Resources, Conservation & Recycling 212 (2025) 107985

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

journal homepage: www.sciencedirect.com/journal/resources-conservation-and-recycling

Full length article

Urine luck: Environmental assessment of yellow water management in buildings for urban agriculture

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ARTICLE INFO

Keywords:
Circular economy
Resource recovery
Decentralized wastewater treatment
Life cycle assessment
Nitrogen fertilizer
Industrial ecology

ABSTRACT

The increasing global demand for agricultural production poses challenges to maintain the needs for critical fertilizers such as nitrogen. This study explores the potential of human urine as a source of renewable nitrogen for fertilizer production. Through a life cycle assessment, three different urine management strategies were compared: (S1) an artificial wetland, (S2) an on-site lab-scale aerobic reactor for nitrogen recovery, and (S3) a centralized wastewater treatment plant. While scenario S2 had the highest impacts in 6 out of 8 categories, an advantage in marine eutrophication was identified. S2 showed high energy demand (750 kg MJ-eq) and ecotoxicity (602 kg 1.4-DCB-eq.) mainly due to energy requirements. Nitrogen production exceeded 2.3 times the yearly nitrogen demands of the building tomato production. Upscaling S2 reduces impacts up to 2 times, lowering the payback time from 29 to 13 years. Therefore, implementing large-scale nitrogen recovery systems in cities is encouraged.

1. Introduction

Global population growth is increasing the demand for fertilizers, posing a challenge to the modern agricultural industry. According to the Food and Agricultural Organization of the United Nations (FAO, 2019), the global demand for nitrogen (N) for its use as fertilizer is increasing by an average of 1 % every year, requiring the production of an additional 1,074 million tons of N fertilizer annually. This N required for fertilization represents 56 % of the total combined demand for N, phosphorus (P) and potassium (K) fertilizers (FAO, 2019). In conventional agriculture, almost 75 % of global N is synthesized from atmospheric N_2 through the Haber-Bosh process, which can account for 10 times more energy use than K and P fertilizer production (Martin et al., 2023; Khan and Hanjra, 2009). Moreover, this process relies on non-renewable energy sources, using natural gas (50 %), oil (31 %) and coal (19 %) as feedstock (Smith et al., 2020) and contributing to 2 % of worldwide energy consumption and 1.6 % of global CO_2 emissions (Zhang et al., 2021). Further, inadequate N and P management and overfertilization in conventional agriculture can be responsible for nutrient leaching, leading to eutrophication of water bodies (Maurer et al., 2003).

One of the waste fractions that is considered highly suitable for fertilizer production is human urine or yellow water, particularly due to its nutrient-rich composition. While only constituting around 1 % of the total wastewater fraction, the majority of nutrients in wastewater come from urine, which contains up to 80 % of N, 50 % of P and 70 % of K (Chatterjee et al., 2019). Nevertheless, in developed nations, domestic wastewater is typically collected and processed in wastewater treatment plants (WWTPs) (Martin et al., 2023) where energy-intensive processes are required to denitrify wastewater to recover $<$10 % of the N content (Esculier et al., 2019). Implementing urine source separation through e.g. dry urinals or urine-diverting toilets is thus crucial to isolate urine from wastewater and prevent nutrient dilution. This circular approach not only facilitates the utilization of urine for fertilizer production but also reduces the reliance on scarce natural resources for fertilizer manufacturing as well as the need for nutrient removal in WWTPs. Yet, source separation does not necessarily translate into less environmental impacts (Remy and Jekel, 2008).

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https://doi.org/10.1016/j.resconrec.2024.107985
Received 13 March 2024; Received in revised form 20 September 2024; Accepted 20 October 2024

Numerous studies have applied life cycle assessment (LCA) to urine source separation and nutrient recovery to understand the potential environmental impacts and benefits of this alternative source of fertilizers and wastewater treatment. Existing LCAs of nutrient recovery range from the macro to the local scale, commonly focusing on the operation phase. For instance, Hilton et al. (2021) compare struvite concentration-pasteurization and precipitation systems through urine diversion with a conventional WWTP scenario for Michigan, Vermont and Virginia, where diversion systems were able to reduce greenhouse gas emissions (GHGs) by 29 %−74 % and energy consumption by 26–41 %. Comparative LCAs have also been conducted at WWTPs that recover nutrients at a local scale, where the impact on climate change decreased by 45 % when a decentralized district system was used (Besson et al., 2021). Others have evaluated urine separation technologies using the Biological Oxygen Demand as a unit of measurement (Lam et al., 2015) or the impacts of treating a specific amount of urine (Kavvada et al., 2017). Regarding the direct impact on agriculture on a regional scale, in Sweden up to 51 liters of water were saved per kilogram of harvested wheat thanks to the use of sprayed urine, reducing energy consumption by 50 % compared to a conventional scenario using synthetic fertilization (Tidåker et al., 2007). Similarly, Medeiros et al. (2020) found that the transport of urine from buildings to remote crop areas was an environmental hotspot.

Yet, the literature shows crucial research gaps in the systemic environmental assessment of nutrient recovery from urine, as it barely explores the interconnections between buildings, urban agriculture and wastewater treatment. On the one hand, green buildings are already applying decentralized wastewater treatment systems for pollutant removal, such as nature-based solutions (e.g., artificial wetlands). Although environmental assessments of green infrastructure are abundant, they are mostly compared with grey infrastructure (Romanovska et al., 2023), whereas novel circular approaches to wastewater management are yet to be compared. Only by doing so, can we effectively determine whether nutrient recovery from yellow water is a more environmentally friendly solution than existing green infrastructure and WWTPs focused on removal. On the other hand, compact urban areas with high crop density hold large potential for reusing recovered nutrients from urine (Trimmer and Guest, 2018). Recovering nutrients in public and private buildings can support the rising number of local food production initiatives aiming to increase food self-sufficiency in cities. With urban agriculture taking innovative forms such as rooftop greenhouses or vertical farms, urine source separation in buildings could help meet the demand for local fertilizers while reducing nutrient losses due to storage and transport (Pimentel-Rodrigues and Siva-Afonso, 2019). Literature on trials and models for using urine in urban agriculture have existed for a few years, aiming to evaluate plant growth and safety, and nutrient demand (e.g., Chrispim et al., 2017; Wielemaker et al., 2018). In the case of hydroponic vertical farms, D'ostuni et al. (2023) estimated the potential of recovering and reusing urine in a building in Amsterdam with a focus on nutrient content and using a theoretical model. However, environmental assessments are missing and, more importantly, they need real data on available on-site nutrient recovery technologies to understand the technical and environmental implications of a circular approach to nutrient production in buildings.

This article analyzes whether recovering urine in buildings to produce fertilizers for urban agriculture is environmentally beneficial compared to conventional end-of-pipe solutions and decentralized green infrastructure. To do so, we study the implementation of a novel aerobic reactor in an integrated rooftop greenhouse in a Mediterranean city using real lab-scale data. Three scenarios are compared using LCA: (1) an existing on-site artificial wetland; (2) an aerobic reactor for nutrient recovery that generates N in the form of nitrate for direct use as fertilizer, and (3) a conventional centralized WWTP. Our study is the first to compare these three systems at the building scale using real data and will serve to inform about the potential environmental impacts and benefits of transforming buildings into nutrient producers and consumers. Equipped with this knowledge, architects, urban planners and local administrations will be able to identify new synergies between the built environment, urban agriculture and wastewater and to adapt to the climate and resource crises.

2. Materials and methods

2.1. Case study description: Decentralized wastewater treatment & recovery

The evaluation of the environmental performance of yellow water management strategies was carried out at the ICTA-ICP building at the Universitat Autònoma de Barcelona in Cerdanyola del Vallès, Catalonia. The ICTA-ICP building consists of office areas and research laboratories, as well as an integrated rooftop greenhouse (iRTG) with hydroponic agriculture systems in which crops are mainly fertilized through mineral and synthetic nutrients (Sanjuan-Delmás et al., 2018). The building obtained a LEED certification, partly due to its onsite management and separation systems for black, grey, and yellow water. Based on ICTA-ICP's current facilities and considering the integration of an N recovery system for the fertilization of the iRTG crops, an attributional LCA is performed following the steps defined in ISO 14044 (2006) to compare three scenarios (Fig. 1):

S1: Decentralized wastewater treatment system through an artificial wetland

This scenario represents the current configuration at the ICTA-ICP building, which is treating wastewater through a biofilter in the front yard where the liquid fraction from toilets and urinals is processed. After the separation of the liquid and solid fractions, the waste fluids pass through an ultraviolet light to remove microorganism contamination. Then, it is piped to a collection pond, where it is diluted with water and finally pumped to a biofilter outside the building. Vegetation performs the water purification process, reducing the flow of wastewater that goes to the corresponding centralized WWTP. Finally, and although this has not been considered here since it is an unusual situation, in cases where overflow exists, a bypass water pipe conveys the excess of wastewater from the wetland to the sewer (i.e., S3). Thanks to this scenario, it is possible to analyze and compare whether the implementation of an N-recovery strategy (S2) that can further retain the value of wastewater by recovering nutrients, also translates into an improved environmental performance.

S2: Decentralized aerobic reactor

In this scenario, the moving bed biofilm reactor (MBBR) implemented collects only the yellow water coming from the urinals of the three toilets located in the southeast side of the ICTA-ICP building in order to transform the urine into liquid N fertilizer through a nitrification process, i.e. the oxidation of ammonia to nitrate (Özel Duygan et al., 2021). Urine with high nutrient rates passes through a 100-liter reactor and the tanks of the system where a flow of aeration and calcium carbonate is initially applied for pH regulation. After 2–3 weeks of residence time, N is ready to be used as liquid fertilizer. Since our aim is to better understand the potential of N recovery for urban agriculture instead of its removal from wastewater, we estimated the upscaling potential of the system to feed a tomato crop in ICTA-ICP's rooftop greenhouse. Literature data was used to determine tomato crops that can be fertilized with a certain amount of N fertilizer and actual data from previous harvests at ICTA-ICP (**Supplementary Material A1**). It is noteworthy to emphasize that this is a lab-scale study to explore the implementation of this system and its benefits.

S3: Centralized WWTP

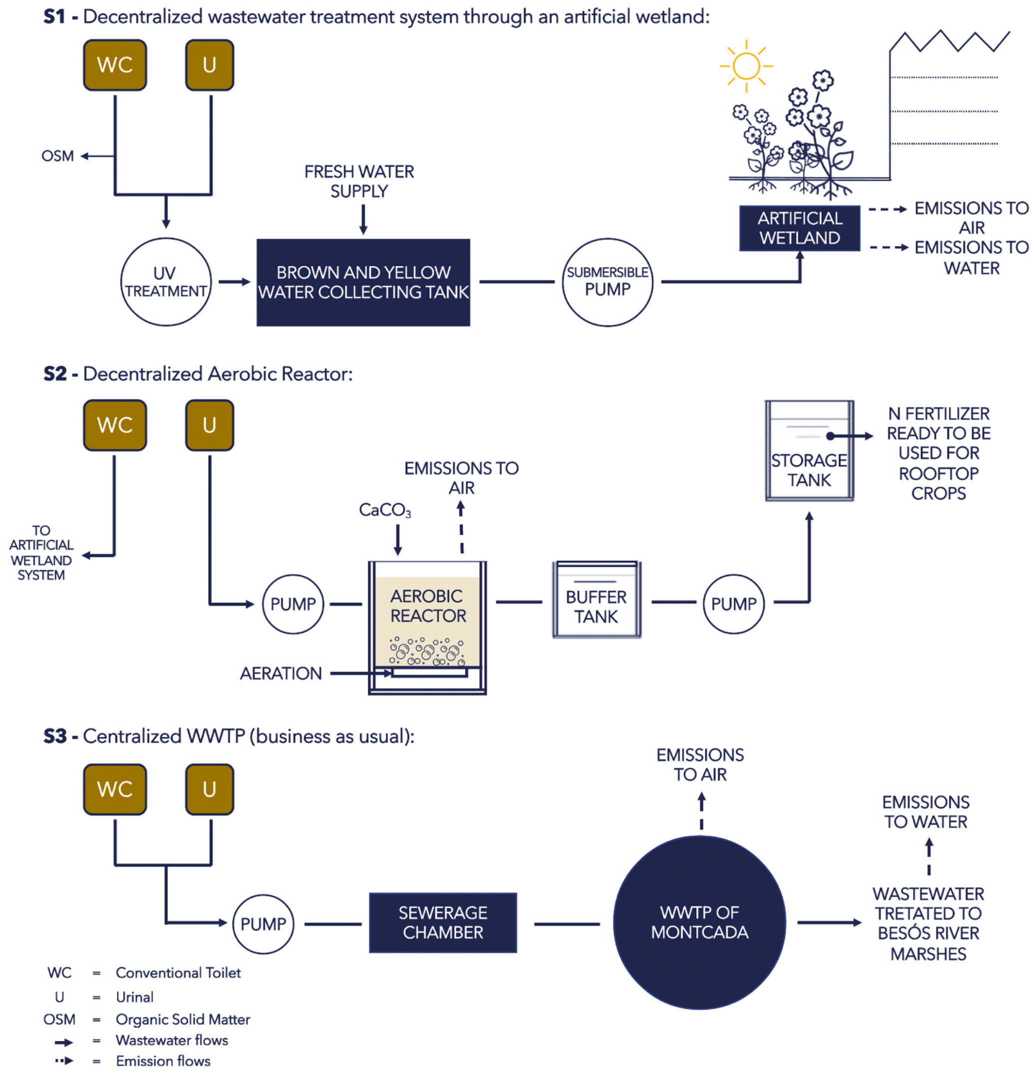

Fig. 1. Configuration and yellow water flows in the three wastewater treatment and recovery scenarios.

A third scenario considers a conventional treatment of yellow and black water, which in this case would be piped to a combined sewer and treated at the Montcada WWTP in Barcelona (specific operational scheme defined in **Supplementary Material B1**). This scenario represents a standard building construction configuration (i.e., business as usual, BAU), that also exists in the ICTA-ICP operating system to convey wastewater from building laboratories or when there is a need to temporally bypass wastewater (e.g., S1 during overflow or maintenance). No nutrient recovery processes are integrated in this WWTP (AMB, 2023; Rufí-Salís et al., 2020).

2.2. Estimation of urine flows

To quantify the amount of urine generated, an anonymous survey of the urinal users was conducted. In particular, the work consisted in counting the number of users using male dry urinals on the southeast side of the ICTA-ICP building (i.e., in 3 out of the 6 men's restrooms) during one working week. This enabled to account for the number of times the urinals were used. Assuming that all users answered the survey when using the urinals, 101 urine discharges were detected in one work week.

The composition and quantity of urine varies from person to person, from men to women, from children to adults (up to half of an adult) and from region to region depending on dietary habits, population size, and climate region. It is therefore not surprising that the literature results vary from 1.25 to 1.59 L/person/day divided into 4–5.4 times per day depending on the average household size (Karak and Bhattacharyya, 2011; Kavvada et al., 2017; Lam et al., 2015; Martin et al., 2023). To avoid oversizing, the excreted volume was taken from a previous LCA developed by Kavvada et al. (2017), assuming a daily urine generation of 1.4 L/person/day in 5.4 times per day, as this was the smallest amount found per discharge. This implies an average of 0.26 liters per flush, which, multiplied by the number of uses detected, gives the estimated volume of urine (**Supplementary Material A.3**).

For scenario S1, all the flush water from the toilets plus urinals was considered (excluding the organic solid matter), which is carried through the system to the artificial wetland. Since it also collects a mixture of wastewater (i.e., grey plus yellow water), physical allocation was used to distinguish the amount of urine transported and treated in the system compared to the total amount of wastewater in the building. Based on the data obtained from the survey and previously estimated building water flows in the ICTA-ICP building (Sanjuan-Delmas, 2017, p. 135), 0.31 % of the biofilter water corresponds to urine from the south-east side of the building. However, the system incorporates water from the conventional supply network to dilute the mixture which is also fed to the wetland. Therefore, a total of 4.38 % of what is processed in the biofilter corresponds to the final processing of the building's yellow water.

2.3. Life cycle assessment (LCA)

2.3.1. Goal and scope

The objective of the study is to identify the environmental impact of different yellow water management strategies in buildings. Since the aerobic reactor scenario (S2) is the only one that recovers nutrients, the functional unit (FU) focused on urine management, that is 1 m^3 of treated yellow water. Note that, at the WWTP, the N concentration of 1 m^3 of treated wastewater does not correspond with of 1 m^3 of treated urine. Nonetheless, it is presumed that if 1 m^3 of urine (FU) is not processed on-site, it will be sent to the Montcada facility, where it will be treated as an additional cubic meter of wastewater. Consequently, for comparative purposes, it is regarded as an equivalent unit. In the case of S3 the corresponding physical allocation of 4.38 % (as explained in Section 2.2) was made to adhere to the established FU.

The system boundaries include raw material extraction, and all the processes required for urine collection, storage, and processing/treatment (Fig. 2). Within the considered system inputs no organic solid matter was taken into account in S1 due to the early separation from the liquid fraction as well as its separate treatment system. The assessment concludes with the treated wastewater as the product. The end of life of the infrastructure is not included since end-of-life options are uncertain, but sludge final treatment in S2 and S3 is considered. The system's lifetime was set to 20 years for all materials and processes, except for the pumps, which need to be replaced every 12 years on average. In S1, the maintenance of the system and the wetland vegetation (and its carbon fixation potential) are outside the system boundaries. In S2, N recovery for fertilizer production is considered as a by-product. Therefore, system expansion was conducted to account for the avoided impacts of reducing the production of synthetic fertilizers. N fertilizer application to crops, food production and harvesting are beyond the scope of the LCA. For the WWTP scenario (S3), only the operation phase was included, as the study provides building-scale solutions and the WWTP is already in place and prepared to treat larger volumes of wastewater (e.g. Lundin et al., 2000).

2.3.2. Life cycle inventory

The life cycle inventories were compiled using a combination of primary data gathered from direct measurements and secondary data from existing databases and literature. For S1, primary design data of the system's materials and the operating processes of the building were used, together with information from literature when needed (the inventory data of the different scenarios can be seen in **Supplementary Material C**). Since S1 also processes grey water, physical allocation was required for greater accuracy in the impacts corresponding to yellow water only (see Section 2.2). In contrast, S2 only processes urine, so no physical allocation was needed. For materials and processes in S2, real data were collected from the lab-scale design project and with the collaboration of the aerobic reactor's installation company. Finally, information was taken from literature for S3 for the treatment of 1 m^3. Data on chemicals, transport and waste from the Calafell WWTP in Spain (Lorenzo-Toja et al., 2016) were used since its processes, size and location can be assimilated to those of Montcada WWTP, where information was not available. The transport distances were adjusted to correspond to those of S3, also considering that the sludge treatment is performed in the Besós WWTP due to the non-existing treatment capacity in the Montcada facilities.

2.3.3. Relevant assumptions

To develop the three proposed scenarios for the treatment of 1 m^3 of yellow water a set of assumptions were made for the N content in urine, the operation of the MBBR and the resulting emissions.

- N content in yellow water and FU: The N content of the chosen FU was set to 7.5 kg N/m^3 as reported by (Kavvada et al., 2017) with a 0.5 % of N volatilization in the concentrated urine nitrification

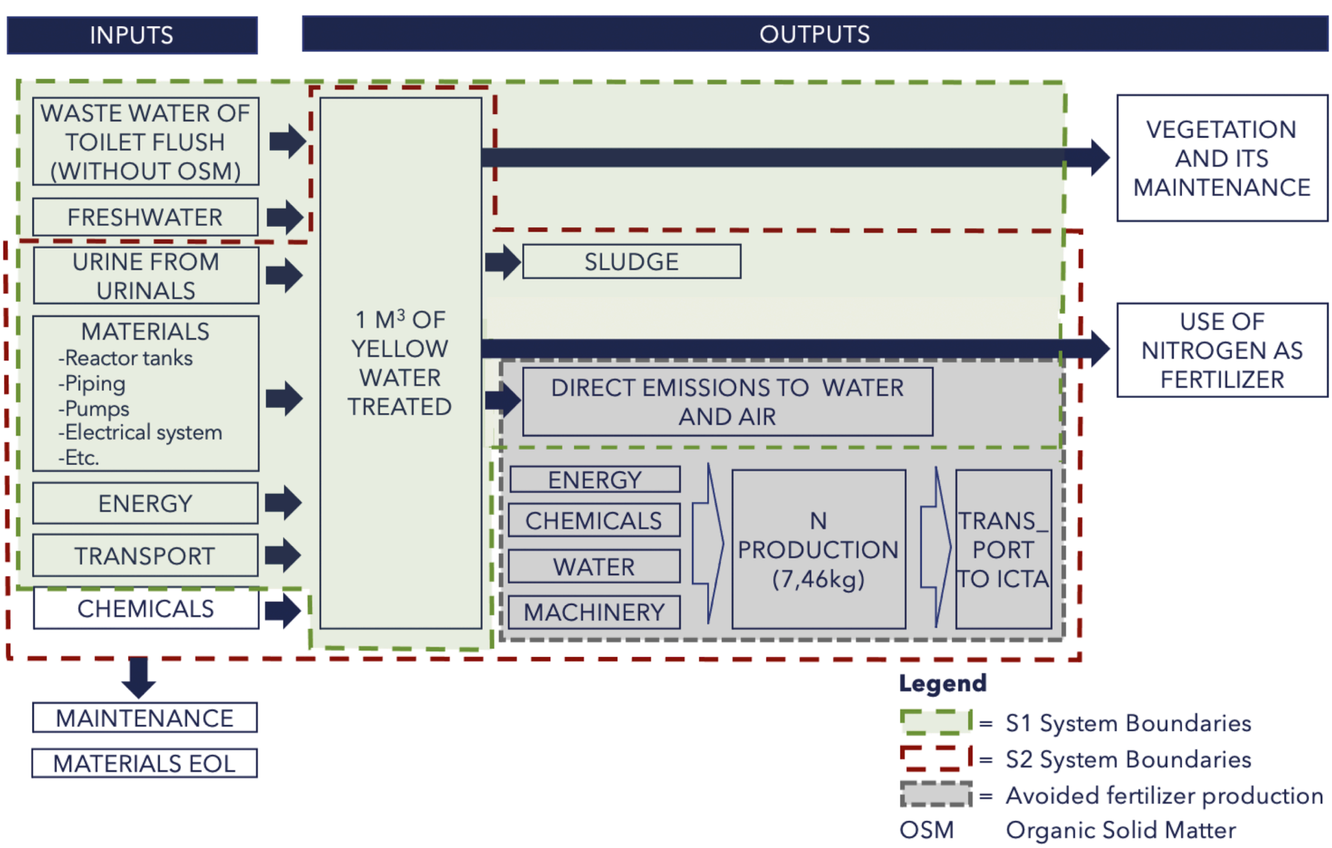

Fig. 2. System boundaries defined for the comparison of scenarios.

processes (Martin et al., 2023), corresponding to a total of 7.46 kg N/m^3. To achieve the chosen FU, an estimation of time has been made with the gathered information from users (further information can be seen in the Supplementary Information), defining 292 days for the production of the necessary yellow water.

- Energy needs: The determination of the energy requirements of the S2 scenario were based on the functional requirements from the pumps and aeration system as well as employed sensors which were specified for the pilot structure. The chosen aeration system is an air compressor with a 1.1 kW consumption. The use of devices was set for the duration of time to achieve the functional unit of 1 m^3 of treated yellow water. Due to the greater uncertainty and high contribution observed in the system energy needs, a sensitivity analysis was performed in Section 3.3 changing the aeration system to a turbine with 0.5 kWh consumption.
- MBBR pH regulation: For the operational needs of S2 scenario, a requirement for a strong base was identified due to the release of 0.107 mol H^+ per 1 g of N. $CaCO_3$ was used, as a standard base employed in WWTPs. The total amount applied for the 7.46 kg of N contained in the FU was set at 39.5 kg.
- Emissions to air and water: Relevant emissions to air and water were identified through existing literature for all scenarios. Emissions from wetlands were identified following research developed by Sovik et al. (2006) and Fuchs et al. (2011). Methane (CH_4) emissions were excluded from S1 due to the low organic matter content in the studied urine. For S2, emission factors were applied following the findings of Martin et al. (2023) for the emissions to air and Spångberg et al. (2014) for emissions to water. Finally, in S3 total emissions were calculated as specified by Lorenzo-Toja et al. (2016).
- Nutrient removal efficiency: Nutrient removal efficiency for the S1 scenario was calculated through the inflow and outflow N concentrations reported by Sovik et al. (2006) and Fuchs et al. (2011). For the WWTP scenario, direct removal information was obtained from the reference installation described in Section 2.3.2.

2.3.4. Impact assessment

The impact assessment includes the mandatory classification and characterization steps. The life cycle impact assessment LCIA Scores tool based on Brightway 2.5 was used (Muñoz-Liesa et al., 2024). Developed at ICTA-UAB, LCIA Scores is based on a Microsoft Excel file with build-in macros that calculates impact assessment scores for ecosphere and technosphere processes, fed from ecoinvent v3.9.1 as a background database. The ReCiPe (H) 2016 v1.03 (Huijbregts et al., 2017) and Cumulative Energy Demand v1.01 methods were used as implemented by ecoinvent 3.10 (Sonderegger and Stoikou,. 2022). The following midpoint impact categories were selected to depict key impacts of wastewater treatment systems: Climate Change (CC), Terrestrial Acidification (TA), Freshwater and Marine Eutrophication (FE, ME), Ozone Depletion (OD), Ecotoxicity (ET) (sum of marine, freshwater and terrestrial), and Water Use Consumption (WC). Additionally, the single-indicator Cumulative Energy Demand (CED) was analyzed. To understand the environmental viability of implementing the N-recovery system, we calculated its environmental payback time using the CC indicator. The equations defined by Rufí-Salís et al. (2022) were considered (see **Supplementary Material A.2**).

3. Results and discussion

3.1. On-site or off-site wastewater management: Comparative analysis

Our comparative analysis shows evidence of the potential benefits and tradeoffs of urine recovery in buildings (Table 1). The aerobic reactor (S2) does not release N-compounds into water bodies by transforming urine into nitrates, which simultaneously avoids the production of synthetic N fertilizers. This translates into beneficial effects on marine eutrophication (ME) with a reduction of 2.8 times compared to a WWTP, leading to -4.73E-03 kg N-eq/m^3, whereas the biofilter (S1) and the WWTP (S3) do contribute with N emissions to water due to their lower N-removal efficiencies, which are 53 % and 26 %, respectively. This

Table 1
Environmental impacts of the three scenarios per FU (1 m^3 of treated yellow water). ANP: avoided nitrogen fertilizer production. The values in bold represent the relative change with respect to S3.

	S1	S2	S2+ANP	S3
Climate change (kg CO_2-eq.)	**1.3** $1.09 \cdot 10^{+0}$	**73.0** $5.91 \cdot 10^{+1}$	**23.4** $1.90 \cdot 10^{+1}$	**1.0** 8.10 · 10e-1
Cumulative energy demand (MJ-eq.)	**1.3** $2.60 \cdot 10^{+1}$	**70.3** $1.46 \cdot 10^{+3}$	**36.1** $7.50 \cdot 10^{+2}$	**1.0** 2.08 · 10e+1
Total ecotoxicity (kg 1,4-DCB-eq.)	**1.9** $1.02 \cdot 10^{+1}$	**166.4** $8.94 \cdot 10^{+2}$	**112.0** $6.02 \cdot 10^{+2}$	**1.0** 5.38 · 10e+0
Water consumption (m^3-eq.)	**1.6** $9.46 \cdot 10^{-3}$	**81.2** $4.82 \cdot 10^{-1}$	**-86.9** $-5.16 \cdot 10^{-1}$	**1.0** 5.93 · 10e-3
Freshwater eutrophication (kg P-eq.)	**3.4** $2.10 \cdot 10^{-3}$	**33.3** $2.09 \cdot 10^{-2}$	**20.7** $1.30 \cdot 10^{-2}$	**1.0** 6.26 · 10e-4
Marine eutrophication (kg N-eq.)	**0.4** $3.53 \cdot 10^{-3}$	**0.3** $2.57 \cdot 10^{-3}$	**-0.6** $-4.72 \cdot 10^{-3}$	**1.0** 8.39 · 10e-3
Ozone depletion (kg CFC-11-eq.)	**1.5** $5.78 \cdot 10^{-6}$	**114.5** $4.35 \cdot 10^{-4}$	**98.0** $3.73 \cdot 10^{-4}$	**1.0** 3.80 · 10e-6
Terrestial acidification (kg SO_2-eq.)	**1.5** $3.68 \cdot 10^{-3}$	**136.6** $3.35 \cdot 10^{-1}$	**68.6** $1.68 \cdot 10^{-1}$	**1.0** 2.45 · 10e-3

Less impact — More impact

negative term, together with the water consumption impact (WC), does not imply that both categories have a net positive impact or that impacts have been sequestered, but rather correspond to the load that is being ceased to be emitted by avoiding the production of synthetic fertilizer which is being displaced in the market.

Notwithstanding both ME and WC benefits, nutrient recovery at the building scale performs worse across the remaining 6 impact categories by up to 2 orders of magnitude with respect to S1 and S2. For instance, in the climate change impact category, accounting for the avoided impacts of synthetic fertilizers (S2 + ANP, with nearly 40 kg CO_2-eq/m^3) does not balance out the environmental impacts of the aerobic reactor (e.g. 59.1 kg CO_2-eq/m^3). A similar trend was observed when comparing the treatment of greywater in constructed wetlands, membrane bioreactors and WWTPs at the community, neighborhood and household scale (Kobayashi et al. 2020). In this case, membrane bioreactors only displayed favorable environmental results when applied at the community scale and when reusing large amounts of greywater. Due to its similar lab-scale set-up of an MBBR for urine nitrification, these results can be partially compared to the findings presented by Badeti et al. (2024). Our study reports 19 kg of net CO_2 eq/m^3 (scenario S2+ANP), whereas Badeti et al. (2024) estimated 25.6 kg CO_2 eq/m^3. However, the assessments cannot be fully compared, as the latter study did not consider the complete life cycle of the system and only evaluated direct air emissions and electricity consumption.

3.2. Contribution analysis

Energy consumption and direct air emissions are the main sources of impact in the aerobic reactor (Fig. 3), but these are also the flows that hold most uncertainty. Although the infrastructure needed to build the aerobic reactor was modelled using real data from the lab set-up, the operation relies on theoretical estimates (see Section 2.3.3). In fact, the results are highly sensitive to energy consumption, both in terms of electricity mix and energy intensity (see sensitivity analysis in Section 3.5). Nevertheless, the literature suggests that our results fall within the typical energy use of an MBBR. The aerobic reactor requires 128 kWh/m^3 of treated yellow water or 17.2 kWh/kg N, whereas similar lab tests reported between 11 and 59 kWh/kg N (Fumasoli et al. 2015) and up to 3 kWh/kg N (Badeti et al. 2024) in the nitrification of urine. In contrast, the operation of the Haber-Bosch process requires 10–14 kWh/kg N to produce synthetic fertilizers (Martin et al., 2023). The operational energy demand is higher in S2 than in S1 and S3. While the biofilter requires a minimal energy input to run the influent pump, the WWTPs generally benefit from economies of scale. Thus, this result must be taken with care, as it refers to a lab-scale design.

Two relevant environmental trade-offs are worth highlighting when comparing the environmental impacts of WWTPs (S3) and constructed wetlands (S1): ecotoxicity impacts and ozone depletion impacts. The former is due to the material impacts of the bioreactor due to the need for additional equipment (e.g., pumps), instrumentation (e.g., sensors and control units) and electrical connections (e.g., cables) to be operated. In turn, the ozone depletion impacts can be mainly found in the direct emissions. Both impact categories, even when incorporating the avoided impacts of the off-site fertilizer production (ANP), can only offset their burden by 24.6 % and 12.6 %, respectively.

Direct air emissions, CH_4 and N_2O also play an essential role, especially in WWTPs in the climate change impact category. In particular, N_2O and NH_3 emissions contribute to 11 % of the impacts of the aerobic reactor (Fig. 4), while in the case of the WWTP, CH_4 and N_2O represent 17 % of the impacts. This has been found despite higher contributions reported by Lorenzo-Toja et al. (2016), who found that direct GHG emissions can make up to 60 % of a WWTP's impact on global warming. This difference could be due to the lack of reported values in some stages of the WWTP studied by Lorenzo-Toja et al. (2016) and added in our study given its proximity and similarity to the case study. Similarly, Badeti et al. (2024) measured that around 60 % of the GHG emissions of an MBBR for urine nitrification were associated to N_2O and CO_2 emissions.

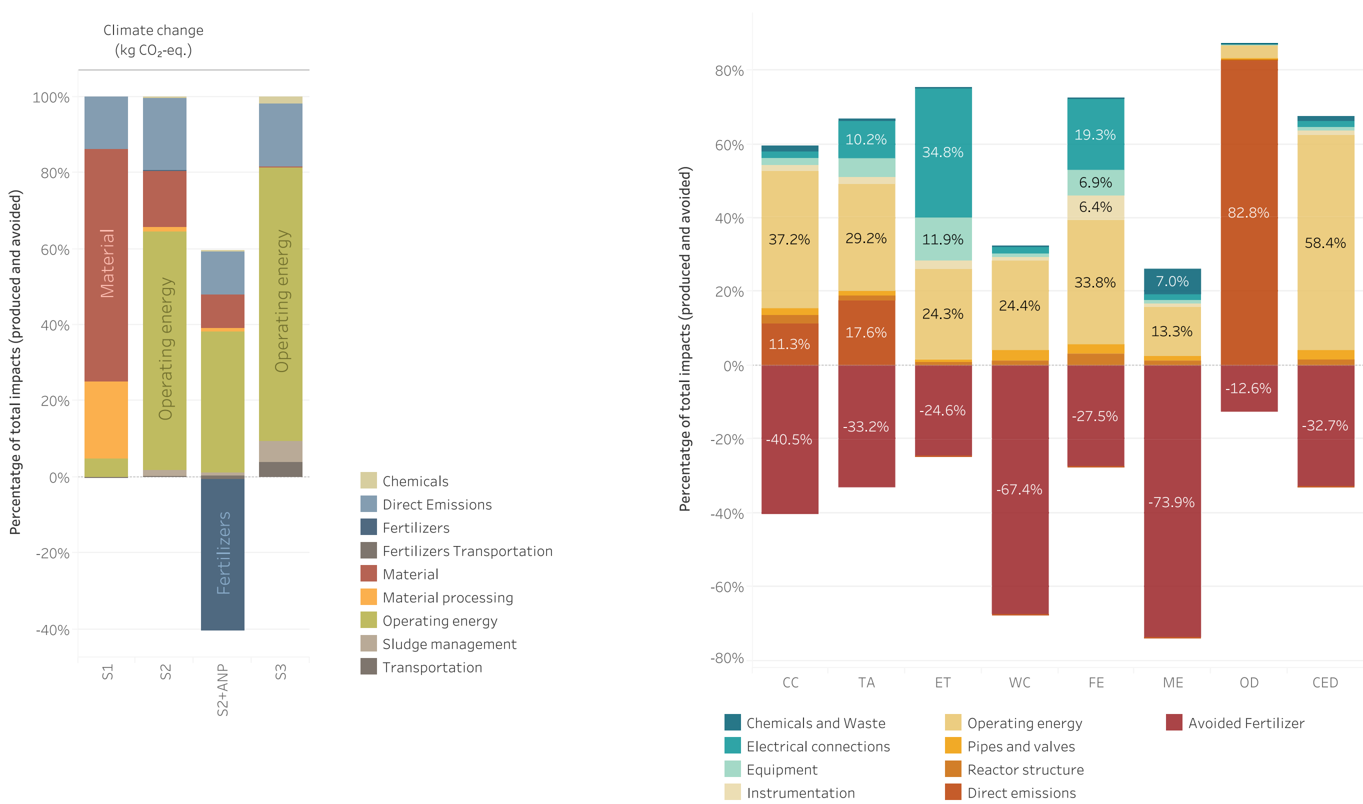

Fig. 3. Contribution analysis of the decentralized aerobic reactor (S2) as a percentage of total produced and avoided impacts across all categories analyzed.

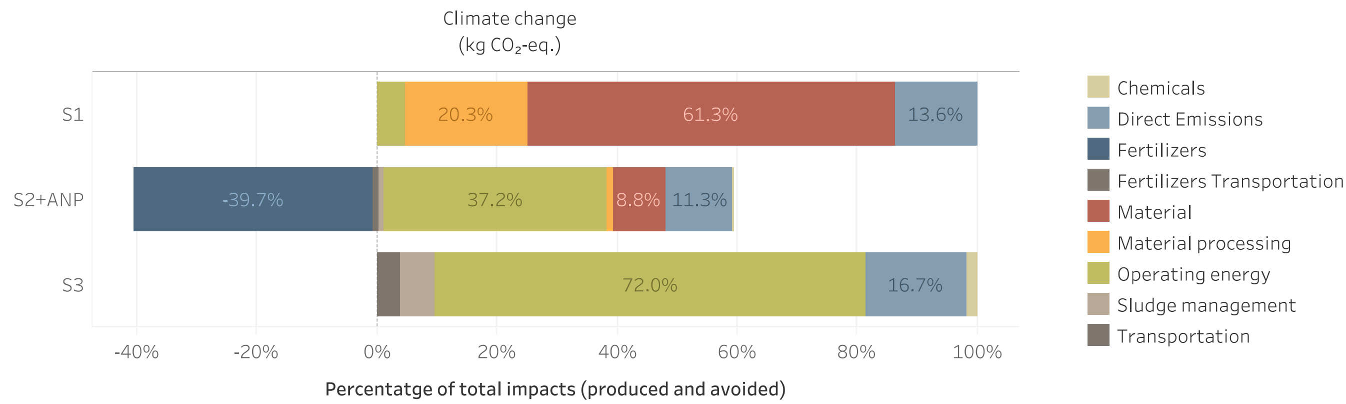

Fig. 4. Contribution analysis of Climate Change (CC) impacts for all three scenarios analyzed.

Finally, the raw material use and processing accounts for 61 % of the climate change impacts in the S1 scenario. This is explained by the need for a collection and storage system for the constructed wetland, which uses pipes and a 2,000-liter tank, adding up to 5.17 kg of polyethylene which represent approximately half of the material impacts. Excluding ANP, materials are the top 3 main contributors in S2, followed by the above-mentioned energy use and direct emissions. In contrast, operating energy and direct air emissions in S3 account for 72 % of CO_2-eq emissions.

3.3. Sensitivity analysis

The results presented here are dominated, and thus highly sensitive, to energy consumption, both in terms of electricity mix and energy intensity. Within the current S2 scenario we assumed that the anaerobic reactor requires a constant aeration pump through an air compression system sourced from the Spanish electricity mix. However, in larger scale settings we assumed that an aeration turbine would be used, increasing the efficiency of the system from 128 to 58 kWh/m^3 of treated yellow water or 7.8 kWh/kg N. By doing so, the environmental impacts in the climate change category will be offset by −217 % compared to S3 scenario (Fig. 5), while it would improve the impacts of the rest of impact categories analyzed. However, considering the worst circumstances with the original energy consumption and the European electricity mix, environmental impacts increase up to 36.8 times the climate change impacts of S3 scenario. Finally, except for ET and OD categories,

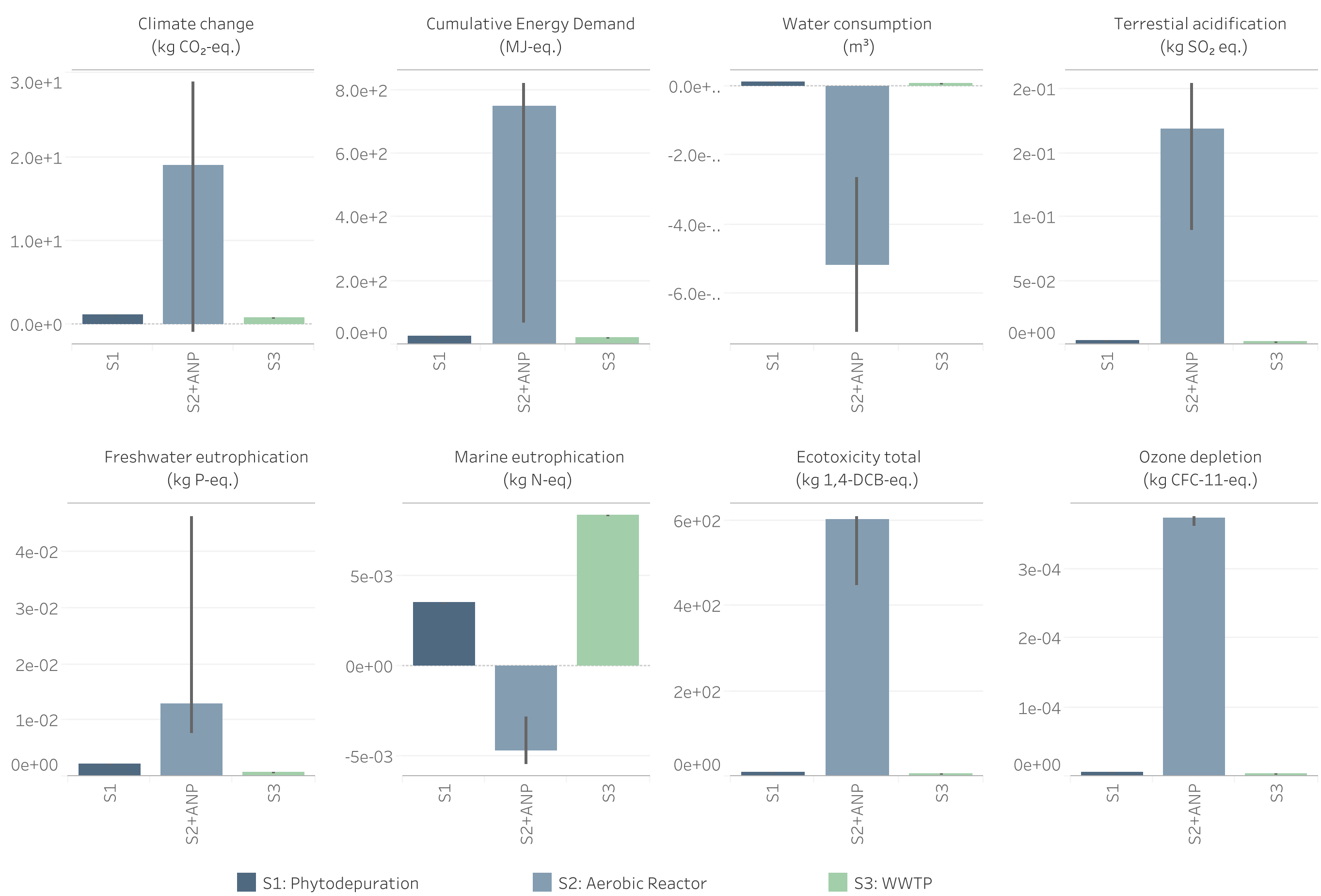

Fig. 5. Comparative analysis considering different electricity consumption and mixes for the S2 scenario for all impact categories and all three scenarios analyzed.

sourcing the electricity of the bioreactor from renewable sources is a key factor to offset the anaerobic reactor impacts and to produce nitrogen from urine in a sustainable manner.

3.4. Applying nutrient recovery to urban agriculture

Although the system boundaries considered here do not include crop production, to get a more tangible idea of the possibilities offered by urine recovery in a building, the potential for crop fertilization with the treatment of 1 year of yellow water is analyzed here. The estimated yields that could be grown on an open field using the urine recovered from the 3 urinals (9.32 kg N/year) are approximately 583 m^2 of tomatoes which correspond with 3,036 kg of tomatoes per year (Karak and Bhattacharyya, 2011). This size is similar to the entire four iRTG at ICTA-ICP (summing up 512 m^2). In addition, if a wastewater separation system was incorporated in the female and male toilets, this amount could increase to up to 0.34 ha of tomatoes per year (54.6 kg N). This would correspond to about 17.8 tons of tomatoes for open crop areas (Pérez-Neira et al., 2018). Currently, at ICTA-ICP an average of 1,338 kg of tomatoes are harvested per year, distributed in 84.34 m^2 of net area for each rooftop greenhouse (15.7 $\pm$ 1.54 kg/m^2) (Muñoz-Liesa et al., 2022; Parada et al., 2021; Rufí-Salís et al., 2020b). If we consider that 0.048 kg N/m^2 are needed for tomato crops inside the building (by scaling the data from Karak and Bhattacharyya, 2011 and Pérez-Neira et al., 2018), 4.07 kg N fertilizer would be needed per year, thus managing to cover the demand by 2.3 times with the current urine production coming from the 3 urinals (see calculations in **Supplementary Material A.1**). This analysis must be further explored using the yields of crops grown in an iRTG using recovered N, which has been barely explored.

Although not further developed in the present research, related literature on the fertilization of crops from a membrane bioreactor effluent fed with source separated urine also describe de content of other nutrients, apart from nitrogen, which are appropriate for plant fertilization. Especially relevant would be P with an indicated content of 0.3 to 0,6 g/L, although presented nutrient contents of the produced effluent also show relevant amounts of K and sulfur (Sohn et al., 2024). These additional nutrients could, as well as N, be avoided when fertilizing with the present effluent.

If the urine treatment of only half of the building urinals is considered, 546.5 kg CO_2-eq. emissions could be avoided every year. Even if this exceeds the cultivation needs of the iRTG, the building can become an exporter of N fertilizer. This scenario can be further improved if the treatment of yellow water from the 6 available urinals is considered, assuming the same number of uses on both sides of the building, the current ICTA-ICP flow rates (Sanjuan-Delmás, 2017), a constant volume of the reactor, and taking into consideration the increase in infrastructure. Thus, by doubling the amount of N fertilizer produced, the impacts on the CC category decrease by up to 1.29 times while the rest of impact categories decrease between 1.01 (ozone depletion) to 2.08 times (water consumption, Table 2). These potential improvements highlight the variability of environmental impacts when prototype and small-scale settings are assessed in LCA, which can decrease by 90 % when scaled up to more industrial systems (Gavankar et al., 2015). Therefore, note that these improvements can significantly change the results compared to well-stablished WWTP technologies here assessed (S3), which already benefit from economies of scale. Nevertheless, the amount of recovered urine is strongly dependent on user perceptions. In the case of ICTA-ICP, for instance, chemical filters were removed from the urinals to collect pure urine, which could generate unpleasant smells and discomfort for restroom users. ICTA-ICP users did not express discomfort, nor did they limit their use of these urinals in favor of conventional toilets, but user feedback did stress the importance of dealing with odors when implementing this system in the long run.

On the other hand, one could consider the additional input of recovered N from WWTPs. The Montcada WWTP is currently not equipped with this recovery system, but facilities around the world are working towards this goal. In the Montcada WWTP, N recovery could be done through a biological reactor for N recovery after pretreatment and primary treatment, which generates struvite, ($MgNH_4PO_4{\cdot}6H_2O$), a crystalline by-product of WWTPs containing high N and P concentrations that forms by spontaneous or induced precipitation (Rahman et al., 2014). Considering an N concentration in struvite of 5.7 % (Arcas-Pilz et al., 2021) and the total amount of water going to the biofilter to be diverted to the recovery plant (1,152 m^3 per year), 0.0035 kg of N could be recovered per m^3 of water every year. This estimate is based on Rufí-Salís et al. (2020) for ICTA-ICP wastewater flows, which in a year represent a production of about 4.03 kg N. If 4.07 kg N are needed annually in the iRTG crops, then the demand is practically covered (99 %).

Table 2
Impact assessment of the decentralized aerobic reactor (S2) per cubic meter of yellow water, considering half or all the urinals available in the ICTA-UAB building are recovered.

Impact categories	Recovery from 50 % of the building's urinals	Recovery from 100 % of the building's urinals	Times the improvement (100 % vs 50 %)
Climate Change (CC) (kg CO2 eq)	$1.90 \cdot 10^{+1}$	$1.47 \cdot 10^{+1}$	1.29
Cumulative Energy Demand (CED) MJ-Eq	$7.50 \cdot 10^{+2}$	$6.69 \cdot 10^{+2}$	1.12
Total Ecotoxicity (ET) (kg 1,4-DCB eq)	$6.02 \cdot 10^{+2}$	$3.03 \cdot 10^{+2}$	1.99
Water Use Consumption (WC) m3	$-5.16 \cdot 10^{-1}$	$-5.58 \cdot 10^{-1}$	2.08
Freshwater Eutrophication (FE) (kg P-eq)	$1.30 \cdot 10^{-2}$	$7.68 \cdot 10^{-3}$	1.69
Marine Eutrophication (ME) (kg N-eq)	$-4.72 \cdot 10^{-3}$	$-4.96 \cdot 10^{-3}$	2.05
Ozone Depletion (OD) (kg CFC-11-eq)	$3.73 \cdot 10^{-4}$	$3.71 \cdot 10^{-4}$	1.01
Terrestrial Acidification (TA) (kg SO_2 eq)	$1.68 \cdot 10^{-1}$	$1.21 \cdot 10^{-1}$	1.39

3.5. Payback period of nutrient recovery

To estimate the time needed to compensate for the environmental impacts caused by the aerobic reactor, we considered the ANP scenario of S2. For this purpose, the total impacts on the CC category during the 20 years of its use (1,478 kg CO_2-eq) have been considered and divided by the impacts avoided in one year (59.1 and 54.8 kg CO_2-eq for 50 and 100 % of building's urinals, respectively). For the climate change impact category, this gives a payback time of 29.4 years for the treatment of wastewater from the southeast urinals in the reactor, and 13.7 years if we consider the production of urine from all urinals in the building to transform the incoming yellow water into N fertilizer (see **Supplementary Material A.2**). This demonstrates that the volume of treated urine plays a critical role in the application of this technology (Kobayashi et al. 2020), and would be thus recommended for highly populated buildings, such as educational facilities or compact neighborhoods.

4. Conclusions

There is an urgent need to meet the global supply needs of fertilizers while complying with current sustainability agendas. This assessment explores the environmental impacts of alternative strategies to manage urine from a case-study building. The LCA results showed that the aerobic reactor (S2) is beneficial in terms of avoided N fertilizer production in 2 of the 8 impact categories evaluated. These categories correspond to ME and WC which performed 0.6 and 86.9 times better than S3, respectively. This improvement is due to the efficient N recovery, limiting emission to water as well as the highly water efficient treatment system compared to S1 and S3.

The current system of wastewater treatment through an artificial wetland (S1) shows a similar performance to the WWTP in most impact categories but marine eutrophication due to its higher recovery rate. Further analysis revealed that S2 had higher impacts in OD and ET, which were 98 and 112 times higher than S3. The main contributions causing these larger impact fall under the electrical installation, the operation energy use and the systems direct emissions. In a similar way, the main contributions to the CC impact category correspond mainly to the energy use (37.2 %) and direct emissions (11.3 %). The importance of the energy use and energy mix is further demonstrated in the sensitivity analysis, showing high variability in most impact categories, especially CC and CED.

The upscale of the MBBR recovery system from 3 to 6 urinals shows a direct improvement in each impact category as well as a direct reduction of the installation payback time from 29 years (with 3 urinals) to 13 years (with 6 urinals). Therefore, the environmental performance of the S2 scenario could be greatly improved by scaling up the nutrient recovery system to a neighborhood or university campus scale, for example. Together with an improved aeration turbine suitable for large scale settings and a renewable energy mix, environmental impacts such as climate change could be reduced two-fold, offsetting the impacts of the urine reactor.

Finally, applying nutrient recovery to urban agriculture shows that the benefits of on-site N recovery are undoubtedly a very advantageous alternative. With 1 m^3 of treated yellow water, 2.4 tons of tomatoes could be cultivated in outdoor cultivation, and if source-separation systems for toilets were implemented throughout the building, >17 tons of tomatoes could be fertilized in one year. These findings can facilitate decision-making processes to optimize wastewater management systems, defining which one to use and promote sustainable practices in buildings. We encourage interdisciplinary research teams to further explore potential techniques to recover N from urine and to demonstrate their agronomic feasibility and food safety in building integrated agriculture. This shall promote the implementation of circular strategies in buildings that facilitate the production of local food at lower environmental costs.

CRediT authorship contribution statement

María Virginia Maiza: Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Formal analysis, Data curation, Conceptualization. **Joan Muñoz-Liesa:** Writing – review & editing, Writing – original draft, Visualization, Validation, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Anna Petit-Boix:** Writing – review & editing, Writing – original draft, Visualization, Supervision, Methodology, Formal analysis, Data curation, Conceptualization. **Verónica Arcas-Pilz:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Funding acquisition, Data curation, Conceptualization. **Xavier Gabarrell:** Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

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

Acknowledgements

Financial support of "Building Integrated Agriculture for an effective ecological transition" (BINAFET) project BINAFET TED2021–130047B-C21 and TED2021–130047B-C21, with financing from the European Union (NextGenerationEU fund) and within the framework of the Recovery, Transformation and Resilience Plan. This research was also supported by the WEF4Build project (2023 Clima 00041) "Boosting Buildings Climate Resilience through Water-Food-Energy Nexus based Solutions", funded by AGAUR, Generalitat de Catalunya. The authors would like to thank Juan Antonio Baeza and Albert Guisasola, ICTA-UAB staff and the aerobic reactor installation company AERIS for their technical support and data provision. J. Muñoz-Liesa would like to acknowledge the Swedish Research Council for Sustainable Development (FORMAS) funding for "ECO-FARM CITY" project (Circular vertical farms to reduce environmental impacts in cities, grant number 2022–02850). A. Petit-Boix acknowledges the support of the María Zambrano grant (MZ 2021–19) under the European Union's "NextGenerationEU" program, funded through the Spanish Ministry of Universities. The authors thank the Catalan agency AGAUR through its research group support program (2021 SGR 00734), and the Spanish Ministry of Economy, Industry and Competitiveness for funding the "María de Maeztu" program for Units of Excellence in R&D (CEX 2019–0940-M).

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2024.107985.

Data availability

Data is provided in the supplementary material

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