSS WWTPs, new infrastructure needs to be installed which adds up to significant costs.

When it comes to food production, Helsingborg performs better overall than Corleone. With regards to the freshwater efficiency, the Helsingborg WWTP (1.7 $m^2/m^3$-linear) is a better option than the Corleone WWTP (0.8 $m^2/m^3$-linear). This implies that nearly twice the land area can be irrigated using a unit linear flow of freshwater in Helsingborg than in Corleone. The freshwater efficiency is calculated by dividing the agricultural land area by the product of the irrigation water demand and the water circularity as shown in Table 1. Corleone's arid climate demands a higher evapotranspiration of a crop than in Helsingborg leading to a higher irrigation water demand (i.e., a high $V_{irr}$). Helsingborg currently does not use any treated wastewater for irrigation with the effluent being discharged into the ocean and Corleone only has a capacity of 25 $m^3$/h to provide treated wastewater for irrigation. Although the circularity ($MCI_{FW}$) of Corleone's agriculture is higher than Helsingborg's the irrigation water required in Corleone is much larger than what this WWTP can currently provide leading to a low freshwater efficiency.

This observation supports the view that treated wastewater reuse for irrigation is much more important in arid and semi-arid regions, such as Corleone, which are prone to water scarcity (Ofori et al., 2021) and have high evapotranspiration needs (Liu et al., 2023). Moreover, most of Corleone's treated wastewater is discharged into the Eleuterio River

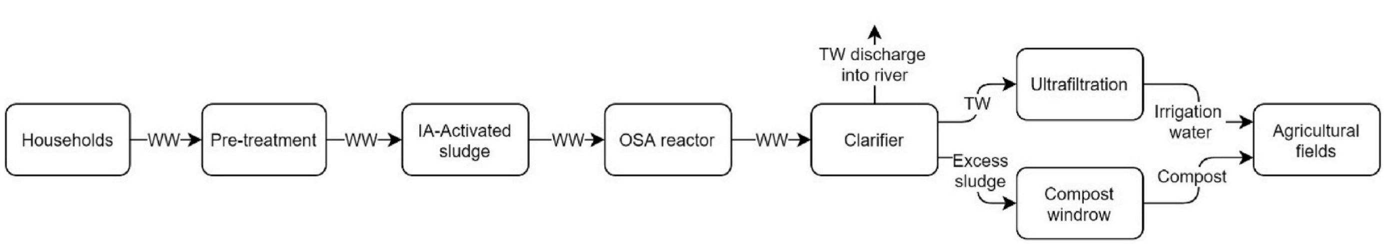


**Fig. 3.** Flowchart depicting the Corleone (conventional) case study. WW-Wastewater, TW-Treated wastewater, IA-intermittent aeration, OSA-Oxic settling anaerobic.

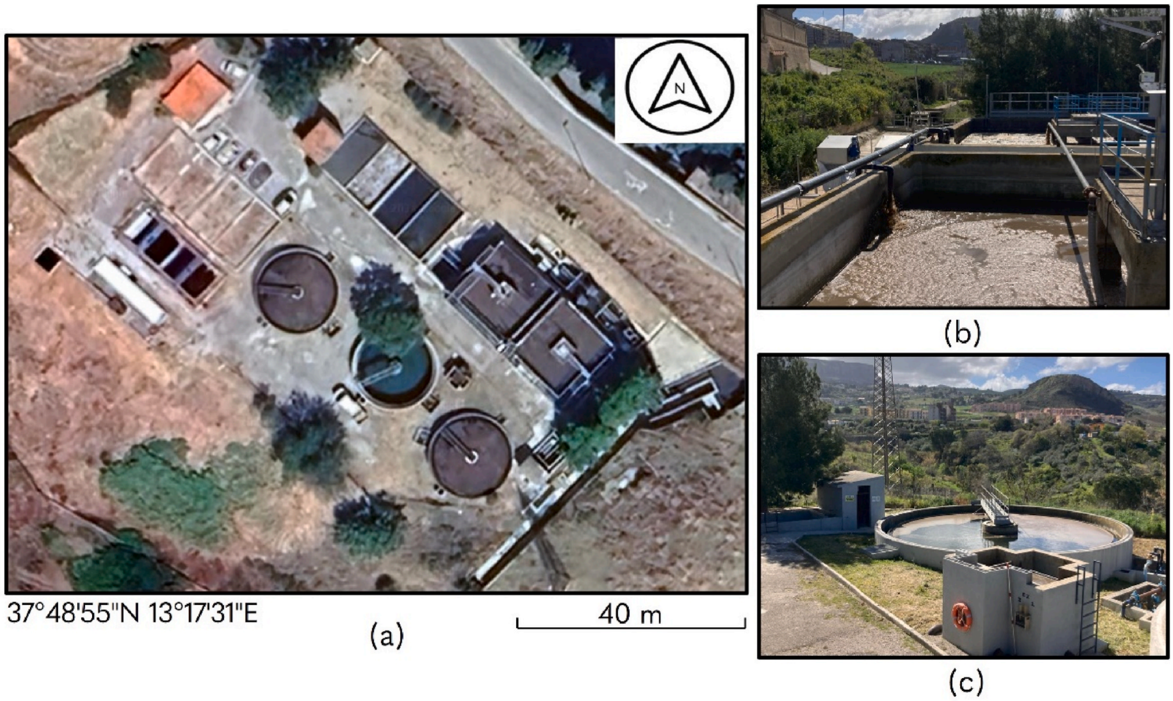


**Fig. 4.** (a) Google Earth. (2024). Corleone wastewater treatment plant: Satellite view. Retrieved December 16, 2024, from https://earth.google.com/), (b) The biological IA reactor, (c) The settling tank.

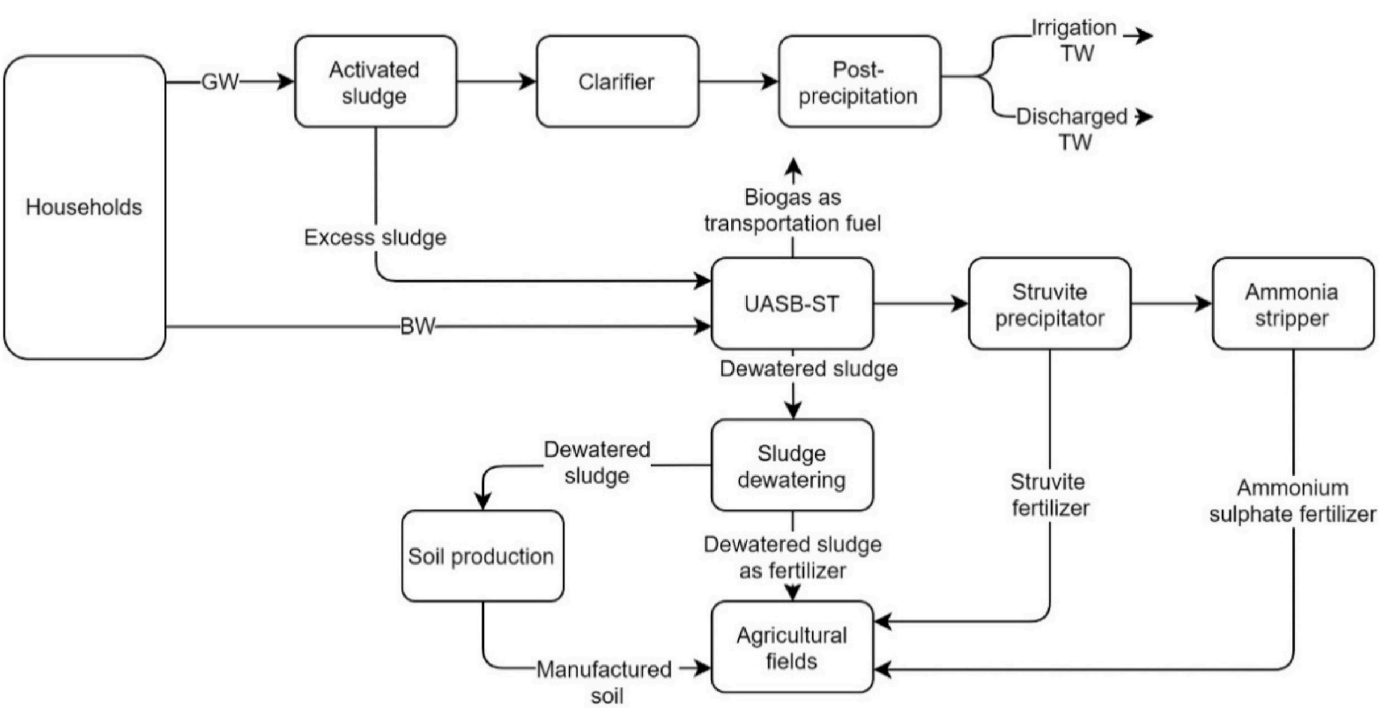


**Fig. 5.** Flowchart depicting the Helsingborg (source separation) case study. GW-Grey water, BW-Black water, TW-Treated wastewater, UASB-ST-Upward anaerobic sludge blanket-septic tank.

which has a relatively small flow rate that is insufficient to dilute the nutrients and organic matter present in the effluent, revealed by the negative freshwater restoration. Therefore, increasing the treated wastewater reuse for irrigation is strongly suggested.

The nutrient efficiency for Helsingborg (80.4 $m^2$/kg-linear) is slightly better than that of Corleone (75.0 $m^2$/kg-linear) but the difference is not substantial. However, this assumed that the nutrients present in the treated wastewater used for irrigation will be taken up by the crops with the same efficiency as the zeolite and ammonium sulphate fertilizer products obtained in Helsingborg. Additionally, the quality of the fertilizer products, in terms of lower heavy metal or organic micropollutants concentrations, are not accounted for by this indicator. For future research, indicators to capture the quality of the recovered products must be developed and included in the nutrient efficiency

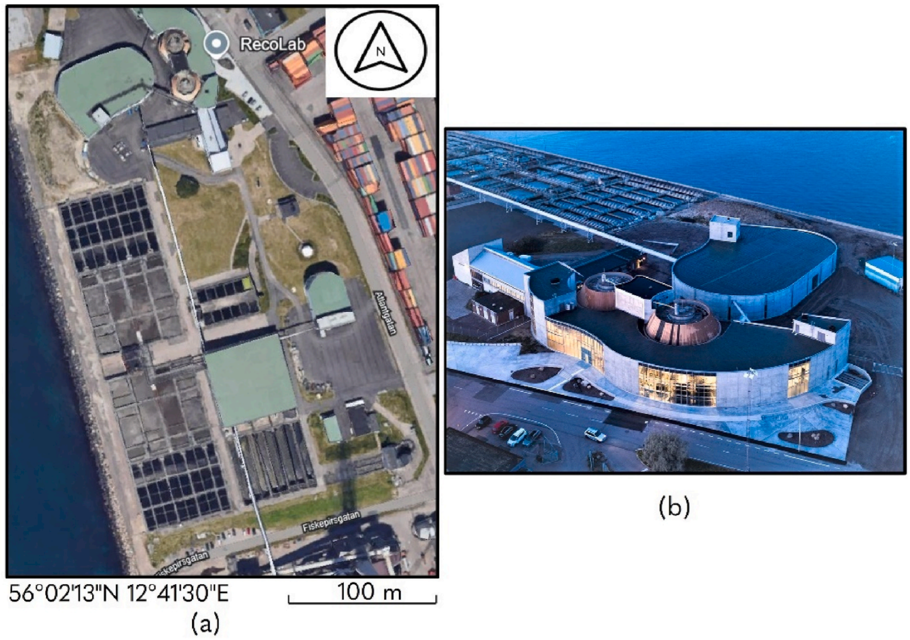


**Fig. 6.** (a) Google Earth. (2024). Recolab, Helsingborg: Satellite view. Retrieved December 16, 2024, from https://earth.google.com/). (b) A photo of the Helsingborg WWTP.

calculations.

The biggest advantage for FP is seen in terms of the energy efficiency of irrigation with Helsingborg (14.1 $m^2$/kWh) performing almost 30 times better than Corleone (0.3 $m^2$/kWh). The energy input ($E_{irr}$), for irrigation, which is the denominator in the equation for energy efficiency as shown in Tables 1 and is much higher in Corleone for two reasons. Firstly, the distance between the WWTP and the point of irrigation is assumed to be 0.1 km in Helsingborg and 2 km in Corleone. Secondly, some irrigation water is assumed to be sourced from underground. The average groundwater depth in Sweden is only 2 m (Barthel et al., 2021) which translates into a much lower pumping energy use when compared to an average of 25 m in Sicily (Morici et al., 2023). Therefore, since much higher energy is required to irrigated a unit area of land in Corleone, its FP energy efficiency is low.

The FP energy efficiency can significantly benefit from the close-distance reuse of treated wastewater. DSS does help to reduce the

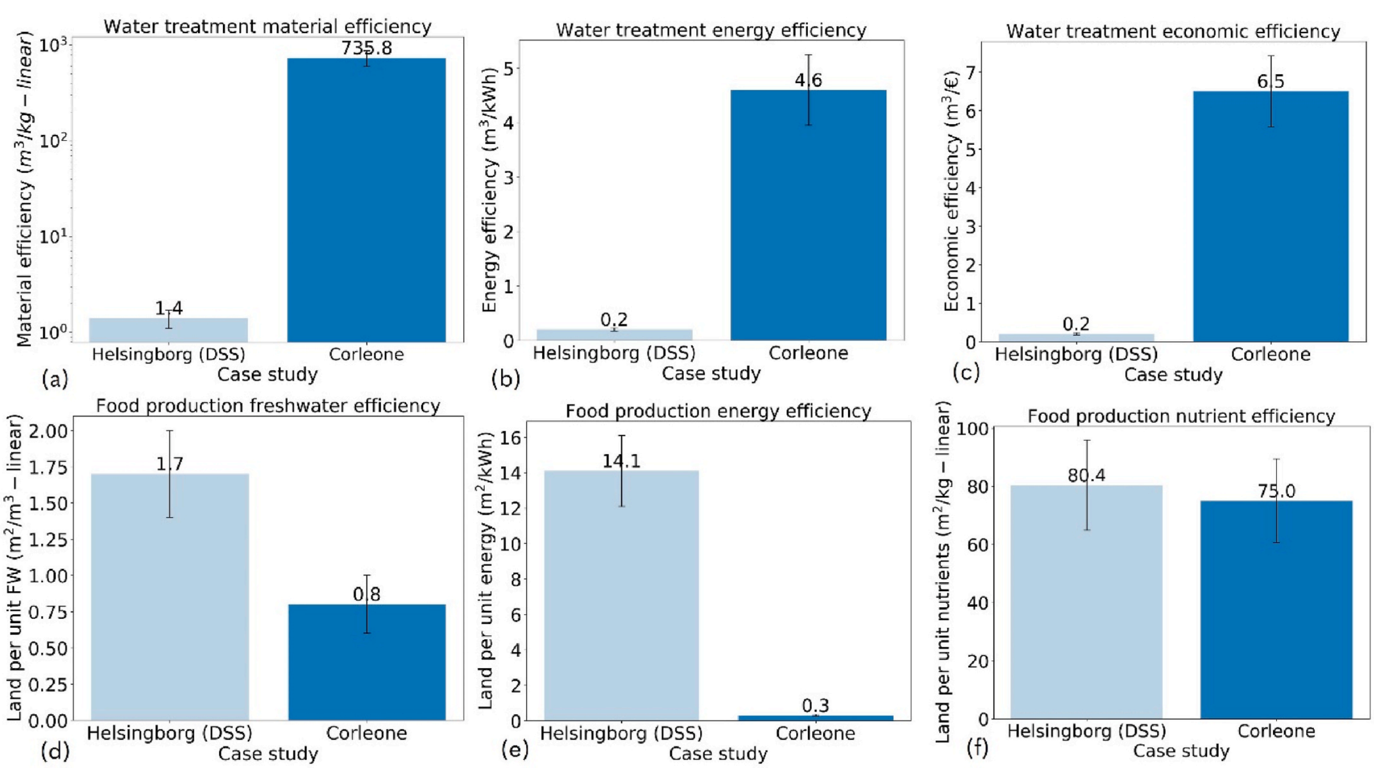


**Fig. 7.** The various efficiencies for the water treatment and the food production processes. (a), (b), and (c) show that Corleone performs better than Helsingborg in terms of the water treatment efficiencies. (d), (e), and (f) show that Helsingborg is the preferred choice with regards to the food production efficiencies.

energy use of transporting irrigation water and thereby favour local water reuse (Capodaglio, 2017). Therefore, to maximise the benefit of treated wastewater reuse for irrigation, reducing the distance between the WWTP and the agricultural fields is crucial and decentralization may help with this. The WT and FP efficiency results are shown in Fig. 7.

The circularities are calculated by subtracting the linear flow fractions in each case study from 1 and the linear flows depend on the sum of virgin resources (V) and unrecovered waste (W) as shown in Table 1. Helsingborg's use of chemical-intensive treatment technologies leads to a very low mixed resource circularity (1 %) compared to Corleone's (40 %). Corleone also performs better (84 %) in comparison to Helsingborg (65 %) in terms of biotic resource circularity. This is because in Helsingborg, the BW and excess sludge are digested to produce energy from biogas, and therefore the unrecovered waste (W) value is high. On the contrary, in the case of Corleone, the excess sludge is composted and thus, the material resources are retained, leading to a low 'W' value. The low biotic circularity of Helsingborg need not be a negative thing as biogas is a renewable energy source and the use of sludge products for soil application may not be suited in all cases. Therefore, in such cases, maximising the circularity is not so important given that biogas production is quite valuable from the sustainability perspective.

The water circularities of both cases are the same (98 %) because the assumption is that only 5 % of the wastewater volume is lost during conveyance and treatment. Helsingborg, however, shows a much better performance for nutrient circularity (95 %) compared to Corleone (58 %). This is consistent with the fact that at Helsingborg, nutrients are recovered using struvite precipitation and ammonia stripping. In the Corleone case study, nutrients are not recovered separately. Yet the nutrient circularity of Corleone is not negligible. Some of the nutrients present in the excess sludge become part of the compost and the nutrients present in the treated wastewater get used in agriculture through irrigation.

For food production, the water circularities of both cases are low because only a small percentage of the irrigation water requirement to produce wheat for a population of 12000 is met by treated wastewater. Both cases largely depend on groundwater leading to high 'V' values in the equation for water circularity shown in Table 1. The FP water and nutrient circularities can be improved by using the recovered treated wastewater and nutrients. However, treated wastewater irrigation is not practised at either location, but it is a part of the future plan.

The FP nutrient circularity of Helsingborg is slightly better (35 %) compared to Corleone (30 %). This is mainly because the quantity of nutrients recovered from the WWTP is much higher for Helsingborg than for Corleone and thus more of the agricultural fertilizer need can be met with the recovered nutrients (i.e., with a lower V). The circularity assessment results of the two case studies are shown in Fig. 8.

When it comes to the three nature reciprocity indicators, Helsingborg is found to be the better option. No freshwater restoration is achieved by Helsingborg because some of the treated wastewater is used for irrigation and the rest is discharged into the ocean. This is not necessarily a negative point as long as the effluent quality does not disturb the ocean ecosystem. Contrary to this, the Corleone WWTP discharges its treated wastewater into the Eleuterio River. However, the FR value is $-8.1 \times 10^7$ $m^3/y$ implying that the treated wastewater requires a large quantity

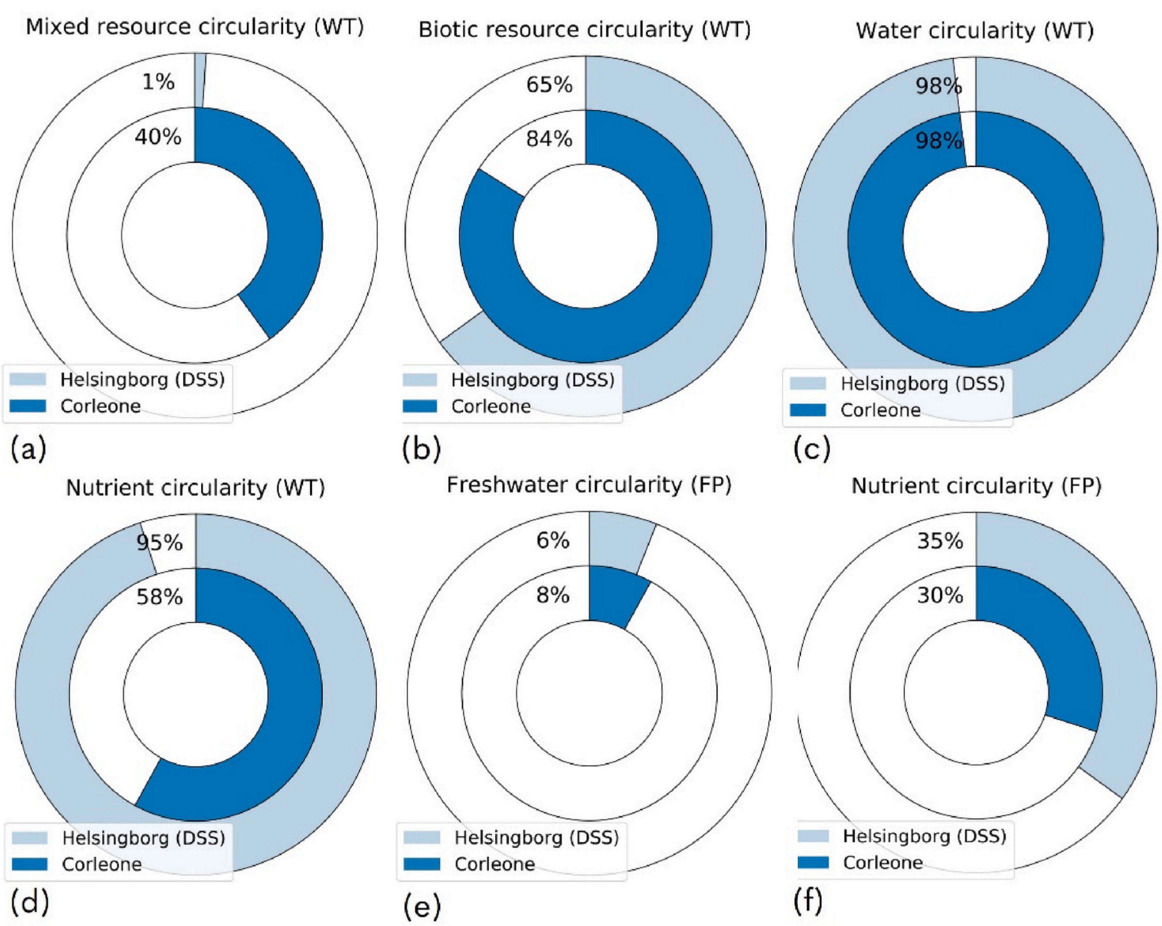


**Fig. 8.** Two case studies circularities for: (a) Mixed resources, (b) biotic resources, (c) water, (d) nutrients for the water treatment process; (e) freshwater, and (f) nutrients for the food production process. As it can be seen from these figures, Corleone performs better in terms of the mixed and the biotic resource categories, and Helsingborg in the nutrient category for the water treatment process. Also, Corleone performs slightly better for freshwater circularity but Helsingborg has a higher nutrient circularity in food production.

of water for its dilution. The phosphorus concentration in the effluent combined with the low stream flow contribute towards a WPL value greater than 1 implying that the stream flow is insufficient for dilution. And since the FR is calculated as a product of $Q_{dis}$ and 1-WPL (see Table 1), a negative value is obtained. Although dilution of the treated wastewater discharge does not consume the river water, the river water quality can suffer through the non-consumptive use of it. The streamflow of Eleuterio is simply insufficient to provide this dilution capacity therefore, reuse of the treated wastewater is strongly recommended instead of the discharge to avoid further degradation in the river water quality. Arid regions like Corleone are expected to have even lower stream flows in the future due to climate change. Therefore, reuse of the treated wastewater for irrigation or other purposes is needed.

The biomass assimilation of the nutrients, calculated as the product of NUE, NRE, and $M_{inf}$ (shown Table 1), is higher ($2.9 \times 10^4$ kg/y) for Helsingborg than Corleone ($1.1 \times 10^4$ kg/y) as expected because the nutrients are recovered as ammonium sulphate and struvite to be used in agriculture. In Corleone, the nutrients are not recovered by a dedicated recovery process but, are used in agriculture by applying treated wastewater and the sludge compost. Consequently, the nutrient recovery efficiency (NRE) of Corleone (43 % for N and 38 % for P) is lower than that of Helsingborg (78 % for N and 98 % for P). Incorporating nutrient recovery is recommended for Corleone to improve the NRE.

The soil organic matter sequestration is calculated by subtracting the volatile solids (VS) component from the mass of organic matter applied to soil as shown in Table 1. The SS value of Helsingborg ($20.8 \times 10^4$ kg/y) is higher than that of Corleone ($9.0 \times 10^4$ kg/y) mainly because the VS component of the sludge after anaerobic digestion (35 % in Helsingborg's case) is lower than that of the compost product (60 % in Corleone's case). The lower VS content leads to more of the organic matter present in the sludge being sequestered into the soil upon application.

Helsingborg achieves an energy self-sufficiency of over 200 % implying that more than twice the energy (electricity + heat) spent on the treatment is recovered according to the equation in Table 1. This was despite the higher energy consumption ($E_{in}$) of the DSS WWTP when compared to the conventional one. Including thermal energy recovery using heat pumps contributes to the high self-sufficiency of Helsingborg. A high thermal energy recovery is favoured by the GW separation as this stream contains most of the heat energy (Larsen, 2015). The thermal energy that can be recovered from DSS WWTPs is estimated to be between 477 kWh/capita/year and 840 kWh/capita/year (Kjerstadius et al., 2016). Thus there is a large variability in this. In any case, the recovery of thermal energy with SS treatment is very promising for energy-positive WWTPs. In contrast, energy is not recovered in Corleone and thus it relies entirely on an external energy supply. The nature reciprocity and the energy self-sufficiency results are shown in Fig. 9.

## 6. Discussion

### 6.1. DSS vs conventional centralized treatment

Using the WFE framework revealed the different pros and cons of the two approaches. Firstly, the material efficiency of WT is strongly dependent on the type of treatment technology used irrespective of the decentralization scale. This means that the use of chemical-intensive treatment will negatively affect the material efficiency of the treatment, and these should be avoided to keep the DSS WWTP's material efficiency high.

Secondly, most of the DSS treatment advantages result from the source separation of BW and GW. These include a higher efficiency of biogas recovery from the BW and excess sludge digestion, a higher thermal energy recovery, and higher nutrient recovery. The main advantage of decentralization is the reduction in energy use for the transport of irrigation water. However, this advantage may not be significant in regions with shallow groundwater depths, as in Helsingborg, since the advantages of treated wastewater reuse depend upon the availability of other water resources (Pannirselvam et al., 2019). This

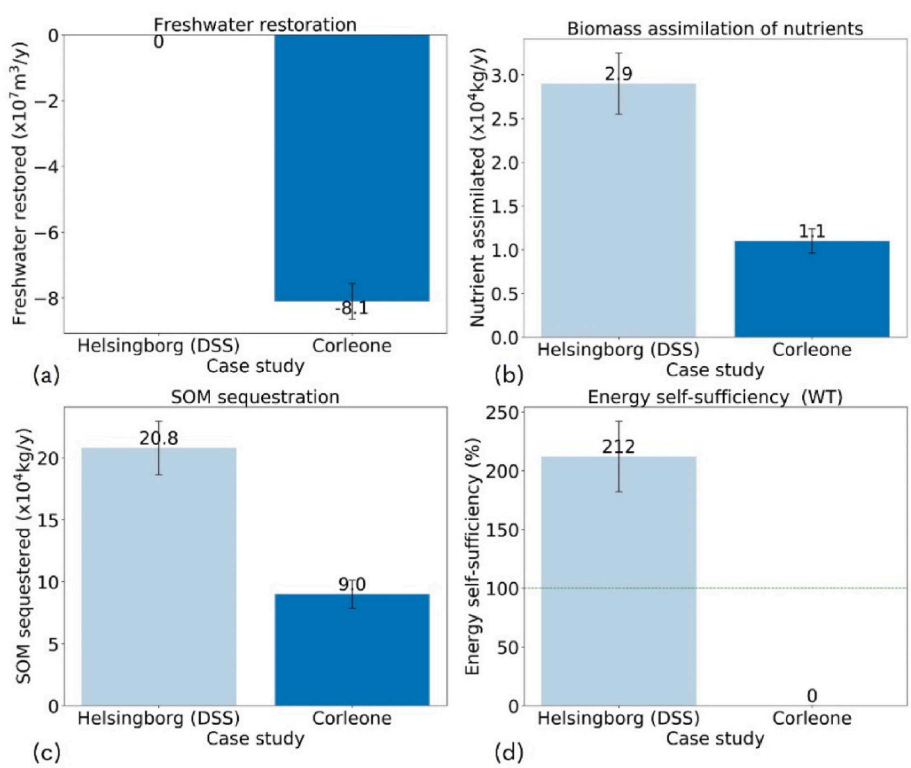


**Fig. 9.** The nature reciprocity and the energy self-sufficiency compared for the Helsingborg and Corleone case studies. (a) Freshwater restoration, (b) Biomass assimilation of nutrients, (c) SOM sequestration, (d) Energy self-sufficiency. Helsingborg performs better on the four criteria.

confirms the view of Rahman et al. (2024) that arid and semi-arid regions stand to gain the most from decentralized wastewater reuse because of seasonal stream flows and low water table depths. Consequently, for the future infrastructure, the local climate should be carefully considered and SS should be implemented before decentralization because of more benefits. However, decentralized treated wastewater reuse may be given equal importance in arid and semi-arid regions.

Thirdly, maximising circularity may not always be desirable, subject to the context. Some biotic material may be lost if the excess sludge is digested for energy recovery, causing a lower circularity. This is the case in Helsingborg which has a biotic resource circularity of 65 % compared to the 84 % in Corleone. Although composting the excess sludge may lead to a higher circularity than biogas production, it can also lead to higher environmental impacts as shown in Morsink-Georgali et al. (2022). Moreover, the recovery of renewable biogas energy is desirable despite reducing the WWTP's circularity. The decision-makers need to consider whether they want to maximise the circularity or recover renewable energy.

Further, the conventional centralized approach performs better on the economic criteria for wastewater treatment as also show in the study of Fort Collins conducted by Cole et al. (2018) and that of Garrido-Baserba et al. (2018). This is attributable to the high initial capital cost for decentralization and source separation. However, as demonstrated in this study, DSS can lead to higher freshwater and energy efficiencies for food production, which could lead to cost savings. Therefore, further studies need to be done to quantify the potential cost savings and extra income for the overall nexus.

### 6.2. Novelty of the WFE framework

The proposed WFE framework has several elements of novelty. Firstly, this is the first WFE framework that incorporates nature reciprocity indicators. The WFE framework of Li & Ma (2020) takes a life-cycle assessment approach to evaluating the environmental damages of the three sectors focussing thereby only on the negative impacts of the three sectors. A more holistic view is provided by the inclusion of nature reciprocity in the novel framework enabling the user to evaluate the balance between the positive and negative environmental impacts of the nexus.

Secondly, this framework includes the most commonly recovered resources from wastewater and its application is not limited to any single resource in contrast to the WFE studies presented by Fetanat et al. (2021), centred on energy recovery and Yao et al. (2018), focused on nutrient recovery and reuse. The inclusion of nutrients, energy, and treated wastewater ensures that the decision makers can assess many more resource recovery options and plan them in an integrated and comprehensive manner.

Thirdly, this framework is meant specifically for planning the resource recovery solutions related to WWTPs and comparing the DSS and conventional approaches to wastewater treatment. To the best of the author's knowledge, this is the first framework to serve this purpose. The adoption DSS and the holistic WFE nexus-based planning have traditionally been separate discussions. This framework bridges the gap between the two consequential topics.

### 6.3. Advantages of the WFE framework

The proposed WFE framework allows us to use the functional units and indicators of direct relevance to each sector thereby facilitating the communication of their benefits. It was shown how the framework makes it possible to evaluate the effect of resource recovery directly on the water treatment efficiency. The efficiency assessment of the WT sector helps to compare the case studies on their performance solely from treating wastewater to the relevant effluent standards. Additionally, the framework can explicitly quantify the benefits to the FP sector. These benefits were shown here in terms of higher nutrients, freshwater circularity, and efficiency in the production of the wheat crop, as well as in the form of a higher energy efficiency of food production as a result of the DSS treatment. These indicators are directly relevant to the FP sector. A lack of clear communication within the nexus results in a lack of integrated planning and management of the resources and an inclusive tool is required to bridge the communication gap (Mohtar and Daher, 2016). The clear communication of the benefits of resource recovery to the water treatment and food production sectors is the first advantage of this WFE framework that will likely lead to a more holistic and integrated resource management.

This framework accounts for the relevant local climate and geographical factors. The FP efficiency assessment is done for the freshwater, the nutrients and the energy flows. The agricultural conditions of different regions demand varying quantities of water based primarily on the evapotranspiration (ET) needs. Treated wastewater may be used to grow a wide variety of crops. Standardizing the crop to the common wheat offers a proxy way of judging the efficiency of the FP sector irrespective of the crop type. It aids in the comparison of different real-life case studies while still retaining the local climate factors in the form of evapotranspiration value. Therefore, a second advantage of the framework is that it can help account for the agricultural water requirements subject to the local climate.

The use of reciprocity indicators as part of the framework helps to ensure that the sectors take responsibility for making a positive impact on the natural environment. The FR indicator can also help to evaluate if discharging the treated wastewater is causing too much pressure on the natural streams as shown in the Corleone case study. A negative FR value informs the decision-maker of the natural stream's insufficient dilution capacity. A lack of acceptance of treated wastewater reuse for irrigation from a certain percentage of the population has been discussed by Verhoest et al. (2022) and Saliba et al. (2018). Verhoest et al. (2022) pointed to the crucial need to make people aware of how their decision to consume treated wastewater-irrigated crops can protect the natural environment. Showing that discharge of the treated wastewater could be having a net negative effect on the natural streams can contribute to this. It can potentially lead to higher acceptance, which is the third advantage of the framework.

### 6.4. Limitations and future outlook

The WFE framework does not account for the revenues generated by a WWTP in return for providing the recovered resources due to the lack of data. This needs to be considered for a more accurate economic efficiency assessment. Furthermore, this framework does not capture the benefit assessment of discharging treated wastewater into the ocean or a lake since only the FR benefits to a stream can be assessed. In the future, methods to assess the other benefits should be developed.

In addition to the above, the circularity assessment in the WFE framework is focused on mixed materials, biotic resources, nutrients, and water. While these resources are mostly targeted for recovery, a few other resources, such as metals, are also sometimes recovered. Methods to assess the circularities of other resources should also be developed for a more comprehensive assessment framework.

Another limitation of this study is that the authors have not accounted for the quality of the recovered products. The chemical- and energy-intensive treatment processes of Helsingborg that were damaging from the energy efficiency and circularity perspectives are important because they also lead to a higher quality of the recovered products. For example, the recovered nutrients contain lower heavy metal and PFAS concentrations. Also, the higher quality of water being used for irrigation is not captured by this framework. For future work, indicators need to be developed that can capture the effect of improved quality of the recovered resources.

Additionally, the economic efficiency assessment in this paper included a comparison between the capex of upgrading an existing centralized WWTP with that of the green-field costs of a DSS treatment plant. This naturally favours the existing infrastructure. The DSS treatment plants can be seen as a replacement for the conventional centralized ones at the end of their service period. Then, the choice would be between constructing a new conventional and a DSS WWTP. Therefore, a life cycle cost comparison is recommended for future work.

Lastly, the WFE framework lacks an assessment of social factors such as public acceptance because the authors do not have expertise in this area. It is recommended that the social dimension be included in the framework for future work.

## 7. Conclusions

The paper covers a literature gap in using a novel WFE framework to compare a conventional centralized and a decentralized source separation approaches to wastewater treatment and resource recovery. The framework with its reproducible assessment methods achieves the following.

- The framework covers multiple dimensions, such as economic performance, nature reciprocity, efficiency of water treatment and food production, energy self-sufficiency of the water treatment process, and circularity.
- It takes into account the local climate and the agricultural conditions by including factors such as the agricultural land use per capita and the evapotranspiration needs of crops.
- It contains indicators specific to the water treatment and food production sectors that make the benefits of resource recovery easier to communicate.
- It can potentially help to increase the acceptability of treated wastewater reuse for irrigation by explicitly showing the positive/negative effect of treated wastewater discharge into a stream.

All this makes the proposed WFE assessment framework more integrated and holistic in nature when compared to the existing assessment frameworks.

The new framework was applied to two case studies, one consisting of a decentralized source separation treatment (Helsingborg) and the other one being a conventional centralized treatment (Corleone). The related assessments of two different approaches resulted in the following observations.

- The Helsingborg approach to heat and electricity recovery leads to an energy-positive water treatment. However, the construction of the new infrastructure leads to a low economic efficiency. Additionally, the chemical- and energy-intensive processes reduce its material and energy efficiencies but also lead to better quality recovered resources and new methods to quantify the resource quality need to be developed. The food production becomes substantially more efficient and circular by the Helsingborg approach.
- In Corleone, the centralized infrastructure is already present and therefore this remains the more economically efficient approach atleast for the duration of the infrastructure life. Due to a low water table in Corleone, replacing groundwater with treated wastewater for irrigation is strongly recommended. This would also reduce the pressure on the local river where the effluent is discharged.
- Lastly, the future of the WWTP infrastructure must be planned based on the local conditions. Source separation may be the first step in the infrastructure upgrade for most places except for arid and semi-arid regions where decentralized treated wastewater reuse is crucial.

## CRediT authorship contribution statement

**Anurag Bhambhani:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Conceptualization. **Oriana Jovanovic:** Writing – review & editing, Validation, Supervision, Conceptualization. **Hamse Kjerstadius:** Writing – review & editing, Validation, Data curation. **Daniele Di Trapani:** Writing – review & editing, Validation, Data curation. **Giorgio Mannina:** Writing – review & editing, Validation, Data curation. **Jan Peter van der Hoek:** Writing – review & editing, Validation, Supervision, Methodology, Funding acquisition, Data curation. **Zoran Kapelan:** Writing – review & editing, Validation, Supervision, Methodology, Funding acquisition, Data curation.

## Declaration of competing interest

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

## Acknowledgements

This work is part of the project WIDER UPTAKE (www.wider-uptake.eu). This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 869283. This article reflects only the author's view. The Commission is not responsible for any use that may be made of the information it contains.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2025.144716.

## Data availability

Data will be made available on request.

## References

Albrecht, T.R., Crootof, A., Scott, C.A., 2018. The Water-Energy-Food Nexus: a systematic review of methods for nexus assessment. Environ. Res. Lett. 13 (4). https://doi.org/10.1088/1748-9326/aaa9c6.

Ampe, K., Paredis, E., Asveld, L., Osseweijer, P., Block, T., 2020. A transition in the Dutch wastewater system? The struggle between discourses and with lock-ins. J. Environ. Pol. Plann. 22 (2), 155–169. https://doi.org/10.1080/1523908X.2019.1680275.

Barroso Soares, R., 2017. Comparative analysis of the energy consumption of different wastewater treatment plants. International Journal of Architecture, Arts and Applications 3 (6), 79. https://doi.org/10.11648/j.ijaaa.20170306.11.

Barthel, R., Stangefelt, M., Giese, M., Nygren, M., Seftigen, K., Chen, D., 2021. Current understanding of groundwater recharge and groundwater drought in Sweden compared to countries with similar geology and climate. Geogr. Ann. Phys. Geogr. 103 (4), 323–345. https://doi.org/10.1080/04353676.2021.1969130.

Besson, M., Berger, S., Tiruta-barna, L., Paul, E., Spérandio, M., 2021. Environmental assessment of urine, black and grey water separation for resource recovery in a new district compared to centralized wastewater resources recovery plant. J. Clean. Prod. 301. https://doi.org/10.1016/j.jclepro.2021.126868.

Bhambhani, A., Jovanovic, O., van Nieuwenhuijzen, A., van der Hoek, J.P., Kapelan, Z., 2024. Introducing a new method to assess the benefits of resources recovered from wastewater to the natural environment. Sustain. Prod. Consum. 46, 559–570. https://doi.org/10.1016/j.spc.2024.03.016.

Bhambhani, A., Kapelan, Z., van der Hoek, J.P., 2023. A new approach to circularity assessment for a sustainable water sector: accounting for environmental functional flows and losses. Sci. Total Environ., 166520 https://doi.org/10.1016/j.scitotenv.2023.166520.

Cairns, R., Krzywoszynska, A., 2016. Anatomy of a buzzword: the emergence of 'the water-energy-food nexus' in UK natural resource debates. Environ. Sci. Pol. 64, 164–170. https://doi.org/10.1016/j.envsci.2016.07.007.

Capodaglio, A.G., 2017. Integrated, decentralized wastewater management for resource recovery in rural and peri-urban areas. Resources 6 (2). https://doi.org/10.3390/resources6020022.

Cardoso, B.J., Rodrigues, E., Gaspar, A.R., Gomes, A., 2021. Energy performance factors in wastewater treatment plants: a review. J. Clean. Prod. 322. https://doi.org/10.1016/j.jclepro.2021.129107.

Chae, K.J., Ren, X., 2016. Flexible and stable heat energy recovery from municipal wastewater treatment plants using a fixed-inverter hybrid heat pump system. Appl. Energy 179, 565–574. https://doi.org/10.1016/j.apenergy.2016.07.021.

Cole, J., Sharvelle, S., Fourness, D., Grigg, N., Roesner, L., Haukaas, J., 2018. Centralized and decentralized strategies for dual water supply: case study. J. Water Resour. Plann. Manag. 144 (1). https://doi.org/10.1061/(asce)wr.1943-5452.0000856.

Corona, B., Shen, L., Reike, D., Rosales Carreón, J., Worrell, E., 2019. Towards sustainable development through the circular economy—a review and critical

assessment on current circularity metrics. Resour. Conserv. Recycl. 151. https://doi.org/10.1016/j.resconrec.2019.104498.
Culha, O., Gunerhan, H., Biyik, E., Ekren, O., Hepbasli, A., 2015. Heat exchanger applications in wastewater source heat pumps for buildings: a key review. Energy Build. 104, 215–232. https://doi.org/10.1016/j.enbuild.2015.07.013.
Dargin, J., Daher, B., Mohtar, R.H., 2019. Complexity versus simplicity in water energy food nexus (WEF) assessment tools. Sci. Total Environ. 650, 1566–1575. https://doi.org/10.1016/j.scitotenv.2018.09.080.
Diaz-Elsayed, N., Rezaei, N., Guo, T., Mohebbi, S., Zhang, Q., 2019. Wastewater-based resource recovery technologies across scale: a review. Resour. Conserv. Recycl. 145, 94–112. https://doi.org/10.1016/j.resconrec.2018.12.035.
El-Gafy, I., 2017. Water–food–energy nexus index: analysis of water–energy–food nexus of crop's production system applying the indicators approach. Appl. Water Sci. 7 (6), 2857–2868. https://doi.org/10.1007/s13201-017-0551-3.
Ellen MacArthur Foundation, 2019. Circularity indicators- an approach to measuring circularity. http://www.ellenmacarthurfoundation.org/circularity-indicators/.
Eurostat, 2024. Agricultural Production - Crops.
Fetanat, A., Tayebi, M., Mofid, H., 2021. Water-energy-food security nexus based selection of energy recovery from wastewater treatment technologies: an extended decision making framework under intuitionistic fuzzy environment. Sustain. Energy Technol. Assessments 43. https://doi.org/10.1016/j.seta.2020.100937.
Firmansyah, I., Carsjens, G.J., de Ruijter, F.J., Zeeman, G., Spiller, M., 2021. An integrated assessment of environmental, economic, social and technological parameters of source separated and conventional sanitation concepts: a contribution to sustainability analysis. J. Environ. Manag. 295. https://doi.org/10.1016/j.jenvman.2021.113131.
Garrido-Baserba, M., Vinardell, S., Molinos-Senante, M., Rosso, D., Poch, M., 2018. The economics of wastewater treatment decentralization: a techno-economic evaluation. Environ. Sci. Technol. 52 (15), 8965–8976. https://doi.org/10.1021/acs.est.8b01623.
Greer, R.A., Hannibal, B., Portney, K., 2020. The role of communication in managing complex water-energy-food governance systems. Water (Switzerland) 12 (4). https://doi.org/10.3390/W12041183.
Guest, J.S., Skerlos, S.J., Barnard, J.L., Beck, M.B., Daigger, G.T., Hilger, H., Jackson, S. J., Karvazy, K., Kelly, L., Macpherson, L., Mihelcic, J.R., Pramanik, A., Raskin, L., Van Loosdrecht, M.C.M., Yeh, D., Love, N.G., 2009. A new planning and design paradigm to achieve sustainable resource recovery from wastewater. Environ. Sci. Technol. 43 (16), 6126–6130. https://doi.org/10.1021/es9010515.
Huysman, S., Sala, S., Mancini, L., Ardente, F., Alvarenga, R.A.F., De Meester, S., Mathieux, F., Dewulf, J., 2015. Toward a systematized framework for resource efficiency indicators. Resour. Conserv. Recycl. 95, 68–76. https://doi.org/10.1016/j.resconrec.2014.10.014.
Kjerstadius, H., Bernstad Saraiva, A., Spångberg, J., 2016. Can Source Separation Increase Sustainability of Sanitation Management.
Kjerstadius, H., Bernstad Saraiva, A., Spångberg, J., Davidsson, 2017. Carbon footprint of urban source separation for nutrient recovery. J. Environ. Manag. 197, 250–257. https://doi.org/10.1016/j.jenvman.2017.03.094.
Lam, L., Kurisu, K., Hanaki, K., 2015. Comparative environmental impacts of source-separation systems for domestic wastewater management in rural China. J. Clean. Prod. 104, 185–198. https://doi.org/10.1016/j.jclepro.2015.04.126.
Larsen, T.A., 2015. CO2-neutral wastewater treatment plants or robust, climate-friendly wastewater management? A systems perspective. Water Res. 87, 513–521. https://doi.org/10.1016/j.watres.2015.06.006.
Larsen, T.A., 2019. Source Separation and Decentralization for Wastewater Management. IWA Publishing.
Li, P.C., Ma, H. wen, 2020. Evaluating the environmental impacts of the water-energy-food nexus with a life-cycle approach. Resour. Conserv. Recycl. 157. https://doi.org/10.1016/j.resconrec.2020.104789.
Liu, Y., Gui, D., Yin, C., Zhang, L., Xue, D., Liu, Y., Ahmed, Z., Zeng, F., 2023. Effects of human activities on evapotranspiration and its components in arid areas. Int. J. Environ. Res. Publ. Health 20 (4). https://doi.org/10.3390/ijerph20042795.
Longo, S., d'Antoni, B.M., Bongards, M., Chaparro, A., Cronrath, A., Fatone, F., Lema, J. M., Mauricio-Iglesias, M., Soares, A., Hospido, A., 2016. Monitoring and diagnosis of energy consumption in wastewater treatment plants. A state of the art and proposals for improvement. Appl. Energy 179, 1251–1268. https://doi.org/10.1016/j.apenergy.2016.07.043.
Maktabifard, M., Zaborowska, E., Makinia, J., 2018. Achieving energy neutrality in wastewater treatment plants through energy savings and enhancing renewable energy production. Rev. Environ. Sci. Biotechnol. 17 (4), 655–689. https://doi.org/10.1007/s11157-018-9478-x.
Mannina, G., Badalucco, L., Barbara, L., Cosenza, A., Di Trapani, D., Gallo, G., Laudicina, V.A., Marino, G., Muscarella, S.M., Presti, D., Helness, H., 2021. Enhancing a transition to a circular economy in the water sector: the eu project wider uptake. Water (Switzerland) 13 (7). https://doi.org/10.3390/w13070946.
Mannina, G., Badalucco, L., Barbara, L., Cosenza, A., Di Trapani, D., Laudicina, V.A., Muscarella, S.M., Presti, D., 2022. Roadmapping the transition to water resource recovery facilities: the two demonstration case studies of Corleone and marineo (Italy). Water (Switzerland) 14 (2). https://doi.org/10.3390/w14020156.
McConville, J.R., Kvarnström, E., Jönsson, H., Kärrman, E., Johansson, M., 2017. Source separation: challenges & opportunities for transition in the Swedish wastewater sector. Resour. Conserv. Recycl. 120, 144–156. https://doi.org/10.1016/j.resconrec.2016.12.004.
Mohtar, R.H., Daher, B., 2016. Water-Energy-Food Nexus Framework for facilitating multi-stakeholder dialogue. Water Int. 41 (5), 655–661. https://doi.org/10.1080/02508060.2016.1149759.
Molajou, A., Pouladi, P., Afshar, A., 2021. Incorporating social system into water-food-energy nexus. Water Resour. Manag. 35 (13), 4561–4580. https://doi.org/10.1007/s11269-021-02967-4.
Morici, S., Gagliano Candela, E., Favara, R., La Pica, L., Scaletta, C., Pecoraino, G., 2023. Hydrogeochemical characterization of the alluvial aquifer of catania plain, sicily (south Italy). Environ. Earth Sci. 82 (6). https://doi.org/10.1007/s12665-023-10816-3.
Morsink-Georgali, P.Z., Kylili, A., Fokaides, P.A., Papadopoulos, A.M., 2022. Compost versus biogas treatment of sewage sludge dilemma assessment using life cycle analysis. J. Clean. Prod. 350. https://doi.org/10.1016/j.jclepro.2022.131490.
Nhamo, L., Mabhaudhi, T., Mpandeli, S., Dickens, C., Nhemachena, C., Senzanje, A., Naidoo, D., Liphadzi, S., Modi, A.T., 2020. An integrative analytical model for the water-energy-food nexus: South Africa case study. Environ. Sci. Pol. 109, 15–24. https://doi.org/10.20944/preprints201905.0359.v1.
Ofori, S., Puškáčová, A., Růžičková, I., Wanner, J., 2021. Treated wastewater reuse for irrigation: pros and cons. Sci. Total Environ. 760. https://doi.org/10.1016/j.scitotenv.2020.144026.
Opher, T., Friedler, E., 2016. Comparative LCA of decentralized wastewater treatment alternatives for non-potable urban reuse. J. Environ. Manag. 182, 464–476. https://doi.org/10.1016/j.jenvman.2016.07.080.
Pannirselvam, M., Shu, L., Griffin, G., Philip, L., Natarajan, A., Hussain, S., 2019. Water Scarcity and Ways to Reduce the Impact. Springer international publishing. http://www.springer.com/series/13085.
Pasciucco, F., Pecorini, I., Iannelli, R., 2022. Planning the centralization level in wastewater collection and treatment: a review of assessment methods. J. Clean. Prod. 375. https://doi.org/10.1016/j.jclepro.2022.134092.
Paulo, P.L., Azevedo, C., Begosso, L., Galbiati, A.F., Boncz, M.A., 2013. Natural systems treating greywater and blackwater on-site: integrating treatment, reuse and landscaping. Ecol. Eng. 50, 95–100. https://doi.org/10.1016/j.ecoleng.2012.03.022.
Poustie, M.S., Deletic, A., Brown, R.R., Wong, T., de Haan, F.J., Skinner, R., 2015. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. Urban Water J. 12 (7), 543–558. https://doi.org/10.1080/1573062X.2014.916725.
Rahman, K.Z., Al Saadi, S., Al Rawahi, M., van Afferden, M., Bernhard, K., Friesen, J., Müller, R.A., 2024. Small decentralized technologies for high-strength wastewater treatment and reuse in arid and semi-arid regions. Environments - MDPI 11 (7). https://doi.org/10.3390/environments11070142.
Roefs, I., Meulman, B., Vreeburg, J.H.G., Spiller, M., 2017. Centralised, decentralised or hybrid sanitation systems? Economic evaluation under urban development uncertainty and phased expansion. Water Res. 109, 274–286. https://doi.org/10.1016/j.watres.2016.11.051.
Sakthivel, S.R., Tilley, E., Udert, K.M., 2012. Wood ash as a magnesium source for phosphorus recovery from source-separated urine. Sci. Total Environ. 419, 68–75. https://doi.org/10.1016/j.scitotenv.2011.12.065.
Saliba, R., Callieris, R., D'Agostino, D., Roma, R., Scardigno, A., 2018. Stakeholders' attitude towards the reuse of treated wastewater for irrigation in Mediterranean agriculture. Agric. Water Manag. 204, 60–68. https://doi.org/10.1016/j.agwat.2018.03.036.
Schoen, M.E., Garland, J., 2017. Review of pathogen treatment reductions for onsite non-potable reuse of alternative source waters. Microbial Risk Analysis 5, 25–31. https://doi.org/10.1016/j.mran.2015.10.001.
Shannak, S., Mabrey, D., Vittorio, M., 2018. Moving from theory to practice in the water–energy–food nexus: an evaluation of existing models and frameworks. Water-Energy Nexus 1 (1), 17–25. https://doi.org/10.1016/j.wen.2018.04.001.
Simpson, G.B., Jewitt, G.P.W., Becker, W., Badenhorst, J., Masia, S., Neves, A.R., Rovira, P., Pascual, V., 2022. The water-energy-food nexus index: a tool to support integrated resource planning, management and security. Frontiers in Water 4. https://doi.org/10.3389/frwa.2022.825854.
Smajgl, A., Ward, J., Pluschke, L., 2016. The water-food-energy Nexus - realising a new paradigm. J. Hydrol. 533, 533–540. https://doi.org/10.1016/j.jhydrol.2015.12.033.
The world bank, 2024. World Development Indicators. World Development Indicators.
Verhoest, P., Gaume, B., Bauwens, J., te Braak, P., Huysmans, M., 2022. Public acceptance of recycled water: a survey of social attitudes toward the consumption of crops grown with treated wastewater. Sustain. Prod. Consum. 34, 467–475. https://doi.org/10.1016/j.spc.2022.10.003.
Wang, H., Yang, Y., Keller, A.A., Li, X., Feng, S., Dong, Y. nan, Li, F., 2016. Comparative analysis of energy intensity and carbon emissions in wastewater treatment in USA, Germany, China and South Africa. Appl. Energy 184, 873–881. https://doi.org/10.1016/j.apenergy.2016.07.061.
Yan, P., Qin, R.C., Guo, J.S., Yu, Q., Li, Z., Chen, Y.P., Shen, Y., Fang, F., 2017. Net-zero-energy model for sustainable wastewater treatment. Environ. Sci. Technol. 51 (2), 1017–1023. https://doi.org/10.1021/acs.est.6b04735.
Yao, Y., Martinez-Hernandez, E., Yang, A., 2018. Modelling nutrient flows in a simplified local food-energy-water system. Resour. Conserv. Recycl. 133, 343–353. https://doi.org/10.1016/j.resconrec.2018.02.022.
Ye, Y., Ngo, H.H., Guo, W., Chang, S.W., Nguyen, D.D., Zhang, X., Zhang, J., Liang, S., 2020. Nutrient recovery from wastewater: From technology to economy. Bioresour. Technol. Rep. 11. https://doi.org/10.1016/j.biteb.2020.100425.
Yi, J., Guo, J., Ou, M., Pueppke, S.G., Ou, W., Tao, Y., Qi, J., 2020. Sustainability assessment of the water-energy-food nexus in Jiangsu Province, China. Habitat Int. 95. https://doi.org/10.1016/j.habitatint.2019.102094.
Zarei, M., 2020. Wastewater resources management for energy recovery from circular economy perspective. Water-Energy Nexus 3, 170–185. https://doi.org/10.1016/j.wen.2020.11.001.