Science of the Total Environment 905 (2023) 167193

Contents lists available at ScienceDirect

Science of the Total Environment

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

Review

Literature review on the potential of urban waste for the fertilization of urban agriculture: A closer look at the metropolitan area of Barcelona

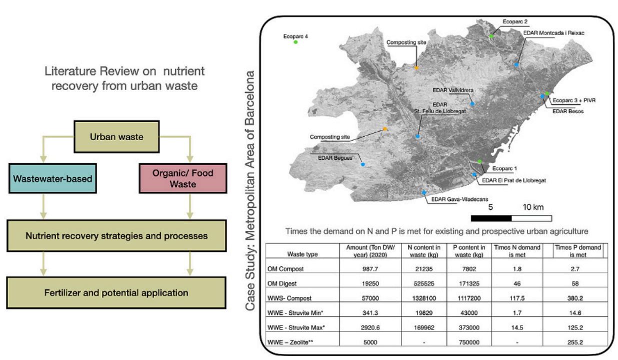
Verónica Arcas-Pilz [a], Xavier Gabarrell [a,b,*], Francesco Orsini [c], Gara Villalba [a,b]

[a] Sostenipra Research Group (2021 SGR 00734), Institut de Ciència i Tecnologia Ambientals ICTA-UAB (CEX2019-0940-M), Z Building, Universitat Autònoma de Barcelona (UAB), Campus UAB, 08193 Bellaterra, Barcelona, Spain
[b] Department of Chemical, Biological and Environmental Engineering, Universitat Autònoma de Barcelona (UAB), Campus UAB, 08193 Bellaterra, Barcelona, Spain
[c] DISTAL—Department of Agricultural and Food Sciences, Alma Mater Studiorum—University of Bologna, 40127 Bologna, Italy

HIGHLIGHTS

- Nutrient circularity is key due to losses into the environment and depleting resources.
- Urban agriculture can serve as a link to increase nutrient circularity in cities.
- Urban waste can be divided into two streams wastewater and organic, bio, food waste.
- Existing structures like WWTP can supply great amounts of key nutrients like N and P.
- Social perception and legal constraints are key in the future of nutrient recovery.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Damia Barcelo

Keywords:
Nutrient circularity
Waste management
Urban agriculture
Nutrient recovery
Wastewater
Food waste

ABSTRACT

Urban agriculture (UA) activities are increasing in popularity and importance due to greater food demands and reductions in agricultural land, also advocating for greater local food supply and security as well as the social and community cohesion perspective. This activity also has the potential to enhance the circularity of urban flows, repurposing nutrients from waste sources, increasing their self-sufficiency, reducing nutrient loss into the environment, and avoiding environmental cost of nutrient extraction and synthetization.

The present work is aimed at defining recovery technologies outlined in the literature to obtain relevant nutrients such as N and P from waste sources in urban areas. Through literature research tools, the waste sources were defined, differentiating two main groups: (1) food, organic, biowaste and (2) wastewater. Up to 7 recovery strategies were identified for food, organic, and biowaste sources, while 11 strategies were defined for wastewater, mainly focusing on the recovery of N and P, which are applicable in UA in different forms.

The potential of the recovered nutrients to cover existing and prospective UA sites was further assessed for the metropolitan area of Barcelona. Nutrient recovery from current composting and anaerobic digestion of urban sourced organic matter obtained each year in the area as well as the composting of wastewater sludge, struvite precipitation and ion exchange in wastewater effluent generated yearly in existing WWTPs were assessed. The

* Corresponding author at: Sostenipra Research Group (2021 SGR 00734), Institut de Ciència I Tecnologia Ambientals (CEX2019-000940-M), Z Building, Universitat Autònoma de Barcelona (UAB), Campus UAB, 08193 Bellaterra, Barcelona, Spain.
E-mail address: xavier.gabarrell@uab.cat (X. Gabarrell).

https://doi.org/10.1016/j.scitotenv.2023.167193
Received 18 April 2023; Received in revised form 20 August 2023; Accepted 16 September 2023
Available online 22 September 2023
0048-9697/

results show that the requirements for the current and prospective UA in the area can be met 2.7 to 380.2 times for P and 1.7 to 117.5 times for N depending on the recovery strategy. While the present results are promising, current perceptions, legislation and the implementation and production costs compared to existing markets do not facilitate the application of nutrient recovery strategies, although a change is expected in the near future.

1. Introduction

1.1. Urban agriculture: reducing distances and optimizing the use of space

In past decades, the increase in population in expanding urban areas has risen the demand for food (United Nations, 2019), putting great pressure on the agricultural industry to supply cities within the existing agricultural land. This additional stress builds over the ongoing challenges in agriculture, including climatic instability and land degradation (Dsouza et al., 2021). These pressures have prompted the need for highly intensified production systems that can further contribute to land degradation, the use of non-renewable resources and greenhouse gas emissions (Chojnacka et al., 2020; Dsouza et al., 2021). Agriculture and the total food system are currently responsible for 21–37 % of all GHG emissions, which are expected to increase by 30 % in 2050 due to population growth, dietary change, income growth and consequentially land-use change (IPCC, 2019). One plausible approach to mitigate the considerable pressures associated with sustaining urban populations is to enhance the resilience and self-sufficiency of urban centres. This approach would foster an environment for food production within the city (Ercilla-Montserrat et al., 2019; Sanyé-Mengual et al., 2018).

The concept of UA has garnered significant attention in recent years and is recognized as a viable means to address food provisioning in urban regions. Additionally, it serves as a catalyst for fostering community cohesion and augmenting the availability of green spaces within cities (Lal, 2020; Wielemaker et al., 2019). The potential benefits of UA also include shortening supply chains and consequently reducing transportation and food losses (Sanyé-Mengual et al., 2015; Sanyé Mengual, 2015; Toboso-Chavero et al., 2019). These benefits may change depending on the typology of UA applied, which can greatly vary between soil-based outdoor agriculture and indoor hydroponic vertical farming.

UA has different appearances and can present itself in different shapes and forms, as well as motivations and functions. Urban residents are often guided by the traditional images of agriculture; therefore, most family and community gardens, as well as social peri-urban farms, are conceived to follow the principles of organic or agroecological farming on soil. While UA globally faces issues in accessing fertile soils in highly dense cities, alternative ways for crop production have been suggested, including building integrated agriculture solutions (Despommier, 2013) or Z-Farming (Thomaier et al., 2015), leading to the colonization of often forgotten and underutilized spaces, as for instance, building rooftops (Appolloni et al., 2021).

The production systems most suitable for rooftop and indoor UA are mainly based on soilless systems with the use of alternative substrates to avoid heavy loads. The practices of hydroponic/aeroponic agriculture and aquaponics have been shown to be great alternatives for building-based agricultural systems, although downsides can also be observed. Hydroponic technologies can entail a great investment in infrastructure and require technical specialization for their manipulation. Besides, these technologies rely on the use of mineral nutrients, commonly dissolved into a nutrient solution supplied to the crop through fertigation. While productivity commonly increases, as compared to soil-based agriculture, these practices imply a great environmental burden, being based on linear systems (Dsouza et al., 2021; Sanjuan-Delmás et al., 2018).

Efforts have been made to reduce the emissions of UA production related to fertilization by closing flows within the crop system, reusing the leachate nutrients for the same crop or as a cascade system on less demanding crops before being discarded or further reused (Rufí-Salís et al., 2020c; Rufí-Salís et al., 2020b). While this can certainly be a plausible solution to minimize fertilizer and water loss into the environment, it does not entirely solve the burden of nutrient extraction or generation to further increase agricultural activity inside the city, such as in the case of phosphate rock mining or synthetic nitrogen production (Cordell et al., 2009; Cordell and White, 2014). On the other hand, additional infrastructure and equipment are installed to close the water flow and to ensure both the stability of the given nutrient solution and the control of pests and diseases, making the system much more complex.

While the notion of UA and the utilization of building rooftops offer enticing prospects for augmenting agricultural pursuits within urban areas, the integration of nutrient flows necessitates thorough investigation. Such exploration holds the promise of facilitating a transition from conventional linear systems to more sustainable and circular alternatives. By delving into the possibilities of nutrient flow integration, cities can potentially embrace more efficient and regenerative practices, ushering in a new era of urban agricultural sustainability.

1.2. Do we need circular nutrient strategies in cities?

The extraction and synthetization of fertilizers has been an ongoing activity since the agricultural green revolution, becoming a necessity to maintain the great production of goods to feed the world. While earlier methods of fertilization included the use of urine, manure, human excreta and guano, the modern agricultural industry relies on the mining and synthetization of plant available nutrients (Sun et al., 2018). This practice has greatly shifted the global nutrient pools with the export and import of these products to ensure fertilization of large land extensions (Villalba et al., 2008).

This nutrient pool shift has generated several consequences not only due to the extraction of nutrients by excessive mining but also via the emissions generated by their transport and application elsewhere. Fertilization is also responsible for the leakage of excess nutrients towards the environment, generating exosystemic problems that include acidification, eutrophication, or emissions of GHGs into the atmosphere (Cordell et al., 2009; Liu et al., 2008). The current anthropogenic sources of N_2O mainly originate from nitrogen over fertilization and poor management, with soil emissions of 3 Mt. of N_2O-N each year (IPCC, 2019). Today, approximately 90 % of the extracted phosphate rock is still used to produce agricultural fertilizer, while 50 % of the nitrogen demanded for agriculture is supplied by synthetic N generated via the Haber-Bosch process (Chojnacka et al., 2020; Zabaleta and Rodic, 2015). The application of P fertilizers is also regarded as greatly inefficient with extensive losses due to erosion and leaching, with only approximately 10 % of the applied P reaching consumers (Chojnacka et al., 2020). This non-renewable resource is expected to be depleted by 20–35 % by 2100 under best estimates if its extraction continues at present rates, reaching its peak production rate between 2030 and 2040 (Möller et al., 2018; Van Vuuren et al., 2010).

Significant strides in diminishing reliance on non-renewable nutrients and curbing the release of P into the environment can be accomplished through the implementation of recovery and reuse strategies. Despite their potential, these strategies are presently underutilized and warrant increased attention and implementation (Chojnacka et al., 2020; Oarga-Mulec et al., 2019). A large share of nutrient losses in urban areas is associated with food and human waste. Estimations of global annual food waste indicate that 931 million tonnes were generated in

2019, approximately 121 kg per capita each year, where approximately 60 % originated from households (Forbes et al., 2021). Approximately 97 % of global food waste is disposed in landfills, where inappropriate management may cause nutrient leaching into the environment, causing eutrophication and accumulation in soils, as well as methane emission and odour (Chojnacka et al., 2020; Ren et al., 2020).

With a circular economy mindset, the potential of the reuse and recycling of wastes is explored, and the capacity for self-sufficiency and resilience is enhanced, escaping from linear production approaches that entail the need for importing and exporting resources into and out of the system (Girotto and Piazza, 2021; de Kraker et al., 2019). Presently, urban nutrient cycles predominantly lack a circular approach to effectively manage and recycle nutrients, resulting in a significant reduction in the self-sufficiency of cities. The import of external manure, or other organic or synthetic nutrients to pursue UA may seem redundant due to an increase in nutrient loss within the urban ecosystem as well as the already existing potential source of nutrients within urban areas (Martin et al., 2019; Wielemaker et al., 2019).

Here is where UA can increase its benefits for sustainable urban development, not only raising its food security with the local production of goods, as well as increasing green spaces and biodiversity, but also serving as a destination for urban recovered nutrients to further close urban material flows (de Kraker et al., 2019). The increasing interest in UA and hydroponic production with the use of recycled nutrients can serve as a marker to further encourage nutrient recovery practices that can also become profitable in the near future (Martin et al., 2019).

The focus on waste disposal and nutrient recovery has been greater in recent years with the impulse of new EU regulations and development goals striving for better waste management strategies with a reduction of landfilling as well as making P recovery in wastewater treatment plants (WWTPs) a requisite (Kratz et al., 2019). The primary objective of this study is to comprehensively explore the various avenues currently being investigated for nutrient recovery in urban settings and define if it's possible to cover the needs of existing or future UA systems.

2. Methodology

2.1. State of the art – nutrient recovery and reuse in urban environments

This work comprises the literature analysis of research on the nutrient recovery of urban waste streams for the nutrient supply for UA. For this purpose, an initial literature search was performed via the search platform Scopus with the keywords "nutrient recovery" AND "Urban agriculture" OR "Urban waste".

This initial search resulted in 26 scientific articles (Table 1) that could be further sorted into two main categories established depending on the waste type or source. These categories were defined as "food-bio-organic waste" and "wastewater". For the literature review, only nutrients that can be sourced in urban areas have been identified, disregarding potential organic material from the agricultural industry or other imported material. Notably, only wastes have been regarded, while products specifically elaborated for fertilization were disregarded.

From these categories, a second search was elaborated with the key words "nutrient recovery" AND "Urban" AND "organic waste" AND NOT "wastewater" (10) "nutrient recovery" AND "food waste" AND NOT "wastewater" (56), "nutrient recovery" AND "bio waste" AND NOT "wastewater" (8) and "nutrient recovery" AND "urban" AND "wastewater" (81). All literature before 2017 was excluded to avoid outdated waste treatment methodologies.

Applying these criteria, the articles obtained for the search of nutrient recovery based on food-bio-organic waste yielded a total of 24 results, while 58 results were retrieved for wastewater sourced fertilizers. These articles were then classified into categories corresponding to the recovery technology, as shown in Table 2 for food-bio-organic waste

Table 1
Waste type, treatment and target nutrient identified with primary search on urban waste-derived nutrient recovery.

	Reference	Waste type/origin	Recovery technology	Target nutrients	Application
1	(Weidner and Yang, 2020)	Organic waste	Composting/insect rearing/anaerobic digestion	NPK	Soil-based agriculture/ hydroponic/aquaponic
2	(Shrestha et al., 2020)	Organic waste	Composting	NP	Soil-based agriculture
3	(de Kraker et al., 2019)	Kitchen waste/garden residue/ Urine	Anaerobic digestion/vermicomposting/ composting/struvite precipitation	NP	Urban agriculture/municipal green/peri-urban agriculture
4	(Kjerstadius et al., 2017)	Centralized and source-separated food waste and wastewater	Struvite precipitation/anaerobic digestion/ biological nitrogen removal/sludge composting	NP	Agriculture
5	(Factura et al., 2010)	Bio-waste and excreta	Terra Preta	NP	Urban agriculture
6	(Suthar and Gairola, 2014)	Garden waste	Vermicompost	NPCa	Soil-based agriculture
7	(Rufí-Salís et al., 2020c)	Nutrient leachate	Leachate recirculation, chemical precipitation, and membrane filtration	P	Hydroponic
8	(Pimentel-Rodrigues and Siva-Afonso, 2019)	Source-separated urine	Urine storage	P	Green roofs
9	(Ali et al., 2023)	Organic waste	Composting	N	
10	(Akoto-Danso et al., 2019)	Domestic wastewater		P	Soil-based agriculture
11	(Podder et al., 2020)	Landfill leachate	Anaerobic digestion/struvite precipitation	NP	
12	(You et al., 2019)	Urban wastewater	Sorption (synthetic Zeolites Ze-CA)	NP	Food production
13	(Schröder et al., 2021)	Organic waste	Compost/vermicompost	NP	Soil-based agriculture/urban agriculture
14	(Guaya et al., 2018)	Urban wastewater	Sorption (synthetic Zeolites)	NPK	Food production
15	(Magwaza et al., 2020)	Domestic wastewater		NP	Hydroponic
16	(Rufí-Salís et al., 2020a)	Urban wastewater	Struvite	NP	Urban agriculture
17	(Calabria et al., 2019)	Domestic wastewater	Sorption (natural Zeolite)	N	Hydroponic
18	(D'ostuni et al., 2023)	Domestic wastewater	Moving bed biofilm reactor (MBBR)	NPK	Hydroponic/building-integrated agriculture
19	(Van Ginkel et al., 2017)	Domestic wastewater	Struvite	NP	Hydroponic/aquaponic
20	(Zhao, 2014)	Domestic wastewater and organic waste	Anaerobic digestion		Urban agriculture
21	(John et al., 2023)	Urban wastewater	Freeze concentration technology	N	Agriculture
22	(Weidner et al., 2020)	Urban wastewater	Membrane bioreactor	NPK	Hydroponic/aquaponic
23	(Tesfamariam et al., 2022)	Wastewater and organic waste	Thermal hydrolysis/anaerobic digestion	NPK	Soil-based agriculture/urban agriculture
24	(Brown et al., 2023)	Wastewater and organic waste	Compost/anaerobic digestion	NP	Urban agriculture

Table 2
Waste type and treatment identified with a secondary search on food-bio-organic waste-derived nutrient recovery (red) and wastewater-derived nutrient recovery (blue).

	Waste type	Treatment	Reference
Food-bio-organic waste	Agro-food waste/ kitchen waste/ Textile sludge/ bio-waste	Anaerobic digestion	(Oarga-Mulec et al., 2019), (Davidsson et al., 2017), (Pleissner et al., 2017), (Ren et al., 2020), (Kumar et al., 2020), (Gienau et al., 2018a,b), (Möller et al., 2018), (Wang and Lee, 2021), (Ravindran et al., 2021), (Reilly et al., 2021), (Weidner and Yang, 2020), (Tian et al., 2019), (Battista et al., 2020), (Rojo et al., 2021), (Mikula et al., 2023), (Kumar et al., 2023), (Feiz et al., 2022), (Bruno et al., 2022), (Liu et al., 2022), (Bareha et al., 2022), (Guruchandran et al., 2022), (Campos et al., 2019), (Toop et al., 2017), (Chojnacka et al., 2022), (Kacprzak et al., 2022), (Eraky et al., 2022)
	Food waste	Thermal treatments: Thermal hydrolysis (TH), Hydrothermal Carbonization (HTC), Hydrothermal liquefaction (HTL)	(Idowu et al., 2017), (Rao et al., 2018), (Gollakota and Savage, 2018), (Wang and Lee, 2021), (Sarrion et al., 2021), (Sudibyo et al., 2022) (Palansooriya et al., 2023), (Sarrion et al., 2023b), (Wu et al., 2023), (Sarrion et al., 2023a), (Chojnacka et al., 2022) (Kacprzak et al., 2022)
	Food waste/ Green Waste/ Sewage sludge/ Urban organic waste	Composting/ Co-composting/ Compost tea	(Mortula et al., 2020), (Awasthi et al., 2020), (Möller et al., 2018), (Ravindran et al., 2021), (Schröder et al., 2021), (Milinković et al., 2019), (Weidner and Yang, 2020), (Shrestha et al., 2020), (Dsouza et al., 2021) (Manga et al., 2022) (Ali et al., 2023), (Fang et al., 2023) (Chojnacka et al., 2022), (Kacprzak et al., 2022), (Rana et al., 2022)
	Onion waste/ Organic waste	Vermicomposting	(Pellejero et al., 2020), (Ravindran et al., 2021), (Schröder et al., 2021), (Milinković et al., 2019) (Biruntha et al., 2020), (Wongkiew et al., 2023)
	Organic waste	BSLF/ Insect rearing	(Magee et al., 2021), (Weidner and Yang, 2020) (Mok et al., 2020)
Wastewater	Treated wastewater (primary & secondary effluent)	Ion Exchange/ Adsorption	(Guaya et al., 2018), (You et al., 2019), (Calabria et al., 2019), (Sheikh et al., 2023), (Sánchez and Martins, 2021), (Reig et al., 2021), (Gowd et al., 2022)
	Treated wastewater (primary effluent)	Hydroponic agriculture	(Magwaza et al., 2020), (Clyde-Smith and Campos, 2023)
	Source-separated urine	Green roofs	(Pimentel-Rodrigues and Siva-Afonso, 2019), (D'ostuni et al., 2023)
	Untreated wastewater	Soil-based agriculture	(Akoto-Danso et al., 2019)
	Filtered untreated wastewater, Urine, Treated wastewater (primary effluent), Treated water (secondary effluent)	Photobioreactor /Photo-fermentation	(Dalvi et al., 2021), (Chatterjee et al., 2019), (Karbakhshravari et al., 2020), (González et al., 2020), (Escudero et al., 2020), (Sánchez-Zurano et al., 2021), (Robles et al., 2020), (González-Camejo et al., 2019), (Yulistyorini, 2017),

(continued on next page)

Table 2 (*continued*)

Waste type	Treatment	Reference
		(Khandan et al., 2020), (Guilayn et al., 2020), (de Morais et al., 2022), (Amaya-Santos et al., 2022), (Panda et al., 2021), (Moges et al., 2020), (Gowd et al., 2022), (Zhu et al., 2022)
Treated wastewater (primary effluent), Source-separated urine, Anaerobic digester centrate	Struvite precipitation	(Karbakhshravari et al., 2020), (Kjerstadius et al., 2017), (Ruff-Salís et al., 2020a), (Rodrigues et al., 2019), (Lorick et al., 2020), (Macura et al., 2019), (Sánchez and Martins, 2021) (Mayor et al., 2023), (Sánchez, 2020), (Podder et al., 2020), (Gowd et al., 2022), (Besson et al., 2021)
Treated water (secondary effluent)	AnMBR	(Jiménez-Benítez et al., 2020a), (Jiménez-Benítez et al., 2020b), (González-Camejo et al., 2019), (Mullen et al., 2019), (Zhou et al., 2022)
Wastewater Treated water (secondary effluent)	Membrane filtration: Reverse Osmosis, Membrane Contractors, Membrane Bioreactor	(Vecino et al., 2019), (Volpin et al., 2019), (Sheikh et al., 2023), (Reig et al., 2021), (Samarina and Takaluoma, 2020), (Deemter et al., 2022), (Yao et al., 2017)
Wastewater sludge	Thermal treatment: Pyrolysis, Eutectic freeze crystallization (EFC)	(Jellali et al., 2021), (Tomasi Morgano et al., 2018), (Guilayn et al., 2020), (John et al., 2023),
Wastewater sludge, Source-separated urine	Anaerobic digester	(Kjerstadius et al., 2017), (Srivastava et al., 2020), (de Kraker et al., 2019), (Firmansyah et al., 2021), (Guilayn et al., 2020), (Lehtoranta et al., 2022), (Brown et al., 2023)
Wastewater sludge,	Composting	(Oarga-Mulec et al., 2019), (Firmansyah et al., 2021), (Chen et al., 2022)
Treated wastewater (primary effluent), Source-separated urine	Stripping	(Bolzonella et al., 2018), (Lorick et al., 2020), (Macura et al., 2019), (Guilayn et al., 2020)
Wastewater Treated water (secondary effluent)	Microbial electrochemical technologies (METs)	(Kashima, 2020), (Matar et al., 2022), (Besson et al., 2021)

and wastewater.

Additional searches on Scopus, Web of Science and Google Scholar were conducted for each nutrient recovery process and waste treatment to encourage a better description and insight.

With the initial literature search, two main nutrient sources for urban nutrient recovery were identified: wastewater-based residues as well as food wastes and other organic and bio wastes produced in urban ecosystems that mainly originated in households, food processing and catering industries or green areas such as gardens and parks (Möller et al., 2018). All recovery technologies identified in Tables 1 and 2 are depicted in Fig. 1 for a better understanding of the waste flows and possible combinations of methodologies within each waste type as well as different combinations of waste types.

3. Results

In this section the two main sources for nutrient recovery identified in literature will be described (Sections 3.1 and 3.2), followed by the recovery technologies identified (Section 3.3).

3.1. Nutrient recovery from food and biowaste

City landscapes and households produce a large amount of biomass throughout the year, up to 400 to 800 g daily per person (Ahmed et al., 2019), which could be increased with the integration of agriculture within urban areas (Dsouza et al., 2021; Manríquez-Altamirano et al., 2020). Biomass and food waste are often underused sources of nutrients, although there is significant potential for the recovery of non-renewable and energy-intensive nutrients such as P and N (Idowu et al., 2017; Zabaleta and Rodic, 2015). The waste generated in the household is of great importance, but the overproduction and further disposal of food surplus is also part of the problem. Previous work has identified that up to 40 % of produced food is wasted, and savings from up to 25 % of mined P can be achieved by the reduction of these food wastes (Drangert et al., 2018). The landfilling and poor management of food and biowaste can lead to large amounts of anaerobic decomposition that causes greenhouse gas emissions, which needs to be avoided and processed in a controlled way (Dsouza et al., 2021). Increasing efforts to reduce the landfilling of organic waste advocate for effective source separation and further treatment of municipal waste to recover and recycle nutrients from the organic fraction (Davidsson et al., 2017). This appears to be highly relevant, considering that in 2018, only 30 % of the European organic fraction is currently source-separated and further recycled (Möller et al., 2018), while the remainder is landfilled or incinerated (Sun et al., 2018).

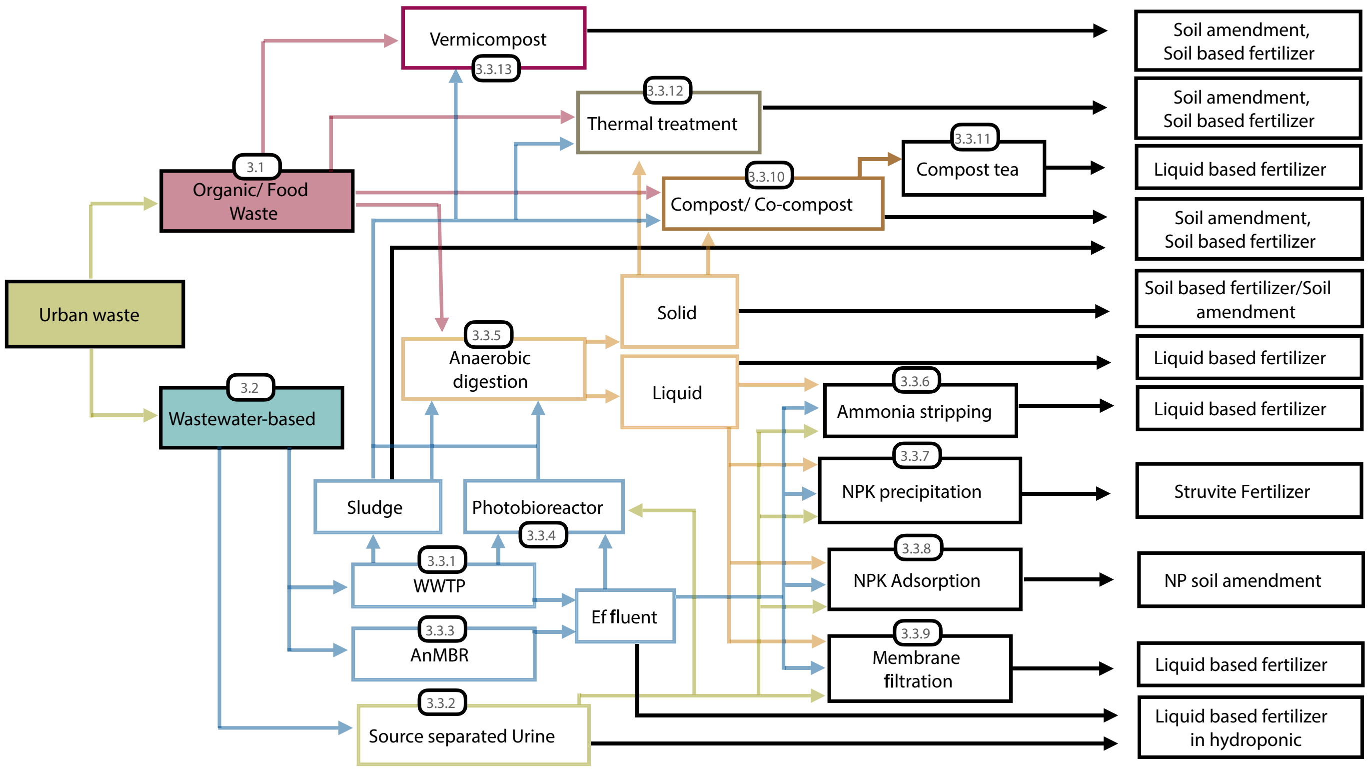


Fig. 1. Main flows of nutrients from urban waste to agricultural fertilizer identified from the studies compiled in Tables 1 and 2.

3.2. Nutrient recovery from wastewater

The perception of WWTPs has been shifting and evolving in recent years, transitioning from waste carrying and removal technologies to resource recovery plants for nutrients and for energy (You et al., 2019). Currently, wastewater and processed sewage sludge and effluent are the main P carriers that are further recycled in agriculture in some European countries, although environmental regulations often do not enable the direct use in conventional agriculture and therefore are often just being incinerated (Möller et al., 2018). Better management of wastewater streams is endorsed in European regulations, pursuing greater circularity in urban areas with larger water and nutrient recovery as well as the reduction of energy consumption and GHG emissions (Marinelli et al., 2021; You et al., 2019). The proximity of WWTPs to urban areas gives them a great advantage to supply nutrients for UA, avoiding great transportation or potential storage problems. The potential of this circularity has been discussed in previous studies: in a neighbourhood in Munich, Germany, to reach savings of 26 % of freshwater resources while promoting UA to produce 66 % and 246 % of fruit and vegetable demand, respectively (Marinelli et al., 2021).

3.3. Nutrient recovery technologies

The following Sub-sections 3.3.1 to 3.3.13 detail the technologies and facilities identified in the literature (Tables 1 & 2).

3.3.1. Nutrient recovery technologies in WWTP

The outline of a WWTP can incorporate certain steps depending on the purpose of the treated water, but wastewater is commonly pretreated to remove any potential solid waste and oils. Then, the wastewater is sent to a primary clarifier to settle the primary sludge and obtain a clearer wastewater that can then undergo primary treatment with activated sludge. While simpler layouts may involve the addition of a secondary clarifier after the primary treatment, other WWTPs incorporate a secondary treatment, which is usually the nitrification and denitrification process (Ostace et al., 2013; Vilanova et al., 2017). Further tertiary processes can be added for further nutrient removal, in which chemical removal or ultrafiltration processes are usually installed (You et al., 2019).

The sludge from the first clarifier, second clarifier, and nutrient removal processes is treated in parallel by mixing, dewatering, anaerobic digestion, or composting systems (Vilanova et al., 2017).

WWTPs have great potential to house nutrient recovery technologies. These installations are the main collectors of domestic wastewater in urban areas and have the sole purpose of bringing the water composition below thresholds established by environmental regulations. The reduction of nutrients in the water is a requisite that can usually be implemented via biological or chemical treatments (Guisasola et al., 2019; Vilanova et al., 2017). Biological treatments have been considered more reliable and effective methods for achieving a substantial reduction in nutrients. However, as thresholds for nutrient emissions into the environment are reduced, chemical treatments are gaining more popularity in water treatment processes (Hospido et al., 2004).

Chemical treatment mainly consists of the reduction of soluble nutrients (mainly P) to particulate nutrients by their binding and precipitation, which can be achieved with the addition of metal salts such as Ca in the form of lime, Fe and Al. Once precipitated, nutrients can be further disposed in the sedimented sludge, or repurposed as fertilizers like the case of struvite. The main constraint for chemical treatments is the amount of chemicals needed to effectively remove the soluble P, which must be in a 1:1 relation to the present P. This method involves a great investment not only for the metal salts themselves but also for the required storage infrastructure (Crini and Lichtfouse, 2019; Foley et al., 2010). On the other hand, the excessive application of these chemicals can lead to unwanted chemical reactions. For the chemical removal of N, several techniques are available, including air stripping, ion exchange or membrane filtration. To increase the sustainability of the process, biological treatment has recently assumed the form of a required step for initial nutrient removal. This step consists of the removal of N and P with microbial activated sludge (Vilanova et al., 2017). The removal of N is based on the decomposition of incoming organic N into ammonia (NH_4^+) through aerobic and heterotrophic bacteria, which can undergo further steps for successful ammonia removal. Most commonly, the produced ammonia can be released in an aerobic environment for a nitrification process generating nitrate (NO_3^-) with the help of autotrophic bacteria. A denitrification step can be added to generate N in the form of gas (N_2) that can be exhausted into the environment (Hou et al., 2021). This step is performed under anaerobic conditions by heterotrophic bacteria that require the addition of organic matter as a carbon source (Vilanova et al., 2017).

The biological removal of P, on the other hand, is based on the addition of phosphate-accumulating organisms (PAOs), which also require anaerobic and aerobic stages. These bacteria can release P in the form of phosphate (PO_4) under anaerobic conditions, while P can be captured in aerobic environments (Close et al., 2021; Guisasola et al., 2019; Hou et al., 2021; Poh et al., 2021; Vilanova et al., 2017). This process can be combined with the activated sludge and nitrification and denitrification stages for organic N removal (Hou et al., 2021; Sarvajith and Nancharaiah, 2022), obtaining biomass that can be further disposed with the settled sludge or further rereleased into an aqueous phase to be chemically precipitated (Anders et al., 2021).

Many of the technologies described in the upcoming sections can be added to the WWTP outline for further nutrient removal, and most importantly, recovery.

3.3.2. Source-separated urine

Human urine has been generally considered highly suitable for fertilizer production due to its high N and P contents, being the greatest nutrient contributor in wastewater (80 %, 50 % and 55 % for N, P and K, respectively) while being only 1 % of the fraction detected in the total wastewater (Chatterjee et al., 2019; Federico Volpin et al., 2018).

To increase waste treatment efficiency, the separation of waste streams at the domestic level has been suggested to reduce separation and nutrient recovery processes. This idea for source separation of household waste streams has already been regarded as an upcoming reality in some countries, such as China and Sweden, enabling new nutrient recycling regulations (Kjerstadius et al., 2017; Pimentel-Rodrigues and Siva-Afonso, 2019). These countries have already established separation technologies with urine diverting toilets for better management and fertilizer production (Pimentel-Rodrigues and Siva-Afonso, 2019). Other ways to increase circularity would not only be separation in households but also the direct application of urine in building green roofs, rooftop agriculture or green facades, directly avoiding nutrient loss due to storage and transport (D'ostuni et al., 2023; Pimentel-Rodrigues and Siva-Afonso, 2019). The direct application of urine in agricultural production has shown promising results compared to wastewater treatment with increased N and P recovery rates as well as a reduced impact on surrounding environments due to eutrophication (Malila et al., 2019).

However, while the application of human urine in agriculture has been performed it has also shown several constraints in present times, containing greater N concentration compared to other nutrients, the potential content of chemicals and pharmaceuticals but also the general negative perception of human urine application by producers and consumers (Ikeda and Tan, 1998; Simha et al., 2017; Simha and Ganesapillai, 2017; Federico Volpin et al., 2018). To avoid these constraints, several technologies have been applied to enhance nutrient recovery while reducing the content of potential impurities in urine (Calabria et al., 2019; Guaya et al., 2018; Rodrigues et al., 2019; Rufí-Salís et al., 2020a). These same technologies can also be applied to WWTP and anaerobic digestion effluents to further remove nutrient content for better water recovery or disposal into water bodies, as further explained in the upcoming sections (Sections 3.3.3 to 3.3.9). The processes that have been mostly explored consist of nutrient precipitation, obtaining mineral fertilizer such as struvite or urine concentration to enhance nutrient removal and serving as liquid fertilizer (Yang et al., 2015; Yao et al., 2017). Other technologies can entail membrane filtration or reverse osmosis, ammonia stripping and ion exchange through resins or sorbents (Federico Volpin et al., 2018). Source-separated urine can also serve as feedstock for other recovery technologies, such as photobioreactors, being used as a nutrient source for algal growth to ensure nutrient recovery (Tuantet et al., 2019; Zhang et al., 2014). Overall, the use of source separated urine as feedstock for other recovery processes has shown a reduction in GHG emissions, especially due to a reduced emission of N_2O from nitrifying processes and from and avoided N fertilizer production (Marinelli et al., 2021).

3.3.3. Anaerobic membrane bioreactor - AnMBR

In recent years, waste treatment has undergone several innovations to reduce the impact of management while obtaining cleaner water and nutrients. One promising solution for wastewater management is the combination of anaerobic digestion with a membrane bioreactor system. This combination is referred to as an anaerobic membrane bioreactor (AnMBR) and has shown promising results for the generation of high-quality effluent and greater energy recovery than anaerobic treatments in WWTPs (Jiménez-Benítez et al., 2020a). AnMBR has only been applied at pilot scales, but previous work on this innovative treatment has shown the potential to combine wastewater with food and organic waste, obtaining lower environmental impacts than other anaerobic-based treatments (Jiménez-Benítez et al., 2020a). The effluent quality has been reported to be high for application in agriculture since its N and P contents are elevated. While this effect has been considered a constraint in AnMBR technology, it can be considered an opportunity for fertigation purposes reducing N and P mineral fertilizer application by 71 % and 39 %, respectively (Jiménez-Benítez et al., 2020b). The obtained effluent from AnMBR can be further processed for nutrient recovery and concentration through ion exchange technology (Mullen et al., 2019) and microalgal biomass production (González-Camejo et al., 2019).

3.3.4. Biological treatment (photobioreactor)

An alternative biological treatment to that commonly observed in WWTPs is nutrient capture via algal biomass growth (Guilayn et al., 2020; Tuantet et al., 2019). This process is regarded as a better solution for wastewater treatment than secondary activated sludge or even secondary nutrient removal technologies (Mennaa et al., 2019; Munasinghe-Arachchige et al., 2020). Nutrient capture is a suitable treatment due to its cost-efficiency for pathogen, BOD, N and P removal and the generation of oxygen for organic N breakdown via photosynthesis, therefore avoiding costly aeration mechanisms (Mennaa et al., 2019). The resulting biomass can then be utilized in several ways, from biomass for biofuel production, for fertilization purposes and even as animal feedstock, being a more circular approach to N dissipating technologies employed in nitrifying and denitrifying processes (Nagarajan et al., 2020). Algal production using wastewater as a nutrient source can be performed in open air ponds with natural light conditions or in photobioreactors, with or without continuous illumination, to further enhance algal growth. Open air production can be less costly but greatly subjected to natural temperature and light conditions, while exterior or indoor photobioreactors can provide more stable environments throughout the year (Nagarajan et al., 2020). Work has been performed on the combination of WWTP processes with algal production, using different wastewater stages as potential feedstock for biomass growth, from untreated wastewater, primary clarified, anaerobically digested, tertiary treated wastewater to even source-separated urine (Amaya-Santos et al., 2022; Samorì et al., 2013; Tuantet et al., 2019; Zhang et al., 2014). The removal rate for N and P varies depending on the algal species and growth conditions (Panda et al., 2021). Previous work has reported recovery rates up to 52 % and 38 % in N and P, respectively, via microalgal growth (Chatterjee et al., 2019), but higher rates can be reached in photobioreactors with up to 80 % nitrogen and total P removal in urine (Tuantet et al., 2019; Zhang et al., 2014) or even 99 % N and P removal levels in the case of *Chlorella sorokiniana* in source-separated blackwater (Moges et al., 2020). Even with these extensive positive traits of algal production for nutrient removal, the application of this technology as a sole large-scale wastewater treatment is still not effective and can be highly energy consuming (Gowd et al., 2022). Potential unsuccessful removal of toxins as well as bacterial contamination make a previous wastewater sterilization necessary, which primarily increases the treatment costs, placing photobioreactors as tertiary treatment stages. On the other hand, bacterial contamination can be avoided with the use of extremophile microalgal species or with the co-culture of beneficial or symbiotic bacterial strains (Rashid et al., 2019), making organic matter removal possible (Robles et al., 2020). Algal biomass sampling can also be crucial, adding an additional step and cost to nutrient removal. This sampling can be performed by centrifugation, filtration or chemical precipitation, which can be expensive and energy-consuming (Mennaa et al., 2019; Robles et al., 2020). The produced algal biomass can further contribute to nutrient recovery processes with the coupling of other technologies like anaerobic digestion for additional biogas production (Panda et al., 2021) or its application as slow release fertilizer in the form of green biomass (Moges et al., 2020).

3.3.5. *Nutrient recovery from anaerobic digestion*

The recycling of nutrients from biowaste via anaerobic digestion has largely been analysed and considered to have great potential in urban waste management it's a mature technology used globally which is capable of generating economic surpluses (Rojo et al., 2021) due to the generation of methane to meet local energy requirements, as well as high nutrient recovery in the digestate with a small fraction loss of phosphorous and nitrogen (approximately 0 to 10 %) (Oarga-Mulec et al., 2019). Apart from macronutrients contained in the digestate, other compounds such as micronutrients, hormones and other organic elements can have a positive effect on plant and soil microorganisms (Ren et al., 2020).

The process of anaerobic digestion has four stages that involve different key microbial communities in an oxygen-deprived environment. During the first stage, hydrolysis and breakdown of the feed component polymers occurs, being reduced to monomers by microbial secreted hydrolases (Rojo et al., 2021; Sikora et al., 2017). Acidogenesis is the second phase of anaerobic digestion, where the hydrolysed compounds undergo acidic fermentation, followed by acetogenesis with the formation of acetate, hydrogen and carbon dioxide, which are further transformed into methane in methanogenesis during the last step (Guruchandran et al., 2022).

The resulting digestate is mainly applied in agricultural fields due to its high nutrient content, especially for nitrogen, phosphorous and potassium (Gienau et al., 2018b). The liquid and solid fraction of the remaining digestate contain mineral and organic forms of nitrogen greatly available for plants, while P is mainly recovered in the form of phosphate (Vögeli et al., 2014; Wang and Lee, 2021; Zabaleta and Rodic, 2015). While the liquid fraction has a greater nitrogen (in the form of dissolved ammonia) and potassium content, the solid fraction has greater amounts of total nitrogen and phosphorous. Compared to sludge or dairy, food waste anaerobic digestate contains greater amounts of nitrogen in the form of ammonia and a greater N:P ratio (Dutta et al., 2021).

Additional treatments can be added to the solid and liquid fraction to further recover P and N, such as thermal treatment of the digestate into biochar as well as fertilizer production through precipitation processes, ion exchange/adsorption processes, ammonia stripping and membrane filtration, (Campos et al., 2019; Eraky et al., 2022; Gienau et al., 2018b; Guilayn et al., 2020; Vaneeckhaute et al., 2017).

While the generated digestate proves to be a good source of nutrient recovery, the generation of volatile nitrogen compounds such as NH_3 can increase the impact of this waste. The composition of the digestate, ammonia release and potential methanogenesis greatly depend on the incoming feed, with suggested C/N ratios of 15–30 (Guruchandran et al., 2022). Previous work on anaerobic digestion treatments of food waste showed C/N ratios of 49, although the content of oils and spices in food wastes can be detrimental to methanogenic activity, slowing its progress (Kumar et al., 2020), co-digestion processes with textile waste has shown C/N of 12.2 (Kumar et al., 2020), while Deinking sludge co-digestion showed C/N over 100 (Bareha et al., 2022).

To obtain a digestate of the required quality for UA, the organic biomass must be collected free of impurities, which is difficult to achieve even through selective organic waste collection (Naroznova et al., 2016). Additional pre-treatment options can be evaluated to increase the digest value and specially to avoid heavy metal contamination for its use as a nutrient source without risk. Previous work on biowaste pre-treatment options detailed three technologies, namely, "biopulp", "screw press" and "disk screen", to reduce impurities in the digestate. The most environmentally favourable technology seems to be the "biopulp" technology, which allows for increasing the digestion value with greater biogas production as well as nutrient recovery (Khoshnevisan et al., 2018). Although biogas digestate is a very attractive option to provide readily available and consistent recovered nutrients for UA (Chojnacka et al., 2022), the need to enforce pre-treatment technologies is necessary to avoid potential contamination into the food production system (Davidsson et al., 2017; Kjerstadius et al., 2017). In the case of wastewater streams source separation of both blackwater and urine can increase nutrient recovery through anaerobic digestion processes (Lehtoranta et al., 2022).

While solid biogas digestate can be applied without processing as well as in a composted or co-composted form as soil amendment and growing bed substrate (Brown et al., 2023; Czekała et al., 2022; Dimambro et al., 2015), work has also been conducted in the application of liquid digestate in hydroponics using the nutrient film technique (Martin et al., 2019; Bergstrand et al., 2020; Weidner and Yang, 2020).

These studies have shown the potential of organic fertilization based on biogas digestate but urge for a better control of nutrients for balanced fertilization as well as heavy metal content (Bergstrand et al., 2020; Ezziddine et al., 2021; Wang and Lee, 2021).

Through life cycle analysis, the environmental performance of anaerobic digestion of food waste has been analysed, showing favourable results in the case of nutrient recovery (Bruno et al., 2022) and overall better performance than incineration and composting processes (Feiz et al., 2022) as well as economic profitability (Liu et al., 2022).

3.3.6. *Ammonia stripping*

Ammonia stripping is an easy process that has been incorporated in wastewater treatment plants for ammonia remediation, favouring the formation of gaseous ammonia (NH_3) through an increase in pH, which is usually made with the addition of lime (Kinidi et al., 2018). While high concentrations of ammonia can be toxic for bacteria and therefore not recommended for biological treatment, ammonia stripping has a high removal success of up to 90 %, being tested already in municipal waste, landfill leachate and wastewater effluent (Kinidi et al., 2018; Zangeneh et al., 2021). The recovery and further use of the stripped ammonia gas can be achieved by adsorption to acid, obtaining ammonium sulphate fertilizer (Lorick et al., 2020). Ammonia stripping can present difficulties with the presence of organic and inorganic fine particles, generating problems of fouling and clogging of the stripping column (Bolzonella et al., 2018). Other methods of ammonia stripping have been developed over the years to avoid high energy and chemical use for this process, combining ammonia stripping processes with electrodialysis and membrane stripping (Federico Volpin et al., 2018).

Based on its comparatively steady performance and capability, ammonium stripping is currently the industry standard technology for highly effective ammonium removal in China. However, due to the high expense of employing caustic soda, hot steam, and sulfuric acid, as well as the creation of the secondary pollutant, the application is restricted economically (Zhu et al., 2022).

3.3.7. Struvite precipitation

The process of P precipitation has been largely analysed and considered a valuable approach to recover P, N and Mg from wastewater and human urine. Precipitation occurs when the struvite components Mg^{2+}:NH_4^+:PO_4^{3-} are present with a molar ratio of 1:1:1 and a pH of approximately 8.5–9.5 (de Kraker et al., 2019; Uysal et al., 2014). The amount of Mg in wastewater and urine is usually insufficient to ensure total precipitation and therefore is usually added, although other precipitation techniques have been developed with the addition of sea water (Shaddel et al., 2020). The precipitated crystalline mineral with the composition $MgNH_4PO_4{\cdot}6H_2O$ is referred to as struvite or magnesium ammonia phosphate (MAP) (Simha and Ganesapillai, 2017) and has been considered a valuable slow release fertilizer due to its low solubility, being also generally regarded as a pollutant and heavy metal-free crystal (de Kraker et al., 2019). Although it has been mostly applied on laboratory scale (Lorick et al., 2020), the precipitation of P has been endorsed in WWTP installations due to the great recovery capacity of P reaching up to 90 % or even a complete recovery under the right conditions (Simha and Ganesapillai, 2017; Federico Volpin et al., 2018) while also enabling for recovering N, although in smaller proportions (Podder et al., 2020). This process can be added as a treatment for primary or secondary effluent as well as source-separated urine and sludge effluent (Sánchez, 2020). The addition of struvite precipitation in WWTPs with enhanced biological phosphorous removal treatment in place does not entail great modifications (Rufí-Salís et al., 2020a), being a further source of nutrient recovery and direct application in soil and hydroponic agriculture (Arcas-Pilz et al., 2021; Carreras-sempere et al., 2021; Liu et al., 2011).

3.3.8. Ion exchange/adsorption

The processes of NH_4-N and PO_4-P capture via ion exchange or adsorption have been investigated and used for nutrient removal. The principle of this removal technology is the fixation of the pollutant into a solid surface through a physicochemical interaction. The ion exchange process involves bringing the solution in contact with the resin, and as ions in the solution come into contact with the resin, they bind to the resin while releasing other ions into the solution. Adsorption is a process in which molecules or ions from a liquid or gas adhere to the surface of a solid material, known as an adsorbent. The interaction is typically due to weak van der Waals forces or other intermolecular forces (Mehta et al., 2015; Zhou et al., 2023). Previous studies focused on nutrient removal through ion exchange technology from wastewater mainly used natural or synthetic zeolites for N and metal-loaded chelating resins, iron-based hydroxide compounds and hydrotalcites for P as the captor or exchange surface (Kuntke et al., 2016; Williams, 2013). The ion exchange process is a simple exchange between the wastewater flow and an exchange material-containing column, and while NH_4-N or PO_4-P is attached to the media column, other cations are released into the wastewater (Williams, 2013). Loaded zeolites can be then regenerated with a NaOH solution for a consequent use (Reig et al., 2021; Sheikh et al., 2023). This method shows a great recovery (between 43 and 80 % and 90–97 %) for both N and P and can then be reversed with salt water, which can then be precipitated as struvite to be used as fertilizer (Gowd et al., 2022; Lohman et al., 2020; Mullen et al., 2019; Federico Volpin et al., 2018; Zhou et al., 2023). Work has also been done on the recovery of P from anaerobic digest from municipal solid waste, sewage sludge and sewage sludge ash through electrodialysis showing promising results (up to 81 % P recovery in municipal solid waste). This process consists of the selective separation of P anions through an ion exchange membrane charged with an electric field (Oliveira et al., 2021). Although ion exchange technologies are promising, their implementation has only been in small scale and laboratory settings (Sánchez and Martins, 2021).

Previous studies using natural zeolites for the recovery of both ammonium and phosphate, applied the charged sorbent as a slow releasing soil amendment, showing promising results for its application in agriculture (Guaya et al., 2018; You et al., 2019).

Because the nutrient-rich sorbent/exchange media may be used directly as a nutrient product in agriculture, adsorption/ion-exchange technologies can be thought of as hybrid nutrient accumulation-nutrient recovery technologies (Mehta et al., 2015).

Adsorption has shown to be adequate for the recovery of nutrients in low concentrations as well as a great design flexibility, although some disadvantages have been reported, especially on the complexity of adsorbent regeneration and the additional waste generation due to adsorbent replacement, generating extra costs (Zhou et al., 2023).

3.3.9. Membrane filtration

Membrane-based separation methods for nutrient recovery entail a great variety of filtration methods, such as nanomicro- and ultrafiltration (NF, MF, and UF), which are usually followed by reverse osmosis (RO) in treated wastewater and in anaerobic digestion liquid fractions (Gienau et al., 2018a). The use of MF and UF is common in WWTP, and further application of NF and RO can be made to obtain non-drinkable clean water (Hube et al., 2020). NF and RO are pressure-driven filtration processes that can incur high overall costs as well as more operational complications compared to other previously explained techniques, such as ammonia stripping (Gienau et al., 2018a; Volpin et al., 2018). Membrane filtration-based processes such as NF and RO can also cause urea and ammonia losses that can lead to poor N recovery (Simha and Ganesapillai, 2017; Federico Volpin et al., 2018).

More recently, membrane technology has been further developed with the application of membrane contractors (MC) which differ from the previously explained pressure-based membrane filtration. This technology enables the transfer of a substance from one phase to the other through a non-selective membrane without the dispersion of the phases. The diffusion occurs through a defined gradient or pressure on the receiving phase which also defines de separation selectivity (Rongwong and Goh, 2020). MC can be used for the diffusion in different phases, liquid-gas, gas-liquid, liquid-liquid, gas-gas. In the case of wastewater recovery of nitrogen in the form of ammonia, liquid-liquid MC has been used. This process requires the increase of the liquid phase pH to obtain ammonia gas to then diffuse through the membrane pores into the gaseous phase (Licon Bernal et al., 2016; Noriega-Hevia et al., 2020; Rongwong and Goh, 2020). On the other side of the membrane sorbents, usually strong acids like sulfuric and nitric acids, can be the used to dissolve the ammonia gas from the membrane into the liquid phase (gas-liquid). Stable liquid fertilizers can be obtained through this process like ammonium sulphate which can be applied in fertigation (Gonzalez-Salgado et al., 2022; Licon Bernal et al., 2016; Noriega-Hevia et al., 2020; Vecino et al., 2020). This technique has been seen to have great recovery potential as well as a better environmental performance with the reduced energy needs compared to air stripping and pressure-bound membrane separation. On the other hand the high content of suspended solids in the liquid can cause clogging and membrane fouling (Licon Bernal et al., 2016; Rongwong and Goh, 2020) therefore it's combination with pre-treatments is important to ensure longer membrane lifetime and operation (Deemter et al., 2022). Previous studies have applied this technology for the production of liquid fertilizers from the aqueous streams obtained from the regeneration step of zeolites used in WWTP tertiary treatment (Mayor et al., 2023; Sancho et al., 2017; Sheikh et al., 2023) as well as adsorption processes (Samarina and Takaluoma, 2020). This combination of emerging technologies has proven to be promising, ensuring a greater ammonia concentration through the loaded zeolites and using NaOH for zeolite regeneration creating a closed loop process, able to recover up to 92 % of N in the form of ammonia (Reig et al., 2021; Sheikh et al., 2023).

3.3.10. Composting/co-composting

A classic and commonly practised nutrient recovery from household biomass is the process of composting, which has been widely employed not only on a larger scale with municipal green and organic waste but also on smaller scales in neighbourhoods and even private gardens

(Dsouza et al., 2021; Shrestha et al., 2020; Ulm et al., 2019). Not only can urban green and food waste be destined for composting sites, but anaerobic digestion as well as sewage sludge can also be composted, as commonly happens with 60–90 % of sewage sludge produced in the UK, Ireland, Spain, France, or Luxemburg (Bastida et al., 2019).

Finland, Denmark, Ireland, The Netherlands, Sweden, and Switzerland prohibit the usage of municipal solid waste-derived compost. While in Austria, UK, Hungary and Italy the municipal solid waste compost is landfilled; France, Spain, and Portugal also use this compost for agriculture. (Kacprzak et al., 2022).

The composting process is divided into three stages: an initial mesophilic stage, a thermophilic phase, and a maturation stage. The names of the stages correspond to the temperatures reached in the compost pile and, therefore, the corresponding bacteria and fungi that are active in each phase (Ravindran et al., 2021). The decomposition of organic waste mostly occurs during the thermophilic phase, where oxygen is consumed by microorganisms, while carbon dioxide and ammonia are released. This phase is also crucial for good compost quality since the high temperatures reached in this phase enable the elimination of potential pathogens (Babu et al., 2021).

In urban areas, compost has not only been regarded to recover nutrients but also as a method for urban soil remediation (Heyman et al., 2019; Kranz et al., 2020; Schwartz et al., 2017). In some cities, compost generated in urban and peri-urban areas is mostly used for landscaping inside the urban area, but only a small fraction is destined for agricultural purposes (Eldridge et al., 2018). Although composting might be the most common way to treat biomass, food waste and sewage sludge for nutrient recovery, it also presents some downsides. During the process of composting, a loss of N can occur due to ammonia volatilization and even emissions in the form of N_2O and N_2, which added to the CH_4 emissions can cancel out the carbon credit for fertilizer substitution (Chen et al., 2022). Usually, good P mineralization is observed, generating low N:P ratios, although this also results in the leaching of P with greater compost applications (Shrestha et al., 2020; Small et al., 2019; Wielemaker et al., 2019). Further N and P losses may also be experienced when inappropriate management is provided, e.g., when the compost pile features too much moisture, high aeration, alkaline pH and a low C/N ratio (Jiang et al., 2011; Kashima, 2020). To achieve a good compost, an initial moisture in the range of 50–60 % and a pH value between 5.5 and 8 must be ensured (Ali et al., 2023). Usually a long process is needed, guaranteeing the elimination of potential pathogens and nutrient availability. This last part might be crucial since the mineralization process of N via composting can be very slow (Zabaleta and Rodic, 2015). While traditional composting can be time-consuming, in-vessel composting systems can be a good way to shorten composting periods while also having overall better control over the composting conditions (Ravindran et al., 2021).

The closing of nutrient loops by composting has been regarded as having great potential, Dsouza et al. (2021) even proposing a direct benefit from compost and plant production with CO_2 enrichment from compost exhaust, compost itself and leached nutrients (Chojnacka et al., 2022). By the addition of amendments such as bulking agents or other urban-sourced materials for co-composting, it is possible to reach optimal pH, particle size, moisture and C/N ratios and to serve as biofilters for potential GHG emissions (Kaudal and Weatherley, 2018; Asquer et al., 2019; Awasthi et al., 2020) like in the case of faecal sludge co-composting with sawdust as bulking agent which has shown promising results to avoid nitrogen losses during the composting process (Manga et al., 2022).

The final compost composition and quality are also highly dependent on the incoming feed, and great differences between two sources are observed. Mechanical separation of organic waste from green wastes that can originate in urban settings is crucial, especially with reference to the composition and impurity content, as well as heavy metal concentrations (Smith, 2009).

The use of compost obtained from food waste in vegetable production has shown great yields when substituting chemical for 30 % of the N requirements, showing better results than the crops fertilized with 100 % N chemical fertilizer (Fang et al., 2023).

3.3.11. Compost tea

Compost and vermicompost tea, originating during the composting and vermicomposting processes or by the addition of water, have also been considered important nutrient sources and have been used in hydroponic production systems, showing promising results in the nutritional content within the plants, although yield could be compromised (García-Villela et al., 2020; Pérez et al., 2012; Preciado-Rangel et al., 2015; Santiago-López et al., 2016). The compost tea quality greatly depends on the compost composition and feed origin, as compost tea from municipal waste is a great source of necessary nutrients for plant growth and is applicable to hydroponic systems via the irrigation system. The increase in bacterial activity by the application of compost tea can also increase pathogenic suppression in the plant substrate (Stewart-Wade, 2020).

3.3.12. Thermal treatment

Waste thermal treatments such as incineration, pyrolysis, and emerging technologies like hydrothermal carbonization (HTC) and hydrothermal liquefaction (HTL) (Sarrion et al., 2021) are processes that can be utilized with all kinds of organic residue, entailing the use of high temperature and pressure to produce ashes in incineration processes, referred to as biochars in the case of pyrolysis, hydrochars in the case of HTC (Möller et al., 2018) or energy dense biocrude in the case of HTL (Gollakota and Savage, 2018). While incineration also produces ashes with inorganic P (Fang et al., 2020; Hartmann et al., 2020; Kirchmann et al., 2017), this process requires high energy inputs while generating high carbon losses. However, incineration is a common practice in waste disposal and the process allows for recovering energy and P (Li et al., 2020), as in the case of Sweden for municipal waste (Ahmed et al., 2019).

The principle of biochar production is the use of pyrolysis to breakdown and reorder the minerals and substances in organic biomass, and while other elements disperse during the process, P and K remain retained (Guilayn et al., 2020). The P retention in the produced biochar depends on the retention time and temperature, with greater retention at 450 °C to 600 °C (Sun et al., 2018). Other processes to produce biochar can be made with lower temperature requirements of approximately 440–500 °C (low temperature pyrolysis) or even 180–250 °C, as in the case of HTC (Dutta et al., 2021; Sun et al., 2018). To generate biochar with great nutrient content, it is important to give a high nutrient-containing feedstock. When exploring the potential of municipal organic waste and sewage sludge, high concentrations of P were observed (Sun et al., 2018). The high temperatures achieved during pyrolysis are favourable for the combustion of pathogens and are also able to immobilize heavy metals when appropriate management and processes are ensured (Sun et al., 2018; Xia et al., 2020). These characteristics make thermal combustion a suitable process to be applied in WWTP (Wang et al., 2020; Zheng et al., 2020), where sludges with high P content can be dried and combusted. In HTC and HTL processes, on the other hand, no previous drying is required, and these processes are potentially more energetically efficient (Rao et al., 2018). Food waste and food waste digestate contain great moisture and are good candidates for HTC and HTL.

Hydrochar also generates a nutrient-rich water as a process by-product, which can be further reused for fertilization purposes (Dutta et al., 2021; Wang et al., 2020; Zabaniotou and Stamou, 2020; Zheng et al., 2020) through recovery techniques like struvite precipitation (Sarrion et al., 2023b). To increase the nutrient content of the process water the resulting hydrochar of the HTC process can be washed with acid to encourage the N, P and K leaching, with up to 100 % and 98 % of N and P transference respectively (Sarrion et al., 2021; Sarrion et al., 2023a).

On a different note, high temperatures and significant amounts of acetic acid are needed under HTL working conditions to regulate the feedstock pH, which may not be economically feasible. Therefore, in order to reduce the operational cost of HTL, future research should find substitutes for acetic acid (e.g., other affordable organic acids, or heterogeneous solid acid catalysts) that can maximize energy and nutrient recovery for digestates utilizing HTL under more accommodating operating circumstances (Sudibyo et al., 2022).

The slow P release of biochar and hydrochar products makes them favourable fertilizers that could avoid further P losses into the soil and water bodies, as in the case of commercial fertilizers or manure (Möller et al., 2018; Sun et al., 2018).

Further benefits from the use of biochar and hydrochar are the promotion of carbon sequestration and plant growth and the increase in microbial communities in the soil (Dutta et al., 2021; Ijaz et al., 2020; Zabaniotou and Stamou, 2020).

Alternative to the previously mentioned thermal treatments, the natural freeze concentration uses the freezing point of water and salt concentrated solutions as a natural separation method. This has been also used on different wastewater streams emulating cold climate conditions for the concentration of nitrate water (John et al., 2023).

3.3.13. Vermicomposting

To increase and stabilize the process of composting, the use of earthworms can be encouraged. The derived compost, also referred to as vermicompost, is the bio-oxidation and stabilization of organic matter by earthworms and other microorganisms (Suthar, 2007). For this process, some earthworms have been identified as detritus feeders. Some examples of the earthworms most characterized in organic waste recycling are *Perionyx excavates* (Perrier), *Eisenia fetida* (Savigny) and *Eudrilus eugeniae* (Kingberg) (Biruntha et al., 2020; Gupta and Garg, 2009; Pattnaik and Reddy, 2010). These species have also been categorized as fast debris feeders and are capable of reducing hazardous waste material (Ahadi et al., 2020; Ravindran et al., 2021). Although it is a very eco-friendly and mostly cost-free addition to composting, vermicomposting can entail some different management skills. While the production of compost can entail 6 to 9 months, the process of vermicomposting can be much faster, ranging between 1 and 4 months Its end product can be more homogenous than that of thermophilic compost, and its nutrient content is also enhanced (Ravindran et al., 2021; Schröder et al., 2021). Previous studies on the vermicomposting of sewage sludge and green waste have reported an increase in nutrient availability and therefore greater yield production as well as the content of humic substances and plant growth-promoting hormones (Biruntha et al., 2020; Hanc and Pliva, 2013; Tognetti et al., 2005). On the other hand, the production of vermicompost needs greater monitoring and skill for its production, maintaining certain conditions to ensure good living conditions for earthworms. The specifications and conditions that have to be maintained are an initial C/N range below 40:1, a temperature range of 18–67 °C, a pH range of 5.9–8.3 and a moisture content of approximately 10 % (Ahmed et al., 2019; Stewart-Wade, 2020). To achieve these conditions, a previous composting phase is often encouraged (Ravindran et al., 2021). It should be noted, however, that temperatures lower than those achieved in the thermolysis stage in the composting process entail a reduced effectiveness to ensure a pathogen-free final product (Biruntha et al., 2020; Hanc and Pliva, 2013; Tognetti et al., 2005). The vermicomposting process can generate a by-product known as vermicompost leachate which contains beneficial microbial communities that can enhance the nutrient uptake by plants when used in soilless production systems (Wongkiew et al., 2023).

4. Case study of nutrient recovery application: exploring the potential of NPK recovery in the AMB

Barcelona is a densely populated city in the Mediterranean area with 16,420 inhabitants km^{-2}, with limited land availability that has prompted the inclusion of agricultural activities inside the urban area, especially focusing on rooftop agriculture systems (Appolloni et al., 2021; Zambrano-Prado et al., 2021c). This interest has led to extensive research and educational activities from Higher Education and Research institutions (e.g., ICTA-UAB, UPC, IRTA) as well as organizations (e.g., Replantem) focusing on the integration and application of agriculture in the city of Barcelona. The city council has also promoted these activities with the creation of 7 hydroponic installations on building rooftops and patios for social and community integration purposes (IMPD project) (Biel, 2019) and hosts an annual green roof contest to endorse projects that propose the creation of rooftop gardens (Ajuntament de Barcelona, 2020). The objective is the creation of 34,100 m^2 of green roofs by 2030 (Zambrano-Prado et al., 2021c).

The potential of the metropolitan area of Barcelona to implement rooftop open air agriculture and rooftop greenhouses has already been identified for several urban and peri-urban areas. Such work has been developed with the help of GIS rooftop databases and remote sensing approaches that mainly focus on larger roof extensions to host these installations, with industrial and retail parks and large social housing neighbourhoods being the best candidates (Fig. 2) (Nadal et al., 2017, 2018; Sanyé-Mengual et al., 2018; Toboso-Chavero et al., 2019, 2021; Zambrano-Prado et al., 2021a; Zambrano-Prado et al., 2021b).

While several areas have been identified as potential UA sites due to the rooftop material and rooftop extension, these have only been contemplated in a theoretical way. However, other rooftop UA areas have been implemented, mainly through the social project “Horts al terrat” from the municipality of Barcelona and the green roof competition. The total potential identified in the literature was 44.44 ha, and the area of existing sites was 0.77 ha, comprising a total of 45.21 ha (452’100 m^2) within the metropolitan area of Barcelona (Table 3). Other typologies of UA, such as indoor or soil-based UA, were not included.

If this area is dedicated to tomato production with an estimated productivity of up to 16.5 kg m^{-2} year^{-1} (Sanyé-Mengual et al., 2018), the potential production could entail up to 7459 t of tomato, which is equal to 13.3 % of the tomato consumption within the metropolitan area (Table 1 in the Supplementary material).

From 2017 to 2021 the LIFE project ENRICH (standing for “Enhanced Nitrogen and phosphorus Recovery from wastewater and Integration in the value Chain”) was developed through the collaboration of institutes, not only dedicated to the recovery of nutrients from wastewater streams in the AMB but also its valorisation in agriculture. This project enabled the creation of a demonstration plant and has defined several outcomes through its lifetime. The applied recovery strategies defined during this project were the struvite precipitation for slow-release solid fertilizer, zeolite adsorption and membrane contractors to produce liquid fertilizer, which were further used in the cultivation of tomatoes (Carreras-sempere et al., 2021). The results obtained from the pilot production and application determined that the environmental benefits and economic viability of struvite crystallization, ion exchange and liquid-liquid MC are promising for future nutrient recovery and valorisation (Mayor et al., 2023). A similar project that has taken place in the same area is the Waste2Product, dedicated to the recovery of nutrients of WWTP through sorbents and nutrient-enriched zeolites to be further applied as slow-releasing soil amendments (Guaya et al., 2018, 2020; You et al., 2019).

On the other hand, projects for the nutrient valorisation of organic wastes have also been developed in this region. In 2004, the AMB began a program to promote composting of domestic waste with the collaboration of the Catalan Waste Agency. Data from 2014 states that 35 municipalities joined the composting program, with the participation of 4250 citizens.

The Horizon 2020 project Desicive, focused on the recovery of nutrients through the composting and anaerobic digestion of bio, organic and food wate based on of its pilot scale demonstration sites in the campus of the Universita Autónoma de Barcelona, within the AMB. Through this project the campus waste production was characterized

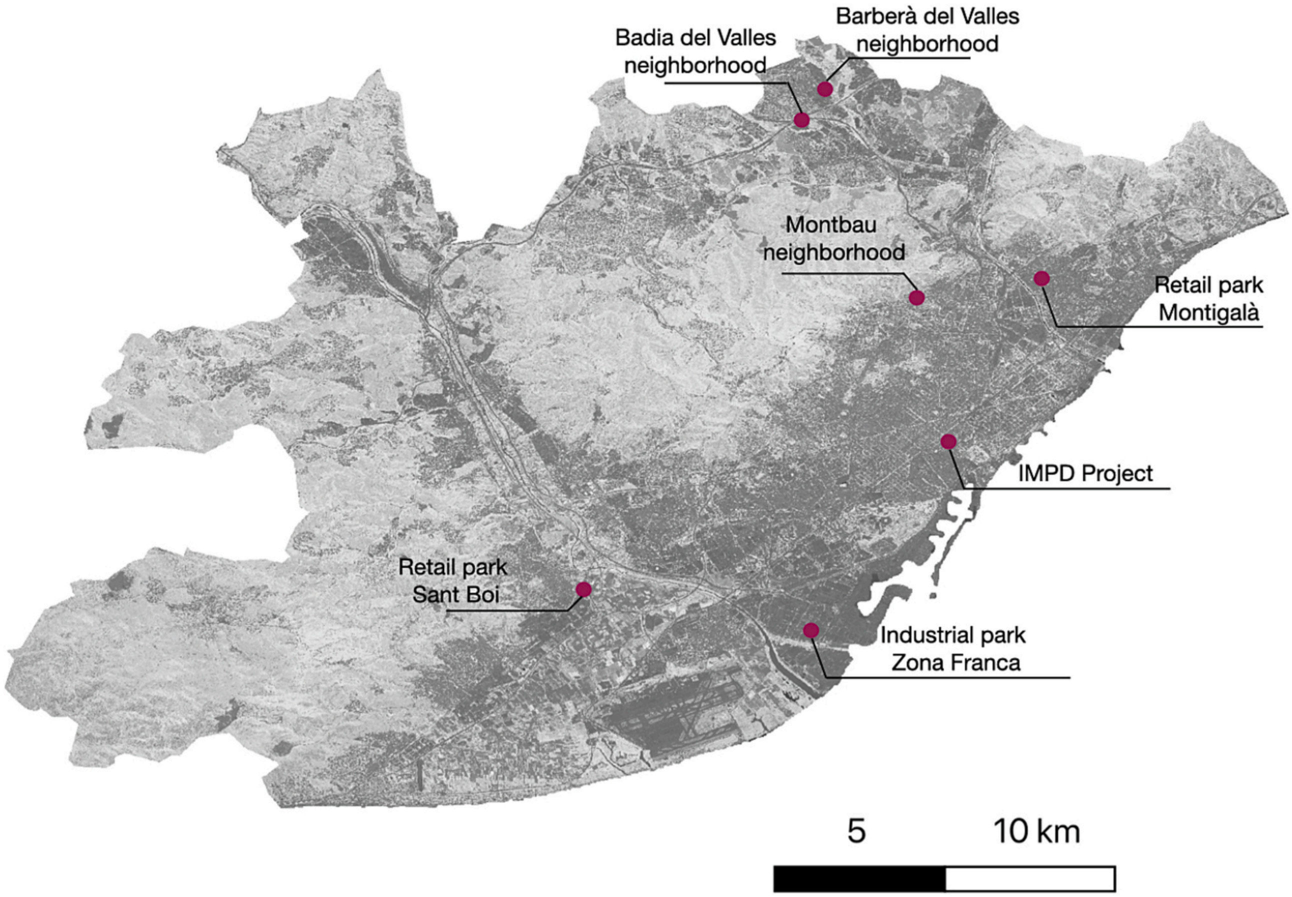


Fig. 2. Map of the Metropolitan Area of Barcelona with identified locations for rooftop agriculture.

Table 3
Areas identified within the metropolitan area of Barcelona for UA on rooftops.

	Area/project name	Building type	Area	Comment	Reference
Potential areas found in literature	Zona Franca	Industrial park	13.06 ha	Approximately 14 % of tomato imported (128,000 people demand year^{-1})	(Sanyé Mengual, 2015)
	Sant Boi	Retail park	5.58 ha	Urban self-supply 3.8 %	Sanyé-Mengual et al., 2018
	Montigalà	Retail park	5.22 ha	Urban self-supply 3.5 %	Sanyé-Mengual et al., 2018
	Montbau	Neighbourhood	0.06 ha	Up to 37 % of tomato self-supply	(Toboso-Chavero et al., 2019)
	Badia del Vallés	Neighbourhood	20.52 ha	Self-sufficiency for tomato 210 % and lettuce 21 % in the neighbourhood	(Zambrano-Prado et al., 2021a; Zambrano-Prado et al., 2021b; Zambrano-Prado et al., 2021c)
	Barberà del Vallés	Neighbourhood			
Implemented areas	IMPD Project "Hort al terrat"	Municipality buildings	0.02 ha	3590 kg year^{-1} of vegetables	(Biel, 2019) (Ajuntament de Barcelona, 2020)
	Green Roof Competition	Private rooftops	0.75 ha	Vegetable and urban green	(Ajuntament de Barcelona, 2020)

and a micro anaerobic digestor was implemented for the generation of energy and fertilizer in the form of composted digestate.

Furthermore, emerging businesses like "Abono km0" are developing a commercial strategy to recovery food waste from restaurants and kitchens to further produce vermicompost (Website: https://abonokm0.com/), although there is no available data on the recovered amount of waste nor compost production.

4.1. Organic waste generation and treatment

While a waste recovery system is generally implemented with classified sorting bins, only approximately 36 % of the waste is separately recovered, with a greater fraction of "rest" or "unclassified" waste. The current goal is to increase the separated fraction up to 50 % in all municipalities, which was achieved by only 16 % of all municipalities by 2018. The generation of separated organic waste in the metropolitan area is divided into three categories: household waste, organic waste from large producers and green waste. The total organic waste collected from the classified sorting bin in 2020 was 184,000 tonnes, of which 78 % was household waste, while only 6 % and 16 % originated from large producers and green waste, respectively. This differentiated waste is then transported to two specialized installation typologies: comprehensive waste handling stations called Ecoparcs (1, 2 and 4) and composting sites. The handling of this waste fraction is similar in Ecoparc, with a pre-treatment to prevent impurities and the mixing of the three organic waste categories for digestion and further production of compost. The composting sites, on the other hand, do not entail an anaerobic digestion step to produce compost. Although the waste is collected in separated bins, the percentage of impurities is still high depending on the municipality of origin. Ecoparcs 1, 2 and 4 presented impurity percentage ranges of 8.7–23 %, 5.4–30.3 % and 1.7–36.6 %, respectively, while the composting sites showed ranges of 2.5–19.2 %.

The locations of the Ecoparcs and composting sites, which are shown in Fig. 3, are mostly located within the metropolitan area of Barcelona except for Ecoparc 4. The destination of the collected organic waste from the differentiated bins in 2020 was mostly in Ecoparc 2 with up to 44 % of the generated waste, while Ecoparc 1, Ecoparc 4 and the composting sites received 32 %, 18 % and 7 %, respectively, of the generated waste. The yearly production of compost almost reaches 30,000 tons, which can be further employed in gardens and surrounding agriculture. The "rest" or "unclassified" fraction produced in 2020 was greater than 800,000 tons, and it was also processed in Ecoparcs 1, 2, 3 and 4, with a distribution of 21 %, 23 %, 23 % and 33 %, respectively, of the total. The processes this "unclassified" waste fraction undergoes is a mechanic separation treatment as well as biologic treatments and a biowaste stabilization phase for the annual production of more than 70,000 tons of stabilized biowaste for soil amendment and landfilling.

4.2. Wastewater generation and treatment

In the metropolitan area of Barcelona, 7 WWTPs (Fig. 3) are responsible for the treatment of approximately 270,000 Mm^3 of wastewater each year. The technologies of all plants vary for both sludge and effluent treatments, between sludge anaerobic digestion and composting and between sludge dewatering and field application, to secondary or tertiary water treatments (Table 4).

The yearly production of sludge is approximately 57,000 tons of dry matter (2020), which is then directly employed in agriculture (24 %) or composting (68 %). The water treatment in the WWTP from El Prat de Llobregat follows five main steps, starting with a pre-treatment for solid and oil separation, a primary treatment where the sludge is removed, a secondary treatment with nitrification and denitrification processes for nitrogen removal, tertiary treatment, denitrification processes and ultrafiltration and reverse osmosis to retrieve and regenerate water. Apart from the WWTP in Gavà I Viladecans, no other WWTP has water regenerating processes.

In conventional WWTPs, approximately 30 % of the influent wastewater nutrients are removed through active sludge separation in the primary treatment, while further nitrification and denitrification processes in the secondary water treatment can reach a removal of up to 70 %. This nutrient removal is applied in 4 WWTPs in the metropolitan area, namely, the WWTP in el Prat de Llobregat, in St. Feliu de Llobregat, in Gavà I Viladecans and in Begues. Although this can be considered a good removal rate, approximately 1200 t of N and 160 t of P are still released every year into the Mediterranean Sea, only considering the WWTP in El Prat de Llobregat. Therefore, the potential for additional nutrient removal is great.

Work on the reduction of P and N in the wastewater effluent in El Prat de Llobregat and Besòs WWTP's has already been done, proposing struvite precipitation or zeolite adsorption. These works consider the recovery of these nutrients a success, obtaining 5000 t $year^{-1}$ of loaded zeolite with a 15 % PO_4^{3-} content or otherwise a range of 43–368 t $year^{-1}$ of P in the form of struvite in the El Prat de Llobregat WWTP when precipitation is considered (depending on the recovery technology).

4.3. Nutrient recovery potential in the metropolitan area of Barcelona

Considering all previously collected information, an estimate of the potential of nutrient recovery with the existing infrastructure of the AMB can be determined (Table 5 and Tables 2, and 3 from the supporting material). This estimation is again based on the assumption that all the defined area are dedicated to tomato production.

The existing generation of food/bio/organic waste and the produced

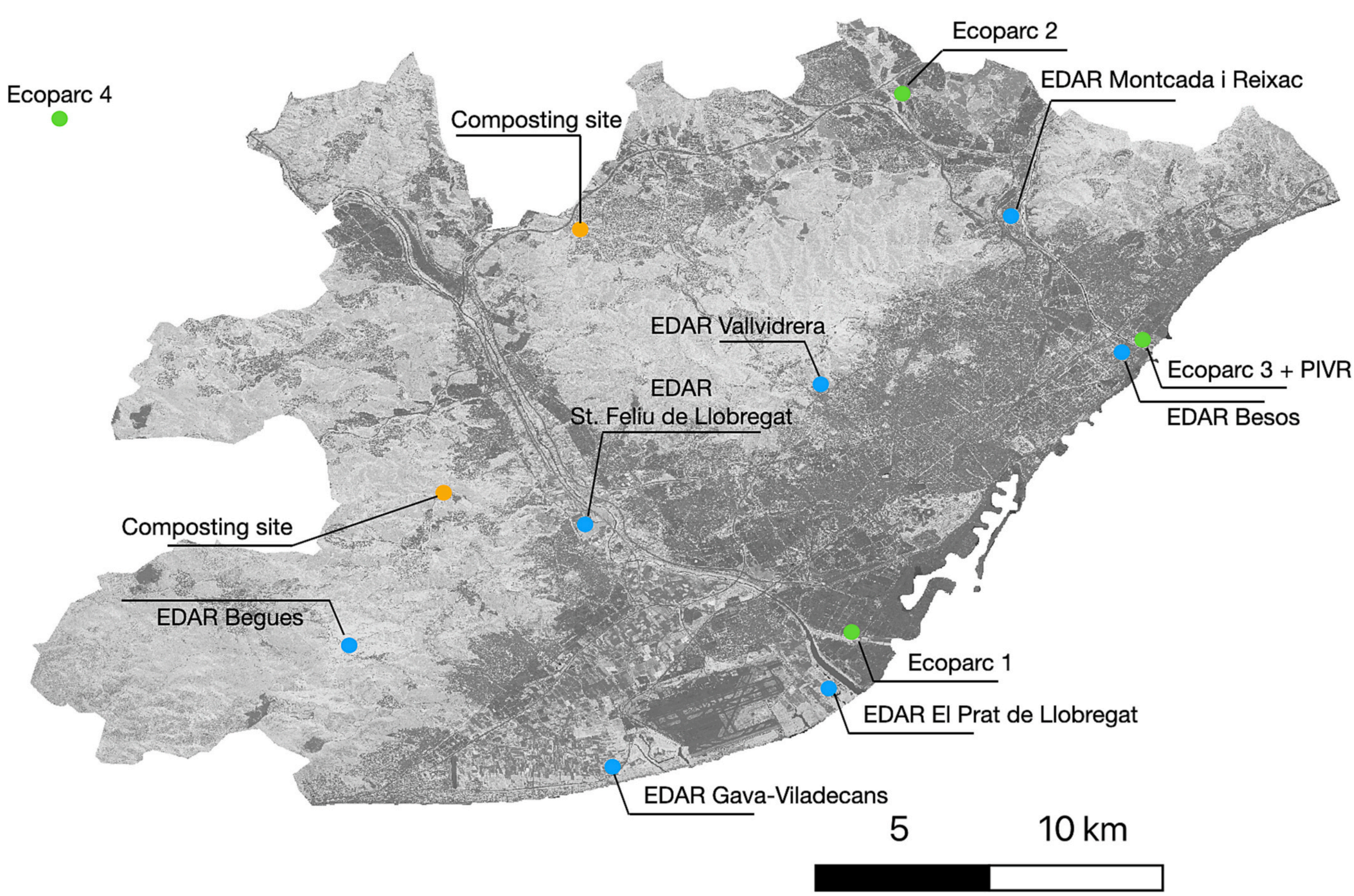


Fig. 3. Map of the metropolitan area of Barcelona with the locations of the currently active WWTPs (blue), Ecoparcs (green) and composting sites (orange).

Table 4
Description of the WWTPs identified in the metropolitan area of Barcelona, yearly treated amount (* obtained from the official website of the metropolitan area of Barcelona: https://www.amb.cat/c/portal/download_file?type=xls&name=aigua_tractada_edar&jsons=aigua_cabal_tractat_edar-0-0), and percentage in relation to the total wastewater treated in the area (** obtained from the technical sheets of the EDAR installations of the official website: https://www.amb.cat/es/web/ecologia/aigua/instalacions-i-equipaments). Estimation of incoming N and P (*** based on You et al., 2019); CAnD – Composted anaerobic Digestion, DW – Dewatering.

	WWTP	Incoming flow (Mm^3 y^{-1}) (2019)*	% from total **	Sludge treatment **	Effluent treatment **	Incoming N (ton y^{-1}) ***	Incoming P (ton y^{-1}) ***
1	El Prat de Llobregat	92.1	36 %	CAnD	Secondary and tertiary treatment	5525 t	626 t
2	Besòs	120.4	45 %	DW	Secondary treatment	7225 t	819 t
3	St. Feliu de Llobregat	18.5	7 %	CAnD	Secondary and tertiary treatment	1114 t	126 t
4	Montcada I Reixac	18.8	6 %	–	Secondary treatment	1129 t	128 t
5	Gavà I Viladecans	14.8	5 %	CAnD	Secondary and second decanter	892 t	101 t
6	Begues	0.3	0.7 %	–	Secondary treatment	21 t	2 t
7	Vallvidrera	0.2	0.5 %	DW	AnMBR	15 t	2 t

Table 5
OM = Organic Material, (* taken from the technical sheet of Ecoparcs and composting site official website: https://www.amb.cat/es/web/ecologia/residus/instalacions-i-equipaments/llistat), compost generated in Ecoparcs 1 to 4 was destined to a digestion process; WWS = wastewater sludge (** obtained from the technical sheets of the EDAR installations of the official website: https://www.amb.cat/es/web/ecologia/aigua/instalacions-i-equipaments), WWE = wastewater effluent, Min = minimum removal of 7 % P with AirPrex technology, and Max = maximum removal of 60 % P with RemNut Technology; ** obtained from Rufí-Salís et al. (2020a), *** obtained from You et al. (2019).

Waste type	Amount (ton DW/year) (2020)	N content in waste (kg)	P content in waste (kg)	Times N demand is met	Times P demand is met
OM Compost*	987.7	21,235	7802	1.8	2.7
OM Digest*	19,250	525,525	171,325	46	58
WWS- Compost**	57,000	1,328,100	1,117,200	117.5	380.2
WWE - Struvite Min***	341.3	19,829	43,000	1.7	14.6
WWE - Struvite Max***	2920.6	169,962	373,000	14.5	125.2
WWE – Zeolite****	5000	–	750,000	–	255.2

compost and digest in the centralized Ecoparcs and composting sites can produce up to 550 t of N-nitrogen and 170 t of P-phosphorous each year, meeting approximately 48 and 60 times the N and P demand, respectively.

Sludge compost generation in all WWTPs can be a great source of nutrients, especially P, with a yearly production greater than 1000 t, which can cover 117 and 380 times the UA requirements of N and P, respectively.

For both recovery strategies no additional logistics would be needed, taking in account the existing infrastructure and existing waste transport. The centralized recovery in the local facilities shown in the metropolitan area would only require an additional transport to the urban gardens.

The generation of struvite can vary between recovery technologies, as defined by Rufí-Salís et al. (2020a), ranging from the lowest recovery of 7 % of P for incoming wastewater P in the AirPrex technology to P recovery of 60 % for incoming P in the RemNut technology. In this case, only the WWTP in El Prat de Llobregat is considered due to the existing necessary installation. The generation of struvite only in this plant can meet the N demand 1.7 to 14.5 times and the P demand by 14.6 to 125.2 times.

The work of You et al. (2019) encourages the possibility of ion exchange processes with loaded zeolites in the El Prat de Llobregat WWTP due to the existing filtration installations. If the struvite precipitation is not considered, yearly recovery of 5000 t of loaded zeolite is defined with an approximate content of 750 t of P, meeting the P demand by 255 times. While the simultaneous nutrient obtention is possible in the defined OM and WWS scenarios, in this case the WWE can undergo different processes that are not considered to happen at the same time, like struvite precipitation and ion exchange process.

Although other technologies could be applied to the nutrient cycle in this area, it would be necessary to further install additional infrastructure, which can also generate further environmental impacts on the nutrient recovery systems (Rufí-Salís et al., 2020a). On the other hand, the potential of struvite as a slow-release fertilizer and the generally positive characteristics to be used in UA due to the lack of smell, low content of heavy metals and potential pathogens suggests the production that could be achieved in all WWTPs (Table 4 from the supporting information). If all WWTPs had the capacity for struvite precipitation between 7 and 60 % of the incoming P, a potential production of 126 to 1082 t of P and 52 to 489 t of N could be recovered yearly.

5. Constraints and obstacles to fulfilling the nutrient recovery potential

The capabilities to recycle and reuse nutrients such as N and P are clear, but why are the advances in the application of these processes so slow? The use of organic waste from urban areas is generally perceived as having a poor quality or containing great amounts of unwanted elements that could be toxic or polluting (Gímenez Lorang et al., 2005). This finding is also true for sewage sludge quality, which was determined to contain great amounts of potentially toxic elements (PTEs) in long-term experiments that could be traced to World War II (Möller et al., 2018). On the other hand, the quality of organic waste and sewage sludge has been increasing for the past decade, making it a great nutrient resource. Nevertheless, due to public or private legislation standards, most of these recovery sources are not permitted, e.g., in organic farming (Awasthi et al., 2020; Möller et al., 2018). Furthermore, the production costs of recovered P can be a constraint to implementing these processes compared to the existing extraction chain of mined phosphate rock (Chojnacka et al., 2020; Oarga-Mulec et al., 2019). Higher initial investment in large scale application and the maintenance costs can also influence the application of technologies like nutrient recovery systems in existing WWTP (Environmental Protection Agency, 2021). Therefore, most of the emerging recovery technologies remain on the laboratory or pilot scale with limited market uptake (Cordell et al., 2021). To encompass most barriers on the up-scaling and application of P recovery technologies Barquet et al., 2020 defines four key points, being 1) the current insufficient knowledge on the technology application and performance (economic, agricultural and environmental); 2) the slow change in existing infrastructure for the implementation of recovery systems, such as WWTP with long-term financial investments, defining lower applicability of decentralized systems such as source-separated systems; 3) the lack of coordination between livestock and crop farms, making the application of mineral and synthetic fertilizer a more economically viable alternative 4) the lack of awareness on the current nutrient dependency and policy implementation. Barquet et al., 2020 further point out the dissociation between the nutrient reduction and recovery strategies, defining that while most policy implementations on nutrient management are directed to the reduction of load thresholds, this does not specifically encourage nutrient recovery for its valorisation as agricultural fertilizer. Although the application of recovery technologies is still not fully considered in waste treatment processes, this state of mind is slowly changing and pushed towards recovery in response to the foreseen P shortages in the coming years as well as the environmental impact of untreated waste. Since 2021 European legislation has considered the use of struvite as an agricultural fertilizer and therefore encouraged its recovery and application. An increase in biowaste-derived fertilizers is expected and estimated to replace up to 30 % of inorganic fertilizers (Chojnacka et al., 2020) and up to 50–60 % of phosphate rock imported into Europe and utilized in agriculture (Möller et al., 2018).

6. Conclusions

The increase in nutrient circularity has been a pressing matter in recent years, and while mainly associated with agriculture and rural areas, the loss of nutrients is also a reality in cities. The present study has identified the literature concerning nutrient recovery technologies from urban waste flows that can be further repurposed in UA.

Three main conclusions can be drawn from this study.

First, two main waste types were identified as most regarded in the literature, namely, organic, bio, and food waste and wastewater. Under these two umbrellas, 24 recovery strategies pertain to organic-bio and food waste and 58 recovery strategies pertain to wastewater.

Second, it can be concluded that the yearly production amount of both waste types can fulfil the N and P requirements for UA in the metropolitan area of Barcelona in ranges of 2,7 to 380,2 and 1,7 to 117,5 times the necessary amount for P and N, respectively, contemplating centralized recovery strategies suitable for the currently existing infrastructures.

Third, the promising results for many recovery strategies are put on hold or restricted to a laboratory scale due to current perceptions, legislation and economic cost for this new market that do not facilitate their application and nutrient repurposing for agriculture. On the other hand, these perceptions could shift in the near future due to the pressing need for P recovery, new emerging technologies and the need to reduce nutrient leaching into surrounding ecosystems.

Overall, the potential of nutrient recovery in urban areas is clear and has been vastly explored in the scientific community in recent years. While currently centralized nutrient recovery can be encouraged in urban settings a shift to source separated and better managed waste disposals must be envisioned in the future making nutrient recovery an integral part of urban waste management strategies.

CRediT authorship contribution statement

All authors were responsible for the conception and design of the study. V. Arcas-Pilz, G. Villalba, X. Gabarrell and F. Orsini, conceived the original idea for the study. V. Arcas-Pilz processed and analysed the data. V. Arcas-Pilz took the lead in writing the manuscript. All authors critically revised the draft for important intellectual content. All authors gave their final approval to the manuscript.

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.

Data availability

Data will be made available on request.

Acknowledgements

The authors are grateful to the Spanish Ministry of Economy, Industry and Competitiveness (Spain) for the grant awarded to V. Arcas-Pilz (FPI-MINECO 2018). This work was supported by the “María de Maeztu” program for Units of Excellence in R&D [CEX2019-0940-M]. The research leading to these results has received funding from the European Research Council under the European Horizon2020 research and innovation program under grant agreement No 862663 (FoodE) as well as, the ERC grant agreement n° 818002 URBAG, awarded to Gara Villalba and the Interreg NWE grant agreement NWE 474 GROOF. Finally, we would like to thank the Ministerio de Ciencia y Innovación (MCIN/AEI/10.13039/501100011033/FEDER, UE) and the EU founding “Next Generation” for the founding of the projects MOVE4EDU (PID2021-126845OB-C21) and BINAFET (TED2021-130047B-C21). The publication reflects the author’s views. The Research Executive Agency (REA) is not liable for any use that may be made of the information contained therein.

References

Ahadi, N., et al., 2020. Remediation of heavy metals and enhancement of fertilizing potential of a sewage sludge by the synergistic interaction of woodlice and earthworms. J. Hazard. Mater. 385 (September 2019).

Ahmed, M., et al., 2019. Innovative processes and technologies for nutrient recovery from wastes: a comprehensive review. Sustainability (Switzerland) 11 (18).

Ajuntament de Barcelona, 2020. Second Green Roofs Competition. https://ajuntament.barcelona.cat/ecologiaurbana/en/green-roof-competition. (Accessed 1 December 2021).

Akoto-Danso, E.K., et al., 2019. Agronomic effects of biochar and wastewater irrigation in urban crop production of tamale, Northern Ghana. Nutr. Cycl. Agroecosyst. 115 (2), 231–247.

Ali, H., Leta, S., Hussen, A., Alemu, T., 2023. Resource recovery potential from source-separated organic municipal solid waste: opportunities for organic fertilizer production and creating sustainable urban agriculture in Ethiopia. J. Mater. Cycles Waste Manag. 25 (4), 2417–2430. https://doi.org/10.1007/s10163-023-01709-5.

Amaya-Santos, G., et al., 2022. Production of *Chlorella vulgaris* using urban wastewater: assessment of the nutrient recovery capacity of the biomass and its plant biostimulant effects. J. Appl. Phycol. 34 (6), 2971–2979.

Anders, A., et al., 2021. Phosphorus recycling from activated sludge of full-scale wastewater treatment plants by fast inversion of the biological phosphorus elimination mechanism. J. Environ. Chem. Eng. 9 (6), 106403. https://doi.org/10.1016/j.jece.2021.106403.

Appolloni, E., et al., 2021. The Global Rise of Urban Rooftop Agriculture: A Review of Worldwide Cases, 296.

Arcas-Pilz, V., et al., 2021. Recovered phosphorus for a more resilient urban agriculture: assessment of the fertilizer potential of struvite in hydroponics. Sci. Total Environ. 799.

Asquer, C., et al., 2019. Biomass and bioenergy biomass ash characterisation for reuse as additive in composting process. Biomass Bioenergy 123 (July 2018), 186–194. https://doi.org/10.1016/j.biombioe.2019.03.001.

Awasthi, S.K., et al., 2020. Changes in global trends in food waste composting: research challenges and opportunities. Bioresour. Technol. 299 (December 2019), 122555. https://doi.org/10.1016/j.biortech.2019.122555.

Babu, R., Prieto Veramendi, P.M., Rene, E.R., 2021. Strategies for resource recovery from the organic fraction of municipal solid waste. Case Stud. Chem. Environ. Eng. 3 (March), 100098. https://doi.org/10.1016/j.cscee.2021.100098.

Bareha, Y., et al., 2022. Evaluating the impact of substrate addition for anaerobic co-digestion on biogas production and digestate quality: the case of deinking sludge. J. Environ. Manag. 319 (June), 115657. https://doi.org/10.1016/j.jenvman.2022.115657.

Barquet, K., Järnberg, L., Rosemarin, A., Macura, B., 2020. Identifying barriers and opportunities for a circular phosphorus economy in the Baltic Sea region. Water Res. 171.

Bastida, F., et al., 2019. The effects of struvite and sewage sludge on plant yield and the microbial community of a semiarid Mediterranean soil. Geoderma 337, 1051–1057. https://doi.org/10.1016/j.geoderma.2018.10.046.

Battista, F., et al., 2020. Food wastes and sewage sludge as feedstock for an urban biorefinery producing biofuels and added-value bioproducts. J. Chem. Technol. Biotechnol. 95 (2), 328–338.

Bergstrand, K.J., Asp, H., Hultberg, M., 2020. Utilizing anaerobic digestates as nutrient solutions in hydroponic production systems. Sustainability (Switzerland) 12 (23), 1–12.

Besson, M., et al., 2021. Environmental assessment of urine, black and grey water separation for resource recovery in a new district compared to centralized wastewater resources recovery plant. J. Clean. Prod. 301.

Biel, C., 2019. Huertos En Azoteas de Edificios Públicos de Barcelona.

Biruntha, M., et al., 2020. Vermiremediation of urban and agricultural biomass residues for nutrient recovery and vermifertilizer production. Waste Biomass Valoriz. 11 (12), 6483–6497. https://doi.org/10.1007/s12649-019-00899-0.

Bolzonella, D., Fatone, F., Gottardo, M., Frison, N., 2018. Nutrients recovery from anaerobic digestate of agro-waste: techno-economic assessment of full scale applications. J. Environ. Manag. 216, 111–119. https://doi.org/10.1016/j.jenvman.2017.08.026.

Brown, Sally, Butman, David, Kurtz, Kate, 2023. Steps to circularity: impact of resource recovery and urban agriculture in Seattle and Tacoma, Washington. J. Environ. Manag. 345 (April), 118648. https://doi.org/10.1016/j.jenvman.2023.118648.

Bruno, Morena, et al., 2022. Ex ante life cycle assessment and environmental cost-benefit analysis of an anaerobic digester in Italy. Clean Waste Syst. 3 (January), 100021. https://doi.org/10.1016/j.clwas.2022.100021.

Calabria, Jorge L., Lens, Piet N.L., Yeh, Daniel H., 2019. Zeolite ion exchange to facilitate anaerobic membrane bioreactor wastewater nitrogen recovery and reuse for lettuce fertigation in vertical hydroponic systems. Environ. Eng. Sci. 36 (6), 690–698.

Campos, José Luis, et al., 2019. Nitrogen and phosphorus recovery from anaerobically pretreated agro-food wastes: a review. Front. Sustain. Food Syst. 2 (January), 1–11.

Carreras-sempere, M., Caceres, R., Viñas, M., Biel, C., 2021. Use of Recovered Struvite and Ammonium Nitrate in Fertigation in Tomato (*Lycopersicum esculentum*) Production for Boosting Circular and Sustainable Horticulture (November).

Chatterjee, Pritha, et al., 2019. Microalgae grow on source separated human urine in nordic climate: outdoor pilot-scale cultivation. J. Environ. Manag. 237 (August 2018), 119–127. https://doi.org/10.1016/j.jenvman.2019.02.074.

Chen, Wenhao, et al., 2022. The GHG mitigation opportunity of sludge management in China. Environ. Res. 212 (PC), 113284. https://doi.org/10.1016/j.envres.2022.113284.

Chojnacka, K., Moustakas, K., Witek-Krowiak, A., 2020. Bio-based fertilizers: a practical approach towards circular economy. Bioresour. Technol. 295 (September 2019), 122223. https://doi.org/10.1016/j.biortech.2019.122223.

Chojnacka, K., Moustakas, K., Mikulewicz, M., 2022. Valorisation of agri-food waste to fertilisers is a challenge in implementing the circular economy concept in practice. Environ. Pollut. 312 (August), 119906. https://doi.org/10.1016/j.envpol.2022.119906.

Close, Kylie, et al., 2021. The storage compounds associated with Tetrasphaera PAO metabolism and the relationship between diversity and P removal. Water Res. 204 (August), 117621. https://doi.org/10.1016/j.watres.2021.117621.

Clyde-Smith, Dominic, Campos, Luiza C., 2023. Engineering hydroponic systems for sustainable wastewater treatment and plant growth. Appl. Sci. (Switzerland) 13 (14).

Cordell, Dana, White, Stuart, 2014. Life's bottleneck: sustaining the world's phosphorus for a food secure future. Annu. Rev. Environ. Resour. 39, 161–188.

Cordell, Dana, Drangert, Jan Olof, White, Stuart, 2009. The story of phosphorus: global food security and food for thought. Glob. Environ. Chang. 19 (2), 292–305.

Cordell, Dana, Brownlie, Will J., Esham, Mohamed, 2021. Commentary: time to take responsibility on phosphorus: towards circular food systems. Glob. Environ. Chang. 71, 102406. https://doi.org/10.1016/j.gloenvcha.2021.102406.

Crini, Grégorio, Lichtfouse, Eric, 2019. Advantages and disadvantages of techniques used for wastewater treatment. Environ. Chem. Lett. 17 (1), 145–155. https://doi.org/10.1007/s10311-018-0785-9.

Czekała, Wojciech, et al., 2022. Biogas plant operation: digestate as the valuable product. Energies 15 (21), 1–11.

Dalvi, Vivek, Chawla, Pushap, Malik, Anushree, 2021. Year-long performance assessment of an on-site pilot scale (100 L) photobioreactor on nutrient recovery and pathogen removal from urban wastewater using native microalgal consortium. Algal Res. 55 (271).

Davidsson, Saraiva, A. Bernstad, Magnusson, N., Bissmont, M., 2017. Technical evaluation of a tank-connected food waste disposer system for biogas production and nutrient recovery. Waste Manag. 65, 153–158. https://doi.org/10.1016/j.wasman.2017.03.052.

Deemter, Dennis, Oller, Isabel, Amat, Ana M., Malato, Sixto, 2022. Advances in membrane separation of urban wastewater effluents for (pre)concentration of microcontaminants and nutrient recovery: a mini review. Chem. Eng. J. Adv. 11 (March), 100298. https://doi.org/10.1016/j.ceja.2022.100298.

Despommier, Dickson, 2013. Farming up the city: the rise of urban vertical farms. Trends Biotechnol. 31 (7), 388–389. https://doi.org/10.1016/j.tibtech.2013.03.008.

Dimambro, Mary, Eco, Cambridge, Rayns, Francis, 2015. Literature review : digestate use in protected horticulture. Horticiltulture 2 (1), 21–45.

D'ostuni, M., et al., 2023. Evaluating the impacts of nutrients recovery from urine wastewater in building-integrated agriculture. A test case study in Amsterdam. Sustain. Cities Soc. 91 (January), 104449. https://doi.org/10.1016/j.scs.2023.104449.

Drangert, Jan-Olof, Tonderski, Karin, McConville, Jennifer, 2018. Extending the European Union waste hierarchy to guide nutrient-effective urban sanitation toward global food security—opportunities for phosphorus recovery. Front. Sustain. Food Syst. 2 (February), 1–13.

Dsouza, Ajwal, Price, Gordon W., Dixon, Mike, Graham, Thomas, 2021. A conceptual framework for incorporation of composting in closed-loop urban controlled environment agriculture. Sustainability (Switzerland) 13 (5), 1–28.

Dutta, Shanta, He, Mingjing, Xiong, Xinni, Tsang, Daniel C.W., 2021. Sustainable management and recycling of food waste anaerobic digestate: a review. Bioresour. Technol. 341 (September), 125915. https://doi.org/10.1016/j.biortech.2021.125915.

Eldridge, Simon M., Yin, Kwong, Nerida, Chan, 2018. Agronomic and economic benefits of green-waste compost for peri-urban vegetable production : implications for food security. Nutr. Cycl. Agroecosyst. 111 (2), 155–173. https://doi.org/10.1007/s10705-018-9931-9.

Environmental Protection Agency, 2021. Life Cycle and Cost Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants (August).

Eraky, Mohamed, et al., 2022. A new cutting-edge review on the bioremediation of anaerobic digestate for environmental applications and cleaner bioenergy. Environ. Res. 213 (June), 113708. https://doi.org/10.1016/j.envres.2022.113708.

Ercilla-Montserrat, Mireia, et al., 2019. Analysis of the consumer's perception of urban food products from a soilless system in rooftop greenhouses: a case study from the mediterranean Area of Barcelona (Spain). Agric. Hum. Values 36 (3), 375–393. http://link.springer.com/10.1007/s10460-019-09920-7 (October 30, 2019).

Escudero, Ania, et al., 2020. Pharmaceuticals removal and nutrient recovery from wastewaters by Chlamydomonas Acidophila. Biochem. Eng. J. 156 (December 2019), 107517. https://doi.org/10.1016/j.bej.2020.107517.

Ezziddine, Maha, Liltved, Helge, Seljåsen, Randi, 2021. Hydroponic lettuce cultivation using organic nutrient solution from aerobic digested aquacultural sludge. Agronomy 11 (8).

Factura, H., et al., 2010. Terra Preta sanitation: re-discovered from an ancient Amazonian civilisation - integrating sanitation, bio-waste management and agriculture. Water Sci. Technol. 61 (10), 2673–2679.

Fang, Le, et al., 2020. Novel recovered compound phosphate fertilizer produced from sewage sludge and its incinerated ash. ACS Sustain. Chem. Eng. 8 (17), 6611–6621.

Fang, Xuejuan, et al., 2023. Closing the food waste loop: analysis of the agronomic performance and potential of food waste disposal products. J. Clean. Prod. 382 (November 2022).

Feiz, Roozbeh, et al., 2022. The biogas yield, climate impact, energy balance, nutrient recovery, and resource cost of biogas production from household food waste—a comparison of multiple cases from Sweden. J. Clean. Prod. 378 (February).

Firmansyah, I., et al., 2021. An integrated assessment of environmental, economic, social and technological parameters of source separated and conventional sanitation concepts: a contribution to sustainability analysis. J. Environ. Manag. 295 (January), 113131. https://doi.org/10.1016/j.jenvman.2021.113131.

Foley, Jeffrey, de Haas, David, Hartley, Ken, Lant, Paul, 2010. Comprehensive life cycle inventories of alternative wastewater treatment systems. Water Res. 44 (5), 1654–1666. https://doi.org/10.1016/j.watres.2009.11.031.

Forbes, Hamish, Quested, Tom, O'Connor, Clementine, 2021. Unep Food Waste Index Report 2021.

García-Villela, K.M., et al., 2020. Ecological nutrient solutions on yield and quality of melon Fruits [Soluciones Nutritivas Ecológicas En La Producción y Calidad de Melón]. Terra Latinoam. 38 (1), 39–44. https://www.scopus.com/inward/record.uri?eid=2-s2.0-85083463107&doi=10.28940%2Fterra.v38i1.527%2F709&partnerID=40&md5=12a4025afcc4caf7a6f0e2ab48661d49.

Gienau, T., Brüß, U., Kraume, M., Rosenberger, S., 2018a. Nutrient recovery from anaerobic sludge by membrane filtration: pilot tests at a 2.5 MWe biogas plant. Int. J. Recycl. Org. Waste Agric. 7 (4), 325–334. https://doi.org/10.1007/s40093-018-0218-6.

Gienau, T., Brüß, U., Kraume, M., Rosenberger, S., 2018b. Nutrient recovery from biogas digestate by optimised membrane treatment. Waste Biomass Valoriz. 9 (12), 2337–2347. https://doi.org/10.1007/s12649-018-0231-z.

Gímenez Lorang, Antonio, Torrentó, Montserrat Soliva I., Huerta, Óscar, 2005. El Mercat Del Compost a Catalunya Oferta i Demanda.

Girotto, Francesca, Piazza, Laura, 2021. Food waste bioconversion into new food: a mini-review on nutrients circularity in the production of mushrooms, microalgae and insects. Waste Manag. Res. 40, 47–53.

Gollakota, Akhila, Savage, Phillip E., 2018. Hydrothermal liquefaction of model food waste biomolecules and ternary mixtures under isothermal and fast conditions. ACS Sustain. Chem. Eng. 6 (7), 9018–9027.

González, Inmaculada, et al., 2020. Wastewater nutrient recovery using twin-layer microalgae technology for biofertilizer production. Water Sci. Technol. 82 (6), 1044–1061.

González-Camejo, J., et al., 2019. Effect of light intensity, light duration and photoperiods in the performance of an outdoor photobioreactor for urban wastewater treatment. Algal Res. 40 (December 2018), 101511. https://doi.org/10.1016/j.algal.2019.101511.

Gonzalez-Salgado, Irene, Guigui, Christelle, Sperandio, Mathieu, 2022. Transmembrane chemical absorption technology for ammonia recovery from wastewater: a critical review. Chem. Eng. J. 444, 1–49.

Gowd, Sarath C., Kumar, Deepak, Lin, Richen, Rajendran, Karthik, 2022. Nutrient recovery from wastewater in India: a perspective from mass and energy balance for a sustainable circular economy. Bioresour. Technol. Rep. 18 (April), 101079. https://doi.org/10.1016/j.biteb.2022.101079.

Guaya, Diana, et al., 2018. Valorisation of N and P from waste water by using natural reactive hybrid sorbents: nutrients (N,P,K) release evaluation in amended soils by dynamic experiments. Sci. Total Environ. 612, 728–738. https://doi.org/10.1016/j.scitotenv.2017.08.248.

Guaya, Diana, et al., 2020. Use of nutrient-enriched zeolite (NEZ) from urban wastewaters in amended soils: evaluation of plant availability of mineral elements. Sci. Total Environ. 727, 138646. https://doi.org/10.1016/j.scitotenv.2020.138646.

Guilayn, F., et al., 2020. 19 Reviews in Environmental Science and Biotechnology. In: Valorization of Digestates From Urban or Centralized Biogas Plants: A Critical Review. Springer Netherlands. https://doi.org/10.1007/s11157-020-09531-3.

Guisasola, Albert, et al., 2019. Long-term stability of an enhanced biological phosphorus removal system in a phosphorus recovery scenario. J. Clean. Prod. 214, 308–318.

Gupta, Renuka, Garg, V.K., 2009. Vermiremediation and nutrient recovery of non-recyclable paper waste employing *Eisenia fetida*. J. Hazard. Mater. 162 (1), 430–439. https://linkinghub.elsevier.com/retrieve/pii/S0304389408007425 (September 7, 2021).
Guruchandran, Srisowmeya, Amanullah, Muthahar Fazila, Muninathan, Chakravarthy, Ganesan, Nandhini Devi, 2022. Recent advances in biorefineries for energy and nutrient recovery from food waste. In: Handbook of Waste Biorefinery: Circular Economy of Renewable Energy, pp. 449–485.
Hanc, Ales, Pliva, Petr, 2013. Vermicomposting technology as a tool for nutrient recovery from kitchen bio-waste. J. Mater. Cycles Waste Manag. 15 (4), 431–439.
Hartmann, Tobias Edward, Möller, Kurt, Meyer, Carsten, Müller, Torsten, 2020. Partial replacement of rock phosphate by sewage sludge ash for the production of superphosphate fertilizers. J. Plant Nutr. Soil Sci. 183 (2), 233–237.
Heyman, Hannah, Bassuk, Nina, Bonhotal, Jean, Walter, Todd, 2019. Compost quality recommendations for remediating urban soils. Int. J. Environ. Res. Public 16, 3191. https://doi.org/10.3390/ijerph16173191.
Hospido, Almudena, Moreira, Ma Teresa, Fernández-Couto, Mercedes, Feijoo, Gumersindo, 2004. Environmental performance of a municipal wastewater treatment plant. Int. J. Life Cycle Assess. 9 (4), 261–271.
Hou, Pengfei, et al., 2021. Simultaneous removal of phosphorous and nitrogen by ammonium assimilation and aerobic denitrification of novel phosphate-accumulating organism *Pseudomonas chloritidismutans* K14. Bioresour. Technol. 340 (July), 125621. https://doi.org/10.1016/j.biortech.2021.125621.
Hube, Selina, et al., 2020. Direct membrane filtration for wastewater treatment and resource recovery: a review. Sci. Total Environ. 710.
Idowu, Ifeolu, et al., 2017. Hydrothermal carbonization of food waste for nutrient recovery and reuse. Waste Manag. 69, 480–491. https://doi.org/10.1016/j.wasman.2017.08.051.
Ijaz, Muhammad, et al., 2020. Eco-friendly alternatives to synthetic fertilizers for maximizing peanut (*Arachis hypogea* L.) production under arid regions in Punjab, Pakistan. J. Plant Nutr. 43 (5), 762–772. https://doi.org/10.1080/01904167.2019.1702203.
Ikeda, Hideo, Tan, Xuewen, 1998. Urea as an organic nitrogen source for hydroponically grown tomatoes in comparison with inorganic nitrogen sources. Soil Sci. Plant Nutr. 44 (4), 609–615.
IPCC, 2019. In: Shukla, P.R., Skea, J., Buendia, E. Calvo, Masson-Delmot, V. (Eds.), Climate Change and Land: An IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems, Research Handbook on Climate Change and Agricultural Law, pp. 423–448.
Jellali, Salah, et al., 2021. Sludge-derived biochars: a review on the influence of synthesis conditions on pollutants removal efficiency from wastewaters. Renew. Sust. Energ. Rev. 144 (June 2020), 111068. https://doi.org/10.1016/j.rser.2021.111068.
Jiang, Tao, et al., 2011. Effect of C/N ratio, aeration rate and moisture content on ammonia and greenhouse gas emission during the composting. J. Environ. Sci. 23 (10), 1754–1760. https://doi.org/10.1016/S1001-0742(10)60591-8.
Jiménez-Benítez, Antonio, Ferrer, Francisco Javier, Greses, Silvia, et al., 2020a. AnMBR, reclaimed water and fertigation: two case studies in Italy and Spain to assess economic and technological feasibility and CO2 emissions within the EU Innovation Deal Initiative. J. Clean. Prod. 270.
Jiménez-Benítez, Antonio, Ferrer, José, Rogalla, Frank, et al., 2020b. Energy and environmental impact of an anaerobic membrane bioreactor (AnMBR) demonstration plant treating urban wastewater. In: Current Developments in Biotechnology and Bioengineering: Advanced Membrane Separation Processes for Sustainable Water and Wastewater Management - Case Studies and Sustainability Analysis, pp. 289–310.
John, Miia, Scott Semken, R., Mikkola, Aki, Häkkinen, Antti, 2023. Natural freeze concentration of wastewater for nutrient recovery. Cold Reg. Sci. Technol. 213 (June), 103943.
Kacprzak, Malgorzata, et al., 2022. Biodegradable Waste Management in the Circular Economy. In: Biodegradable Waste Management in the Circular Economy. Wiley and Sons Ltd All.
Karbakhshravari, Mohsen, et al., 2020. Recovery of struvite from hydrothermally processed algal biomass cultivated in urban wastewaters. Resour. Conserv. Recycl. 163 (April), 105089. https://doi.org/10.1016/j.resconrec.2020.105089.
Kashima, Hiroyuki, 2020. In: Tojo, Seishu (Ed.), Nutrient Recovery From Wasted Biomass Using Microbial Electrochemical Technologies, Recycle Based Organic Agriculture in a City. Springer Nature Singapore Pte Ltd., pp. 69–90
Kaudal, Bhawana Bhatta, Weatherley, Anthony J., 2018. Agronomic effectiveness of urban biochar aged through co-composting with food waste. Waste Manag. 77, 87–97. https://doi.org/10.1016/j.wasman.2018.04.042.
Khandan, N., et al., 2020. 44 Lecture Notes in Civil Engineering. In: Sustainable Autarky of Food-energy-water (Safe-water).
Khoshnevisan, Benyamin, et al., 2018. Life Cycle Assessment of Different Strategies for Energy and Nutrient Recovery From Source Sorted Organic Fraction of Household Waste, 180(2018), pp. 360–374.
Kinidi, Lennevey, et al., 2018. Recent development in ammonia stripping process for industrial wastewater treatment. Int. J. Chem. Eng. 2018.
Kirchmann, Holger, Gunnar, Börjesson, Cohen, Yariv, Katterer, Thomas, 2017. From Agricultural Use of Sewage Sludge to Nutrient Extraction : A Soil Science Outlook, pp. 143–154.
Kjerstadius, H., Bernstad Saraiva, A., Spångberg, J., Davidsson, 2017. Carbon footprint of urban source separation for nutrient recovery. J. Environ. Manag. 197, 250–257. https://doi.org/10.1016/j.jenvman.2017.03.094.
de Kraker, Jeltsje, Kujawa-Roeleveld, Katarzyna, Villena, Marcelo J., Pabón-Pereira, Claudia, 2019. Decentralized valorization of residual flows as an alternative to the traditional urban waste management system: the case of Peñalolén in Santiago de Chile. Sustainability (Switzerland) 11 (22), 1–26.
Kranz, Christina N., et al., 2020. The Effects of Compost Incorporation on Soil Physical Properties in Urban Soils – A Concise Review, 261.
Kratz, S., Vogel, C., Adam, C., 2019. *Agronomic performance of P recycling fertilizers and methods to predict it: a review*. In: *Nutrient Cycling in Agroecosystems*, 115. Springer Netherlands.
Kumar, Nitin, Gupta, Sunil Kumar, Yadav, Brahmdeo, 2023. Optimisation of process parameters of a thermal digester for the rapid conversion of food waste into value-added soil conditioner. Waste Manag. Res. 1–17. https://doi.org/10.1177/0734242X231167078.
Kumar, Pushpendar, Samuchiwal, Saurabh, Malik, Anushree, 2020. Anaerobic digestion of textile industries wastes for biogas production. Biomass Convers. Biorefin. 10 (3), 715–724.
Kuntke, P., Schaetzle, O., Loos, K., 2016. Clinoptilolite-based Mixed Matrix Membranes for the Selective Recovery of Potassium and Ammonium, 90, pp. 62–70.
Lal, Rattan, 2020. Home Gardening and Urban Agriculture for Advancing Food and Nutritional Security in Response to the COVID-19 Pandemic, pp. 871–876.
Lehtoranta, Suvi, et al., 2022. Circular economy in wastewater management—the potential of source-separating sanitation in rural and peri-urban areas of Northern Finland and Sweden. Front. Environ. Sci. 10 (February), 1–18.
Li, Jiang Shan, et al., 2020. Current Developments in Biotechnology and Bioengineering: Resource Recovery from Wastes. In: Phosphorus (P) Recovery and Reuse as Fertilizer From Incinerated Sewage Sludge Ash (ISSA). Elsevier B.V. https://doi.org/10.1016/B978-0-444-64321-6.00014-8
Licon Bernal, E.E., Maya, C., Valderrama, C., Cortina, J.L., 2016. Valorization of ammonia concentrates from treated urban wastewater using liquid-liquid membrane contactors. Chem. Eng. J. 302, 641–649. https://doi.org/10.1016/j.cej.2016.05.094.
Liu, Bo, et al., 2022. Option of treatment scenario for food waste anaerobic digestion by-products based on comprehensive evaluation and scenario promotion benefits analysis. J. Clean. Prod. 374 (August), 133861. https://doi.org/10.1016/j.jclepro.2022.133861.
Liu, Yi, Villalba, Gara, Ayres, Robert U., Schroder, Hans, 2008. Global phosphorus flows and environmental impacts from a consumption perspective. J. Ind. Ecol. 12 (2), 229–247.
Liu, Ying Hao, et al., 2011. Eco-friendly production of maize using struvite recovered from swine wastewater as a sustainable fertilizer source. Asian Australas. J. Anim. Sci. 24 (12), 1699–1705.
Lohman, Hannah A.C., et al., 2020. Advancing sustainable sanitation and agriculture through investments in human-derived nutrient systems. Environ. Sci. Technol. 54 (15), 9217–9227.
Lorick, Dag, et al., 2020. Effectiveness of struvite precipitation and ammonia stripping for recovery of phosphorus and nitrogen from anaerobic digestate: a systematic review. Environ. Evid. 9 (1), 1–20. https://doi.org/10.1186/s13750-020-00211-x.
Macura, Biljana, et al., 2019. Effectiveness of ecotechnologies for recovery of nitrogen and phosphorus from anaerobic digestate and effectiveness of the recovery products as fertilisers: a systematic review protocol. Environ. Evid. 8 (1), 1–9. https://doi.org/10.1186/s13750-019-0173-3.
Magee, Kieran, Halstead, Joe, Small, Richard, Young, Iain, 2021. Valorisation of organicwaste by-products using black soldier fly (Hermetia Illucens) as a bio-convertor. Sustainability (Switzerland) 13 (15), 1–17.
Magwaza, Shirly Tentile, Magwaza, Lembe Samukelo, Odindo, Alfred Oduor, Mditshwa, Asanda, 2020. Hydroponic technology as decentralised system for domestic wastewater treatment and vegetable production in urban agriculture: a review. Sci. Total Environ. 698, 134154. https://doi.org/10.1016/j.scitotenv.2019.134154.
Malila, R., Lehtoranta, S., Viskari, E.L., 2019. The role of source separation in nutrient recovery – comparison of alternative wastewater treatment systems. J. Clean. Prod. 219, 350–358. https://doi.org/10.1016/j.jclepro.2019.02.024.
Manga, Musa, Evans, Barbara E., Ngasala, Tula M., Camargo-Valero, Miller A., 2022. Recycling of faecal sludge: nitrogen, carbon and organic matter transformation during co-composting of faecal sludge with different bulking agents. Int. J. Environ. Res. Public Health 19 (17).
Manríquez-Altamirano, Ana, Sierra-Pérez, Jorge, Muñoz, Pere, Gabarrell, Xavier, 2020. Analysis of urban agriculture solid waste in the frame of circular economy: case study of tomato crop in integrated rooftop greenhouse. Sci. Total Environ. 734, 139375.
Marinelli, Enrico, et al., 2021. Water–energy–food–climate nexus in an integrated peri-urban wastewater treatment and reuse system: from theory to practice. Sustainability (Switzerland) 13 (19).
Martin, Michael, Poulikidou, Sofia, Molin, Elvira, 2019. Exploring the environmental performance of urban symbiosis for vertical hydroponic farming. Sustainability (Switzerland) 11 (23), 10–12.
Matar, Gerald, et al., 2022. Modelling the benefits of urine source separation scenarios on wastewater treatment plants within an urban water basin. Water Sci. Technol. 86 (3), 482–495.
Mayor, Álvaro, et al., 2023. Life-cycle assessment and techno-economic evaluation of the value chain in nutrient recovery from wastewater treatment plants for agricultural application. Sci. Total Environ. 892, 164452. https://linkinghub.elsevier.com/retrieve/pii/S0048969723030735 (June 6, 2023).
Mehta, Chirag M., et al., 2015. Technologies to recover nutrients from waste streams: a critical review. Crit. Rev. Environ. Sci. Technol. 45 (4), 385–427.
Mennaa, Fatima Zahra, Arbib, Zouhayr, Perales, José Antonio, 2019. Urban wastewater photobiotreatment with microalgae in a continuously operated photobioreactor: growth, nutrient removal kinetics and biomass coagulation–flocculation. Environ. Technol. (U.K.) 40 (3), 342–355.

Mikula, Katarzyna, et al., 2023. Efficient anaerobic digestate valorization: nutrient recovery strategy for enhancing soil fertility in arid agricultural regions. J. Environ. Chem. Eng. 11 (5).
Milinković, Mira, et al., 2019. Biopotential of compost and compost products derived from horticultural waste—effect on plant growth and plant pathogens' suppression. Process Saf. Environ. Prot. 121, 299–306.
Moges, Melesse Eshetu, Heistad, Arve, Heidorn, Thorsten, 2020. Nutrient recovery from anaerobically treated blackwater and improving its effluent quality through microalgae biomass production. Water (Switzerland) 12 (2).
Mok, Wai Kit, Tan, Yong Xing, Chen, Wei Ning, 2020. Technology innovations for food security in Singapore: a case study of future food systems for an increasingly natural resource-scarce world. Trends Food Sci. Technol. 102 (March), 155–168.
Möller, Kurt, et al., 2018. Improved phosphorus recycling in organic farming: navigating between constraints. Adv. Agron. 147, 159–237.
de Morais, Etiele G., et al., 2022. Selenium recovery from wastewater by the green microalgae chlorella vulgaris and Scenedesmus sp. Sci. Total Environ. 851 (August), 158337. https://doi.org/10.1016/j.scitotenv.2022.158337.
Mortula, Md Maruf, et al., 2020. Sustainable management of organic wastes in Sharjah, UAE through co-composting. Methods Protoc. 3 (4), 1–12.
Mullen, Patrick, Venkiteshwaran, Kaushik, Zitomer, Daniel H., Mayer, Brooke K., 2019. Ion exchange nutrient recovery from anaerobic membrane bioreactor permeate. Water Environ. Res. 91 (7), 606–615.
Munasinghe-Arachchige, S.P., Abeysiriwardana-Arachchige, I.S.A., Delanka-Pedige, H. M.K., Nirmalakhandan, N., 2020. Algal pathway for nutrient recovery from urban sewage. Algal Res. 51, 102023. https://linkinghub.elsevier.com/retrieve/pii/S2211926420302162 (September 6, 2021).
Nadal, Ana, et al., 2017. Building-integrated rooftop greenhouses: an energy and environmental assessment in the Mediterranean context. Appl. Energy. 187, 338–351.
Nadal, Ana, et al., 2018. Rooftop greenhouses in educational centers: a sustainability assessment of urban agriculture in compact cities. Sci. Total Environ. 626, 1319–1331. https://doi.org/10.1016/j.scitotenv.2018.01.191.
Nagarajan, Dillirani, Lee, Duu Jong, Chen, Chun Yen, Chang, Jo Shu, 2020. Resource recovery from wastewaters using microalgae-based approaches: a circular bioeconomy perspective. Bioresour. Technol. 302 (January), 122817. https://doi.org/10.1016/j.biortech.2020.122817.
Naroznova, Irina, Møller, Jacob, Scheutz, Charlotte, 2016. Global warming potential of material fractions occurring in source-separated organic household waste treated by anaerobic digestion or incineration under different framework conditions. Waste Manag. 58, 397–407. https://doi.org/10.1016/j.wasman.2016.08.020.
Noriega-Hevia, G., et al., 2020. Nitrogen recovery using a membrane contactor: modelling nitrogen and PH evolution. J. Environ. Chem. Eng. 8 (4).
Oarga-Mulec, Andreea, Hanssen, Jon Fredrik, Jenssen, Petter D., Bulc, Tjaša Griessler, 2019. A comparison of various bulking materials as a supporting matrix in composting blackwater solids from vacuum toilets. J. Environ. Manag. 243 (April), 78–87. https://doi.org/10.1016/j.jenvman.2019.05.005.
Oliveira, Veronica, et al., 2021. A novel approach for nutrients recovery from municipal waste as biofertilizers by combining electrodialytic and gas permeable membrane technologies. Waste Manag. 125, 293–302. https://doi.org/10.1016/j.wasman.2021.02.055.
Ostace, George Simion, et al., 2013. Development and economic assessment of different WWTP control strategies for optimal simultaneous removal of carbon, nitrogen and phosphorus. Comput. Chem. Eng. 53, 164–177.
Palansooriya, Kumuduni Niroshika, et al., 2023. Converting food waste into soil amendments for improving soil sustainability and crop productivity: a review. Sci. Total Environ. 881 (January).
Panda, Sandeep, Mishra, Srabani, Akcil, Ata, Kucuker, Mehmet Ali, 2021. Microalgal potential for nutrient-energy-wastewater nexus: innovations, current trends and future directions. Energy Environ. 32 (4), 604–634.
Pattnaik, Swati, Reddy, M. Vikram, 2010. Nutrient status of vermicompost of urban green waste processed by three earthworm species— Eisenia Fetida, Eudrilus Eugeniae, and Perionyx Excavatus. Appl. Environ. Soil Sci. 2010, 1–13.
Pellejero, G., et al., 2020. Onion waste recycling by vermicomposting: nutrients recovery and agronomical assessment. Int. J. Environ. Sci. Technol. 17 (6), 3289–3296. https://doi.org/10.1007/s13762-020-02685-1.
Pérez, L.S., et al., 2012. Yield, nutritional quality, phenolic content and antioxidant capacity of hydroponic green fodder of greenhouse produced corn under organic fertilization [Re Ndimiento, Calidad Nutricional, Contenido Fenólico y Capacidad Antioxidante De Forraje Verde Hidro]. Interciencia 37 (3), 215–220. https://www.scopus.com/inward/record.uri?eid=2-s2.0-84860360765&partnerID=40&md5=8ca42a098dbdb99b0e8edb9dbbef28c8.
Pimentel-Rodrigues, Carla, Siva-Afonso, Armando, 2019. Reuse of resources in the use phase of buildings. Solutions for water. IOP Conf. Ser.: Earth Environ. Sci. 225 (1).
Pleissner, Daniel, Lau, Kin Yan, Lin, Carol Sze Ki, 2017. Utilization of food waste in continuous flow cultures of the heterotrophic microalga Chlorella pyrenoidosa for saturated and unsaturated fatty acids production. J. Clean. Prod. 142, 1417–1424. https://doi.org/10.1016/j.jclepro.2016.11.165.
Podder, Aditi, Reinhart, Debra, Goel, Ramesh, 2020. Integrated leachate management approach incorporating nutrient recovery and removal. Waste Manag. 102, 420–431. https://doi.org/10.1016/j.wasman.2019.10.048.
Poh, Phiak Kim, et al., 2021. Tropical-based EBPR process: the long-term stability, microbial community and its response towards temperature stress. Water Environ. Res. 93 (11), 2598–2608.
Preciado-Rangel, P., et al., 2015. Nutraceutical quality of cantaloupe melon fruits produced under fertilization with organic nutrient solutions [Calidad Nutracéutica de Melon Cantaloupe Producido Bajo Fertilización Con Soluciones Nutritivas Orgánicas]. Cienc. Investig. Agrar. 42 (3), 475–481. https://www.scopus.com/inward/record.uri?eid=2-s2.0-84981738097&doi=10.4067%2FS0718-16202015000300015&partnerID=40&md5=76b80ff0107f12734fd296ceca852241.
Rana, Shubham, Kumar, Amit, Arora, Himanshu, 2022. Nitrogen flow through waste management systems in an academic campus. Mater. Today: Proc. 57, 1938–1943. https://doi.org/10.1016/j.matpr.2022.03.046.
Rao, Unnati, et al., 2018. Coupling hydrothermal liquefaction and membrane distillation to treat anaerobic digestate from food and dairy farm waste. Bioresour. Technol. 267 (July), 408–415.
Rashid, Naim, Selvaratnam, Thinesh, Park, Won Kun, 2019. Microalgae Cultivation for Biofuels Production. In: Resource Recovery From Waste Streams Using Microalgae: Opportunities and Threats. Elsevier Inc. https://doi.org/10.1016/B978-0-12-817536-1.00021-7
Ravindran, Balasubramani, et al., 2021. Cleaner production of agriculturally valuable benignant materials from industry generated bio-wastes: a review. Bioresour. Technol. 320 (PA), 124281. https://doi.org/10.1016/j.biortech.2020.124281.
Reig, M., et al., 2021. Study of the operational parameters in the hollow fibre liquid-liquid membrane contactors process for ammonia valorisation as liquid fertiliser. Sep. Purif. Technol. 255 (September 2020), 117768. https://doi.org/10.1016/j.seppur.2020.117768.
Reilly, Matthew, et al., 2021. Electrocoagulation of food waste digestate and the suitability of recovered solids for application to agricultural land. J. Water Process Eng. 42 (April), 102121. https://doi.org/10.1016/j.jwpe.2021.102121.
Ren, Ai Tian, et al., 2020. Nutrient recovery from anaerobic digestion of food waste: impacts of digestate on plant growth and rhizosphere bacterial community composition and potential function in ryegrass. Biol. Fertil. Soils 56 (7), 973–989.
Robles, Ángel, et al., 2020. Performance of a membrane-coupled high-rate algal pond for urban wastewater treatment at demonstration scale. Bioresour. Technol. 301 (December 2019), 122672. https://doi.org/10.1016/j.biortech.2019.122672.
Rodrigues, Denis Manuel, et al., 2019. Recovery of phosphates as struvite from urine-diverting toilets: optimization of PH, Mg:PO4 ratio and contact time to improve precipitation yield and crystal morphology. Water Sci. Technol. 80 (7), 1276–1286.
Rojo, Elena, et al., 2021. Biomass, Biofuels, Biochemicals: Circular Bioeconomy-Current Developments and Future Outlook. In: Environment and Material Science Technology for Anaerobic Digestion-based Circular Bioeconomy. Elsevier Inc. https://doi.org/10.1016/B978-0-12-821878-5.00024-6
Rongwong, Wichitpan, Goh, Kunli, 2020. Resource recovery from industrial wastewaters by hydrophobic membrane contactors: a review. J. Environ. Chem. Eng. 8 (5), 104242. https://doi.org/10.1016/j.jece.2020.104242.
Rufí-Salís, Martí, Brunnhofer, Nadin, et al., 2020a. Can wastewater feed cities? Determining the feasibility and environmental burdens of struvite recovery and reuse for urban regions. Sci. Total Environ. 737, 139783. https://doi.org/10.1016/j.scitotenv.2020.139783.
Rufí-Salís, Martí, Parada, Felipe, et al., 2020b. Closed-loop crop cascade to optimize nutrient flows and grow low-impact vegetables in cities. Front. Plant Sci. 11 https://doi.org/10.3389/fpls.2020.596550/full.
Rufí-Salís, Martí, Calvo, Milena J., et al., 2020c. Exploring nutrient recovery from hydroponics in urban agriculture: an environmental assessment. Resour. Conserv. Recycl. 155 (November 2019), 104683. https://doi.org/10.1016/j.resconrec.2020.104683.
Samarina, Tatiana, Takaluoma, Esther, 2020. Ammonia recovery from wastewaters by adsorption with subsequent transmembrane chemical absorption. Proc. World Congr. New Technol. https://doi.org/10.11159/icepr20.146, 146-1-146–2. 146-1-146–2.
Samorì, Giulia, Samorì, Chiara, Guerrini, Franca, Pistocchi, Rossella, 2013. Growth and nitrogen removal capacity of *Desmodesmus communis* and of a natural microalgae consortium in a batch culture system in view of urban wastewater treatment: part I. Water Res. 47 (2), 791–801.
Sánchez, Antonio Santos, 2020. Technical and economic feasibility of phosphorus recovery from wastewater in São Paulo's Metropolitan Region. J. Water Process Eng. 38 (July), 101537. https://doi.org/10.1016/j.jwpe.2020.101537.
Sánchez, Antonio Santos, Martins, Gesaína, 2021. Nutrient recovery in wastewater treatment plants: comparative assessment of different technological options for the metropolitan region of Buenos Aires. J. Water Process Eng. 41 (April).
Sánchez-Zurano, Ana, et al., 2021. Annual assessment of the wastewater treatment capacity of the microalga *Scenedesmus almeriensis* and optimisation of operational conditions. Sci. Rep. 11 (1), 1–11. https://doi.org/10.1038/s41598-021-01163-z.
Sancho, I., et al., 2017. Recovery of ammonia from domestic wastewater effluents as liquid fertilizers by integration of natural zeolites and hollow fibre membrane contactors. Sci. Total Environ. 584–585, 244–251. https://doi.org/10.1016/j.scitotenv.2017.01.123.
Sanjuan-Delmás, David, et al., 2018. Environmental assessment of an integrated rooftop greenhouse for food production in cities. J. Clean. Prod. 177, 326–337.
Santiago-López, G., et al., 2016. Organic nutrient solutions in production and antioxidant capacity of cucumber fruits. Emir. J. Food Agric. 28 (7), 518–521. https://www.scopus.com/inward/record.uri?eid=2-s2.0-84965064194&doi=10.9755%2Fejfa.2016-01-083&partnerID=40&md5=91e780f03566e820edddda2f515298e4.
Sanyé Mengual, Esther, 2015. Sustainability Assessment of Urban Rooftop Farming Using an Interdisciplinary Approach. *TDX (Tesis Doctorals en Xarxa)*. http://www.tesisenred.net/handle/10803/308336 (February 19, 2019).
Sanyé-Mengual, Esther, Oliver-Solà, Jordi, Montero, Juan Ignacio, Rieradevall, Joan, 2015. An environmental and economic life cycle assessment of rooftop greenhouse (RTG) implementation in Barcelona, Spain. Assessing new forms of urban agriculture from the greenhouse structure to the final product level. Int. J. Life Cycle Assess. 20 (3), 350–366.

Sanyé-Mengual, Esther, et al., 2018. Urban horticulture in retail parks: environmental assessment of the potential implementation of rooftop greenhouses in European and South American cities. J. Clean. Prod. 172, 3081–3091. https://www.sciencedirect.com/science/article/pii/S0959652617327762?via%3Dihub (February 19, 2019).

Sarrion, A., Elena, Diaz, Angeles de la Rubia, M., Mohedano, Angel F., 2021. Fate of nutrients during hydrothermal treatment of food waste. Bioresour. Technol. 342, 125954. https://doi.org/10.1016/j.biortech.2021.125954.

Sarrion, A., Ipiales, R.P., et al., 2023a. Chicken meat and bone meal valorization by hydrothermal treatment and anaerobic digestion: biofuel production and nutrient recovery. Renew. Energy 204 (January), 652–660. https://doi.org/10.1016/j.renene.2023.01.005.

Sarrion, A., Medina-Martos, Enrique, et al., 2023b. Life cycle assessment of a novel strategy based on hydrothermal carbonization for nutrient and energy recovery from food waste. Sci. Total Environ. 878 (March), 163104. https://doi.org/10.1016/j.scitotenv.2023.163104.

Sarvajith, M., Nancharaiah, Y.V., 2022. Enhancing biological nitrogen and phosphorus removal performance in aerobic granular sludge sequencing batch reactors by activated carbon particles. J. Environ. Manag. 303 (November 2021), 114134. https://doi.org/10.1016/j.jenvman.2021.114134.

Schröder, Corinna, Häfner, Franziska, Larsen, Oliver Christopher, Krause, Ariane, 2021. Urban organic waste for urban farming: growing lettuce using vermicompost and thermophilic compost. Agronomy 11 (6).

Schwartz, Miles, Bassuk, Nina, Van Es, Harold, Rakow, Don, 2017. Long-term remediation of compacted urban soils by physical fracturing and incorporation of compost. Urban For. Urban Green. 24 (April), 149–156. https://doi.org/10.1016/j.ufug.2017.03.023.

Shaddel, Sina, et al., 2020. Struvite crystallization by using raw seawater: improving economics and environmental footprint while maintaining phosphorus recovery and product quality. Water Res. 173, 115572. https://doi.org/10.1016/j.watres.2020.115572.

Sheikh, M., et al., 2023. Ammonia recovery from municipal wastewater using hybrid NaOH closed-loop membrane contactor and ion exchange system. Chem. Eng. J. 465 (April), 142859. https://doi.org/10.1016/j.cej.2023.142859.

Shrestha, Paliza, Small, Gaston E., Kay, Adam, 2020. Quantifying nutrient recovery efficiency and loss from compost-based urban agriculture. PLoS ONE 1–15. https://doi.org/10.1371/journal.pone.0230996.

Sikora, Anna, Detman, Anna, Chojnacka, Aleksandra, Blaszczyk, Mieczyslaw K., 2017. Anaerobic digestion: I. A common process ensuring energy flow and the circulation of matter in ecosystems. II. A tool for the production of gaseous biofuels. Ferment. Process. 271–302. https://doi.org/10.5772/64645 (InTech, 2017).

Simha, Prithvi, Ganesapillai, Mahesh, 2017. Ecological sanitation and nutrient recovery from human urine: how far have we come? A review. Sustain. Environ. Res. 27 (3), 107–116. https://doi.org/10.1016/j.serj.2016.12.001.

Simha, Prithvi, Lalander, Cecilia, Vinnerås, Björn, Ganesapillai, M., 2017. Farmer attitudes and perceptions to the re–use of fertiliser products from resource–oriented sanitation systems – the case of Vellore, South India. Sci. Total Environ. 581–582, 885–896. https://doi.org/10.1016/j.scitotenv.2017.01.044.

Small, Gaston, et al., 2019. Excess Phosphorus From Compost Applications in Urban Gardens Creates Potential Pollution Hotspots Excess Phosphorus From Compost Applications in Urban Gardens Creates Potential Pollution Hotspots.

Smith, Stephen R., 2009. A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge. Environ. Int. 35 (1), 142–156. https://doi.org/10.1016/j.envint.2008.06.009.

Srivastava, Vaibhav, Vaish, Barkha, Singh, Anita, Singh, Rajeev Pratap, 2020. Urban Ecology. In: Nutrient Recovery From Municipal Waste Stream: Status and Prospects.

Stewart-Wade, Sally M., 2020. Efficacy of organic amendments used in containerized plant production: part 1 – compost-based amendments. Sci. Hortic. 266 (October 2019), 108856. https://doi.org/10.1016/j.scienta.2019.108856.

Sudibyo, Hanifrahmawan, Pecchi, Matteo, Tester, Jefferson William, 2022. Experimental-based mechanistic study and optimization of hydrothermal liquefaction of anaerobic digestates. Sustain. Energy Fuels 6 (9), 2314–2329.

Sun, Daquan, et al., 2018. Phosphorus recovery and reuse by pyrolysis: applications for agriculture and environment. Chemosphere 194, 682–691. https://linkinghub.elsevier.com/retrieve/pii/S0045653517320039 (September 7, 2021).

Suthar, Surendra, 2007. Vermicomposting potential of Perionyx sansibaricus (Perrier) in different waste materials. Bioresour. Technol. 98 (6), 1231–1237.

Suthar, Surindra, Gairola, Shailja, 2014. Nutrient recovery from urban forest leaf litter waste solids using Eisenia Fetida. Ecol. Eng. 71, 660–666. https://doi.org/10.1016/j.ecoleng.2014.08.010.

Tesfamariam, Eyob Habte, Cogger, Craig, Zvimba, John Ngoni, 2022. Impact of Treatment of Biodegradable Waste on Nutrient Recovery. In: Biodegradable Waste Management in the Circular Economy: Challenges and Opportunities, pp. 419–431.

Thomaier, Susanne, et al., 2015. Farming in and on urban buildings: present practice and specific novelties of zero-acreage farming (ZFarming). Renew. Agric. Food Syst. 30 (01), 43–54.

Tian, Xiujun, et al., 2019. Efficient nutrient recovery/removal from real source-separated urine by coupling vacuum thermal stripping with activated sludge processes. J. Clean. Prod. 220, 965–973.

Toboso-Chavero, Susana, et al., 2019. Towards productive cities: environmental assessment of the food-energy-water nexus of the urban roof mosaic. J. Ind. Ecol. 23 (4), 767–780.

Toboso-Chavero, Susana, Villalba, Gara, Durany, Xavier Gabarrell, Madrid-López, Cristina, 2021. More than the sum of the parts: system analysis of the usability of roofs in housing estates. J. Ind. Ecol. 25 (5), 1284–1299.

Tognetti, C., Laos, F., Mazzarino, M.J., Hernández, M.T., 2005. Composting vs. vermicomposting: a comparison of end product quality. Compost Sci. Util. 13 (1), 6–13.

Tomasi Morgano, Marco, et al., 2018. Screw pyrolysis technology for sewage sludge treatment. Waste Manag. 73, 487–495. https://doi.org/10.1016/j.wasman.2017.05.049.

Toop, Trisha A., et al., 2017. AgroCycle - developing a circular economy in agriculture. Energy Procedia 123, 76–80. https://doi.org/10.1016/j.egypro.2017.07.269.

Tuantet, Kanjana, et al., 2019. Optimization of algae production on urine. Algal Res. 44.

Ulm, Florian, et al., 2019. Sustainable Urban Agriculture Using Compost and an Open-pollinated Maize Variety, 212, pp. 622–629.

United Nations, 2019. 12 Demographic Research. In: World Urbanization Prospects The 2018 Revision.

Uysal, Ayla, et al., 2014. Optimization of struvite fertilizer formation from baker's yeast wastewater: growth and nutrition of maize and tomato plants. Environ. Sci. Pollut. Res. 21 (5), 3264–3274.

Van Ginkel, Steven W., Igou, Thomas, Chen, Yongsheng, 2017. Energy, water and nutrient impacts of California-grown vegetables compared to controlled environmental agriculture systems in Atlanta, GA. Resour. Conserv. Recycl. 122, 319–325. https://doi.org/10.1016/j.resconrec.2017.03.003.

Van Vuuren, D.P., Bouwman, A.F., Beusen, A.H.W., 2010. Phosphorus demand for the 1970-2100 period: a scenario analysis of resource depletion. Glob. Environ. Chang. 20 (3), 428–439. https://doi.org/10.1016/j.gloenvcha.2010.04.004.

Vaneeckhaute, Céline, et al., 2017. Nutrient recovery from digestate: systematic technology review and product classification. Waste Biomass Valoriz. 8 (1), 21–40.

Vecino, X., et al., 2019. Liquid fertilizer production by ammonia recovery from treated ammonia-rich regenerated streams using liquid-liquid membrane contactors. Chem. Eng. J. 360 (December 2018), 890–899. https://doi.org/10.1016/j.cej.2018.12.004.

Vecino, X., et al., 2020. Integration of liquid-liquid membrane contactors and electrodialysis for ammonium recovery and concentration as a liquid fertilizer. Chemosphere 245, 125606. https://doi.org/10.1016/j.chemosphere.2019.125606.

Vilanova, Ramon, Santín, Ignacio, Pedret, Carles, 2017. Control and operation of wastewater treatment plants (I). RIAI - Rev. Iberoam. Autom. Inform. Ind. 14 (3), 217–233.

Villalba, Gara, Liu, Yi, Schroder, Hans, Ayres, Robert U., 2008. Global phosphorus flows in the industrial economy from a production perspective. J. Ind. Ecol. 12 (4), 557–569.

Vögeli, Yvonne, et al., 2014. Sandec: Department of Water and Sanitation in Developing Countries. In: Anaerobic Digestion of Biowaste in Developing Countries. http://www.eawag.ch/forschung/sandec/publikationen/swm/dl/biowaste.pdf.

Volpin, F., et al., 2019. Optimisation of a forward osmosis and membrane distillation hybrid system for the treatment of source-separated urine. Sep. Purif. Technol. 212 (August 2018), 368–375.

Volpin, Federico, et al., 2018. Simultaneous phosphorous and nitrogen recovery from source-separated urine: a novel application for fertiliser drawn forward osmosis. Chemosphere 203, 482–489. https://doi.org/10.1016/j.chemosphere.2018.03.193.

Wang, Hui, et al., 2020. Phosphorus recovery from the liquid phase of anaerobic digestate using biochar derived from iron−rich sludge: a potential phosphorus fertilizer. Water Res. 174.

Wang, Wei, Lee, Duu Jong, 2021. Valorization of anaerobic digestion digestate: a prospect review. Bioresour. Technol. 323 (November 2020), 124626. https://doi.org/10.1016/j.biortech.2020.124626.

Weidner, Eckhard, Deerberg, Görge, Keuter, Volkmar, 2020. Biological Transformation. In: Biological Transformation, pp. 1–398.

Weidner, Till, Yang, Aidong, 2020. The potential of urban agriculture in combination with organic waste valorization: assessment of resource flows and emissions for two European cities. J. Clean. Prod. 244, 118490. https://doi.org/10.1016/j.jclepro.2019.118490.

Wielemaker, Rosanne, Oenema, Oene, Zeeman, Grietje, Weijma, Jan, 2019. Fertile cities : nutrient management practices in urban agriculture. Sci. Total Environ. 668, 1277–1288. https://doi.org/10.1016/j.scitotenv.2019.02.424.

Williams, Allen T., 2013. Ion Exchange Nutrient Recovery From Municipal Wastewater.

Wongkiew, Sumeth, et al., 2023. Effects of vermicompost leachate on nitrogen, phosphorus, and microbiome in a food waste bioponic system. J. Environ. Manag. 339 (April), 117860. https://doi.org/10.1016/j.jenvman.2023.117860.

Wu, Renming, et al., 2023. Combined hydrothermal and biological treatments for valorization of fruit and vegetable waste into liquid organic fertilizer. Environ. Res. 221 (January), 115262. https://doi.org/10.1016/j.envres.2023.115262.

Xia, Yunxue, Tang, Yuanyuan, Shih, Kaimin, Li, Bang, 2020. Enhanced phosphorus availability and heavy metal removal by chlorination during sewage sludge pyrolysis. J. Hazard. Mater. 382 (March 2019), 121110. https://doi.org/10.1016/j.jhazmat.2019.121110.

Yang, Linyan, et al., 2015. Application of hydroponic systems for the treatment of source-separated human urine. Ecol. Eng. 81, 182–191. https://doi.org/10.1016/j.ecoleng.2015.04.013.

Yao, Song, et al., 2017. On-site nutrient recovery and removal from source-separated urine by phosphorus precipitation and short-cut nitrification-denitrification. Chemosphere 175, 210–218. https://doi.org/10.1016/j.chemosphere.2017.02.062.

You, Xialei, Valderrama, César, Cortina, José Luis, 2019. Nutrients recovery from treated secondary mainstream in an urban wastewater treatment plant: a financial assessment case study. Sci. Total Environ. 656, 902–909. https://doi.org/10.1016/j.scitotenv.2018.11.420.

Yulistyorini, Anie, 2017. A mini review on the integration of resource recovery from wastewater into sustainability of the green building through phycoremediation. AIP Conf. Proc. 1887 (2017).

Zabaleta, Imanol, Rodic, Ljiljana, 2015. Recovery of essential nutrients from municipal solid waste - impact of waste management infrastructure and governance aspects. Waste Manag. 44, 178–187. https://doi.org/10.1016/j.wasman.2015.07.033.

Zabaniotou, Anastasia, Stamou, Katerina, 2020. Balancing Waste and Nutrient Flows Between Urban Agglomerations and Rural Ecosystems: Biochar for Improving Crop Growth and Urban Air Quality in the Mediterranean Region, pp. 1–32.

Zambrano-Prado, Perla, Muñoz-Liesa, Joan, et al., 2021a. Assessment of the food-water-energy nexus suitability of rooftops. a methodological remote sensing approach in an urban mediterranean area. Sustain. Cities Soc. 75.

Zambrano-Prado, Perla, Pons-Gumí, David, et al., 2021b. Perceptions on barriers and opportunities for integrating urban agri-green roofs: a European Mediterranean compact city case. Cities 114, 103196. https://doi.org/10.1016/j.cities.2021.103196.

Zambrano-Prado, Perla, Orsini, Francesco, et al., 2021c. Potential key factors, policies, and barriers for rooftop agriculture in EU cities: Barcelona, Berlin, Bologna, and Paris. Front. Sustain. Food Syst. 5 (September).

Zangeneh, Arezoo, et al., 2021. Ammonia removal form municipal wastewater by air stripping process: an experimental study. S. Afr. J. Chem. Eng. 36 (October 2020), 134–141. https://doi.org/10.1016/j.sajce.2021.03.001.

Zhang, Shanshan, et al., 2014. Urban nutrient recovery from fresh human urine through cultivation of Chlorella sorokiniana. J. Environ. Manag. 145, 129–136. https://doi.org/10.1016/j.jenvman.2014.06.013.

Zhao, Guang Wei, 2014. Decentralised sanitation and energy reuse in the built environment. Appl. Mech. Mater. 511–512, 787–790.

Zheng, Nai Yun, Lee, Mengshan, Lin, Yi Li, Samannan, Bharath, 2020. Microwave-assisted wet co-torrefaction of food sludge and lignocellulose biowaste for biochar production and nutrient recovery. Process Saf. Environ. Prot. 144, 273–283. https://doi.org/10.1016/j.psep.2020.07.032.

Zhou, Yifan, et al., 2023. A comprehensive review on wastewater nitrogen removal and its recovery processes. Int. J. Environ. Res. Public Health 20 (4).

Zhou, Yuxiao, et al., 2022. Environmental impacts and nutrient distribution routes for food waste separated disposal on large-scale anaerobic digestion/composting plants. J. Environ. Manag. 318 (June), 115624. https://doi.org/10.1016/j.jenvman.2022.115624.

Zhu, Baojun, Wei, Dong, Pohnert, Georg, 2022. The thermoacidophilic red alga Galdieria sulphuraria is a highly efficient cell factory for ammonium recovery from ultrahigh-NH4+ industrial effluent with co-production of high-protein biomass by photo-fermentation. Chem. Eng. J. 438 (February), 135598. https://doi.org/10.1016/j.cej.2022.135598.