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A sustainable performance assessment framework for circular management of municipal wastewater treatment plants

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ABSTRACT

Municipal wastewater treatment plants (WWTPs) could become valuable contributors to a circular economy by implementing the 3R principles (reduce, reuse, and recycle). While reducing the pollution load of sewage is the primary objective of a WWTP, this process generates several potentially valuable byproducts including treated effluent, biogas, and sludge. The effluent can be reused in various end use applications and biogas can be reused as a fuel (for electricity generation, transportation, and cooking) or a chemical feedstock. The sludge can either be directly recycled as soil conditioner or via thermochemical/biochemical processing routes to recover material (e.g., hydrochar), energy (e.g., heat, and syngas), and resource value (phosphorus). This work presents a five-layered assessment framework for quantitatively evaluating the sustainable value of municipal WWTPs by using life cycle assessment (LCA) and life cycle costing assessment (LCCA) tools. In addition, indicators reflecting potential benefits to stakeholders and society arising from investments into municipal WWTPs such as the private return on investment (PROI) and the environmental externality costs to investment ratio (EECIR). The framework is validated in a hypothetical case study where the sustainable value of a circularly managed municipal WWTP is evaluated in situations involving multiple byproduct utilization pathways. Four future circular options (FCOs) are examined for a 50,000 m^3/d capacity WWTP treating sewage up to tertiary standards. The FCOs mainly differ in terms of how biogas is reused (to meet the WWTP's internal energy demands, as cooking fuel, or as fuel for city buses after upgrading) and how sludge is recycled (as soil conditioner or by producing hydrochar pellets for electricity generation). The FCO in which treated effluent is reused in industry, biogas is used as cooking fuel, and sludge is used as a soil conditioner provides the greatest sustainable value (i.e., the lowest private costs and environmental externality costs (EEC) together with high revenues), the highest PROI, and the lowest EECIR. The strengths and limitations of the proposed assessment framework are also discussed.

1. Introduction

The value of the global water and wastewater market was estimated to be USD 263.07 billion in 2020 and is projected to reach USD 500 billion by 2028 (Statista, 2021). Despite significant advancements in wastewater treatment technologies in recent decades, urban authorities in many cities continue to face significant challenges in wastewater management for three main reasons: (i) it is often impossible to channel the large volumes of wastewater from new urban settlements through existing centralized wastewater treatment plant (WWTP) infrastructure; (ii) existing municipal WWTPs may not efficiently remove contaminants of emerging concern (CECs); and (iii) the challenges of handling of sewage sludge are increasing.

In 2020, the proportion of sewage that was treated safely (i.e. in compliance with at least secondary standards) varied substantially across the world, from 25% to 80% (Alabaster et al., 2021). This range is so wide because of the different situations in developing and developed nations; the impact of urbanization on sewage management is a grave concern for developing countries, where sewage generation rates have

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Abbreviations

WWTP	Wastewater Treatment Plant
CEC	Contaminants of Emerging Concern
LCA	Life Cycle Assessment
LCCA	Life Cycle Costing Assessment
CBM	Compressed Biomethane
LBM	Liquefied Biomethane
MFC	Microbial Fuel Cell
MEC	Microbial Electrolysis Cell
AD	Anaerobic Digestion
BOD	Biochemical Oxygen Demand
IPR	Indirect Potable Reuse
DPR	Direct Potable Reuse
PE	Person Equivalent
BAU	Business As Usual
FCO	Future Circular Option
USD	United States Dollar
TPC	Total Private Costs
TPR	Total Private Revenues
PROI	Private Returns on Investment
GWP	Global Warming Potential
HH-NCP	Human Health-Non Carcinogenic Potential
HH-CP	Human Health-Carcinogenic Potential
PMFP	Particulate Matter Formation Potential
POFP	Photochemical Oxygen Formation Potential
AP	Acidification Potential
FEP	Freshwater Eutrophication Potential
FETP	Freshwater Ecotoxicity Potential
WSI	Water Scarcity Index
CED	Cumulative Energy Demand
EEC	Environmental Externality Costs
DALY	Disability Adjusted Life Years
EECIR	Environmental Externalists Costs to Investment Ratio

already surpassed the treatment capacities of existing WWTPs in many cities. The generation of high sewage volumes together with limited financial support for the creation of new WWTPs has often led to large direct discharges of raw sewage into fresh water systems (Suwarno et al., 2014; Wang et al., 2014). Some developed countries also are dealing with this problem. For example, in the United States, 2400 out of 16,000 functional WWTPs (15%) have already exceeded their treatment capacity, with the remainder operating at 81% of their maximum capacity on average (ASCE, 2021). While it is relatively uncommon for WWTPs to directly discharge untreated sewage into the environment, the frequency at which such incidents are reported has recently increased (Brown, 2020). Poor wastewater management practices have been reported to cause ~842,000 deaths globally, imposing an annual economic burden of USD 260 billion (IWA, 2016; UNESCO, 2017).

The rising concentrations of CECs in raw sewage have also become a significant problem because conventional WWTPs are not designed for their removal, necessitating the use of expensive and energy-intensive advanced tertiary treatment techniques such as electro-peroxone treatment (Mustafa, 2020), ozonation, and membrane filtration (Baresel et al., 2015a; Naturvårdsverket, 2018). The exact steps needed to achieve acceptable CEC levels in treated wastewater will depend on the types and levels of target and non-target compounds, which vary spatially and temporally (Tran et al., 2018).

The complexity of sewage sludge management has also increased in recent years. Global sewage sludge production is predicted to reach 127.5 million tons by 2030 (Ephyra, 2020). Such large quantities of sludge cannot be landfilled without causing environmental harm, nor would it be economically viable to do so. Sewage sludge can be used as a soil conditioner, but only if its contents of heavy metals and organic contaminants do not exceed legally mandated limits (SEPA, 2016). However, due to the lifestyle of urban populations, the heavy metal and emerging contaminant budget of WWTPs is constantly increasing.

Because of the challenges outlined above, the core philosophies of sewage treatment will have to evolve rapidly. In particular, there is a globally recognized need to shift from a conventional functionality-centric viewpoint to a holistic circularity-focused one. In a functionality-centric viewpoint, municipal WWTPs are seen as end-of-pipe solutions for reducing wastewater pollution whose main function is to treat sewage to adequate standards and discharge treated effluent into the environment. The side streams (biogas and sludge) have limited applications; biogas is consumed internally to meet the WWTP's energy needs and the sewage sludge is either landfilled or used as soil conditioner, providing modest economic returns. Conversely, in a circularity-focused view, municipal WWTPs must adhere strictly to the 3R principles of the circular economy, i.e., reduce, reuse, and recycle (Heshmati, 2015). The ultimate goal of a circularity-focused management approach is to enhance the sustainable contribution of municipal WWTPs to society. A circularly-managed WWTP therefore aims to: (a) at minimum treat sewage to meet regulatory standards and ideally completely eliminate its pollution load, including removing CECs; (b) reuse treated effluent in multiple ways depending on its quality, avoiding discharge to the environment; (c) reuse biogas as a source of energy, fuel, or chemical feedstock; and (d) recycle sludge as soil conditioner (if permitted) or extract material/fuel/energy/resource value by recycling the material thermochemically or biochemically.

Implementing the 3R principles in municipal WWTPs is a challenging process for cities and depends on three factors. The first is financial strength: the city must be able to afford a four-stage wastewater treatment process including primary, secondary, tertiary (nutrient removal), and advanced tertiary (CEC removal) stages. Ideally the city would be able to invest in advanced tertiary treatment technologies that produce a high-quality effluent that can even be reused in potable applications. The second factor is the availability of infrastructure for upgrading biogas into compressed/liquefied biomethane and recycling sludge through thermochemical or biochemical pathways. The third factor is establishing pathways for utilizing WWTP byproducts, i.e., recycling treated effluent, biogas, and dried biosolids into the technosphere. For example, the city could find ways of using treated effluent in irrigation or industry, while biogas could be reused as an energy source to meet the internal energy demands of the municipal WWTP or upgraded to biomethane for external use as transportation fuel. The complexity of these factors necessitates the development of an informed decision-making framework to help municipal authorities formulate strategies for improving circular management of WWTPs based on quantitative sustainability performance data.

This work therefore introduces a systems-oriented assessment framework for evaluating the sustainable value of a centralized municipal WWTP that is either partially or fully aligned with the 3R principles of the circular economy. The sustainable value of a municipal WWTP is a combined measure of its environmental and cost performance, which are quantified using life cycle assessment (LCA) and life cycle costing assessment (LCCA) tools, respectively. Because of the wide interest in increasing the sustainability of WWTPs and integrating them into a circular economy, a number of holistic assessment studies have been published in this area, mainly using LCA methods. Broadly speaking, these studies tend to focus on either *performance measurement* or *facilitating informed decision-making.* Examples of LCA studies focusing on performance measurement include works quantifying the environmental impacts of advanced tertiary treatment systems for removing emerging contaminants (Li et al., 2019; Rahman et al., 2018), evaluating the reuse of treated effluent as cooling tower makeup in industrial facilities (Theregowda et al., 2014) or for irrigation (Kanaj et al., 2021), comparing the use of biogas as an energy source to meet the internal energy demand of WWTPs to using it as vehicle fuel for freight transportation (Shanmugam et al., 2018), recovering phosphorus from sewage sludge (Egle et al., 2016), and producing hydrochar pellets from sewage sludge for use as biocoal to replace fossil coal for electricity generation (Mohammadi et al., 2020). Studies of the second type focus on the challenges associated with using LCA to make sustainable decisions about municipal WWTPs. Their key findings are relevant to the framework concept proposed in this paper and are discussed in more detail below. While these studies emphasize the importance LCA and LCCA-based decision-making methods for increasing the circularity and sustainability of municipal WWTPs, no assessment framework designed for this purpose has yet been presented.

We sought to address this research gap by developing an assessment framework that can be used as a decision support tool to formulate strategies for improved circular management of WWTPs based on their sustainable value. Compared to previously reported frameworks, ours has three key strengths. First, it is comprehensive in that its assessment of sustainable value accounts for every process and output of a WWTP including wastewater treatment to reduce pollution load, reuse of treated effluent, biogas reuse, and sludge recycling. Second, it is modular, meaning that each activity is addressed in a separate layer – for example, the effluent reuse layer is separate from the biogas reuse layer, so changes made in one layer (e.g., if the city decides to sell treated effluent to industry rather than using it for irrigation) do not affect other layers. Third, it prioritizes WWTP byproduct utilization pathways based on the overall sustainable value that is generated.

The rest of the paper is divided into five sections. The section immediately below presents a circularity-focused viewpoint of a municipal WWTP including a block diagram showing details of key processing pathways and final products. This is followed by a literature review of published systems-level studies (with an emphasis on LCA studies) on WWTPs in a circular economy. The proposed assessment framework is then described and validated in a case study. Finally, the conclusions of this work are summarized.

2. Circular viewpoint of a municipal WWTP

A block diagram outlining a circularity-focused view of municipal WWTPs is shown in Fig. 1. It includes the various wastewater and sludge treatment stages, in-house byproduct processing options such as upgrading biogas to compressed biomethane (CBM), external byproduct processing options such as sludge pyrolysis and incineration, and a list of economically valuable end products.

A four-tiered wastewater treatment process is deemed essential for reducing the chemical and biological pollutant load in the sewage (including CECs) to safe and acceptable levels. Urban sewage entering the WWTP first undergoes preliminary treatment to remove silt and grit. Suspended solids and biodegradable organics are then removed in a series of primary and biologically active secondary treatment stages. Secondary treatment steps include aerobic systems such as stabilization ponds, activated sludge process, sequential batch reactors, and moving bed biofilm reactors, as well as anaerobic systems such as anaerobic ponds and upflow anaerobic sludge blanket systems (Sperling, 2007). However, it is becoming increasingly common to use integrated secondary treatment systems that combine aerobic or anaerobic processes with a microbial fuel cell (MFC) (Hiegemann et al., 2016; Liu et al., 2011) or microbial electrolysis cell (MEC) (Katuri et al., 2019). While standalone MFC/MECs can be effectively used in small-scale, modular packaged plants (AQUACYCL, 2021) and decentralized systems (Foley et al., 2010), they typically fall short of the treatment capacities required by large centralized WWTPs. However, their integration into conventional municipal WWTPs could increase circularity in two ways. First, they can increase the energy self-sufficiency of municipal WWTPs because their electricity demand can be either completely or partially met using MFC/MECs in an integrated treatment setting (Hiegemann et al., 2016; Katuri et al., 2019). Second, municipal WWTPs can produce cleaner fuels like hydrogen and value added chemicals through integration with bioelectrochemical systems (Kaider et al., 2020).

Tertiary treatment methods including chemical precipitation with metal salts such as $FeCl_3$ and nitrification-denitrification or Anammox treatment are commonly used to remove phosphorus (Shewa and Dagnew, 2020) and nitrogen, respectively, from sewage (McCarty, 2018). Integrated treatment systems such as A^2O (Anaerobic/Anoxic/Oxic)

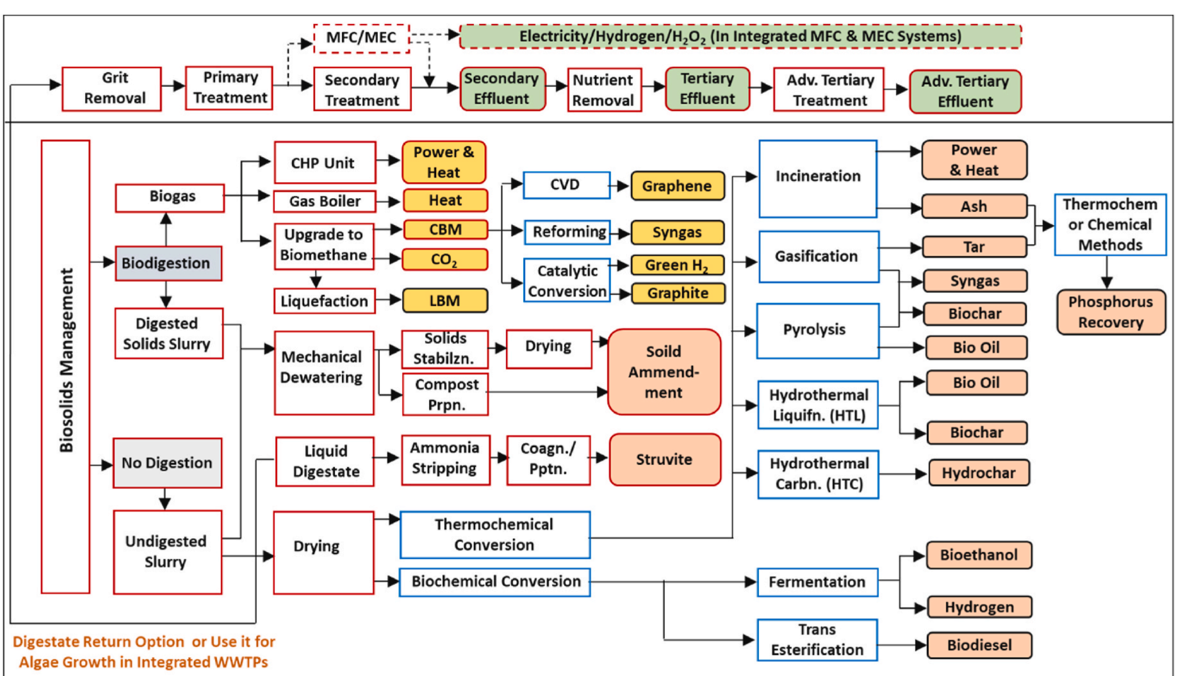


Fig. 1. A circularity-focused view of a circular municipal WWTP and the byproduct value chain.

Table 1A
Value added products directly obtained from municipal WWTPs and their applications.

Byproduct and Specific Application Route	Other Relevant Details and Supporting Examples of Byproduct Use by Cities
Treated Effluent for Industrial Use	• Kalasin Province in Thailand uses 48 million m^3/year of treated wastewater as makeup for power plant cooling towers (Chanamai and Phaowanich, 2018)
• For cooling tower makeup, process water, and boiler water for steam generation	• Singapore recycles its treated effluent for use in chemical and process industries (PUB, 2021).
Treated Effluent for Irrigation • Reuse in agriculture (Ungureanu et al., 2020), agroforestry (Moussaoui et al., 2019) and urban farming (Robbie et al., 2017). Effluent must meet strict quality criteria with respect to fecal coliform count, biochemical oxygen demand (BOD), salts contributing to the sodium absorption ratio of soils and concentration of emerging micro pollutants (Helmecke et al., 2020; Sanz and Gawlik, 2017).	• Globally, 7.1 billion m^3 of treated wastewater effluent is reused (5% of treated wastewater capacity and 0.18% of water consumption), with 50% being used for irrigation (Ungureanu et al., 2020)
Using Treated Effluent as Potable Water • Advanced tertiary effluent meeting potable water standards may be reused indirectly (Indirect potable reuse, or IPR) or directly (Direct Potable Reuse, or DPR).	• Some cities use an IPR strategy to introduce advanced tertiary effluent into groundwater aquifers or upstream of a river/reservoir. Windhoek in Namibia uses a DPR strategy, introducing effluent into a water treatment plant (Tortajada, 2020).
Treated Effluent for Commercial Use • Reuse within cities to water parks or in heating and ventilation systems, etc.	• Air cooling systems in Singaporean public buildings use advanced tertiary treated effluent (NEWater) (Tan, 2018)
Treated Effluent Aquifer Recharge	• Pathogen reduction is important for recharging (Levantesi et al., 2010).
• Recharged (via managed aquifer recharge/aquifer storage) recovery concepts.	• Other factors like arsenic immobilization may be important depending on the aquifer's geochemical characteristics (Fakhreddine et al., 2020).
Electricity/Fuel/Chemicals Production • Municipal WWTPs integrated with MFC/MEC can produce electricity, hydrogen, or H_2O_2 (Kaider et al., 2020).	• MFC modular systems as large as 1000 L have been successfully pilot tested suggesting real time integration possibilities in future (Liang et al., 2018).
Biogas as an Energy Source	• Marselisborg WWTP, Aarhus, Denmark uses electricity generated from biogas to power its water treatment plant (Karath, 2016)
• 1 m^3 biogas (60–65% CH_4 content) has an energy value of approximately 23.3 M/m^3 (Andreoli et al., 2007). WWTPs normally use biogas to meet their internal energy needs, but some WWTPs use electricity from biogas symbiotically.	• Singapore uses biogas from WWTP synergistically in its integrated waste management facilities (NEA, 2020).
Biogas as Cooking Fuel	• Investing in sewage biogas-based cooking infrastructure yields a positive net present value (NPV) (Mohammed et al., 2017).
• Biogas from municipal WWTPs is a good source of cooking fuel.	• Biogas from human waste is used for cooking in Nairobi (Shimanyula, 2014)
CBM and LBM as Transport Fuel	• Several Scandinavian and UK cities operate city buses on CBM obtained from WWTP either directly or co-digested with food waste (BioActiv, 2016; Kotimaki and Hallila, 2017)
• Biogas is upgraded to CBM and used as fuel, mainly for passenger transportation (cars and buses)	• Onsite LBM is currently used at two WWTPs in French cities (Suez, 2016)
• CBM can be converted to liquefied biomethane (LBM) and used as fuel for freight transport.	
CO_2 for Multiple Applications • Biogenic CO_2, a side stream from an upgradation plant, serves multiple industrial needs	
Liquid Anaerobic Digestate • Deammonified liquid digestate at pH 9.0 to 9.3 facilitates struvite precipitation (with recovery rates of 85–92% and 13–21% PO_4^3 and NH_4^+ ions). Struvite is a good alternative to slow-release mineral fertilizer (Tuszynska and Czerwionka, 2021).	• Struvite is commercially produced from AD centrate at some municipal WWTPs in Europe (StateofGreen, 2020)
Dried Digested Sludge as Biofertilizer • Land spreading as organic fertilizer provided its toxic contaminant load is within the safe limits.	

biological aerated filters (Sun et al., 2020; Wang et al., 2015) or removal through algal cultivation (Mohsenpour et al., 2021) are also used for combined removal of nitrogen and phosphorus. Fourth stage treatments are advanced tertiary treatment techniques meant to remove CECs including pharmaceutical compounds and antibiotic resistance genes using ozonation (Iakovides et al., 2019; Naturvårdsverket, 2018), electro-peroxone treatment (Mustafa, 2020), and membrane-based technologies (Heo et al., 2020), possibly in combination with granular activated carbon adsorption (Naturvårdsverket, 2018).

The treated effluent of municipal WWTPs can be reused in multiple ways depending on its quality. When treated to advanced tertiary standards using membrane filtration and ultraviolet disinfection, the effluent can be used in potable applications, either directly by being allowed to flow into drinking water treatment plants or indirectly by being allowed to enter drinking water reservoirs (Tortajada, 2020). Municipal WWTPs lacking advanced tertiary treatment infrastructure can directly reuse treated effluent in non-potable industrial applications (e.g., as cooling tower makeup or process water) or for irrigation. Reuse of secondary treated effluent is also common in WWTPs in some cities in developing nations to minimize regional freshwater withdrawal rates. However, secondary effluent reuse may be undesirable because of negative consequences including alteration of soil properties that may hinder the growth of certain crops (Paulion and Tonetti, 2021) and scaling or corrosion problems when used as industrial makeup water, which may necessitate increased use of chemicals (Dzombak et al., 2012).

Sewage sludge is predominantly managed using anaerobic digestion (AD) processes but may be handled without digestion in some municipal WWTPs. AD of raw sludge produces biogas and digested biosolids as byproducts. Most municipal WWTPs burn this biogas to meet their internal energy demands. However, external uses of biogas outside the boundary of the WWTP as a transportation fuel or a chemical feedstock after upgrading to biomethane are seen as promising sustainable alternatives. For biosolids, reuse as a soil conditioner is a simple and cost-effective option. However, biosolids may be landfilled if they cannot meet the strict regulatory requirements applied in many countries. Recycling via thermochemical or biochemical processing is preferable in the long run because it allows recovery of material/energy/fuel/nutrient value from biosolids, with the potential to replace non-renewable alternative value sources. For WWTPs without an AD facility, raw sludge must be recycled somehow to avoid landfilling. The value-added products that can be directly obtained from a municipal WWTP and their end use applications are summarized in Table 1A. Table 1B lists the value-added products obtainable after additional processing outside the WWTP boundary and their end use applications.

Table 1B
Value added products from municipal WWTPs requiring extended processing and their final applications.

Byproduct and Specific Application Route	Other Relevant Details and Supporting Examples of Byproducts Use by Cities
Electricity and Heat • Energy is recovered from incineration of dried raw or digested sludge	• 558–1068 kWh and 315–608 kWh can be recovered per ton of raw and digested sludge incinerated, respectively (Singh et al., 2020)
Production via Syngas • Biomethane (formed by upgrading biogas from WWTP) undergoes reforming (steam catalytic/ membrane) to produce bio syngas. • Gasification and pyrolysis of dried sludge (raw or digested) produces bio syngas. Bio syngas can be used as a source of energy or as a feedstock for producing biomethanol, bio-petroleum fuels, and bio hydrogen	• A commercial scale gasification plant is used to recover energy from sewage sludge in Tokyo, Japan, (BureauofSewarage, 2010)
Biological Phosphorus Recovery • Direct recovery from dried sludge via chemical leaching or from incinerated ash through chemical and thermo-chemical routes (Jossa and Remy, 2015; Meng et al., 2019). • Tar obtained from sewage sludge gasification contains 20 wt % of phosphorus (as P_2O_5), which can be extracted chemically.	• Saskatoon city in Canada has a commercial plant for phosphorus recovery from wastewater sludge (Ostara, 2013), and other cities including Amsterdam are on verge of commercialization (DutchWater, 2013).
Biofuels (Bio Oil/Biodiesel/ Bioethanol) Production • Fast pyrolysis of sewage sludge produces bio oil with properties equivalent to petroleum-based heavy fuel and superior to other bio oils (Djandja et al., 2020). • Dried sludge undergoes transesterification to produce biodiesel (Olkiewicz, 2015) and fermentation to produce bioethanol (Godoy et al., 2018).	• Daegu city municipality in South Korea (in collaboration with the Danish company SludgeX) is pursuing pyrolysis as an option to address its sewage sludge challenges (SludgeX, 2021).
Carbonaceous Materials	• Biochar/hydrochar has diverse applications (e.g., as a filler in polymer composites or a sorbent in water treatment)
• Biochar is obtained as byproduct from pyrolysis of sewage sludge. Alternately, hydrochar can be produced by hydrothermal carbonization (HTC) of sludge (Djandja et al., 2020; Lundqvist et al., 2020) • Sewage biogas is a potential renewable precursor for graphene production (CORDIS, 2019).	• From 2022 to 2026, 12 biographene production units using biogas as a precursor material will be installed across Europe (CORDIS, 2019).
Green Hydrogen • Bio syngas (from sludge gasification/ pyrolysis, biomethane, and treated wastewater (Stock, 2021) are promising sources for production of Green Hydrogen.	• Green hydrogen production via catalytic conversion of biomethane has been established on a commercial scale in Australia (Maisch, 2019).

3. LCA for circular management of WWTPs - literature review

Table 2 summarizes the findings of published studies explaining how LCA-guided decisions can be made to improve the circular economy prospects of municipal WWTPs.

4. Sustainable performance assessment framework

Accommodating at least some of the above suggestions, the five-layered assessment framework outlined in Fig. 2 is proposed to facilitate informed decision-making by cities aiming for better circular management of municipal WWTPs.

The assessment framework consists of five steps, which are described below.

4.1. *Describe business as usual (BAU) and future circular options (FCOs)*

The framework assumes that the municipal authority is seeking to change the way its WWTPs operate in order to adhere to the 3R principles and increase circularity. This may be voluntary or involuntary (e.g., it may be necessitated by new regulations or other constraints). To achieve this, stakeholders have three options: the city can modify the secondary treatment configuration of its existing WWTPs (e.g., by replacing conventional activated sludge systems with membrane bioreactors or installing advanced tertiary treatment facilities such as ozonation or membrane units. Alternatively, it can change its byproduct

Table 2
Key findings concerning the use of LCA to facilitate informed decision making on circular management of WWTPs.

LCA Stage	Key Findings
Functional Unit Definition Stage	• Volume of sewage treated is the preferred functional unit when comparing different treatment systems in the same geographical region, but person equivalents (PE) are preferred when comparing systems across regions (Byrne et al., 2017) • The functional unit must be defined to reflect specific conditions in the region being evaluated (e.g., consider 60 g BOD/PE and 40–50 g BOD/PE in developed and developing nations, respectively) (Schmid and Tarpani, 2019)
Life Cycle Inventory Modeling Stage	• Ambiguity exists in reported LCAs for WWTPs in developing nations because of limited LCI data on micropollutants and sludge management stages (e.g., impacts of heavy metals due to land spreading of sludge) (Schmid and Tarpani, 2019).
Life Cycle Impact Assessment Stage	• Environmental tradeoffs between globally focused impacts such as increases in GWP and localized impacts such as those pertaining to human health must be carefully weighed and factored in decision-making. For localized impacts, calculating relatively concrete outcomes in terms of damage done to the environment is encouraged. For this reason, integration of LCA with other tools such as quantified risk assessment is strongly recommended (Byrne et al., 2017). • Integrating LCA with tools like exergy (exergoenvironmental) analysis is recommended to account for thermodynamic inefficiencies and improve the accuracy of environmental impact assessments (Aghbashlo et al., 2021). • Exergoenvironmental analysis can be used to select thermodynamically sustainable pathways for using byproducts such as biogas, e.g. for combustion or in cooking (Banerjee and Tierney, 2011)
Results, Interpretation & Communication with Stakeholders	• Discuss only the most relevant impact categories. Explain results in simplified form using a graphical format and provide a comparative perspective (i.e., present multiple options with rankings) (Schneider et al., 2018) • Ecoefficiency is a reliable indicator used by authorities to make informed decisions (Alizadeh et al., 2020). The ecoefficiency of WWTPs can be calculated on the basis of emergy analysis, a combination of LCA and LCCA results, or a combination of LCA and data envelopment analysis (Alizadeh et al., 2020; Hu et al., 2019). However, the LCA and LCCA approach is recommended because it includes damage costs (Alizadeh et al., 2020).

usage plans, e.g., by installing an upgradation unit to upgrade biogas to CBM for use as a fuel for public transport instead of combusting biogas to meet the internal energy needs of WWTPs. Finally, it can construct a new WWTP with a new treatment configuration and byproduct usage options. To decide which options is best, information on the BAU case and the FCOs planned by stakeholders for a municipal WWTP must be consolidated. The consolidated information should include (i) technical data on factors such as the design capacity of the WWTP, influent and effluent water quality, and land area requirements, along with operational data on the consumption of energy and chemicals, etc.; (ii) information on the reuse of treated effluent and biogas, sludge recycling pathways, and incumbent products that are avoided (as summarized in Fig. 3); and (iii) additional data needed to develop sustainability performance indicators specific to byproducts. For example, depending on the utilization pathways being considered, data such as annual freshwater consumption or the total cost of ownership per vehicle-km of city buses may be needed to develop indicators reflecting the sustainability benefits of treated effluent and biogas reuse (by upgrading to CBM for use as transport fuel), respectively.

4.2. Evaluate environmental performance of BAU and FCOs proposed for a WWTP

The environmental performance of BAU and all FCOs considered by stakeholders is quantified by conducting a life cycle assessment (LCA) study. From a conventional product-based LCA perspective, layers 1 and 2 in Fig. 2 (i.e., wastewater and biosolids treatment) correspond to the cradle to gate stages of the life cycle, while layers 3 through 5 represent the gate to grave stages. The overall environmental performance is the sum of the environmental impacts attributed to WWTP operations (based on m^3 or population equivalents (PE) of sewage treated) minus the water depletion impacts avoided by reuse of treated effluent (m^3) together with the sum of the subsequent use stage impacts (e.g., use stage impacts of (tank to wheel) CBM driven city buses, if biogas from WWTP upgraded to CBM and sold to transport provider) minus the impacts due to incumbent products (e.g., life cycle (well to wheel) impacts of incumbent diesel driven buses for equivalent amount of bus-km) that are avoided under the FCO. Also included is the sum of the impacts of the sludge recycling products under the FCO minus the impact of the incumbent sludge products that is avoided.

4.3. Monetize environmental impacts of BAU and FCOs proposed for a WWTP

The EECs are determined by monetizing the environmental impacts using an LCA approach. Unit externality prices for most impact categories at the midpoint and endpoint levels are well established for the European region (EuropeanCommission, 2019; Trinomics, 2020) and have also been developed for most other regions (Karkour et al., 2020). The EEC of a particular impact is obtained by multiplying its absolute score by a designated unit environmental externality price for the relevant impact category. The net EEC for a given WWTP scenario is then calculated using equation (1).

$$NetEEC_{WWTP} = \left[EEC_{opns}\right] + \left[\left(-EEC_{afw}\right) + \left(EEC_{gpu} - EEC_{aiu}\right) + \left(EEC_{srp} - EEC_{ap}\right)\right] \quad (1)$$

Here.

EEC_{opns} = Environmental externality costs of WWTP Operations (layers 1 & 2 in Fig. 2).

$-EEC_{afw}$ = Environmental externality costs of avoided freshwater use. This includes avoided costs attributed to water depletion impact and processing of freshwater (e.g., deionization) for a designated use. The negative sign indicates a credit because externality costs of treated effluent are already accounted for under WWTP operations.

EEC_{gpu} = Environmental externality costs of gas product use phase (e.g., use of CBM/LBM as transport fuel).

EEC_{aiu} = Environmental externality costs of avoided incumbent use (e.g., incumbent fuels avoided when CBM/LBM is used).

EEC_{srp} = Environmental externality costs of any sludge recycling products as shown in Fig. 3.

EEC_{ap} = Environmental externality costs of incumbent product avoided due to use of products obtained from sludge recycling.

All EECs in equation (1) are quantified using equation (2):

$$EEC = \sum_{i=1}^{n}\left[\left(Env.IS_{ic}\right) \times \left(UExt.P_{ic}\right)\right] \quad (2)$$

Here.

$Env.IS_{ic}$ = Environmental impact score (quantified using LCA) for a given impact category.

$UExt.P_{ic}$ = Environmental externality price per unit impact for a particular impact category.

4.4. Perform LCCA of BAU and FCOs proposed for a WWTP

The total private costs (TPC) and total private revenues (TPC) from sale of byproducts for the BAU case and the FCOs under consideration are quantified in the third step. The TPC and TPR of municipal WWTP for a given option are calculated using equations (3) and (4).

$$TPC_{wwtpopns} = \left[UC_{opns} \times Q_{tre}\right] \quad (3)$$

$$TPR_{ByprodSale} = \left[R_{eff} + R_{gp} + R_{ds}\right] = \left[\left(USP_{eff} \times Q_{eff}\right) + \left(USP_{gp} \times A_{gp}\right) + M_{ds}\left(USP_{ds}\ or - UDC_{ds}\right)\right] \quad (4)$$

Here.

TPC_{opns} = Total Private Cost (USD) of WWTP Operations (layers 1 and 2 of Fig. 3); UC_{opns} = unit cost of WWTP operations (USD/m^3) per cubic meter of wastewater treated; and Q_{tre} = water volume treated (m^3) per unit time.

R_{eff} = revenues (USD) from sale of treated effluent; USP_{eff} = unit selling price (USD/m^3) for effluent; and Q_{eff} = amount of effluent sold (m^3) per unit time.

R_{gp} = revenues (USD) from sale of gas products (electricity & externally for CBM/LBM);

USP_{gp} = unit selling price of gas products (USD/kWh for electricity & USD/kg for CBM/LBM); and A_{gp} = amount of gas product sold in kWh for electricity or kg for CBM/LBM.

R_{ds} = revenues (USD) from selling dry sludge as soil conditioner; M_{ds} = mass of dry sludge (tons)/time; USP_{ds} = unit selling price of dry sludge as soil conditioner (USD/ton).

DC_{ds} = Disposal costs (USD) of dried sludge at a given time; and UDC_{ds} = unit disposal cost of dry sludge (USD/ton).

DC_{ds} becomes relevant in scenarios when dry sludge fails to meet the regulatory requirements for use as soil conditioner or if sludge undergoes extended processing (e.g., incineration) where tipping fees will be collected. An LCC exercise is performed to calculate all private costs included in equation (1).

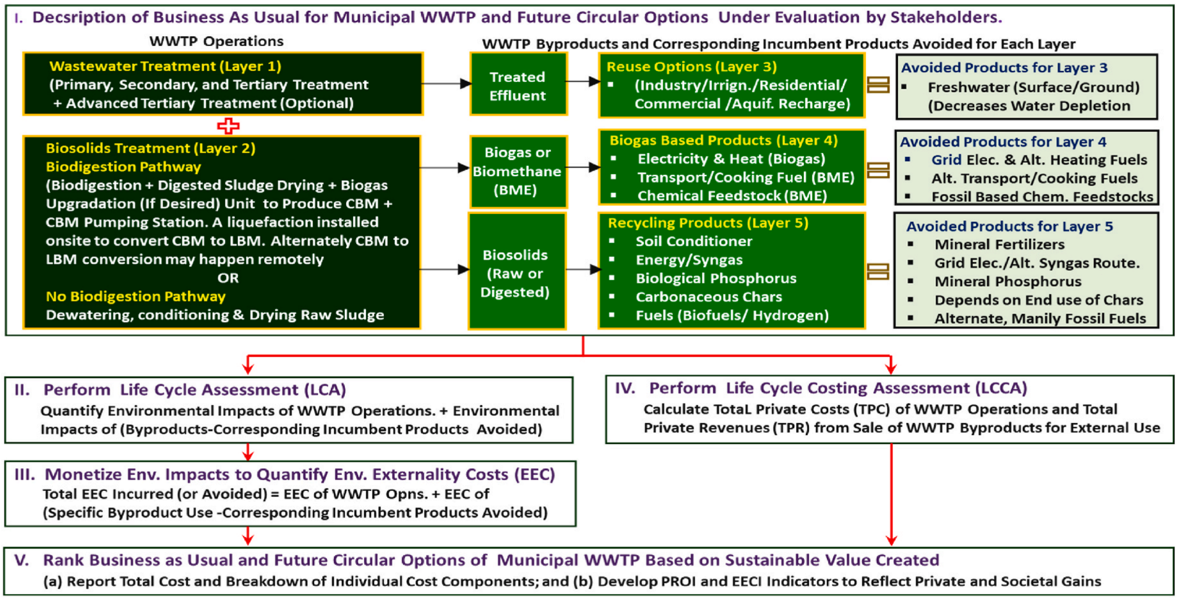
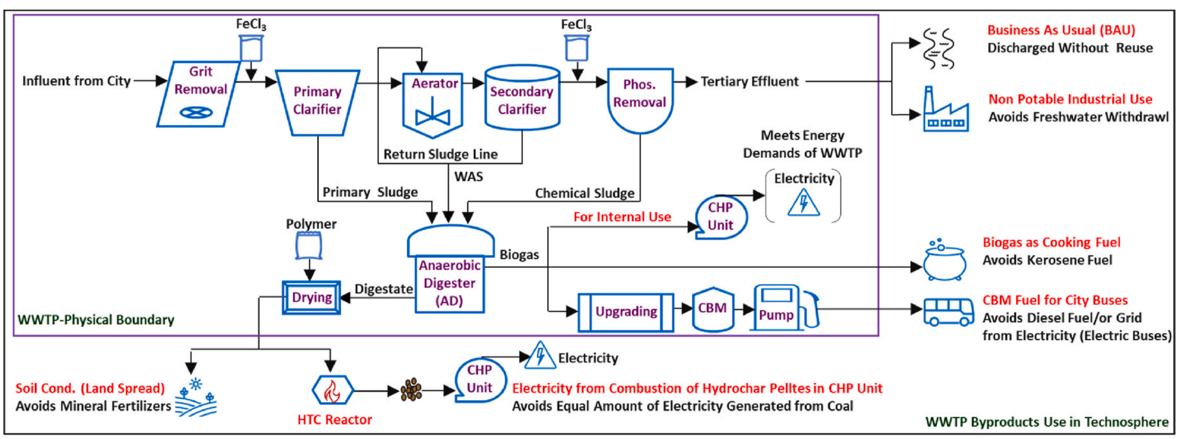


Fig. 2. Framework for improving circularity of municipal WWTPs based on environmental and economic performance assessment.



Options	Wastewater Treatment	Biosoilds Treatment via Anaerobic Digestion (AD)	Effluent Reuse	Biogas Reuse	Sludge Recycling
BAU	Tertiary Treatment	Biogas combusted in CHP unit	No Reuse	Meets Internal Energy Needs	Soil Conditioner
FCO1	Tertiary Treatment	Biogas combusted in CHP unit	Industrial Reuse	Meets Internal Energy Needs	Soil Conditioner
FCO2	Tertiary Treatment	Biogas is filled into gas cylinders	Industrial Reuse	Cooking Fuel (Avoids Kerosene)	Soil Conditioner
FCO3	Tertiary Treatment	Biogas upgraded to CBM and then filled into buses	Industrial Reuse	Transport Fuel (Avoids Diesel)	Soil Conditioner
FCO4	Tertiary Treatment	Biogas upgraded to CBM and then filled into buses	Industrial Reuse	Transport Fuel (Avoids Grid Elec.)	Energy from Hydrochar

BAU and FCO1: Biogas combusted in CHP unit to meet internal enerticity and heat needs of WWTP. Remaining electricity is sourced from electrical grid.
FCO2: Heating needs of AD unit met by combusting biogas in gas boiler. Residual stream of biogas is filled into cylinders (to be used as cooking fuel) via onesite pumping station
FCO3 and FCO4: Heat needs of AD unit met by combustion biogas in gas boiler, Residual stream of biogas upgraded to CBM and filled into buses via onsite pumping station

Fig. 3. System boundary showing business as usual (BAU) for a municipal WWTP and four future circular options (FCOs) under consideration by stakeholders.

4.5. Rank BAU and FCOs for a municipal WWTP based on their sustainable value

Finally, the results of the performance assessment must be presented to stakeholders in a simplified format that is easily understood. For this purpose, the sustainable value of each evaluated option is calculated as the sum of $TPC_{wwtpopns} + TPR_{ByprodSale} + NetEEC_{WWTP}$. In addition to the sustainable value, indicators such as private returns on investment (PROI) and EEC to investment ratio (EECIR) are used to help explain the results. Some authors of previous studies recommended the use of ecoefficiency for this purpose because it presents the unit cost of EEC to society as a tradeoff per unit cost incurred by stakeholders. While ecoefficiency is an effective method tool for communicating LCC and LCA results, stakeholders will be interested to know what return they can expect from their investment, especially when they have to make decisions relevant to improving the circularity of municipal WWTPs. We therefore suggest using the private return on investment (PROI) and EEC to investment ratio (EECIR) as indicators: the former reflects stakeholder profitability, while the latter reflects societal gains resulting from increasing the circularity of municipal WWTPs. The PROI and EECIR are calculated using equations (5) and (6), respectively.

$$PROI_{wwtp} = \left(\frac{TPR_{Byprod.Sale} - TPC_{wwtpopns}}{TPC_{wwtpopns}} \right) \tag{5}$$

$$EECIR_{wwtp} = \left(\frac{NetEEC_{wwtp}}{TPC_{wwtpopns}} \right) \tag{6}$$

The BAU and FCOs are then ranked based on their respective sustainable value, PROI and EECIR data. Additional indicators specific to byproduct utilization may also be used in the ranking and discussed with stakeholders if desired – examples may include the regional water stress index to quantify the benefits of reusing treated effluent or the transport quality index (TQI) if an incumbent fuel (e.g., diesel for transit buses) is replaced with CBM obtained from WWTPs.

5. Application of the framework to a municipal WWTP: a case study

The proposed framework must be validated to demonstrate its practical utility as a decision-making tool for stakeholders seeking to improve the circular performance of WWTPs. The five stages of the framework were therefore applied sequentially to a specific municipal WWTP. The analysis focuses on a hypothetical municipal WWTP located in a major city within a developing nation, namely Chennai in India. Chennai is a relevant location for the case study for three reasons. First, the city generates 1500 MLD (million liters/day) of sewage, of which only 28% is treated; the remaining 1073 MLD is discharged directly into natural water bodies without treatment (Goutham, 2017). Second, Chennai is classified as an extremely water-stressed city (Viswanath, 2019) where reusing wastewater is of pivotal importance. Third, Chennai recycles its treated wastewater for non-potable applications on a large scale (Ahluwalia and Kesavan, 2021). The outcome of this case study will thus offer useful insights to the municipal authorities of Chennai and many other cities facing similar freshwater shortages and wastewater treatment challenges.

5.1. Description of BAU and four FCOs proposed for a municipal WWTP in Chennai, India (stage I of the framework)

A system boundary showing WWTP operations (wastewater treatment, sludge treatment, biogas combustion/upgradation, and sludge drying) and byproduct utilization options for the BAU case and four FCOs (FCO1, FCO2, FCO3, and FCO4) under consideration by stakeholders are shown in Fig. 3.

Unlike the BAU case, the four FCOs all involve industrial reuse of the tertiary treated effluent from the municipal WWTP. This generates revenue for the WWTP and may generate freshwater savings for the city, which is important given its extreme water stress (Viswanath, 2019). Accordingly, reuse of treated wastewater for industrial applications is already practiced on a large scale in Chennai; most of the city's municipal WWTPs sell their tertiary treated effluent to industries where it is used in various non-potable applications such as makeup water for cooling towers or steam generation. Although reuse of tertiary effluent in industries generates freshwater savings, it also raises important questions about accountability for residual materials such as phosphorus, BOD, and other pollutants in tertiary effluent as well as the potential for untreated micropollutants to escape the applied treatment. However, industrial reuse undoubtedly reduces the human health and ecotoxicological impacts that would otherwise result from the direct discharge of tertiary effluent to the environment. In particular, the behavior of CECs, and their fate and transport to the environment depends heavily on how the tertiary effluent is used within the industry. The presence of micropollutants in WWTP effluents reused in industry is yet to be thoroughly investigated. However, the maximum permitted concentrations of some emerging contaminants in effluent wastewater for industrial reuse are 0.8 mg/l for Diclofenac, 3 mg/l for Ibuprofen, and 5 mg/l for Carbamazepine (Baresel et al., 2015b). Thus, relatively high concentrations of certain CECs can be tolerated in selected industrial reuse applications. Treating wastewater to advanced tertiary standards using techniques such as membrane filtration and ozonation is highly desirable before industrial reuse. However, many municipal WWTPs lack the financial resources to purchase the necessary systems and therefore sell tertiary effluent to industries as part of their circular operation model. Industries also often prefer to purchase tertiary effluent from WWTPs at a lower cost, although some may apply advanced tertiary treatments before reuse depending on the end use application. Consequently, the environmental burden (human health and ecotoxicity) associated with discharge of residual pollutants and untreated micropollutants escaping from tertiary effluent is only considered in the BAU case where the effluent is directly discharged into natural water bodies; it is disregarded in the FCO models.

FCO1 differs from BAU in only one respect, namely the industrial reuse of tertiary effluent. FCO2 and FCO3 each differ from BAU in two ways: like FCO1 they both include industrial reuse of tertiary effluent, but they also include external utilization of biogas, either as cooking fuel (FCO2) or as fuel for buses after upgrading to CBM (FCO3). Finally, FCO4 differs from BAU in three respects: it includes industrial reuse of tertiary effluent, use of biogas as a source of CBM to fuel buses (avoiding the use of electric buses and the combustion of biogas in a CHP unit), and it also includes generation of electricity from hydrochar pellets produced from dry sludge rather than applying dry sludge to agricultural land. The sustainability dimensions of the BAU case and the four FCOs were evaluated as shown in Fig. 4. FCO4 is considered to be the option that is best aligned with the future plans for Chennai city, where diesel buses are expected to be replaced with electric buses and stringent restrictions may be imposed on recycling sludge as soil conditioner. The analysis of the four FCOs provide answers to the following three questions relating to the use of byproducts from WWTP.

a) What are the economic and environmental benefits of reusing tertiary treated effluent when compared to BAU?
b) What is the most sustainable pathway for reusing biogas: to continue reusing it internally to meet the energy demands of the WWTP, to divert it for external use as a cooking fuel, or to upgrade it to CBM for use as a fuel to power city buses? Also, what is the overall benefit of using CBM buses instead of diesel or electric buses, especially if the electricity to power the latter comes from a coal-intensive grid?
c) What is the contribution of digestate recycled as soil conditioner to the overall sustainable value proposition of BAU and FCOs? Also, if the use of digestate as soil conditioner is restricted in future, what are the benefits of converting sludge into hydrochar pellets and using them for energy generation, thus avoiding the equivalent amount of electricity produced from coal?

5.2. LCA of BAU and four FCOs proposed for the WWTP (stage II of the framework)

5.2.1. Goal and scope of LCA study

The functional unit of this LCA study is the volume of wastewater treated (m^3) by the WWTP in one year. The WWTP is assumed to receive and treat 50,000 m^3 of influent wastewater per day. The influent BOD and TSS loads are assumed to be 225 mg/L and 436 mg/L, respectively. The WWTP uses metal salt ($FeCl_3$) precipitation to remove phosphorus, aiming to reduce its concentration in treated effluent to no more than 1 mg/L. A system boundary diagram representing BAU and the four FCOs is presented in Fig. 3.

5.2.2. Life cycle inventory (LCI) modeling details

LCI data for the BAU case and the four FCOs are presented in Fig. 3 and in sections S1 and S2 of the supporting information (SI). LCI modeling was done using the SimaPro (PhD v 9.0) LCA software package (Pre, 2017). The foreground LCI data on WWTP operations and byproduct utilization in the external boundary was developed based on literature data and engineering design calculations. The background LCI data were mainly obtained from the ecoinvent (version 3.2) database (Ecoinvent, 2015). For electricity consumption, the LCI dataset representing the Indian electrical grid mix was used.

5.2.3. Life cycle impact assessment methodology (LCIA)

The life cycle environmental burdens of the following ten midpoint categories are quantified: Global Warming Potential (GWP) as kg CO_2 eq.; Human Toxicity Potential - Non-Cancer (HTP-NC) and Human Toxicity Potential - Cancer (HTP-C) as CTU_h; Particulate Matter Formation Potential (PMFP) as kg $PM_{2.5}$ eq.; Photochemical Oxidant Formation Potential (POFP) as kg NMVOC eq.; Acidification Potential (AP) as mol H^+ eq.; Freshwater Eutrophication Potential (FEP) as kg P eq.; Freshwater Ecotoxicity Potential (FEXP) as CTU_e; Water Scarcity Index (WSI) in m^3; and Cumulative Energy Demand (CED) as MJ. The GWP (100 years) is determined using the method proposed by the Intercontinental Panel for Climate Change (IPCC), CED is determined using a method developed by Frischknecht and coauthors (Ecoinvent, 2007), and the WSI is quantified using method of Boulay et al. (2011). The remaining seven midpoint categories are quantified using ILCD-2011 Midpoint+ (v. 1.08) (International Reference Life Cycle Data System) (EuropeanCommission, 2012).

5.2.3.1. Human freshwater ecotoxicity impacts of CECs included in BAU option. In the BAU case, the tertiary effluent is discharged into a river without treatment to remove CECs. The concentrations of 30 pharmaceutical contaminants commonly found in treated WWTP effluents from Indian cities (Balakrishna et al., 2017; Mathew and Kanmani, 2020) are shown in section S2 of the SI. Impact characterization factors for these compounds obtained from a study reported by Li et al. (2019) are summarized in section S3.

5.2.3.2. Land spreading sludge and human toxicity impacts caused by heavy metals. The HTP impact (NC and C) caused by heavy metal contaminants entering soil due to spreading of digested dried sludge on agricultural land is largely overestimated by LCIA methods such as ILCD (Harder et al., 2017). For this reason, we use the CFs endpoint developed by Harder et al. (2017), which is quantified in Disability adjusted life years (DALY)/kg ton dried sludge, and multiply this value by the previously reported concentration of heavy metals (g/ton) in digested sludge in India (Usharani and Vasudevan, 2016). The resulting impact outcome (DALY/ton) is divided by factors of 0.37 and 0.086 to obtain HTP-NC and HTP-C impacts, respectively, because 1 CTUh of HTP-NC is equivalent to 2.7 DALYs and 1 CTUh of HTP-C corresponds to 11.5 DALYs (Arendt et al., 2020)). Finally, the HTP-NC and HTP-C impacts obtained as CTUh for soil heavy metal contamination are added to the overall impact score for spreading sludge on land (stage 7). Additional details of these calculations are provided in S3.

5.2.4. Uncertainty analysis

An uncertainty analysis is performed to understand the extent of variation in the environmental performance under BAU and the four FCOs. Data perturbations are introduced by defining minimum and maximum limits for key parameters governing the LCI models for BAU and each FCO as shown in Fig. 3. A more detailed discussion of data uncertainties and the supporting assumptions is presented in section S2. The uncertainty analysis was conducted within SimaPro using Monte Carlo simulations with 5000 steps and a 95% level of confidence.

5.2.5. Scenario analysis

A scenario analysis is also conducted to determine how the predicted impacts change in the event that: (a) removal of CECs becomes regulated in future (even if effluent is reused), necessitating the inclusion of an advanced tertiary system in the wastewater treatment train, and (b) external use of biogas as a fuel source is beneficial to society at large. It is important to note that adding advanced tertiary stages for treatment of CECs increases the energy consumption of a WWTP (Naturvårdsverket, 2018); if the necessary electricity is drawn from coal-intensive grids, the benefit of using biogas as a fuel will be diluted. Therefore, in the scenario analysis it is assumed that an ozonation stage is added to the WWTP to remove CECs but the electricity supply to the WWTP is sourced from an off-grid solar power facility. The treated effluent is reused in industry and the biogas is used as cooking fuel or upgraded into fuel for buses, while the digestate is converted into hydrochar pellets and used as a solid fuel in a CHP to generate electricity.

5.3. Monetization of environmental impacts (stage III of framework)

The environmental impacts for the BAU case and FCOs obtained from the LCA study are monetized using unit environmental externality prices obtained from the literature for each impact category (Karkour et al., 2020; Trinomics, 2020). These prices are listed in the SI (S4) for reference.

5.3.1. Converting PMFP impact to disease incidence unit (DIU) for monetization

ILCD-2011 Midpoint + quantifies PMFP impact as kg $PM_{2.5}$ eq., and this impact score is converted into disease incidence units (DIU) using a fractional intake approach recommended in the literature (Humbert et al., 2011; Verones et al., 2016). This approach uses characterization factors (CFs) as deaths/kg $PM_{2.5}$ emitted, which depends on the fractional intake (kg intake/kg emitted) of $PM_{2.5}$ emissions. The ILCD quantified midpoint impact score in kg $PM_{2.5}$ eq. is thus multiplied by a CF in deaths/kg $PM_{2.5}$ emitted for a given fractional intake to obtain the impact as "all-cause premature mortality" (Verones et al., 2016). Determining PMFP impact as DIU is especially relevant when quantifying the externality cost of the PMFP impact because the CFs used for particulate matter emissions above ground (e.g., stack emissions from coal power plants with tall stacks) differ from those for ground-level emissions (e.g., from mobile sources such as city buses) (Verones et al., 2016). The CFs (Deaths (kg emitted) for $PM_{2.5}$ depend on the fractional intake of the corresponding pollutant, which is lower for pollutants emitted above ground than for ground-level emissions. The chosen method thus avoids the risk of overestimating the PMFP externality cost, which may happen when directly monetizing the PMFP midpoint impact as kg $PM_{2.5}$ eq. * USD/kg $PM_{2.5}$ eq. (Trinomics, 2020). Furthermore, using CF as deaths/kg $PM_{2.5}$ emitted enables the incorporation of regionalized factors when quantifying the impact.

The intake fraction for $PM_{2.5}$ emitted from a stationery source for an unknown stack height is determined using the following empirical equation (Humbert et al., 2011): $IF_{Rural} = 2.6 \times 10^{-8} \times Pop.Density_{Rural} + 7.9 \times 10^{-8}$ The rural population density of India in 2019 was 460 inhabitants per km^2 (Worldometer, 2020), giving an intake fraction of 4.21 E−06. This value is close to that of 5.2 E−06 obtained for an outdoor urban environment with very high stack emissions using the average slope method (Verones et al., 2016). The corresponding CF for an intake fraction of 5.2 E−06 is 3.4E-05 deaths/kg emitted. This value is assumed for all stationary source emissions considered in our LCA study. Although emissions from stationary sources (e.g., power plants supplying electricity to WWTP) originate from rural areas, the population density of India is high, making it reasonable to use the CF corresponding to the intake fraction of $PM_{2.5}$ emitted into the outdoor urban compartment with very high stack emissions in accordance with the empirical equation above. The CF for ground-level mobile sources (namely buses fueled with CBM or diesel as the incumbent fuel) in the CS-2 scenario is 2.4E-04 according to the same literature source (Verones et al., 2016). The CF for indoor environments (for cooking with CBM and incumbent kerosene fuel) in the CS-3 scenario is 8.2E-04 (Verones et al., 2016).

5.4. LCCA of BAU and four FCOs proposed for WWTP (stage IV of framework)

An LCCA is performed to calculate the TPC of WWTP operations (i.e., layers 1 and 2 in Fig. 3) and the TPR from sale of byproducts (layers 3–5 of framework in Fig. 3) for BAU and the FCOs, as shown in Fig. 4. All capital costs are annualized and their cost per year is reported because the functional unit of this study is one year. Annualized capital costs are estimated using a capital recovery factor (CRF) calculated as $CRF = \frac{i\times(1+i)^n}{(1+i)^n-1}$, where i = interest rate (taken to be 6% in this work) and n = number of annual payments. When calculating land costs, n is taken to be 20 years. For all equipment except the boiler (n = 6 years) and CBM pumping unit (n = 20 years), n is taken to be 25 years. All costs are adjusted for inflation using prices in the year 2021 as a baseline.

6. Results and discussion

6.1. Baseline scenario results

The results of ten midpoint environmental impacts (normalized to highest score in each category) under the BAU case and the four FCOs for the municipal WWTP are shown in Fig. 4. A contribution analysis of each option (i.e., BAU and FCOs 1–4) is presented in S4 of the SI.

The environmental performance of FCO1 is superior to that of BAU in all impact categories. This was expected because in FCO1, combustion of biogas in an on-site CHP unit provides 88% of the electricity required by the municipal WWTP operations, significantly reducing the amount of electrical energy drawn from India's coal-intensive electric grid, which generates high environmental impacts. The LCI modeling conditions of BAU and FCO1 are quite similar with respect to the biogas (reused internally) and sludge (recycled as soil conditioner) usage pathways. Therefore, their performance is identical in six of the ten impact categories. However, the industrial reuse of tertiary effluent reduces HH-NCP, HH-CP, and FEP impacts under FCO1 by 56–81% compared to the BAU option. The LCA results for FCO1 and FCO2 suggest that the environmental benefits (impact credits) realized by replacing kerosene with biogas as a fuel for cooking in FCO2 or replacing diesel for transit buses with CBM in FCO3, cannot offset the environmental burden caused by WWTP operations. This is because external utilization of biogas leads to a situation in which the municipal WWTP must rely completely on electricity drawn from the Indian electrical grid mix, which increases the overall environmental burden of FCO2 and FCO3. The life cycle environmental impacts credited to FCO2 by avoiding kerosene-based cooking range are in the range of 33–48% for GWP and CED, and 15–24% for PMFP and POFP. Similarly, avoiding the use of diesel in city buses reduces the overall burden under FCO3 by 52% for POFP, 32% for CED, and 12–18% for GWP and AP. However, these reductions are diluted by significant increases in the environmental impacts of the WWTP, which are largely attributed to the dependence on the Indian electrical grid mix to meet the WWTP's electricity demand.

Of the four FCOs, FCO4 is the second-best option with respect to overall environmental performance after FCO1. This is due to impacts credited for substituting incumbent electric buses with CBM alternatives and using hydrochar pellets to replace coal for energy generation. While electric city buses are cleaner than diesel-powered buses because they have zero tailpipe emissions, their sustainability potential in countries like India is questionable because operating them with coal-heavy electricity simply leads to a shift of the environmental burden rather than its reduction. Accordingly, the results for FCO4 show that in an Indian context, driving CBM-fueled city buses is environmentally preferable to introducing electric buses: for the same annual driving distance (in bus-km), the overall environmental impact of CBM buses is 12–40% lower than that of buses powered with electricity from the Indian electrical grid, with impacts in the GWP, PMFP, and CED categories falling by 32–40%. Also, the results indicate that for municipal WWTPs in Indian cities, the material substitution of coal with hydrochar pellets for electricity generation is a more environmentally sustainable pathway for recycling dry digestate as agricultural soil conditioner and thus avoiding mineral fertilizers. Generating electricity by burning hydrochar pellets in a CHP (and avoiding consuming the equivalent amount of coal-derived electricity) reduces environmental impacts in nine of the ten impact categories by 6–40% under FCO4 (the only category without reductions was WSI), whereas using digestate as soil conditioner in lieu of mineral fertilizer yielded impact reductions of only 7–16%.

6.2. Uncertainty analysis results

Fig. 5 shows the results of the uncertainty analysis of the WWTP options.

The tertiary effluent is discharged into natural water systems in the BAU option whereas it is assumed to be reused in industry in all four FCOs. Therefore, the residual pollutant load (BOD, phosphorus etc.) and untreated CECs in the tertiary treated effluent are accounted for in the

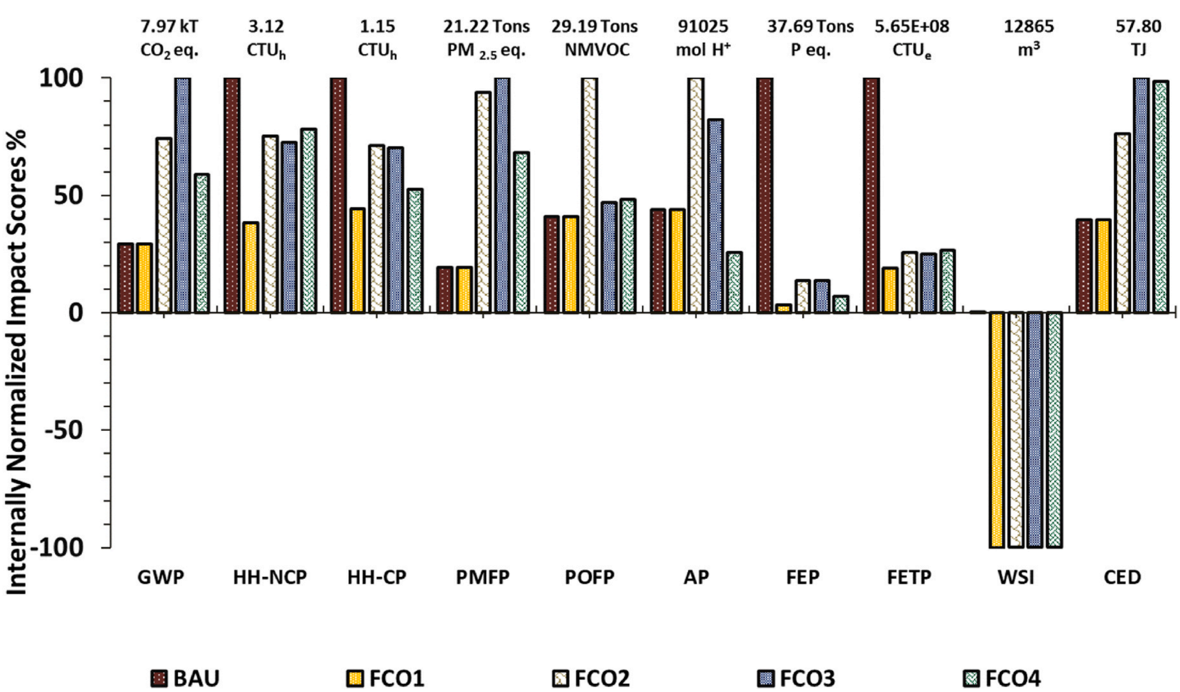


Fig. 4. Life cycle environmental impacts of the studied WWTP under the BAU case and FCOs 1–4.

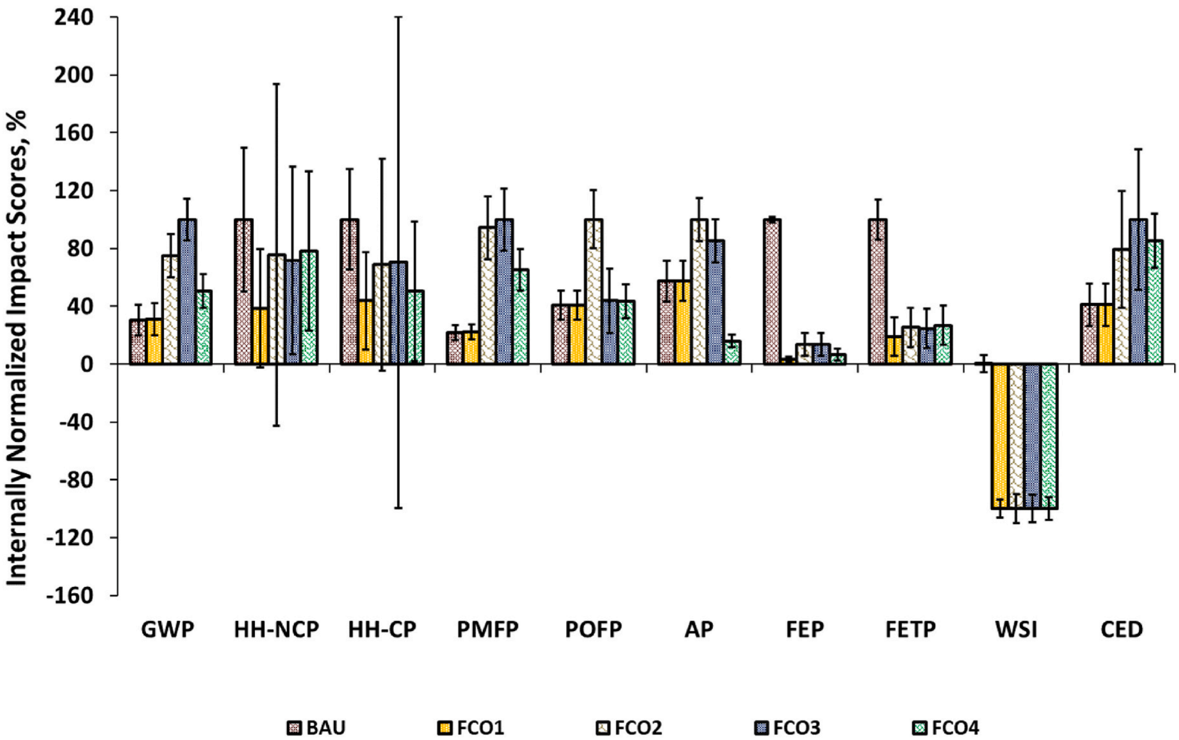


Fig. 5. Uncertainty analysis of BAU and the FCOs for the municipal WWTP case study.

LCI dataset for the BAU option but are not considered for the FCOs. As a result, the uncertainty analysis is not relevant for the FEP FETP, and WSI impacts because the BAU option will necessarily have higher impact scores than the FCOs in these categories. Excluding these categories, BAU and FCO1 have the lowest GWP, PMFP and CED burdens, while the AP burden under FCO4 will always be lower than that for all other options including BAU. The POFP impact of BAU and FCO1 is always less than for FCO2, but the POFP impact scores of FCO3 and FCO4 overlap because they are sensitive to the chosen variation (particularly variation in bus-km avoided when widening the range of fuel economy values assumed for diesel and electric buses in FCO3 and FCO4). Finally, the HH-NCP and HH-CP impacts of all options vary widely, mainly because

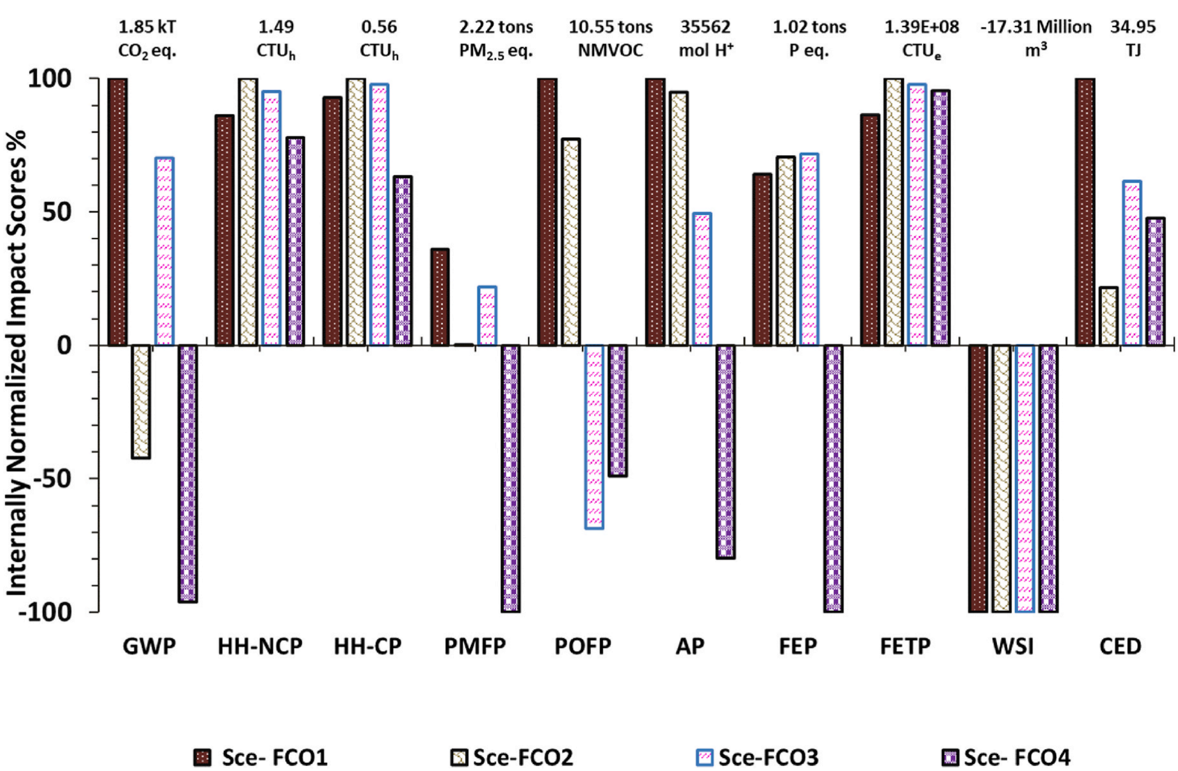


Fig. 6. Scenario analysis of municipal WWTP FCOs with modified modeling parameters.

of the background uncertainty associated with the ecoinvent LCI dataset for the Indian electrical grid mix. Despite this huge variation, FCO1 is likely to have a lower impact in these two categories than the other options.

6.3. Scenario analysis results

A scenario analysis was conducted to evaluate the effects of two major changes in the life cycle of the WWTP under the four FCOs. These changes are: (i) adding an ozonation stage for removal of CECs; and (ii) sourcing the electricity for the WWTP from a solar grid. For scenario analysis of FCO1, the biogas generated at the plant is still combusted in a CHP and used to meet internal energy demand as in the FCO base case, but any additional electricity consumed at the WWTP is sourced from a solar powered energy grid. For FCO2, FCO3, and FCO4, the entire electricity demand of the WWTP is met by a solar powered grid. The results of the scenario analysis are presented in Fig. 6.

The removal of micropollutants from the influent wastewater is not considered in the base case FCOs. However, in the scenario analysis, ozonation is used as an advanced tertiary treatment to remove CECs. An electricity input of 3650 MWh (0.2 kWh/m^3) is needed for ozonation and meeting this demand using electricity from India's coal-intensive national grid will significantly increase some environmental impacts. However, the scenario analysis suggests that new FCOs exhibit superior overall environmental performance when compared to baseline scenarios without ozonation if electricity is obtained from a solar grid. This clearly shows that incorporating advanced tertiary methods into municipal wastewater treatment trains will only be environmentally feasible if WWTPs in India achieve grid independency and rely more on decentralized energy solutions such as onsite solar power panels. The outcome of the scenario analysis also supports our hypothesis that if the electricity for the WWTP is supplied from a renewable source, external applications of biogas will have a more favorable sustainability impact. The Sce-FCO4 scenario provides the best performance of the four options considered in scenario analysis: its environmental burden is negative (because avoidance credits exceed the corresponding burdens) in six impact categories and is significantly lower than that for Sce-FCO1. The opposite is true in the base case, where the impact of FCO4 is greater than that of FCO1. Despite having the highest impact in the HH-NCP, HH-CP and FETP categories, Sce-FCO2 appears to be the best option after Sce-FCO4 because of its negative GWP (−42%) and its lower CED and PMFP impacts when compared to Sce-FCO3 and Sce-FCO1.

Table 3
Net EEC of BAU and FCOs evaluated for municipal WWTP case study.

Impact	EEC in Million USD/Functional Unit)				
	BAU	FCO1	FCO2	FCO3	FCO4
Global Warming (GWP)	0.287	0.287	0.726	0.980	0.578
Human Health-Non Cancer (HH-NCP)	0.612	0.234	0.460	0.444	0.479
Human Health-Cancer (HH-CP)	1.255	0.555	0.895	0.881	0.658
Respiratory Effects (DIU)	0.132	0.132	−1.629	0.462	0.518
Photochemical Smog (POFP)	0.017	0.017	0.042	0.020	0.020
Acidification (AP)	0.016	0.016	0.038	0.031	0.010
Freshwater Eutrophication (FEP)	0.087	0.003	0.012	0.012	0.006
Freshwater Ecotoxicity (FETP)	0.026	0.005	0.007	0.006	0.007
Water Scarcity Index (WSI)	0.001	−0.693	−0.692	−0.692	−0.692
Cumulative Energy Demand (CED)	0.036	0.036	0.069	0.090	0.089
Total EEC	2.468	0.592	−0.074	2.235	1.673

6.4. Environmental externality costs (EECs) of BAU and the proposed FCOs

Table 3 shows the EECs derived from the quantified environmental impacts for each of the municipal WWTP options evaluated in the base case. The unit environmental externality prices used for monetization of impacts and EEC calculations are given in S5.

Monetization of environmental impacts provides an entirely different perspective on the five options (BAU and the four FCOs) considered for the hypothetical municipal WWTP. The total EEC of the BAU option is higher than the four FCOs for two reasons: (a) zero credit is given for freshwater depletion because treated effluent is discharged into natural water bodies; and (b) the tertiary effluent contaminant load attributed to residual phosphorus and untreated CECs is accounted for under the BAU option but not under the FCOs for the reason given in section 4.1. Therefore, BAU has higher HH-NCP, HH-CP, FEP and FETP impacts than the FCOs, leading to higher EECs for these impact categories.

Of the four FCOs, FCO3 has highest total EEC because the environmental costs incurred by complete reliance on the Indian electrical grid exceed the combined externality costs savings realized by replacing diesel with CBM buses and the costs attributed to avoidance of impacts caused by using mineral fertilizers.

The total EECs of FCO1 and FCO4 are 26% and 70% lower, respectively, than that of FCO3, reinforcing the conclusion that municipal WWTPs in Indian cities will be more able to contribute to a sustainable circular economy if the energy content of biogas and dry digestate is used to minimize consumption of electricity from India's electrical grid. Electricity generated by burning solid fossil fuels like coal and lignite imposes a serious health burden on society: a recent study showed that a country's lung cancer incidence risk increases by 59% for every kW of coal-fired generating capacity (Lin et al., 2019). Coal mining activities and coal-fired power plants also contribute significantly to many non-cancer risks including risks of neurological, mutagenic, and reproductive disorders due to the release of heavy metals such as mercury and other airborne emissions (Burt et al., 2013). The human toxicity impact of coal-fired electricity generation is 0.67–1.7 disability adjusted life years (DALY)/GWh, imposing a significant economic burden on countries like India that have a medium human development index (UNDP, 2020) and relatively high cancer incidence rates (Menon et al., 2019) even though the cost-effective threshold for aversion is less than their GDP per capita (Daroudi et al., 2021).

Interestingly, FCO2 has the lowest total EEC of all the considered options including FCO1. This is despite the fact that FCO1 has the best overall environmental performance (as shown in Fig. 4) and the externality costs of FCO2 being 25–77% higher than those of FCO1 in eight out of ten impact categories, with the exceptions being WSI and PMFP (measured as DIU). Because the savings realized from avoiding freshwater depletion are identical for FCO1 and FCO2, it is clear that the externality costs attributed to PMFP impact are the main factor responsible for the low total EEC of FCO2; an EEC of USD 1.69 million per year can be saved by replacing kerosene with biogas fuel for cooking. The disease incidence unit (deaths/kg emitted) of $PM_{2.5}$ indoor emissions is 8.2 E−04 (Verones et al., 2016), and the PMFP impact of substituting kerosene with biogas is −3064 kg $PM_{2.5}$ eq., corresponding to −2.51 DIU. Because the unit price per DIU is EUR 784126 (or USD 940951), this gives a saving of USD 2.34 million. A similar value is obtained if the calculation is based on the indoor intake fraction and disability adjusted life years (DALY) per kg of $PM_{2.5}$ inhaled; the intake fraction of $PM_{2.5}$ in indoor settings with low air circulation can be as high as 62,000 ppm (0.062 kg inhaled/kg emitted) (Fantke et al., 2019), corresponding to an exposure of 15,500 ppm assuming a reasonable exposure time of 25% for the dwellings of the urban poor in Indian cities. Moreover, there is a loss of 30 DALYs/kg inhaled in typical domestic indoor environments, corresponding to 0.465 DALY/kg $PM_{2.5}$ emitted. Therefore, the avoidance of 3064 kg $PM_{2.5}$ eq. due to kerosene burning results in an avoided loss of 1425 DALYs/kg emitted. The average cost per DALY avoided in India is 1430 USD (ICMR and IndiaStateLevelAirPollutionCollaborators, 2019), so the externality costs savings from avoiding kerosene amount to 2.03 million USD, which is within the range we obtained based on monetization of DIUs. The externality cost savings from avoiding kerosene fuel for cooking would be reduced if the indoor inhalation fractions of $PM_{2.5}$ were lowered, or the exposure time was reduced. However, the total EEC of FCO2 would be lower than that of FCO1 even if the EEC savings attributed to PMFP impact under FCO2 were reduced by 40%.

6.5. Assessment framework results for BAU and the four proposed FCOs (stages IV and V of the framework)

The cost breakdown analysis and sustainability performance indicators developed for the BAU option and the FCOs for the hypothetical municipal WWTP in the base case are summarized in Table 4. A detailed discussion of individual TPC and TPR calculations is given in section S6 of the SI.

The PROI reflects stakeholders' potential profits from the sale of byproducts. For example, the PROI of FCO2 is 1.92, meaning that for every dollar invested in WWTP operations (wastewater treatment and sludge management), the stakeholder (i.e., Chennai city municipality in the case study) will earn USD 1.92 from the sale of tertiary effluent to industrial customers, biogas as cooking fuel to replace kerosene, and sludge as a soil conditioner. Among the options considered, PROI decreases in the order FCO1 (2.31) > FCO2 (1.92) > FCO3 (1.67) > FCO4 (1.64). However, it should be noted that the high PROI of FCO1 is due to its low $TPC_{WWTPOpns}$ rather than to high total revenues ($TPR_{Byprodsale}$). The $TPC_{WWTPOpns}$ of FCO1 is 1.75 million USD, which is 0.49, 0.60, and 0.60 million USD lower than the corresponding values for FCO2, FCO3, and FCO4, respectively. On the other hand, the total revenues $TPR_{Byprodsale}$ generated under FCO1 are 0.76, 0.50, and 0.42 million USD lower than those for FCO2, FCO3 and FCO4, respectively. Thus, from a revenue generating standpoint FCO2 is preferable to FCO1 even though FCO1 has a higher PROI.

The EECIR measures the reduction in EEC due to the beneficial use of municipal WWTP byproducts. The EECI of FCO2 (−0.032) is the lowest of all the considered WWTP options. This value means that FCO2 incurs a *Net* $EECP_{WWTP}$ of −0.032 USD for every dollar of $TPC_{WWTPOpns}$ spent on municipal WWTP operations. The negative value implies that the EEC corresponding to FCO2 are completely negated because the EEC savings realized through byproduct use exceed the environmental damage costs incurred by WWTP operations. The negative EEC of FCO2 is largely driven by the reduction in indoor air pollution externality costs due to particulate emissions that results from replacing kerosene with biogas as cooking fuel. FCO1 has second lowest EECI (0.338) followed by FCO4 (0.710) and FCO3 (0.949). Therefore, based on results summarized in Table 2 the most to least preferred options suggested for Chennai city municipality are FCO2>FCO1>FCO4>FCO3>BAU, with FCO2 giving the greatest economic and environmental benefits.

Table 4
TPC, TPR, PROI and EECIR results of evaluated municipal WWTP options.

Cost/Revenue in Million USD	BAU	FCO1	FCO2	FCO3	FCO4
$TPC_{WWTPOpns}$	1.7512	1.7512	2.2409	2.3534	2.3534
R_{eff}	0	5.7213	5.7213	5.7213	5.7213
R_{gp}	0	0	0.7558	0.4905	0.4905
R_{ds}	0.0716	0.0716	0.0716	0.0716	0
$TPR_{Byprodsale}$	0.0716	5.7929	6.5479	6.2879	6.2118
Net $EECP_{WWTP}$	2.468	0.592	−0.073	2.235	1.673
PROI	−0.96	2.31	1.92	1.67	1.64
EECIR	1.408	0.338	−0.032	0.949	0.710

7. Conclusions

In conclusion, a systems-oriented assessment framework has been developed to help stakeholders such as city municipalities formulate strategies for improving the circular performance of municipal WWTPs based on the concept of sustainability. The framework is aligned with the 3R principle of the circular economy and evaluates the sustainable value of initiatives undertaken by municipalities with the intention of reducing the pollution load of sewage entering the WWTP while effectively reusing WWTP byproducts (reusing treated effluent and biogas, and recycling sludge) within the technosphere for beneficial end use applications. For a given degree of circularity, the framework quantifies the environmental impacts of the WWTP, monetizes them on the basis of an LCA, and determines private costs and potential revenues earned by selling byproducts. The features of the framework along with its strengths and limitations are summarized in Fig. 7.

The proposed assessment framework has four key strengths.

a) First, it can be flexibly applied to any circumstance defined by stakeholders. The framework can accommodate a wide range of modeling conditions including (for example) a change in the secondary treatment system whereby a conventional secondary clarifier is replaced with an MBR, adding ozonation to an existing wastewater treatment train as an advanced tertiary step to remove CECs, or recycling sludge for energy generation via incineration instead of using it as a soil conditioner.
b) Second, the framework has data modularity, meaning its layers are independent and changes in one layer do not affect other layers. For example, if in future it were decided that WWTP effluent was to be reused for irrigation rather than sold to industry, the resulting change in layer 3 would not affect other layers and the results obtained by modifying layer 3 could be appended to the overall assessment scores.
c) Third, the assessment framework is based on standard LCA and LCCA methodology and could be integrated with other modeling approaches such as exergy at the LCI modeling stage (Aghbashlo et al., 2021; Banerjee and Tierney, 2011), QRA at the impact assessment stage (Byrne et al., 2017), or urban metabolism at the interpretation of results and decision making stage in order to better understand how circularly managed WWTPs are positioned from a sustainable city perspective (Garcia and Dias, 2019; Guaita et al., 2018).
d) Fourth, the outputs of the framework are presented in terms of total costs consisting of private costs attributed to WWTP operations (layers 1 and 2), revenues from selling byproducts (layers 3 to 5), and net monetized environmental damages caused by WWTP operations plus byproduct applications minus the cost of incumbent options that are avoided under the proposed options. This information is used to develop indicators like PROI and EECI, which stakeholders can use as a basis for making decisions to improve the circularity of municipal WWTP while respecting their technical and financial constraints.

The assessment framework also has some notable major limitations:

a) It fails to address conflicting regulatory priorities. The framework considers the municipal WWTP as a circularly operating entity in that it assumes the 3R principles are strictly followed. This means treated effluent is assumed to be reused without discharge to the environment. However, in some countries, water consumers (e.g., industries) may be required by law to directly purchase treated effluent from WWTPs irrespective of its quality. This is because from a regulatory perspective, the primary role of municipal WWTP is to treat sewage to a stipulated standard and discharge it to natural water bodies. The framework fails in such cases because indicators such as PROI will always be lower.
b) The framework is ineffective in evaluating sustainability aspects of nature-based solutions for sewage treatment. Climate resiliency and

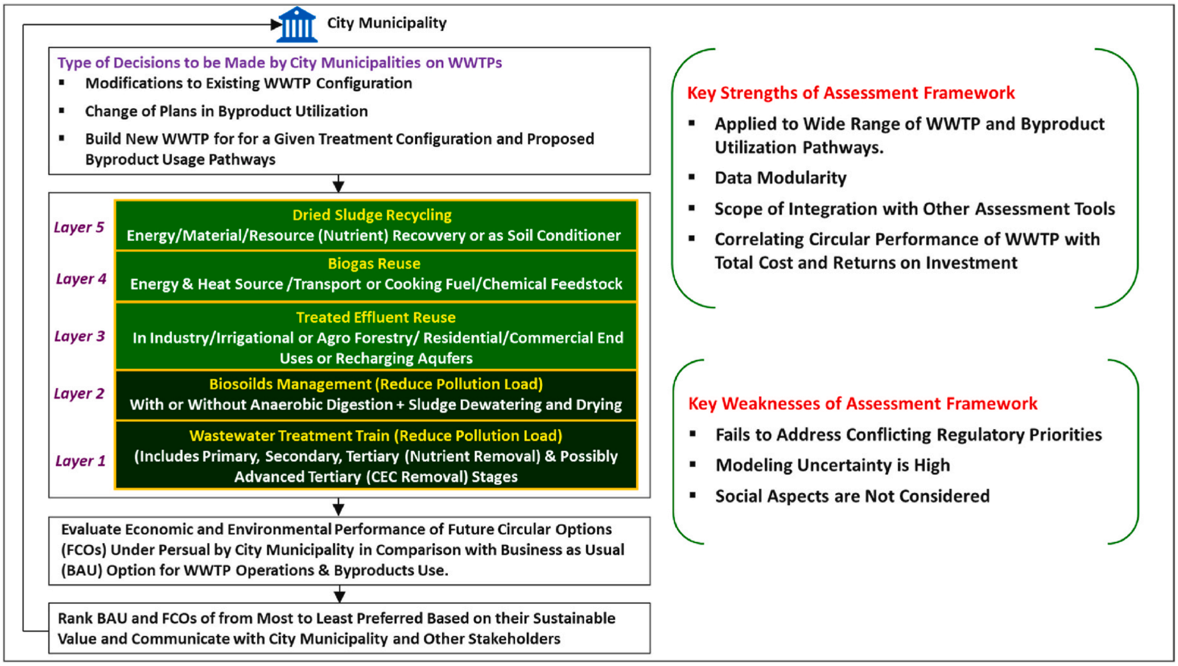


Fig. 7. Overview of the assessment framework including its strengths and limitations.

biodiversity enhancement are two key objectives of nature-based solutions like constructed wetlands (ref) but the framework in its current form cannot analyze these parameters (Stefanakis et al., 2021).

c) The modeling assumptions are rigid and static, making the framework valid for a specific set of modeling assumptions (e.g., biogas is used as cooking fuel and this use avoids the alternative use of kerosene) at a given point of time. Significantly different results may be obtained if (for example) biogas instead replaces a different fuel such as wood biomass. Also, modeling assumptions made during discussions with stakeholders concerning the utilization of byproducts may become invalid as time progresses. For instance, CBM could initially replace diesel as fuel for city buses, but biodiesel could be used at a later stage, which might significantly change the results of the assessment.

Finally, the sustainability performance assessment framework discussed in this work can be used as an algorithm. If integrated with other fields and concepts such as urban planning and urban metabolism, the framework could serve as the basis of a decision support software tool to help stakeholders formulate effective strategies for improving the circularity performance of urban WWTPs.

CRediT authorship contribution statement

Kavitha Shanmugam: Conceptualization, Investigation, Methodology, and, Software. **Venkataramana Gadhamshetty:** Writing – review & editing. **Mats Tysklind:** Writing – review & editing, Funding acquisition. **Debraj Bhattacharyya:** Conceptualization. **Venkata K.K. Upadhyayula:** Conceptualization, Investigation, Methodology, Software, Writing – original draft.

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

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