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Introducing a new method to assess the benefits of resources recovered from wastewater to the natural environment

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ABSTRACT

Resources recovery can improve the economic efficiency and reduce the negative environmental impacts of municipal wastewater treatment plants (MWWTP). The recovered resources can also actively benefit the natural environment enabling a reciprocal relationship between human society and nature. Focusing on these benefits can reveal new resources recovery opportunities. Moreover, for certain environmental impact categories such as emissions of reactive nitrogen, mere damage reduction is insufficient because these emissions are already beyond planetary limits. However, quantitative methods to assess nature benefits are lacking. A new method is developed to calculate the potential nature benefits in three categories: Freshwater restoration, biomass assimilation of nutrients, and soil organic matter sequestration and it is demonstrated on a real-life MWWTP. Focusing on resources recovery helps to purify the wastewater sufficiently for discharge and to benefit the natural environment. Treated wastewater discharge into a river can support freshwater restoration depending on the effluent quality. High quality is achieved by the sufficient removal of the nutrients and organic matter and discharging into a high-flow stream. The recovery of nutrients helps to close the nutrient cycle through biomass assimilation. To maximize this benefit, the nutrient recovery efficiency from the MWWTP must be maximized. But, increasing the nutrient uptake efficiency in agriculture is also crucial, especially for nitrogen. The wastewater sludge products can be applied to soil to sequester organic matter and the products with low volatile solids should be preferred. The development of the new method is a start to recognizing and assessing the potentially positive role of humans in nature.

1. Introduction

Municipal wastewater treatment plants (MWWTPs) protect the natural environment and humans from the discharge of untreated domestic wastewater which is a health hazard (Mo and Zhang, 2013; Van Der Hoek et al., 2016). However, wastewater is also a source of valuable resources such as nutrients and organic matter which may be recovered to replace virgin resources (Chrispim et al., 2020).

Resources recovery can reduce the negative environmental impacts of MWWTPs such as carbon footprint and eutrophication (Cornejo et al., 2016). This can result from lower energy use within MWWTPs or the avoided burden of extracting virgin resources. Furthermore, recovery of resources can improve the economic efficiency of MWWTPs by either reducing treatment costs or generating extra revenue. However, resources recovery may not only reduce the environmental and economic costs but also actively benefit the natural environment.

Using the recovered resources, human society can provide a reciprocal service to the natural environment in return for the resources and services that nature provides. Assessing the potential nature benefits will help foster a symbiotic relationship between human society and the natural environment. This can also create a more holistic view of wastewater treatment and thereby reveal more resources recovery opportunities (Trimmer et al., 2019). Furthermore, it is not sufficient to merely think of damage reduction, we need to explore the potential for actively benefiting the natural environment wherever possible (Bhambhani et al., 2022). However, the assessment of this nature-benefitting aspect of resources recovery from wastewater has largely been neglected (Trimmer et al., 2019).

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The services provided by the natural environment to human society are often considered a one-way flow of benefits and Comberti et al. (2015) point to the need to reconsider this idea. The same authors mention the need to introduce 'reciprocal benefits' (i.e., from humans to nature) within the concept of sustainability. Moreover, two human viewpoints can be distinguished: One in which humans are a part of nature and another wherein humans are considered to be autonomous entities that rule over nature. The former is a viewpoint that more people find congruous with their experiences (Jax et al., 2018). Yet most common sustainability discourses are based on a unidirectional flow of benefits from nature to humans and methods backed by a reciprocal view must be developed (Jax et al., 2018). More specifically, in the wastewater treatment sector, Trimmer et al. (2019) contributed a conceptual framework that explains the potential of the recovered resources from MWWTPs to enhance the ecosystem services. However, the framework does not provide a method to quantitatively assess the enhancement and Trimmer et al. (2019) suggest the development of an assessment method for future work, resulting in a corresponding research gap that is addressed in this paper.

Therefore, the objective of this paper is to develop a novel method to assess the potential nature benefits of the resources recovered from municipal wastewater. This study focusses on certain key resources that include water, nutrients, and organic matter. This is not meant to replace the eco-efficiency assessment methods but to complement them as also suggested by Jax et al. (2018). This paper presents nature benefits as the next step towards sustainability.

In Section 2, the main resources present in domestic wastewater are introduced. This is followed by a discussion on the potential benefits of resources recovery including improved economic and eco-efficiencies. Further, the potential nature benefits from the resources recovered from wastewater and the importance of assessing these benefits are discussed. In Section 3, the novel method for quantitatively assessing the nature benefits is explained and a real-life case study is used to demonstrate the method, including uncertainty and sensitivity analyses. Section 4 contains a discussion of the results of the case study application. In Section 5, the wider implications of the assessment are discussed along with the limitations of the novel method. Finally, the conclusions about the method and its application are presented in Section 6.

2. Literature review

2.1. Resources present in domestic wastewater

The focus of MWWTPs has traditionally been the removal of pollutants from sewage, but now includes resources recovery (Renfrew et al., 2022; Van der Hoek et al., 2016; Wang et al., 2015). This is necessary because important resources are becoming scarce with a rise in human population (Van der Hoek et al., 2016).

Several resources recovery pathways have been studied but a few of them gather the most attention. Mo and Zhang (2013) discussed three main pathways for resources recovery, namely water reuse, on-site energy generation, and nutrient recycling. Trimmer et al. (2019) listed the three most common categories of resources to be recovered from wastewater: Water for reuse, nutrients, and organic matter. Energy recovery, organic carbon (C), and nutrient recovery were also discussed by Puchongkawarin et al. (2015). Kehrein et al. (2020) discussed the potential of recovering P as struvite and organic C (expressed as COD, chemical oxygen demand) as energy or biopolymers.

The upper limit of the quantity of waste that earth can sustain can be summarized by the nine planetary boundaries of Rockström et al. (2009). Four out of the nine planetary boundaries (species extinction rate, atmospheric CO_2 concentration and the emissions of reactive N and P) are transgressed already. Three of the four transgressed boundaries can be traced to the mismanagement of C, N, and P (Slootweg, 2020). The recovery of nutrients and organic matter is urgently needed to prevent further emissions of C, N, and P. Furthermore, organic matter, nutrients, and water are the most valuable resources that can be recovered from wastewater (Lee et al., 2013; Mo and Zhang, 2013; Verstraete et al., 2009) and data regarding the mass balances of COD, N, and P are relatively easy to estimate (Nowak et al., 1999).

Given the above, the resources most commonly recovered from MWWTPs include water for reuse, organic matter, and nutrients (mainly N and P). Hence, they will be the focus of this paper.

2.2. Benefits of resources recovery

Numerous benefits of recovering resources from a MWWTP have been discussed including economic value generation, resource circularity, reduced eutrophication, reduced ecotoxicity, improved energy efficiency and carbon footprint offset (Coma et al., 2017; Gherghel et al., 2019; Kehrein et al., 2020; Lam et al., 2022; Ruiken et al., 2013). These benefits can be classified broadly under two categories: Improved eco-efficiency and enhanced economic efficiency of a MWWTP. Methods to quantify these benefits have also been developed and continue to be the focus of studies.

Eco-efficiency is the ratio between the service delivered by a process and the negative environmental impacts of the process (Hauschild, 2015). Therefore, the eco-efficiency of a MWWTP can be defined as the ratio between the volume of wastewater (m^3/y) treated to discharge standards and the environmental impacts of the treatment process (e.g., the climate change impact measured in kg CO_2 eq.). Most conventional MWWTPs have a net negative environmental impact due to their high resources use intensity (Hao et al., 2019; Schaubroeck et al., 2015). However, resources recovery can reduce the negative environmental impacts of MWWTPs (Cornejo et al., 2013; Hao et al., 2019) thereby, improving their eco-efficiency. E.g., Cornejo et al. (2013) discussed the reduced eutrophication potential of a MWWTP because of treated wastewater (TW) reuse for fertigation (application of fertilizers via irrigation), reduction in the carbon footprint, and embodied energy resulting from energy recovery.

When it comes to economics, generally, resources recovered from MWWTPs have a higher cost than the virgin resources they replace. However, the higher cost can be offset by the reduced operational costs of the MWWTP. E.g., in the Amsterdam West MWWTP, an investment cost of € 4 million was estimated for struvite recovery which can result in an expected yearly saving of about € 400,000 for maintenance (van der Hoek et al., 2017). MWWTPs can also generate revenue by selling the recovered resources thus, generating extra revenue (Tarpani and Azapagic, 2018). Therefore, resources recovery can improve the economic efficiency of a MWWTP by reducing the operational costs of the water treatment or generating revenue from the sale of the recovered resources.

2.3. Nature benefits from resources recovery

Along with a lower negative environmental impact, positive effects on the natural environment can also be achieved using the recovered resources. Soil fertility, microbial biomass, and soil enzyme activity can be improved using sewage sludge application (Boudjabi and Chenchouni, 2021; Dhanker et al., 2021). TW discharge can help with improvement in surface water quality, bank stabilization, and the return of pollution-sensitive aquatic species (Bischel et al., 2013).

A method to assess these potential benefits is necessary. First, this will support a cycle of reciprocal benefits between human society and nature (Trimmer et al., 2019). The natural environment provides numerous services to human society that can be conceptualized using the ecosystem services (ES) framework (Wallace, 2007). Human society can also potentially provide benefits to the natural environment. An example of this is the indigenous communities enhancing the soil fertility of the Amazon forests by adding charcoal, bones, and manure (Comberti et al., 2015). Mutually beneficial relationships (symbiosis) among organisms and between organisms and their natural environments are common. Yet, human society's consideration of their beneficial role in nature has remained limited. The research focus remains on the reduction of negative impacts. The opportunity for benefiting nature that resources recovery provides us should be explored further.

Second, assessing the potential nature benefits will lead to a more holistic view of wastewater treatment and thereby may reveal more resources recovery opportunities (Trimmer et al., 2019). To illustrate, adding organic matter can improve soil structure and reduce erosion with secondary benefits such as improved water retention and enhanced vegetative growth. In some contexts, these benefits can be highly valuable and should not be ignored in favour of a directly recognizable benefit to humans, such as energy generation (Trimmer et al., 2019).

Third, only damage reduction is insufficient for sustainability (Bhambhani et al., 2022). This is especially true for the emissions that have crossed sustainable planetary limits (Bhambhani et al., 2022). For example, the anthropogenic emissions of reactive N and P have crossed the limits that planet earth can sustain (Sandström et al., 2023; Steffen et al., 2015). In such cases, an active approach towards repairing the nutrient flows is required. Despite the knowledge of potential nature reciprocity and the need to assess it, methods incorporating the assessment of enhanced ecosystems are rare (Trimmer et al., 2019) and usually qualitative. Therefore, a holistic method for the quantitative assessment of natural environmental benefits is needed.

The concept of actively providing benefits to the natural environment has to be defined first. In this paper, the natural environment is represented by stock and flux models of water, nutrients (N, P), and carbon. The resources move in between these stocks by natural or artificial processes e.g., N present in the atmosphere flows to the soil stock through natural fixation. It is not the intention to comprehensively describe the natural environment using a limited number of elements but to focus our attention on the most relevant parts of nature that a MWWTP can affect.

Next, we conceptualize environmental damage as an excess build-up of a resource in a particular stock or an excess removal of a resource from a stock. E.g., whereas climate change can be looked upon as an excess build-up of carbon in the atmospheric stock (Ajani et al., 2013), water scarcity can be conceptualized as an excess removal of water from a groundwater stock.

Nature reciprocity can be defined as a re-balancing of the resource stocks. This re-balancing can be achieved by directing a resource from one stock to another that could benefit from it e.g., excessive use of fertilizers has caused a build-up of reactive nitrogen (Nr) species in European rivers (Blaas and Kroeze, 2016). Future Nr emissions can be redirected to living biomass with the help of MWWTPs. The authors do not suggest that pristine natural conditions can be reached again but the focus is on including the assessment of nature benefits as a next step in the sustainability pursuit.

3. Methods

The focus of this paper is on three resource categories and three pathways through which they can benefit nature. The resources are treated wastewater (TW), nutrients (N and P), and organic matter (OM). The TW can be used to restore freshwater; the recovered nutrients can be used for nutrient cycling through the pathways of biomass assimilation; and the recovered organics can support carbon cycling through soil organic matter (SOM) addition. The links between the recoverable resources and their potential nature benefits are shown in Fig. 1 and are based on the work of Trimmer et al. (2019). As shown in the subsequent sections, the key novelty of this paper is the development of a new method to assess the potential nature benefits of wastewater-recovered resources. This is the first step in this direction and the indicators are kept simple to capture sufficient details but maintain ease of calculation for the decision-makers.

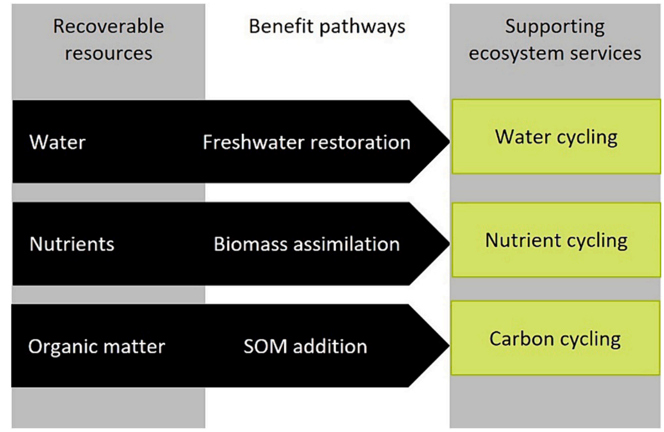

Fig. 1. The link between resources recovered from domestic wastewater and the potential positive effects on the natural environment through enhancement of supporting ecosystem services, based on Trimmer et al. (2019).

3.1. *Water cycling through freshwater restoration*

By 2050, between a third to a half of the global population is likely to face water scarcity (Boretti and Rosa, 2019; He et al., 2021) mainly driven by growing demand, a reduction in water resources, and pollution (Boretti and Rosa, 2019). The global withdrawal of blue water (groundwater and surface water) should remain under 4000–6000 km^3/year to avoid the irreversible collapse of ecosystems (Rockström et al., 2009). Thus, preserving and restoring freshwater reservoirs is critical and MWWTPs can play a crucial role here.

As the quality of the TW improves with technology, MWWTPs can be seen as significant contributors to freshwater reservoirs (Verstraete et al., 2009; Wang et al., 2017). Stream flow augmentation using TW discharge can restore freshwater and improve habitats for aquatic ecosystems (Plumlee et al., 2012). However, the quality of the TW is crucial to the restoration of the freshwater reservoirs. Assuming the wastewater treatment is achieved to meet the discharge standards, the pollutants present still require a certain quantity of freshwater to be diluted to background concentrations. Therefore, the nature benefit is here defined by the quantity of the discharged water multiplied by a factor that accounts for the water quality.

The concepts of gray water footprint (GWF) and water pollution level (WPL) developed by Hoekstra et al. (2011) are used here. The GWF refers to the volume of freshwater required to dilute a given pollutant concentration to the background concentration in the stream (Mekonnen and Hoekstra, 2015) and is calculated as follows:

$$GWF_i = \frac{L_i}{C_{max} - C_{nat}} \tag{1}$$

where GWF_i is the gray water footprint of the MWWTP discharge stream in month i, L_i is the pollutant load (kg/month) for month i, C_{max} is the maximum acceptable concentration (kg/m^3) of a pollutant obtained from the EU water directive (EC, 2000), and C_{nat} is the natural background concentration of the pollutant in the receiving stream when there was no human disturbance in the catchment. If C_{nat} is not known, Hoekstra et al. (2011) suggest using 0 kg/m^3.

A few special conditions should be discussed. It is assumed that in most cases, the C_{max} of a pollutant will be greater than C_{nat} and then Eq. (1) applies. In the case of C_{max} being equal to C_{nat}, the GWF value would be undefined. However, this situation is unlikely to occur because maximum concentration standards are usually not set equal to the natural background concentrations (Hoekstra et al., 2011). Also, it is unlikely that the C_{max} of a pollutant is specified to be lower than the C_{nat}. Still, if that happens then Eq. (1) should not be used.

The WPL is the ratio between the GWF of a MWWTP discharge stream and the stream runoff (m^3/y) (Mekonnen and Hoekstra, 2015) and is calculated as follows:

$$WPL_i = \frac{GWF_i}{R_{act\,i}} \tag{2}$$

where GWF_i is the gray water footprint of the MWWTP discharge calculated using Eq. (1) in the month i, and $R_{act\ i}$ is the actual discharge of the stream receiving the TW (m^3/month) in the month i. A smaller WPL value means a better discharge quality. The WPL calculation has to account for the seasonal stream discharge variations and a monthly estimation is enough for this (Hoekstra et al., 2011).

The benefit indicator is here defined as the quantity of TW discharged into nature multiplied by a quality factor, resulting in freshwater restoration (FR) as follows:

$$FR = \sum_{i=1}^{12} \left(Q_{dis_i} \times (1 - WPL_i) \right) \tag{3}$$

where FR is the freshwater restored in m^3/y, $Q_{dis\ i}$ is the volume of treated water discharge in m^3/month for month i, and WPL_i is the water pollution level calculated using Eq. (2).

A negative FR implies that there is a net consumption of freshwater for the dilution of the pollutants in the TW. Two examples demonstrate the effect of the WPL on the FR values. The WPL of TW can vary significantly and values >1 and as low as 0.08 have been reported in Wang et al. (2020) describing the status of N discharge from MWWTPs in Shenzhen, China. Suppose MWWTP1 and MWWTP2 each discharge 100,000 m^3/month to a river. The N WPL for MWWTP1 and MWWTP2 are 1.2 and 0.08. The FR values achieved by them are shown below.

$$\mathrm{FR}_{\mathrm{MWWTP1}} = 100,000 \times (1 - 1.2) = -20,000\ \mathrm{m}^3/\mathrm{month}$$

$$\mathrm{FR}_{\mathrm{MWWTP2}} = 100,000 \times (1 - 0.08) = 92,000\ \mathrm{m}^3/\mathrm{month}$$

The MWWTP1 discharge requires more water for dilution leading to a net decrease of freshwater by 20,000 m^3/month. In contrast, MWWTP2 with a low WPL restores 92,000 m^3/month of freshwater.

3.2. Nutrient cycling through biomass assimilation

Recovering N and P from wastewater is important for three main reasons. First, N and P are crucial for crop fertilization and the production of these fertilizers is energy and resource intensive. A 4 % annual increase in fertilizer production is projected until 2050 to feed a growing human population (Xie et al., 2016). The Haber Bosch (HB) process used to obtain reactive N from the atmosphere to manufacture fertilizers consumes 1–2 % of the global energy expenditure (Houlton et al., 2019). Contrary to N, P is a non-renewable resource obtained from mining phosphate rocks (van der Hoek et al., 2018). Given the non-renewability of these rocks, P was designated a critical raw material by the EU in 2014 (Hukari et al., 2016). The high energy consumption of the HB process and the excessive mining of the phosphate rocks can be avoided by nutrient recovery from wastewater.

Second, recovering nutrients can prevent eutrophication from TW discharge (Babcock-Jackson et al., 2023; Singh et al., 2023). The loss of reactive N through TW discharge causes direct human health damage such as asthma and cancer and disrupts ecosystems inducing a loss of biodiversity and ecosystem services (Bodirsky et al., 2014). The economic costs of Nr loss to the environment have been estimated to be between € 75 and € 485 billion in the EU (Van Grinsven et al., 2013) and between $ 80 and $ 441 billion in the US (Sobota et al., 2015). P in TW discharge is another major contributor to eutrophication, fish death and ecosystem destruction (Patyal et al., 2022), and causes human health issues such as metabolic bone disease (Gao et al., 2020).

Third, nutrient products derived from wastewater usually have a higher nutrient uptake efficiency (NUE) than conventional fertilizers (Babcock-Jackson et al., 2023; Saliu and Oladoja, 2021; Santos and Pires, 2018). About 85 % of the Nr created using the HB process and 90 % of the mined P are used for food production (Galloway et al., 2003; Kanter and Brownlie, 2019). However, only between 20 % and 30 % of the N in the fertilizers is taken up by crops with the rest leaching into groundwater, volatilizing as ammonia, or running off in streams (Naz and Sulaiman, 2016). Similarly, most of the P is lost to the environment since the crop uptake of P is known to be under 25 % (Roberts and Johnston, 2015). One of the ways to deal with the low NUE is by using slow-release fertilizers (Babcock-Jackson et al., 2023). The industrial manufacturing of slow-release fertilizers is limited by the high production cost and the need for petroleum-based polymers (Vejan et al., 2021). Here, wastewater-recovered nutrients offer an advantage because these are usually in an adsorbed or encapsulated form ensuring their slow release (Vejan et al., 2021).

N is active in the natural environment until either sequestered or converted to N_2 (Galloway et al., 2021). Since biomass assimilation can combat excess reactive nutrient species in the environment (J. Xu and Shen, 2011) and the nutrient products derived from wastewater tend to have a higher NUE, the MWWTPs can provide a nature benefit by efficiently assimilating nutrients into plant biomass.

The MWWTP nutrient recovery efficiency (NRE) can vary between the different recovery techniques (Xie et al., 2016). Also, different nutrient recovery products have varying NUE (Sigurnjak et al., 2016). Thus, both these factors will impact nutrient assimilation. An equation to measure the biomass assimilation of nutrients is presented below:

$$BA = M_{inf} \times NRE \times NUE \tag{4}$$

where BA is the biomass assimilation of nutrients in kg/y, M_{inf} is the mass of nutrients entering a MWWTP with the influent in kg/y, NRE is the nutrient recovery efficiency (0–100 %), NUE is the nutrient uptake efficiency (0–100 %).

Suppose two MWWTPs are compared, one with a phosphorus recovery efficiency of 54 % as discussed in Blöcher et al. (2012) and another with 99 % discussed in Gong et al. (2018). While the NUE of the fertilizer product recovered from MWWTP1 is only 20 %, that of the product from MWWTP2 is 90 %. Suppose both MWWTPs have an inflow of 1000 kg P/month.

$$\mathrm{BA}_{\mathrm{MWWTP1}} = 1000 \times 0.54 \times 0.20 = 108\ \mathrm{kg}$$

$$\mathrm{BA}_{\mathrm{MWWTP2}} = 1000 \times 0.99 \times 0.90 = 891\ \mathrm{kg}$$

Through a combination of a high NUE and NRE, MWWTP2 leads to a much higher BA compared to MWWTP1.

3.3. Carbon cycling through soil organic matter addition

Rising greenhouse gas emissions have led to an increase in the average earth's surface temperature of 1.1 °C compared to the late nineteenth century (Viswanaathan et al., 2022). The negative effects of climate change in the form of extreme weather conditions such as more frequent heat waves, droughts, floods, and wildfires are evident. It is clear that even reaching net zero GHG emissions will only stabilize the warming and not reverse the damage nor eliminate the risks already caused by the risen temperatures (Rogelj et al., 2019). This proves the need for carbon sequestration.

Soil organic carbon (SOC) restoration can play a significant part in reversing climate change (Lehmann et al., 2020; Sommer and Bossio, 2014). About 26 % of the SOC is estimated to have been lost from the top 30 cm of the soil globally due to land use changes (Sanderman et al., 2017) as a consequence of an increasing rate of organic matter volatilization due to higher temperatures (Lugato et al., 2021). Accordingly, the soil in the EU countries is declining in OM (Ferreira et al., 2022; Lugato et al., 2014). In the Netherlands, the trend of the soil organic matter (SOM) differs between regions (Hanegraaf et al., 2009).

The addition of SOM can sequester SOC subject to the local environmental conditions (Navarro-Pedreño et al., 2021). The most crucial factor for carbon sequestration is the stabilization of the OM. (Navarro-Pedreño et al., 2021). For example, sewage sludge biochar can be stored in soil for a much longer time than untreated sludge (Zhao et al., 2023). The degree of stabilization of the wastewater-derived organic matter can vary significantly (Bożym and Siemiątkowski, 2018; Sánchez-Monedero et al., 2004), and must be accounted for. The SOM sequestration benefit is here defined using the following equation:

$$SS = (1 - VS/100) \times OM_{soil} \tag{5}$$

where SS is the SOM sequestration in kg/y, VS is the volatile solids content of the recovered product (%) representing the labile carbon fraction, and OM_{soil} (kg/y) is the mass of organic matter applied to the soil.

Assuming 1000 kg of sludge produced by MWWTP1 and MWWTP2 is to be used for soil application. Whereas MWWTP1 employs anaerobic digestion, MWWTP2 employs aerobic digestion for sludge stabilization. The VS destruction achieved by both processes is assumed to be the same based on Metcalf and Eddy and AECOM (2014): At 50 %. The OM percentages of the digested sludge products are 49.3 % and 71.6 % for MWWTP1 and MWWTP2 respectively as in Černe et al. (2019). Then, the SS values of the two MWWTPs are as follows:

$$SS_{MWWTP1} = (1 - 0.50) \times 493 = 246.5 \text{ kg}$$

$$SS_{MWWTP2} = (1 - 0.50) \times 716 = 358 \text{ kg}$$

Thus, the MWWTP2 achieves a larger SS due to a higher percentage (71.6 %) of OM in the sludge product.

As shown in this section, the method can calculate three nature benefit indicators based on the mass flows of the recovered resources through the MWWTP and the nature compartment where the resource is applied (freshwater streams, soil, or biomass). To ascertain the required mass flows, models or literature data can be used. A schematic diagram of the method is shown in Fig. 2.

3.4. Case study

To demonstrate the new method, a MWWTP planned for construction in Wilp, the Netherlands is used. The mass flows are based on a pilot study which is presented in Stowa (2023). To further understand the technical innovations of this treatment plant, the reader is directed to the same report.

Wilp treats the wastewater using predominantly physio-chemical processes in contrast to the biological processes most commonly employed in the Netherlands. Only the sludge is biologically treated using anaerobic digestion. This is a novel type of MWWTP but, the method introduced is generally applicable. A schematic diagram of the MWWTP is shown in Fig. 3. As can be seen, the MWWTP uses such processes as sieving, electro-coagulation (EC), dissolved air floatation (DAF), nanofiltration, and ion-exchange to recover the resources present in the wastewater while simultaneously meeting the national effluent quality standards. The MWWTP concept has been successfully tested in a pilot and a full-scale plant with a capacity of 100,000 p.e. is in planning for construction. The treated effluent of the full-scale plant will be discharged into the IJssel River.

Currently, different recovery processes for P and organic matter are being explored in the Wilp MWWTP. Thus, this study uses a base case of the MWWTP with only the recovery of N as ammonium sulphate. The first scenario (Scn. 1) includes the recovery of N as ammonium sulphate, OM in the forms of cellulose fibres using a fine sieve and anaerobically digested sludge for soil application. The second scenario (Scn. 2) has an additional recovery of P in the form of struvite from the ash of the incinerated sludge. The third scenario (Scn. 3) includes the recovery of N as ammonium sulphate, OM as cellulose fibres and sludge digestate, and P recovery as vivianite using magnetic separation. These scenarios were chosen by the authors and the case study owners. However, the method presented is generally applicable to other resources recovery scenarios.

3.4.1. Mass flows

The mass flows of organic matter, P, and N are calculated using the Substance flow analyser (STAN) (Cencic, 2008) software developed by TU Wien based on the information provided by the case study owners.

First, the organic matter (COD) mass flows are described here. In the base case, about 30 % of the influent COD is transferred from the water to the sludge phase using a drum and a fine sieve. The sieves mainly recover cellulose which constitutes about 30 % of the influent COD (Reijken et al., 2018). Following this, about 22 % of COD entering the EC-DAF process is separated from the water phase through coagulation. Further, a nanofiltration unit removes nearly 85 % of the COD. The remaining COD in the water passes through the ion exchanger and gets discharged with the effluent. The sludge undergoes anaerobic digestion where about 65 % of the influent COD gets converted into biogas according to Wan et al. (2016). The remaining COD is incinerated for energy production.

In Scn. 1, the cellulose is separated from the water through sieving and used in construction. The EC-DAF sludge undergoes anaerobic digestion which leads to the COD being transferred into a gas fraction containing CH_4 and CO_2 and a solid digestate which is assumed to be applied to agricultural land as soil amendment. The mass flows of the OM in this scenario (blue arrows) as well as the base case are shown in Fig. 4.

Next, the mass flows of N are presented in Fig. 5. Approximately 10 % of the total nitrogen (TN) is removed by the nanofiltration process and 98 ± 2 % is removed by the ion-exchanger. This removal efficiency of the ion-exchange process falls within the commonly cited range of 80–100 % (Feng and Sun, 2015; Huang et al., 2020; Sica et al., 2014). The N is recovered in the form of ammonium sulphate upon the resin regeneration using sulphuric acid and is used as an agricultural fertilizer. The BA was calculated based on the average NUE of N fertilizers in Dutch agriculture, which is approximately 48 % (CBS, 2022).

Next, the P mass flows are described and visualized in Fig. 6. In the base case, most of the P is removed from the water through the EC-DAF process. A 98 ± 1 % recovery efficiency is achieved for P using the EC-DAF process which falls within the 97–98 % range reported in the literature (Bhoi et al., 2023; Inan and Alaydin, 2014; Yang et al., 2022). The recovered P is part of the sludge and passes through the anaerobic digestion process. About 10 % of the P is assumed to be discharged with the digestion supernatant. 90 % enters the sludge incinerator with the digestate. Nearly 100 % of the P in the digestate ends up in the incinerator ash (Petzet et al., 2011) which is used in road construction.

In Scn. 2, along with the OM recovery, the P is recovered as struvite ($NH_4MgPO_4 \cdot 6H_2O$). Firstly, the influent P is incorporated into sludge

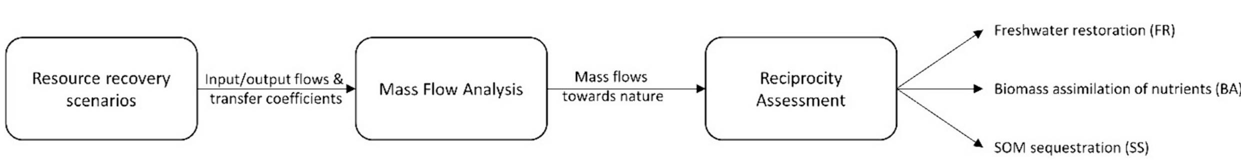

Fig. 2. A schematic diagram explaining the connection between the resources recovery scenarios, the mass flow analysis of the resources and the reciprocity assessment indicators.

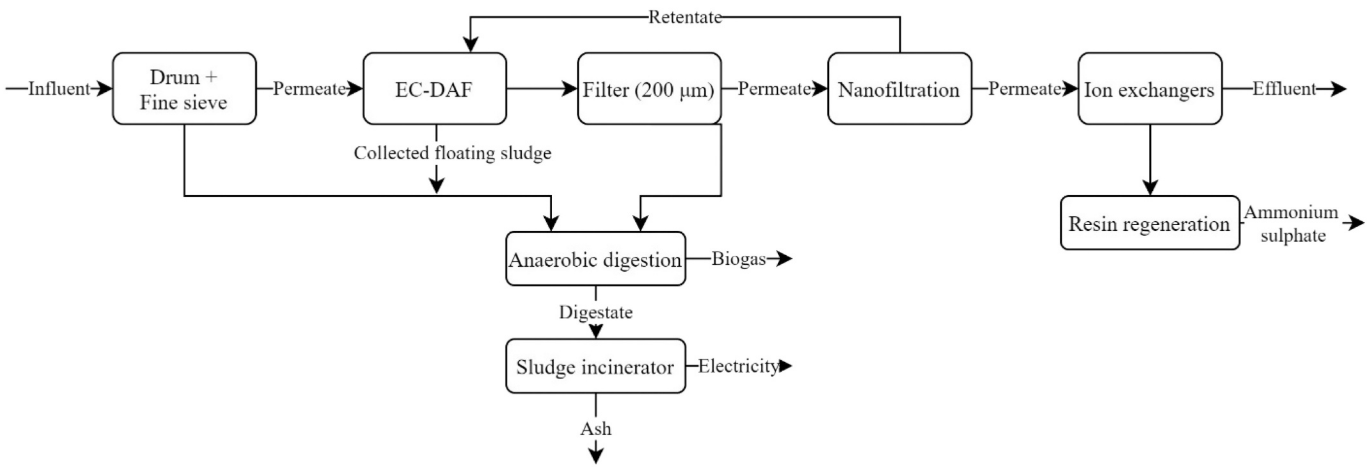

Fig. 3. A schematic diagram of the Wilp MWWTP based on the physio-chemical treatment of domestic wastewater (EC-DAF: electro-coagulation dissolved air flotation).

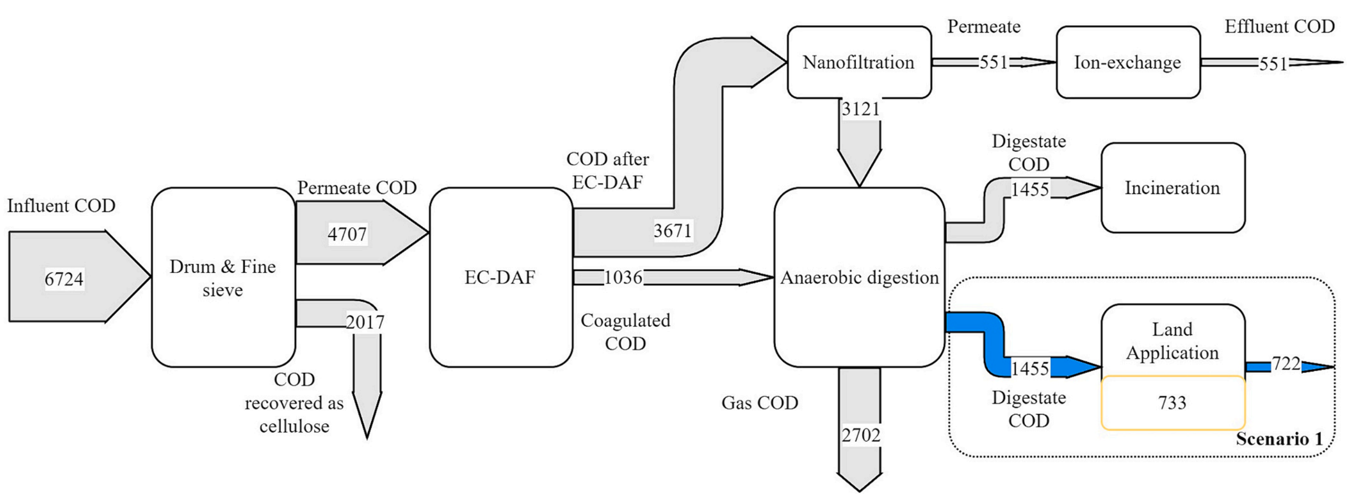

Fig. 4. The mass flows of organic matter through the Wilp MWWTP in the base case in Scn. 1, expressed in $\times 10^3$ COD kg/year. The blue coloured arrows represent Scn. 1.

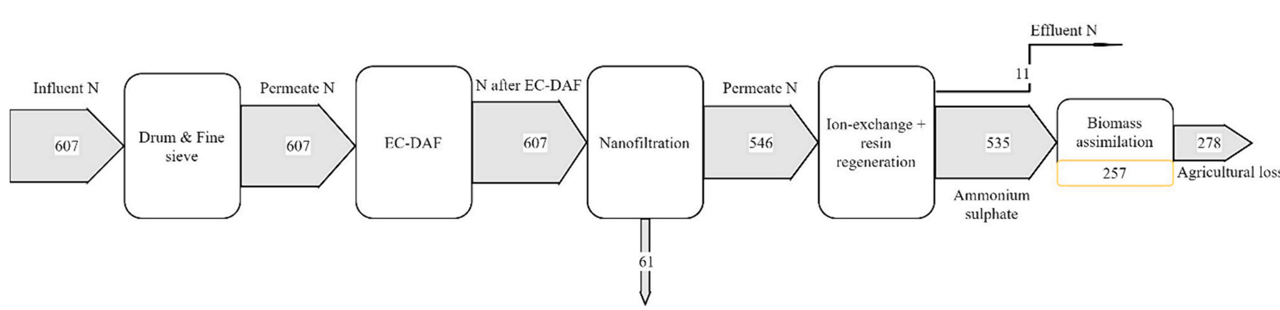

Fig. 5. The annual mass flow of total nitrogen (TN) through the Wilp MWWTP expressed in $\times 10^3$ kg/y (EC-DAF: electro-coagulation dissolved air flotation).

using the EC-DAF process. 99 % of the P present in the sludge gets transferred to the anaerobic digestate which goes to an incinerator. From the incinerator ash, acid leaching is used to recover struvite. A recovery efficiency between 80 % to 95 % can be found in the literature (Krüger and Adam, 2014; Petzet et al., 2011; H. Xu et al., 2012). In this study, a recovery efficiency of 90 ± 5 % is assumed. The NUE of struvite is assumed to be 80 % which is the average for P-fertilizers in Dutch agriculture (CBS, 2022).

In Scn. 3, instead of recovering P from the sludge ash, magnetic separation is used to recover vivianite ($Fe_3(PO_4)_2 \cdot 8H2O$) from the digestate. The efficiency of P recovery as vivianite using magnets is about 60–64 % of the total influent P (Wijdeveld et al., 2022). Here, a recovery efficiency of 64 ± 5 % was assumed. Although vivianite has been reported to have a lower NUE compared to struvite (Ayeyemi et al., 2023), more research is needed to draw stronger conclusions. For now, an uptake efficiency equal to struvite (i.e., 80 %) was assumed.

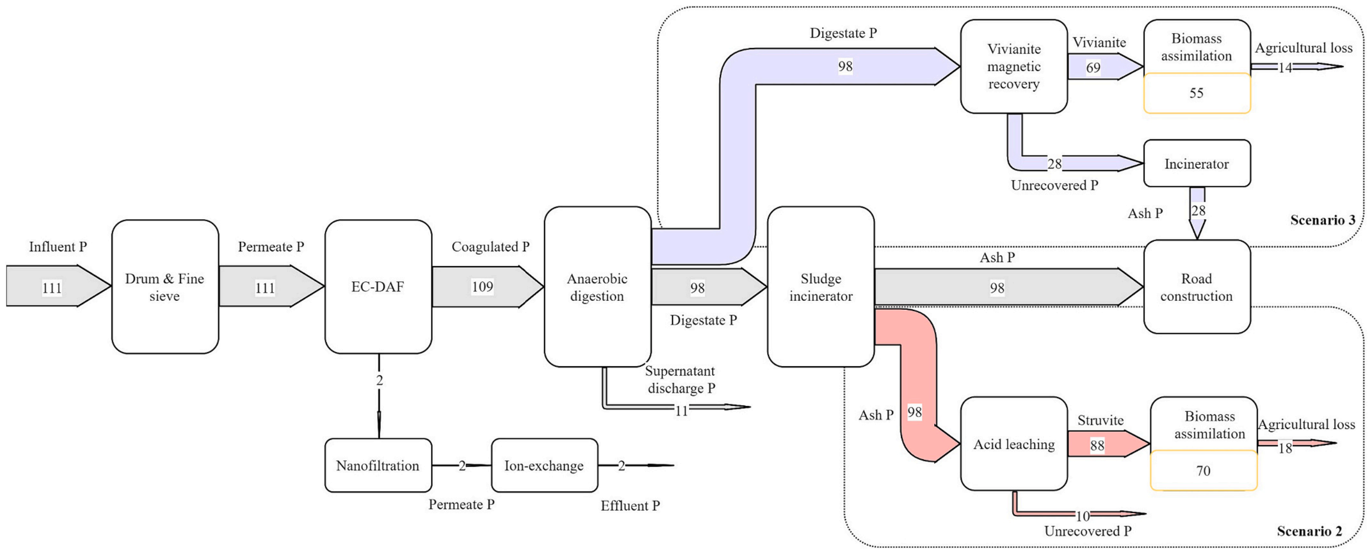

Fig. 6. The annual flow of phosphorus through the Wilp MWWTP Scn. 2, expressed in $\times 10^3$ kg TP/y. The Scn. 3 flows are shown in light blue and the red coloured arrows show the flows of Scn. 2.

3.4.2. *Nature reciprocity assessment*

3.4.2.1. Freshwater restoration. For all three scenarios, the FR is equal because the effluent quality remains constant. To calculate the FR, the GWF is estimated for the different pollutants. While the GWF of the COD is 1.18×10^7 m^3/month, those of TN and TP are 3.85×10^5 m^3/y and 4.81×10^5 m^3/y respectively. Hence, the organic matter was found to have the largest GWF. This GWF was divided by the monthly streamflow of the River IJssel. River Ijssel is a distributary of the River Rhine (Hurkmans et al., 2022) and hence their discharges are correlated. The IJssel discharge was obtained using the Rhine discharge (based on Booij (2017)) and the empirical relationship of Hurkmans et al. (2022). The ratio between the monthly GWF of organic matter and the monthly river runoff was calculated. Based on this monthly WPL and TW discharge, the FR was calculated using Eq. (3). The calculation can be found in the supplementary material (Tables S5, S6, and S7).

3.4.2.2. Biomass assimilation of nutrients. To calculate the nutrient assimilation, the removal efficiencies of the different treatment steps were obtained from the case study owners and verified using the literature. The inflowing N and P masses were calculated by multiplying the influent wastewater volume with the nutrient concentrations. 6.07×10^5 kg N/y and 1.11×10^5 kg P/y were the inflowing mass flows of the nutrients. Based on the MFA, the recovery efficiency of N was found to be 88 % under all scenarios. The recovery efficiency of P (Struvite) was found to be 79 % in Scn. 1. In Scn. 2, a 63 % recovery efficiency was found for P (Vivianite). Further, the NUE of N and P were assumed to be 48 % and 80 % respectively based on CBS (2022). Thereafter, the BA of nutrients was estimated for N and P separately using Eq. (4). For detailed calculations, the reader is directed to the supplementary material.

3.4.2.3. Soil organic matter sequestration. Here, it was assumed that the volatile organic components would be lost in a short time upon soil application. Therefore, only the non-volatile organics were assumed to be sequestered. The volatile component remaining in the sludge after anaerobic digestion was estimated using the Liptak equation that estimates the volatile solids reduction (Dagnew and Parker, 2021; Metcalf and Eddy and AECOM, 2014) as shown below:

$$VS_{reduction} = 13.7 \times Ln(SRT) + 18.9 \quad (6)$$

where SRT is the solid retention time of the anaerobic digester.

An SRT of 10 days was assumed and using Eq. (6) a 50.4 % reduction in the volatile solids was found. The quantity of OM that can be applied to soil was estimated to be 1.45×10^6 kg/y using the MFA. The SS was calculated using Eq. (5) and the calculations are shown in the supplementary material. Note that the calculations made here are specific to this case study, this would be different for other technologies such as aerobic digestion, composting, and incineration.

3.4.3. *Uncertainty analysis*

The uncertainty in the reciprocity indicators could be caused by several factors. The exact NUE will depend on a lot of parameters including the farming practices. The VS in the soil application product can be determined experimentally but often may be estimated by equations which introduce certain uncertainty. Likewise, the recovery efficiencies used in the case study were based on a pilot study and can vary for the full-scale plant. Therefore, an uncertainty analysis was conducted to provide a range for the reciprocity indicators based on the variation in the input parameters.

In STAN, all uncertain inputs are normally distributed with a mean and a standard deviation that can be specified by the user (Laner et al., 2014). The mean entered usually originates from literature or an educated guess and not from a data sample making the nature of the uncertainty epistemic and not random. Consequently, STAN converts the entered standard deviation into the standard error of the mean (SEM) using the following equation.

$$\sigma_X = \frac{\sigma}{\sqrt{N}} \quad (7)$$

where σ_X is the standard error of the mean, σ is the standard deviation specified by the user, and N is the number of data points.

Since only the lower and the upper boundaries of the transfer coefficients are specified here based on literature and an educated estimation of the case study owners, the number of data points (N) in this study is 2. STAN then makes use of the Gaussian error propagation (GEP) method for calculating the resulting uncertainties (Laner et al., 2014). For more details about the GEP method, the readers are directed to Lo (2005).

3.4.4. *Sensitivity analysis*

A sensitivity analysis was used to evaluate the effect of changing the parameters, such as the NRE and the river discharge. For the FR sensitivity, 20 % lower and 20 % higher COD loads in the MWWTP effluent were used. Also, 20 % higher and 20 % lower river discharges were evaluated. To analyze the BA sensitivity, the N recovery efficiency of 88 % estimated by the case study owners was changed by 10 % in both directions. A range of N and P recovery and uptake efficiencies were used, as shown in Tables S11 and S12. Lastly, for analysing the SS sensitivity, the VS content was modified to 30 % and 70 %, along with the original value of 49.6 %.

4. Results

4.1. Reciprocity indicators

The nature reciprocity indicators along with their uncertainties are presented in Table 1. The FR remains the same for the four scenarios and is equal to 7.53 × 10^6 m^3/y because the effluent concentrations remain constant. The OM resulted in the highest GWF (1.18 × 10^7 m^3/month). However, compared to the runoff of the river IJssel (9.77 × 10^8 m^3/month on average), the GWF of the Wilp effluent was insignificant. Consequently, the WPL was very small. Therefore, a large portion (~98 %) of the discharged water (7.53 × 10^6 m^3/y) can be considered as freshwater restored into the river.

Through the recovery of ammonium sulphate and its application in Dutch agriculture, a biomass N assimilation of 2.57 ± 0.04 × 10^5 kg/y is achieved. Since only one pathway of N recovery is used, the biomass N assimilation remains the same for all the scenarios.

In the base case, no P is recovered and consequently, the biomass assimilation of P is 0. In Scn. 2, the biomass assimilation of P was found to be 7.03 ± 0.28 × 10^4 kg/y. In comparison, a value of 5.55 ± 0.31 × 10^4 kg/y was estimated for Scn. 3. Thus, the recovery pathway and the form of P recovered (Struvite or Vivianite) can substantially affect the biomass assimilation.

In the base case, the OM was not recovered and thus the SS is equal to 0. In the other three scenarios, the SS was 7.34 ± 0.75 × 10^5 kg/y. The SS for the base case was found to be 0 because the OM was partly used for biogas production and the rest was incinerated with the ashes being used in road construction. In the other three scenarios, part of the COD was recovered as cellulose fibres and used in construction. This part of the COD did not contribute to the SS. Another part of the COD was converted to biogas which also did not contribute. However, the digestate applied to the soil led to the sequestration of about 50 % of the total OM.

Considering nature benefits, Scn. 2 is the preferred scenario among the four. This is because it provides the same FR and BA of N as the other alternatives and leads to an SS equal to Scn. 1 and Scn. 3. However, Scn. 2 provides the highest BA of P.

Fig. 7 visualizes the reciprocity of the base case and the three alternative scenarios using a spiderweb chart. The indicator value of each scenario has been normalized to that of the best performing one in that category (e.g., Scn. 2 for the biomass assimilation of nutrients) which is represented by 100.

The reciprocity indicators can be contrasted with the environmental

Table 1
Wilp MWWTP base case compared to three different resources recovery scenarios. The uncertainty values shown were calculated as explained in Section 3.4.3.

Reciprocity indicators	Base case	Scn. 1	Scn. 2	Scn. 3
Freshwater restoration (×10^6 m^3/y)	7.53	7.53	7.53	7.53
Biomass assimilation of nitrogen (×10^5 kg/y)	2.57 ± 0.04	2.57 ± 0.04	2.57 ± 0.04	2.57 ± 0.04
Biomass assimilation of phosphorus (×10^4 kg/y)	0	0	7.03 ± 0.28	5.55 ± 0.31
Soil organic matter sequestration (×10^5 kg/y)	0	7.34 ± 0.75	7.34 ± 0.75	7.34 ± 0.75

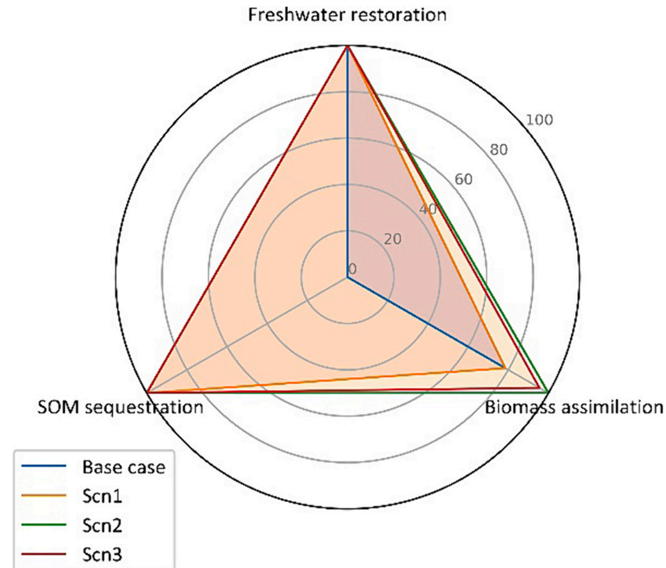

Fig. 7. A comparison of the four scenarios on their nature reciprocity performance. The reciprocity values of the alternatives have been normalized relative to the highest value for that indicator (represented by 100).

damage type indicators commonly used. The study of Tarpani et al. (2020) can be used to contrast the two types of assessments. Tarpani et al. (2020) conducted an LCA to compare treatment methods to recover sewage sludge for different applications. They found the climate change potential of applying anaerobically digested sludge to agricultural soil to be −174 kg CO_2 eq./1000 kg DM. This value results from adding the CO_2 emissions of the electricity used in the anaerobic digestion and the CH_4 emissions after the digested sludge is applied to soil and subtracting the avoided burden of manufacturing industrial fertilizers. The avoided climate change impact of manufacturing fertilizers outweighs the impact of digesting the sludge and applying it to the soil, which led to a negative value for the climate change potential.

However, while an avoided burden can lead to negative values, this is distinct from the physical removal of pollutants from the environment (Tanzer and Ramírez, 2019). The negative value from avoided burden represents a potential reduction in carbon emissions. In contrast, the SS benefit indicator represents the carbon in the wastewater that is sequestered in the soil. While the LCA indicator measures the reduced environmental damage (a result of reduced greenhouse gas emissions), the benefit indicator measures a positive effect on the carbon cycle and the soil environment by physical SOM sequestration.

The reciprocity assessment can lead to different conclusions than an LCA. For example, there could be a treatment option with a higher energy/resource use that is able to contribute much more positively to nature e.g., implementing the nutrient recovery technologies are known to usually increase the global warming potential of MWWTPs (Mayer et al., 2021; Pausta et al., 2024; Pradel and Aissani, 2019) but they also enable the restoration of nutrient cycles which needs to be included in the sustainability discussion. In a study by Xu et al. (2014), anaerobic digestion followed by incineration was found to have a lower negative environmental impact compared to agricultural application of the digested sludge. However, basing a decision solely on lowering the negative environmental impacts may lead to ignoring the potentially positive effects of the agricultural application of the digested sludge in that case. Including the positive effects in the conversation may lead to the decision of using the digested sludge for agricultural application while trying to reduce the negative impacts of doing this (that is revealed by the LCA). In this way, the reciprocity assessment proposed here can be seen as a complementary tool to the LCA.

4.2. Sensitivity analysis

In Table 2, the percentage changes in the BA values are shown when the nitrogen recovery efficiencies and the nitrogen uptake efficiencies are changed from the case study values of 88 % and 48 %. This is an example and the complete sensitivity analysis can be found in the supplementary material (Tables S10–S13).

Modifying the COD load of the effluent by 20 % and the River IJssel discharge by 20 % (Table S10) did not have any significant effect on the FR of the MWWTP. This was because of the large flow of the River Ijssel (1.19×10^{10} m^3/y) in comparison to the MWWTP discharge (7.63×10^6 m^3/y).

For N (as shown in Table 2), a recovery efficiency value of 88 % is already high. Improving it to 98 % can lead to an improvement in the BA by about 11 %. However, a much higher improvement in the BA can be achieved by increasing the NUE, which lies around 48 %. Whereas increasing the NUE to 60 % can lead to an increase in the BA by 25 %, increasing it to 80 % (equal to that of P), can lead to a BA improvement of 67 %. An 86 % increase in BA is possible when both a high N recovery efficiency of 98 % and a high N uptake efficiency of 80 % are achieved.

For P (Table S12), which already has a high NUE in Dutch agriculture (~80 %), improving the NUE to 90 % has a limited effect on the BA, improving it by 13 %. On the other hand, improving the P recovery efficiency from the MWWTP from 79 % to 90 % can improve the BA by 43 %. High recovery and uptake efficiency values of 90 % can improve the BA by 61 %.

Analysing the sensitivity of the soil carbon sequestration values (Table S13), decreasing the VS content of the soil amendment products by 20 % can improve the SS by 40 %.

5. Discussion

5.1. Reciprocity assessment

The novel method helps quantify potential positive effects on the natural environment from the resources recovered from MWWTPs using a life cycle approach. The indicators are calculated using parameters related to a MWWTP (e.g., recovery efficiency) and also those related to the application of the resources (e.g., NUE for nutrients). This will encourage decision-makers to think about the resources recovery solutions down to the application process and thereby help prevent burden shifting. Moreover, the assessment relies on data such as recovery efficiencies and OM content that are easily available/calculable for decision-makers. Thus, a major advantage of this method is the ease of calculation. Certainly, more complex models can be developed to calculate the indicators but, the method captures sufficient details to differentiate between the different resources recovery options (e.g., vivianite or struvite recovery). The different resources scenarios that the Wilp MWWTP adopts can notably vary the kind and extent of the nature benefits.

The method was demonstrated on a novel type of MWWTP that relies mostly on physio-chemical treatment. However, MWWTPs can have a variety of configurations and the method proposed here can easily be applied to these as well. This is because the method is generic, i.e., independent of the treatment process involved and can be used as long as the mass flows of the relevant resources can be calculated.

Table 2
The sensitivity of the biomass assimilation of N to changes in the nitrogen uptake and recovery efficiencies. The percentage changes are relative to the case study values of 88 % recovery efficiency and 48 % uptake efficiency.

		Nitrogen uptake efficiency			
		30 %	**48 %**	**60 %**	**80 %**
Nitrogen recovery efficiency	**78 %**	−45 %	−11 %	11 %	48 %
	88 %	−38 %	0 %	25 %	67 %
	98 %	−30 %	11 %	39 %	86 %

Wilp can restore 7.53×10^6 m^3/y (92 % of the influent) of freshwater into the IJssel River. The discharge water had a very low WPL (monthly average of 1.23×10^{-2}). Consequently, a negligible portion of the annual streamflow is required to dilute this effluent. Therefore, the FR achieved by the MWWTP is almost equal to the discharged effluent. Two important observations should be made. First, the low WPL shows that focussing on the recovery of OM and nutrients from wastewater leads to effluent quality with a very low WPL. Second, the high streamflow rate (9.77×10^8 m^3/month on average) of the IJssel River is an important factor leading to a high FR value. The FR values were sensitive to neither the COD load in the MWWTP effluent nor the Ijssel flow rate (± 20 %). High removal rates of the organic matter and the nutrients help to restore natural stream flows and maintain their quality.

Using the recovered N from Wilp, $2.57 \pm 0.04 \times 10^5$ kg TN/y can be assimilated into plant biomass. Furthermore, $7.03 \pm 0.28 \times 10^4$ kg TP/y and $5.55 \pm 0.31 \times 10^4$ kg TP/y can be assimilated in scenarios 2 and 3. The nutrients excreted by humans would be dissipated in the natural environment (soil and water bodies) unless collected from the domestic sewage and recycled. By actively sequestering the nutrients into plant biomass, a MWWTP can provide a crucial nature benefit.

The BA achieved by a MWWTP depends on the efficiency of the nutrient recovery technology and the nutrient uptake of the fertilizer. The P recovery from the ash after incineration offers a higher recovery efficiency (~80 %) compared to the vivianite recovery using magnetic separation (~64 %). Furthermore, struvite fertilizers have a higher P uptake efficiency and thus contribute to better cycling of nutrients compared to conventional P fertilizers (Li et al., 2019; Uysal et al., 2014). Therefore, to remove the excess reactive nutrient species from the natural environment, two factors are essential. The NRE as well as the NUE of the recovered nutrients must be high. Furthermore, to increase the BA of N, improving the N uptake efficiency in agriculture should be the focus. In contrast, P already has a relatively high uptake efficiency and thus the focus should be more on achieving high recovery efficiencies. This study showed that struvite recovery from sludge ash is the most promising pathway from the perspective of biomass assimilation, as also noted by Egle et al. (2016).

SOM restoration can improve desirable soil properties and also help to sequester carbon to mitigate climate change. However, in the Netherlands, there is no clear trend in the SOM and careful consideration is needed to decide where to use the sludge-derived products. Assuming the location-suitability, an addition of $7.34 \pm 0.75 \times 10^5$ kg/y SOM can be achieved by Wilp after the anaerobic digestion of its sludge. From a climate change perspective, this number is likely to be insignificant to offset the CO_2 emissions. However, even a small addition of SOM can have a significant positive effect on the local environment and agricultural productivity in soils with declining organic content (Hanegraaf et al., 2009). Furthermore, a sludge stabilization method other than anaerobic digestion may retain more biodegradable organic content that can be applied to the soil. Decreasing the VS content of the sludge product can increase the carbon sequestered in the soil. Thus, where higher organic matter sequestration is required, a sludge stabilization process can be selected that retains more organic matter, such as lime stabilization (Yoshida et al., 2018).

It is important to clarify that providing nature benefits alone does not qualify a MWWTP as sustainable. The chemical use, energy consumption, and emissions of the MWWTPs are crucial factors also to be considered and an LCA can help assess these. The Wilp is a predominantly physio-chemical MWWTP which has both disadvantages and advantages when compared to a conventional activated sludge MWWTP. An LCA conducted by Stowa (2023) reports that Wilp uses more electricity (~2.5 times more) and higher dosages of chemicals and raw materials than conventional activated sludge MWWTPs. However, the Wilp MWWTP also has zero direct emissions of CO_2 and N_2O, which implies that higher proportions of OM and N present in the wastewater can be used for benefiting nature through the mechanisms discussed in this paper.

5.2. Limitations

Resources recovered from wastewater contain heavy metals, pathogens, and organic micropollutants that also damage the natural environment. This study ignores their presence because the focus was on assessing the nature benefits. For assessing the negative effects of such substances, damage units developed by Egle et al. (2016) can be used.

When working with nutrient uptake efficiency, the values can vary widely based on the type of fertilizers, soil characteristics, climate, and agricultural practices. In this study, the average efficiency values were obtained for the Netherlands and a sensitivity analysis was used to cover a certain range of the uptake efficiencies. However, more studies are needed to quantify the improved nutrient use efficiencies of slow-release fertilizers obtained from wastewater.

In this study, one of the resources recovery pathways included the use of a sludge-derived product as a soil amendment. In the Netherlands, sewage sludge products are not applied to agricultural soils (Racek et al., 2020). However, this pathway was included because soil application of sludge (and its derived products) is practised in many parts of the world including other EU countries (Hudcová et al., 2019).

6. Conclusions

In this paper, a novel method was developed to assess the potential nature benefits of the resources recovered from wastewater. The following conclusions can be drawn:

- The proposed method works well to quantify key benefits to the natural environment from wastewater-based resources from a life cycle perspective. In the planning and assessment of the resources recovery solutions, the focus should not be limited to reducing the negative environmental impacts. Instead, the nature benefits that can be obtained through the recovered resources should be included in the overall sustainability assessment.
- Focussing on maximizing the recovery of the organic matter and the nutrients in domestic wastewater can significantly improve the effluent quality. The discharge of treated effluent into a stream with a high dilution capacity due to high flow rates can help restore the quality and quantity of freshwater in nature.
- MWWTPs also help to transform the waste nutrients into fertilizers with high uptake efficiencies, thus contributing towards more effective biomass assimilation of nutrients. This can help to reduce the reactive nutrient emissions below the planetary assimilation limits.
- MWWTPs can also help to restore soil organic matter, which can mitigate climate change and improve soil quality. The stabilization of the organic matter achieved by the MWWTP will decide the extent of sequestration. Including the reciprocity assessment in the decision-making process can help uncover the advantages of certain resources recovery pathways that are not yet in practice.

In conclusion, the method proposed in this study is a start towards recognizing and quantifying the potentially positive role of humans in the natural environment through resources recovery solutions.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.spc.2024.03.016.

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