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The use of municipal solid waste in peri-urban and urban agriculture

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Research Paper

Closing the nutrient cycle in urban areas: The use of municipal solid waste in *peri*-urban and urban agriculture

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ABSTRACT

Cities face the challenges of supplying food and managing organic municipal solid waste (OMSW) sustainably amid increasing urbanization rates. Urban agriculture (UA) can help with this effort by producing local crops that are fertilized with nutrients recovered from compost generated from OMSW. This research aims to determine the potential of OMSW compost to supply the nitrogen-phosphorus-potassium (NPK) demand of UA and the environmental benefits of replacing mineral fertilizer from a life cycle perspective. The Metropolitan Area of Barcelona (AMB) serves as the case study given its commitment to reuse biowaste according to the Revised Waste Framework Directive and to promote UA as a signing member of the Milan Urban Food Policy Pact. Based on crop requirements and farmer surveys, we find that the annual NPK demands of the agricultural fields of the AMB that cover 5,500 ha and produce 70,000 tons of crops are approximately 769, 113, and 592 tons of NPK, respectively. Spatial material flow analysis and life cycle assessment were applied to found that the current waste management system can potentially substitute 8 % of the total NPK demanded by UA with compost, reduce the impacts by up to 39 % and yield savings in global warming of 130 %. The more ambitious future scenario of 2025 can potentially substitute 21 % of the total NPK demand and reduce environmental impacts up to 1,049 %, depending on the category considered. Avoiding processing of mixed OMSW, mineral fertilizer replacement and cogeneration of electricity from biogas are the major contributors to these environmental savings.

1. Introduction

Urbanization is progressing rapidly, as more than half of the world population will be living in cities by 2050 (UN, 2015). Urban areas are facing major challenges, such as a growing dependency on agricultural imports and low food security (FAO, 2002). Additionally, traditional agricultural systems and food logistics feeding cities worldwide have been determined to be major sources of environmental impacts (IPCC, 2014; Spiertz, 2010). In this context, agricultural provision systems located in or at the limits of urban areas, so-called *peri*-urban and urban agriculture (Smit et al., 2001) (hereafter referred to as UA for simplicity) has emerged as a practice that, when complemented with circularity strategies, can help cities reduce resource consumption and mitigate environmental impacts while aiming for sustainability (EEA, 2015; Orsini et al., 2020; Rufí-Salís et al., 2020b). UA, considered in this study as for horticultural production only, has been recently developed to improve self-sufficiency through local production and city planning (Inayatullah, 2011; Toboso-Chavero et al., 2019), boosts local economy (de Zeeuw, 2011), increases resilience of food supply chains (Mok et al., 2014), lessens pressure on agricultural land (Specht et al., 2014), and provides ecosystem services (Artmann and Sartison, 2018; Langemeyer et al., 2021).

UA can exploit local sources of fertilization given its proximity to nutrient-rich waste, such as bio-waste (biodegradable waste from green spaces, households, commercial establishments, etc. (EU, 2018, 2008)), referred to in this study as organic municipal solid waste (OMSW),

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reducing the extraction of nonrenewable phosphorus and the energy needed for inorganic mineral fertilizers (Rittmann et al., 2011; Sanjuan-Delmás et al., 2018; Zabaleta and Rodic, 2015). Concerns about urbanization and OMSW generation have led to the Revised Waste Framework Directive (EU, 2018, 2008), which obliges EU member states to collect it separately from other fractions for recycling and nutrient recovery processes by 2023 (EEA, 2020), prohibiting dumping in landfills. Compost resulting from OMSW recycling is nutrient rich (Chojnacka et al., 2019) and can be used as fertilizer in agricultural fields (Ayari et al., 2010). This use is favorable to the plant-soil system (Martínez-Blanco et al., 2013) and is characterized by long-known benefits such as increased organic matter content, water holding capacity, and improved structure (J. C. Hargreaves et al., 2008).

The use of OMSW as an alternative fertilizer for UA can provide environmental benefits (Martínez-Blanco et al., 2013), which have been analyzed using life cycle assessments (LCAs). LCAs enable meaningful comparisons among alternatives and help in the detection of critical activities (Hellweg and Milà i Canals, 2014). While many LCA studies explored the environmental impacts and trade-offs of using OMSW compost as an alternative fertilizer (Blengini, 2008; Boldrin et al., 2009; Hansen et al., 2006; Laurent et al., 2014; Martínez-Blanco et al., 2009; Quirós et al., 2015, 2014; Tonini et al., 2013), few integrated geographically-explicit material flow analysis (MFA) with LCA to spatially determine the potential benefits and tradeoffs of nutrient circularity through OMSW compost between waste management and urban agriculture.

In Moinard et al. (2021), the recycling of organic byproducts (including biowaste compost) was estimated to represent 14–85 % of the nutrient fertilizer demand of a *peri*-urban territory, depending on the nutrients. However, despite establishing a potential for nutrient circularity, no assessment of associated environmental burdens was conducted. In contrast, Sanjuan-Delmás et al. (2021) and Tonini et al. (2020) combined LCA with socioeconomic factors and found that anaerobic digestion with energy and nutrient recovery is a preferred alternative when improving the sustainability of OMSW systems, depending on the urban area. However, while these studies consider nutrient recovery potential and the sustainability of it in OMSW urban treatment, they do not assess the full application of these recovered nutrients for crop production while meeting specific spatial nutrient needs of UA.

In light of this research gap, the novelty of the present study is to use geographically explicit and primary data to determine the potential of and the environmental impacts and benefits of supplying nitrogen (N), phosphorus (P), and potassium (K) (from now on referred to as NPK) from OMSW compost to meet UA nutrient demand by plot and crop type, replacing mineral fertilizer while recycling waste. The Metropolitan Area of Barcelona (AMB) serves as the area of study for its aim to expand the total UA surface area from 8 % to 16 % through the Urban Master Plan (AMB, 2023), its commitment to develop sustainable urban food systems that promote local crop production through the Milan Urban Food Policy Pact (MUFPP, 2015), and the alignment of its metropolitan waste management program with the European Waste Framework Directive, requiring OMSW recycling and nutrient recovery, for which they aim to increase selective OMSW collection from 33 % to 55 % and adapt more treatment facilities for compost production by 2025.

The objective of this waste-to-fork study is to shed light on how local crop production and sustainability within a city can benefit from integrating two urban systems, waste and agriculture. To do so, we use spatial MFA and LCA to calculate the impacts and trade-offs associated with various potential scenarios for supplying the total yearly NPK demand of the 5,568 ha that is dedicated to UA in the AMB, combining both mineral fertilizer and OMSW compost. This study serves to establish strategic guidelines about the potential, limitations, and benefits of nutrient circularity in urban areas.

2. Case study: The Metropolitan Area of Barcelona

The area of study is the Metropolitan Area of Barcelona (AMB for its Catalan acronym), in the western Mediterranean region of Spain with an estimated population of 3.2 million and a population density of 5,093 inhabitants/km^2 (AMB, 2021). The AMB is regulated by the Metropolitan Program for the Prevention and Management of Municipal Resources and Waste 2019–2025 (PREMET25 for its acronym in Catalan) (AMB, 2019) and aims to achieve 55 % MSW valorization and carbon neutrality of the metropolitan waste system by 2025 (PREMET25 AMB, 2019a).

In 2016, 33 % (464,695 tons) of all generated MSW consisted of OMSW (PREMET25 AMB, 2019b), and only 33 % (154,555 tons) of this OMSW was selectively collected and of sufficient quality to be composted in composting plants (CP) or in mechanical biological treatment plants (MBT) that were adapted with a separate line using anaerobic digestion prior to composting (PREMET25 AMB, 2019c). There are six centralized MSW treatment facilities in the AMB, as shown in Fig. 1. Only CP1, CP2, and MBT1 were adapted to treat OMSW separately in 2016 and composted 51 % (78,800 tons) of the selectively collected OMSW for a total of 5,016 tons of compost (PREMET25 AMB, 2019c). See SI5 in supporting information (SI) for more details.

According to the AMB agricultural map, developed through the Integrated System Analysis of Urban Vegetation and Agriculture (URBAG) project (Mendoza Beltran et al., 2022), the AMB has a total of 5,568 ha of UA cropland (8 % of the total surface) and produces a total of 70,000 tons of cereals, fruits, and vegetables annually, which have been aggregated into ten categories, as shown in Fig. 1 (See SI1 and SI3 in the SI for more details). The reference year for this study is 2016, as the UA data used were developed from the latest land coverage map of Catalonia (2015) (Mendoza Beltran et al., 2022) and 2016 is the year of the primary OMSW information that was reported directly by the studied waste management facilities through questionnaires performed by the AMB and the PREMET25.

3. Methods

The first step in this study was to develop a spatially explicit material flow analysis to determine the nutrient demand of UA and the nutrient supply potential from OMSW compost based on current practices. The demand was determined by associating a crop NPK requirement to all the crop fields represented on the URBAG agricultural map of the AMB (Mendoza Beltran et al., 2022). The availability of nutrients in the compost was determined from compost characterization reports obtained from the OMSW treatment facilities through the AMB. Then, various scenarios of nutrient recovery for applications in agricultural fields were defined based on the current and future practices of the waste management system dictated by policy. Finally, a life cycle assessment was conducted to understand the environmental impacts and trade-offs of the scenarios.

3.1. Nutrient demand of urban agriculture

The URBAG AMB agricultural map was processed via geographic information system (GIS) ArcMap software, allowing the spatial aggregation of the 4,581 different agricultural polygons into 69 crop categories (López Bellido et al., 2010) and later into 10 different crop groups (Freixa et al., 2009; Mercadé and del Delgado, 2010). Calculations for nutrient demand were performed separately for each polygon and nutrient (e.g., N, P, and K) before aggregation by multiplying the area (ha) of each agricultural crop polygon by the fertilization rate for each nutrient available from different sources: for N, the fertilizer rates per area of each type of crop (kg N/ha) were extracted from (Mendoza Beltran et al., 2022); for P, the fertilizer rate in terms of kg P_2O_5/ha was only available for some crops (i.e. apple, vegetables, orchard, greenhouse crops, artichokes) from López Bellido et al. (2010). For the rest of

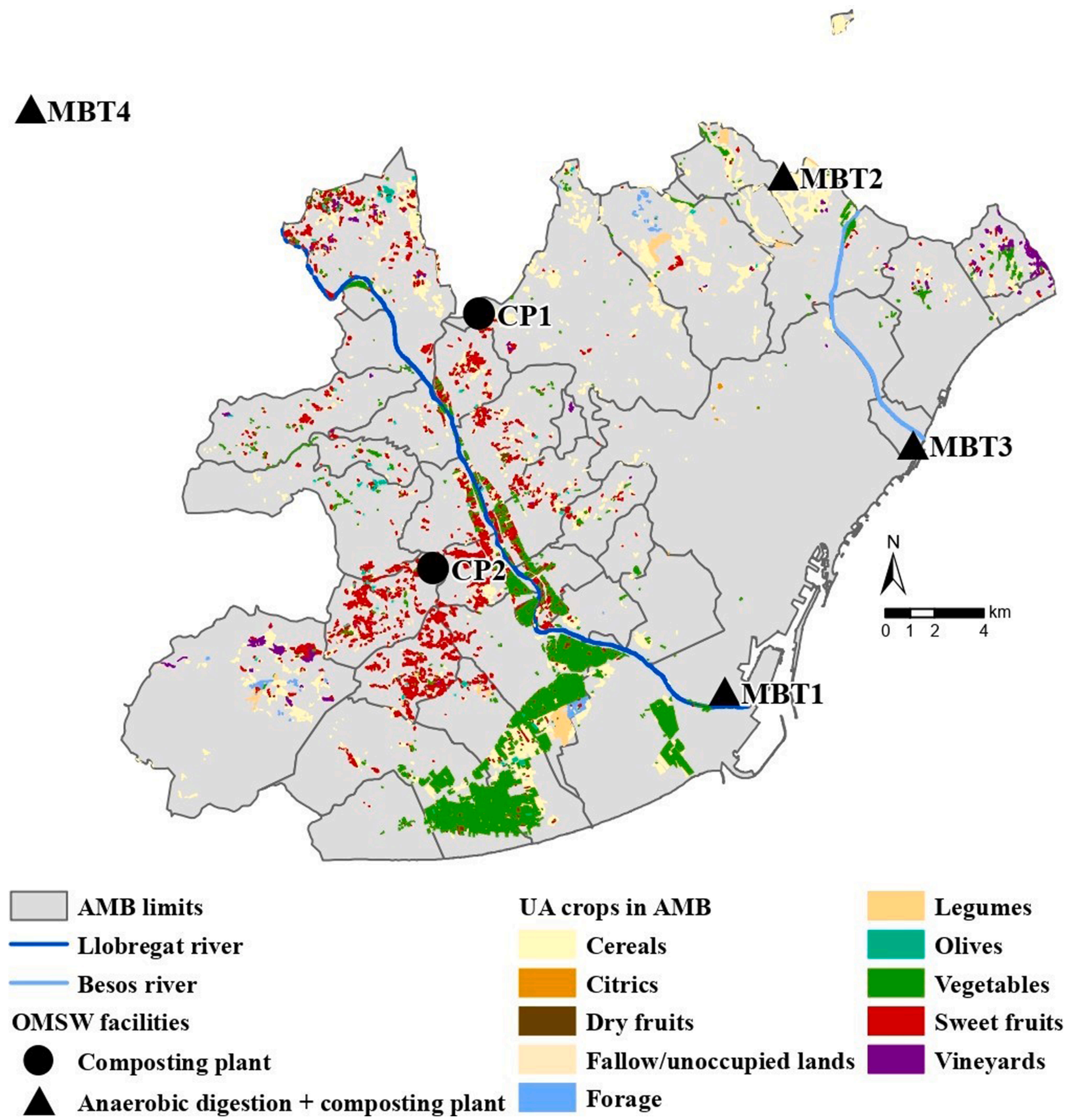

Fig. 1. AMB crops shown on the URBAG agricultural map (Mendoza Beltran et al., 2022) were aggregated based on (Freixa et al., 2009; Mercadé and del Delgado, 2010). CP, composting plant and MBT, mechanical biological treatment plant (anaerobic digestion + composting).

crops (such as non-citrics fruits, almond, apricot, barley, olives, etc.) we used P assimilation rates available from FAO (2013), López Bellido et al. (2010), and Ruralcat (2019), that reported kg P_2O_5 required per ton of crop. We then multiplied those P assimilation rates by the yields (ton of crop/ha) calculated from MAPA (2016) and the URBAG map (Mendoza Beltran et al., 2022) obtaining kg P_2O_5/ha for each crop. The conversion factor 1P = 2.29 P_2O_5 allowed us to put the rates in terms of P/ha; for K, the nutrient contents per area (kg K_2O/ha) were obtained from (FAO, 2017; López Bellido et al., 2010; Ruralcat, 2019) and the amounts of K in K_2O (using the conversion factor 1 K = 1.204 K_2O) (See SI1 in SI for detailed calculations).

3.2. Nutrient supply from organic municipal solid waste

The AMB provided monthly reports that characterized the compost produced by each OMSW facility, with content of dry matter (%), N Kjeldahl (% over dry weight), P (% over dry weight), and K (% over dry weight) (AMB, 2016a, 2016b, 2016c). The yearly amount of compost produced by each facility was multiplied by the annual average percentage of each component for each facility, resulting in the annual amounts of N, P, and K (in tons) that could be supplied from the OMSW compost from each facility. Afterward, a mineralization rate was applied for each nutrient, since these nutrients are not immediately available for plant uptake when supplied from OMSW compost and are released more slowly than mineral fertilizers due to several physicochemical properties (Hargreaves et al., 2008; Viaene et al., 2016). This rate can vary significantly (Amlinger et al., 2003; Moretti et al., 2020), particularly for N, and thus introduces a degree of sensitivity to the study that is later explored with a separate analysis in Section 3.4.4. For N, a mineralization rate of 40 % was used according to Catalonia-specific studies on compost production and its use in agriculture (Martínez-Blanco et al., 2013; Quirós et al., 2015). For P and K, a more generic rate of 95 % was used as in Tonini et al. (2013), based on their transformation to more readily available forms for plant uptake in the composting process (See SI6 of SI for calculations).

3.3. Nutrient supply scenarios for agricultural fields

Three fertilizer supply scenarios in the AMB were designed to represent current practices and future plans according to the metropolitan waste program (PREMET25) (AMB, 2019) (See SI8 and SI9 of SI for calculations). The scenarios are defined as follows:

In *baseline scenario (SC1)*, only mineral fertilizer is applied. The nutrient demand for UA was assumed to be supplied entirely by mineral fertilizers. Although compost is currently being produced in AMB waste management facilities, it is applied as a soil stabilizer or landfilled. In this scenario, compost is assumed to be all landfilled. SC1 is thus the case that most closely represents the current state according to various communications with farmers associations. *Scenario SC2* is a combination of applying both compost and mineral fertilizer. This scenario assumes that the current total of OMSW compost produced (5,016 tons) is applied to the urban agricultural fields of the AMB, substituting mineral fertilizer based on the OMSW selective collection rate (33 %), treatment capacity and compost production infrastructures of the reference year. Only facilities CP1, CP2 and MBT1 are considered here. MBT2 treated selectively collected OMSW in 2016 but was not adapted for its separate treatment and compost production. MBT3 and MBT4 did not treat or adapt to produce compost from OMSW. *Scenario SC3* is a combination of future compost production and mineral fertilizer. This scenario assumes that there will be more nutrient recovery due to the increased compost production projected for 2025 (14,194 tons) by the PREMET25 strategy (PREMET25 AMB, 2019a). This strategy includes an increased selective collection rate (from 33 % to 55 %) of the OMSW, increased treatment capacity in CP2, and the adaptation of the three remaining MBT facilities to produce compost (MBT2, MBT3, and MBT4) based on future OMSW generation estimates from PREMET25.

3.4. Life cycle assessment

The environmental assessment was conducted according to the ISO 14040 and 14,044 standards (ISO, 2006a, 2006b) for the LCA framework. An attributional LCA with a cradle-to-grave approach was adopted to quantify the life cycle impacts associated with the stages of the scenarios considered, from the production of raw materials (e.g., mineral fertilizer production or OMSW collection) to the disposal or recycling of waste (e.g., OMSW compost used as alternative fertilizer).

3.4.1. Goal and scope

The goal of the LCA is to determine the environmental impacts and trade-offs associated with meeting the nutrient demand of UA in the AMB with the NPK recovered from OMSW compost. Three different nutrient supply scenarios are assessed based on a simplification of the current mineral fertilization practices and their combination with compost from current and future production. The function of the system is to supply the total amounts of N, P, and K that are required by the UA of the AMB on a yearly basis. Hence, based on the calculations resulting from Section 3.1, the functional unit (FU) is defined as supplying 769, 113, and 592 tons of N, P, and K per year, respectively, for a total of 1,474 tons of NPK needed annually for UA of the AMB.

The system boundaries are illustrated by Fig. 2, which includes: OMSW selective collection, compost and fertilizer production, distribution, and application on UA fields. Electricity cogenerated from biogas substituting conventional production and the use of OMSW to produce compost substituting non-selectively mixed treatment is included.

3.4.2. Life cycle inventory

The life cycle inventory (LCI) data for foreground processes was collected from primary sources and the background processes were compiled from the Ecoinvent cutoff database version 3.9.1 (Wernet et al., 2016). Data sources are summarized as follows: (i) OMSW collection (Iriarte et al., 2009); (ii) compost production of each facility (AMB, 2016a, 2016b, 2016c), direct emissions to the air from composting (Colón et al., 2012; Møller et al., 2009), avoided mixed OMSW

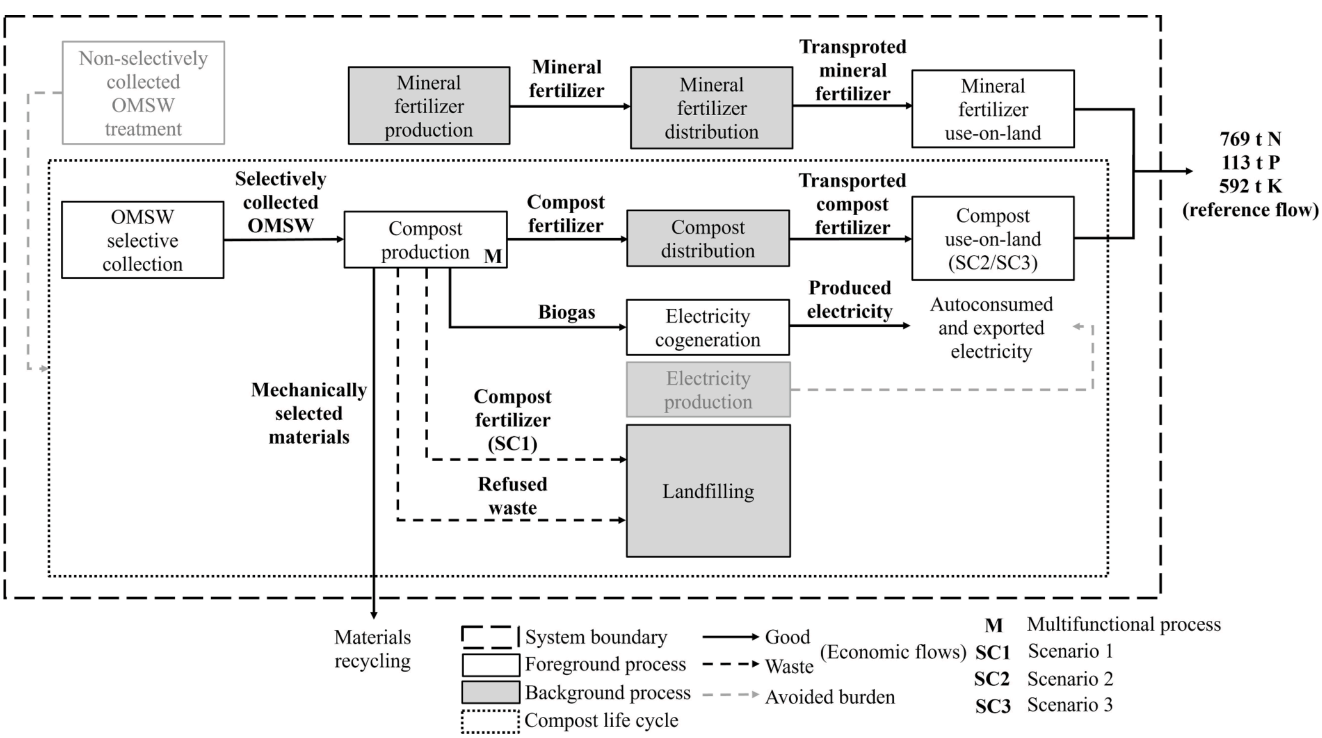

Fig. 2. The system boundary of the assessment is represented by the dotted line. The following stages are considered in the system: OMSW collection, compost and mineral fertilizer production, distribution, and use-on-land, landfilling of refused waste from compost production in all scenarios and of compost in SC1, and avoided burdens associated with cogeneration of electricity from biogas and processing of mixed OMSW. OMSW, organic municipal solid waste; N, nitrogen; P, phosphorus; and K, potassium.

treatment (AMB, 2016c; Kennedy et al., 2009), and fugitive emissions from biogas flaring (Møller et al., 2009); (iii) direct emissions to air and water from compost use-on-land (Bruun et al., 2006; Hansen et al., 2006); (iv) and direct and indirect emissions to air and water from mineral fertilizer use-on-land (Hergoualc'h et al., 2019; Nemecek et al., 2019, 2007), including geographically explicit emission factors (N_2O/ha·year) based on previous study conducted by the group (Mendoza Beltran et al., 2022). The remaining were compiled from Ecoinvent.

An extensive documentation of all calculations, estimations, and assumptions is presented in SI11-SI22 of SI to facilitate reproducibility and application in similar urban areas (see Section 5.1). The complete LCI is presented in SI23, referred to one ton of compost and mineral fertilizer supplied per year. A data quality assessment via a pedigree matrix is included following the recommendations of Guinée et al. (2002).

3.4.3. *Impact assessment*

The software used for the assessment calculations was SimaPro 9.5 (PRé Consultants, 2023) following the impact assessment method ReCiPe midpoint (H) version 1.08 (Huijbregts et al., 2016). The impact categories were selected based on relevance to compost (Colón et al., 2012; Sardarmehni et al., 2021), crop production (Rufí-Salís et al., 2020b; Sanjuan-Delmás et al., 2018) and authors' expertise. Global warming (GW – kg CO_2 eq) and terrestrial acidification (TA – kg SO_2 eq) for emissions from compost and mineral fertilizer production and use-on-land; freshwater eutrophication (FE – kg P eq) and marine eutrophication (ME – kg N eq) due to N and P depletion in agricultural systems; ecotoxicity (ET – kg 1,4-DCB) for persistence of emissions in the environment; and mineral resource scarcity (MRS – kg Cu eq) and fossil resource scarcity (FRS – kg oil eq) due to extraction of finite natural resources and energy-related impacts, respectively. Furthermore, GW, TA, FE, ME, and ET are highlighted as relevant to existing local fertilizer regulations limiting the concentration of nutrient inputs in soils (Real Decreto 1051, 2022).

3.4.4. *Interpretation: Sensitivity analysis*

OMSW compost is prone to release nutrients more slowly than mineral fertilizers, which introduces the need to address the levels of sensitivity associated with nutrient mineralization. N mineralization can vary significantly due to the nature of the organic matter it contains (Hargreaves et al., 2008), the microbial processes involved in its decomposition, and the stabilization of nutrients within it (Viaene et al., 2016). For this study, mineralization rates of 20 % and 60 % were selected as the minimum and maximum variations of the 40 % rate assumed by (Martínez-Blanco et al., 2013; Quirós et al., 2015) and were thus applied to the nutrient supply potential and impact assessment results to determine any relevant sensitivity effects.

4. Results

The 5,568 ha dedicated to UA in the AMB require an estimated total of 1,474 tons of NPK annually, which consist of 769, 113, and 592 tons of N, P, and K, respectively, as illustrated in the left panel of Fig. 3. Most of the NPK needed is used by cereals (42 %), vegetables (38 %), and sweet fruits (11 %) crops, followed by legumes (5 %) and the remaining crop groups (4 %). This is due to the extension (ha) of these crops within the UA and, in some cases, because of the nutrient requirement per area of each crop. For instance, legume crops (322 kg N/ha) have the highest

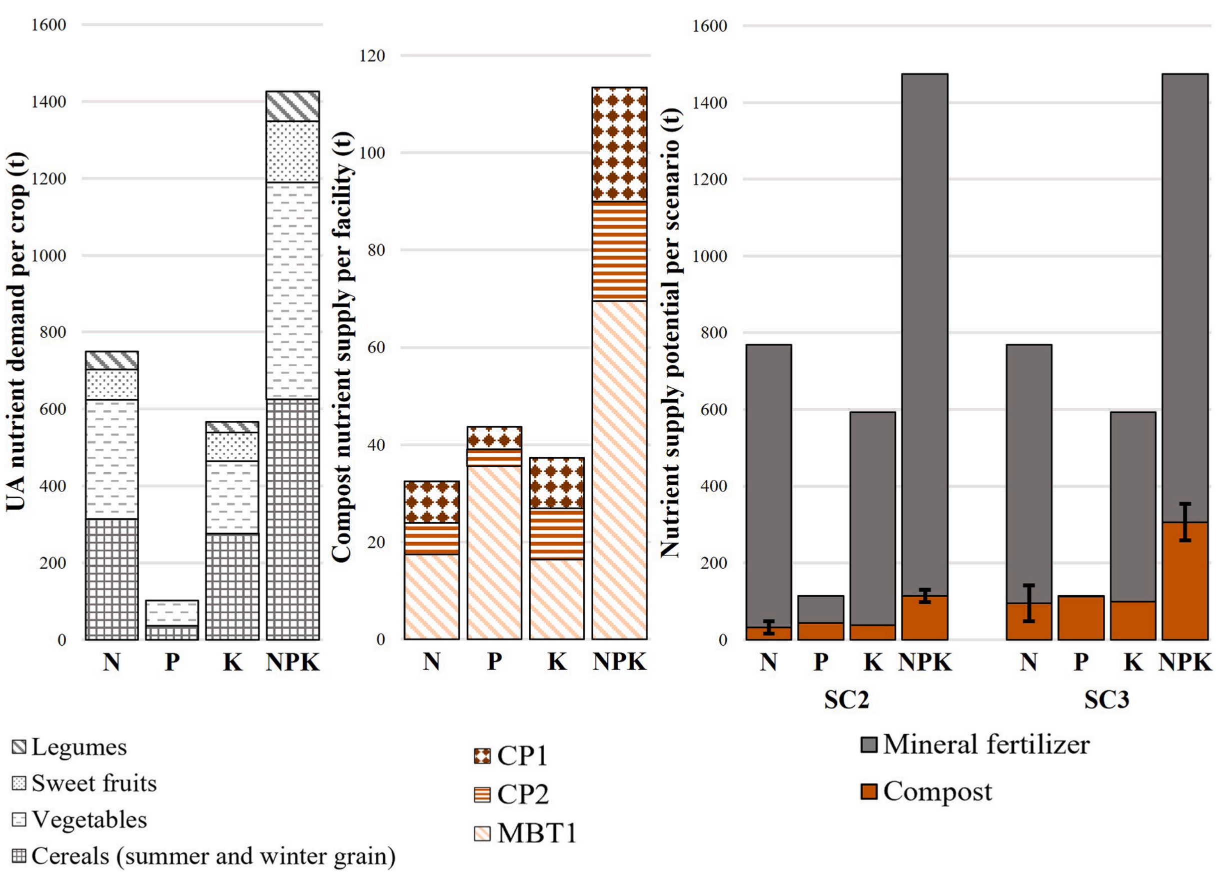

Fig. 3. UA NPK demand per crop group (left), compost NPK supply per facility (center), and potential NPK from compost to meet the UA demand per scenario (right). Nutrients: N, Nitrogen; P, Phosphorus; and K, Potassium. Scenarios: SC2, combination of current compost and mineral fertilizer; SC3, combination of future compost and mineral fertilizer. In the left panel, the crop groups that represent less than 5% of the demand are not displayed. In the right panel, the error bars represent the sensitivity of N mineralization between the alternative rates considered, namely 20% (lowest) and 40% (highest).

aggregated N requirements among the crop groups, followed by citrus fruits (197 kg N/ha) and vegetables (165 kg N/ha) crops. For the P requirements, vegetables (34 kg P/ha), legumes (32 kg P/ha), and forage crops (27 kg P/ha) are ranked the highest. For K, legume (196 kg K/ha) crops are followed by olive (149 kg K/ha) and cereal (141 kg K/ha) crops.

Nevertheless, cereals, vegetables, and sweet fruit crop groups amount to 91 % (5,085 ha) of all AMB UA areas (ha) and are therefore those that demand more NPK in this analysis. Moreover, N and K are mostly needed by cereals (41 and 47 %) and vegetables (40 and 32 %). Most of the P is needed by vegetable crops (57 %), which account for one-third (34 %) of the total surface area dedicated to UA.

Next, from the 5,106 tons of compost produced from OMSW yearly (2016) in the AMB, 32, 43, and 37 tons of N, P, and K, respectively, were estimated as potentially readily available for UA crop uptake annually, amounting to 113 tons of NPK, as shown in the middle panel of Fig. 3. The N supply is the lowest of all three, given its mineralization rate (40 %) compared to P and K (95 %), as described in Section 3.2. This counteracts N's higher content in OMSW compost compared to P and K, as seen in the characterization reports from each facility studied, which is consistent with typical nutrient contents in composts derived from food waste (Boldrin et al., 2009; Sardarmehni et al., 2021).

For the nutrient supply per facility, most of the NPK is provided by MBT1 (61 % – 69 tons), mainly due to the amount of compost produced by it (64 % – 3,212 tons), compared to CP1 (21 % – 1,075 tons) and CP2 (15 % – 728 tons). However, MBT1 is not the most efficient facility in doing so, as it receives the lowest-quality OMSW of all treatment plants and has a waste-to-compost ratio (OMSW needed to produce 1 ton of compost) of 23.0, compared to 7.3 and 4.0 for CP1 and CP2, respectively.

4.1. Potential of the nutrient supply from compost to meet the urban agricultural demand

The two scenarios SC2 and SC3 can potentially meet 8 % (113 total tons NPK) and 21 % (306 total tons NPK) of the UA nutrient demand, respectively, as illustrated in the right panel of Fig. 3. This potential nearly triples from SC2 to SC3 due to the increased compost production between these scenarios, from 5,016 tons (SC2) to 14,194 tons (SC3), a direct effect of PREMET25 (AMB, 2019) actions. First, by increasing the selective OMSW collection rate from 33 % (SC2) to 55 % (SC3), more of this organic fraction will be available and have high enough quality to be treated separately through composting. Moreover, the expansion of CP2 and adaptation of 3 new facilities (e.g., MBT2, MBT3, and MBT4) for compost production potentially increase the capacity of selectively treating collected OMSW from the 84,723 tons treated in 2016 (SC2) to an estimated 271,023 tons by 2025 (SC3) according to (PREMET25 AMB, 2019d); thus, more compost could be produced and would be available for potential nutrient recovery.

In both SC2 and SC3, P was the nutrient with the highest supply percentage of its total demand (38 % and 99 %, respectively), a finding to further consider given the reports of agricultural compost benefits associated with effective P recycling in response to mineral P scarcity (Martínez-Blanco et al., 2013).

4.2. Environmental impact assessment among scenarios

Not surprisingly, the life cycle impact assessment (LCIA) results suggest that meeting the UA demand with OMSW compost fertilizer (SC2 and SC3 scenarios) is environmentally preferable to the scenario that uses only mineral fertilizers (SC1) and has fewer impacts and even provides environmental savings in most of the categories considered. Overall, SC2 and SC3 resulted in reducing impacts by up to 95 % and in environmental savings of up to 1,049 % when compared to SC1, depending on the category and scenario considered. In these categories, the impacts decrease by between 18–3,064 % when compared to SC2 to SC3, mainly due to lesser amounts of mineral fertilizer being produced and used on land and the benefits associated with increased compost production (cogeneration of electricity from biogas and avoided mixed treatment of OMSW). SC1 had favorable results only in the TA category when compared to SC3, where it registered 25 % lower impacts. This is mainly due to the direct emission to air of NH_3 from compost production, specifically from CP2, where the kg of SO_2 eq are 6.8E + 01 per 1 ton of compost produced, compared to 1.8E + 00 and 1.3E + 01 from CP1 and MBT1, respectively. This is associated with CP2 being an open facility, thus lacking gas treatment, as reported by (Colón et al., 2012).

An analysis of impacts disaggregated by stages and sources indicates that most of the impacts in SC1 came from mineral fertilizer N production (e.g., MRS and FRS), mineral fertilizer use-on-land of N (e.g., TA and ME), and compost production in MBT1 (e.g., GW, FE, and ET). In SC2 and SC3, compost production in MBT facilities contributed the most in almost all categories, depending on the scenario. This is mainly due to the share of compost produced by this type of facility and the impacts associated with it, landfilled refused waste (e.g., GW, FE, and ME), direct air emissions to the air (e.g., TA), and electricity consumption (e.g., ET, MRS, and FRS). Nevertheless, these facilities (MBT) also contributed the most, with savings from avoided OMSW treatment in GW, TA, FE, ME, and ET and avoided electricity production in the remaining categories. The total impact assessment results per scenario are presented in Table 1 and a breakdown of the impacts and stage contributions to these is presented in Fig. 4. (See SI25 and SI26 of SI for partial LCIA results).

4.3. Environmental impact assessment by nutrients

The LCIA results were disaggregated by nutrients and their potential sources (compost or mineral fertilizer) to learn which alternative is more beneficial in terms of nutrient supply and how significant reducing mineral NPK consumption is in terms of depleting nonrenewable sources and eutrophication-related emissions (Rufí-Salís et al., 2020b).

To determine this, the impacts of supplying total OMSW compost from each facility were extrapolated to supply 1 ton of each nutrient, and the mineral fertilizer impacts were allocated to their generic markets for inorganic N, P, and K assumed in the inventory. The results of this analysis, illustrated in Fig. 5, indicate that nutrient recovery from MBT1 is favorable for all nutrients in the ET, MRS, and FRS categories, and only N and P in GW are associated with the avoided burdens of the compost production stage. In the eutrophication-related categories (e.g., FE and ME), this alternative falls behind CP1, mainly because of the amount of refused waste that results from the composting process, which ends up being treated in a landfill.

In contrast, CP2 is the least preferable alternative when compared to CP1, MBT1 and mineral fertilizer, associated with the lack of gaseous emissions treatment from its composting technique (turned windrow composting), as reported by (Colón et al., 2012).

Regarding this, TA is highlighted as a category where untreated NH_3 emissions from composting in CP2 significantly exceed the impacts of

Table 1
Life cycle impact assessment results per scenario. Scenarios: SC1, mineral fertilizer only; SC2, combination of current compost and mineral fertilizer; and SC3, combination of future compost and mineral fertilizer.

Impact category	Unit	Scenario		
		SC1	SC2	SC3
Global warming	kg CO2 eq	2.5E + 06	−7.5E + 05	−2.4E + 07
Terrestrial acidification	kg SO2 eq	1.5E + 05	1.5E + 05	1.9E + 05
Freshwater eutrophication	kg P eq	2.3E + 03	1.4E + 03	−2.9E + 03
Marine eutrophication	kg N eq	1.3E + 04	1.2E + 04	9.8E + 03
Ecotoxicity	kg 1,4-DCB	1.8E + 07	1.5E + 07	−5.6E + 07
Mineral resource scarcity	kg Cu eq	6.6E + 04	5.9E + 04	3.2E + 03
Fossil resource scarcity	kg oil eq	1.8E + 06	1.7E + 06	6.8E + 05

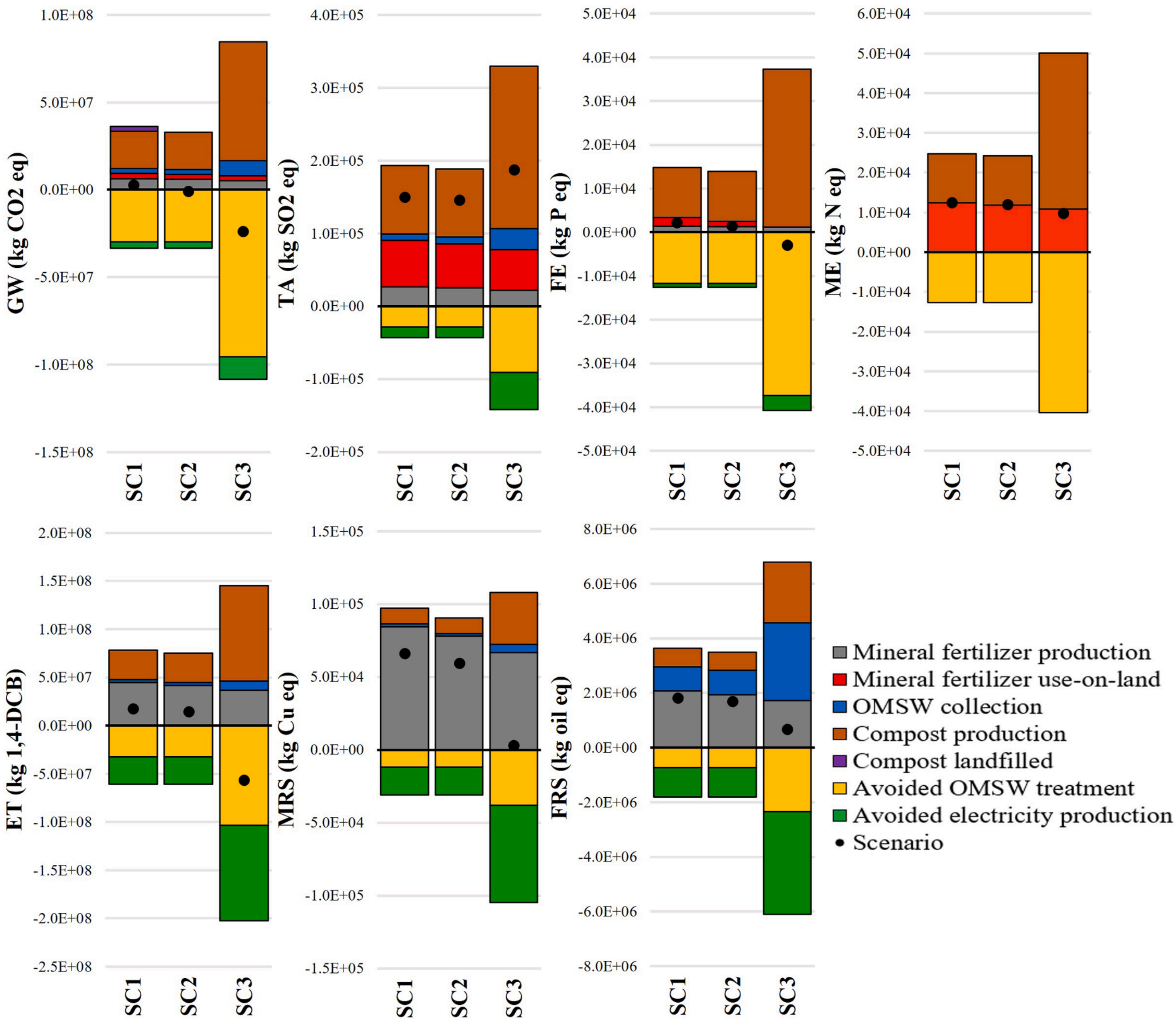

Fig. 4. Total and partial environmental impacts per category and scenario. Impact categories: GW, global warming; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; ET, ecotoxicity; MRS, mineral resource scarcity; and FRS, fossil resource scarcity. Scenarios: SC1, mineral fertilizer only; SC2, combination of current compost and mineral fertilizer; and SC3, combination of future compost and mineral fertilizer. Those stages that contribute less than 4% to the total impact are not displayed. The circles represent the total environmental impact of each scenario. Avoided burdens are represented by negative contributions to the total impact.

other supply sources. (See SI27 and SI28 of SI for complete LCIA per tonne of nutrient).

4.4. Sensitivity analysis: Nitrogen mineralization from compost

The sensitivity in N supply due to mineralization is represented by the error bars in the right-hand panel of Fig. 3. Depending on the N mineralization rate, the amount of N demanded by UA crops that could be met by OMSW compost ranges from a total of 16 to 48 tons in SC2 and from 47 to 141 tons in SC3. Overall, this sensitivity translates into a total NPK supply potential for UA as low as 7 % (in SC2 considering N mineralization of 20 %) and as high as 24 % (in SC3 considering N mineralization of 60 %). Consequently, this sensitivity is reflected in the environmental impacts, most significantly on MRS, GW, FRS, and ME, which can vary +/- 102 %, 20 %, 12 %, and 8 %, respectively, depending on the scenario considered.

The impacts are heavily influenced by the sensitivity introduced by the potential range of N mineralization rates. The production of N-based mineral fertilizers is characterized by energy-intensive processes that involve the combustion of fossil fuels and greenhouse gas (GHG) emissions (Krzyżaniak et al., 2019; Spinelli et al., 2013) and natural resource extraction, contributing to MRS, GW, and FRS. Moreover, the use-on-land of N mineral fertilizers can lead to N emissions when used in excess (Gong et al., 2022; Mendoza Beltran et al., 2022), further contributing to GW and ME as it is emitted to the air or leached to water bodies, respectively. See SI29 in SI for total and partial LCIA under mineralization rates.

5. Discussion

5.1. Nutrient recovery potential through compost

The potential nutrient recovery from compost, even in the most ambitious scenario, is still far from being able to substitute for mineral fertilizer. For the case of the AMB, the future waste management implementation of increasing selective waste practices as reflected by

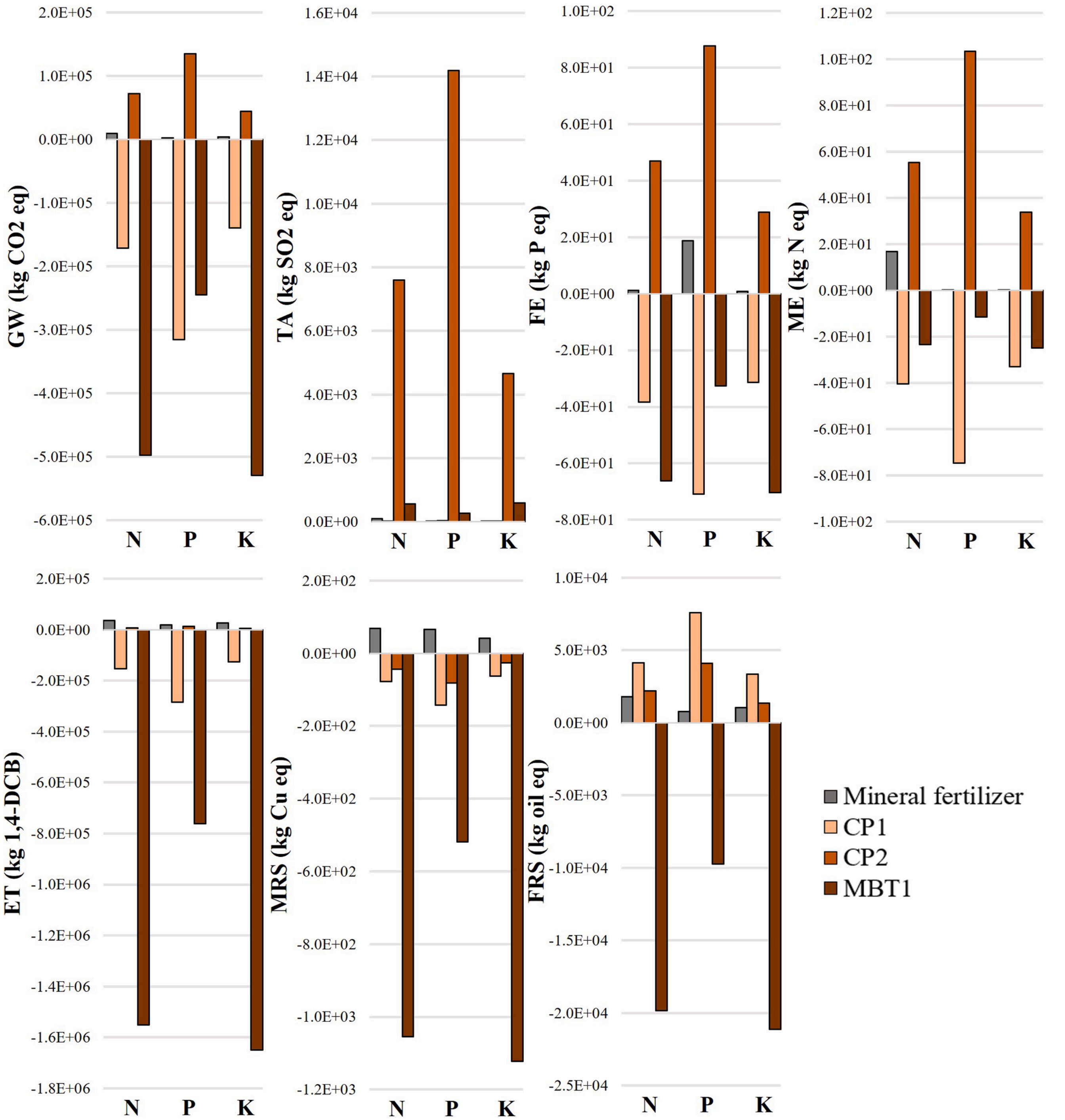

Fig. 5. Total environmental impact of supplying 1 ton of nutrient per category and nutrient. Impact categories: GW, global warming; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; ET, ecotoxicity; MRS, mineral resource scarcity; and FRS, fossil resource scarcity. Scenarios: SC1, mineral fertilizer only; SC2, combination of current compost and mineral fertilizer; and SC3, combination of future compost and mineral fertilizer. Avoided burdens are represented by negative contributions to the total impact.

SC3 is able to close the urban nutrient cycle by 21 %. While significant volumes of nutrient-rich waste are generated in urban areas, it was found that far more ambitious waste management strategies need to be in place for nutrient circularity from compost to significantly substitute mineral fertilizer in UA. The estimates indicate that only 42 % of the 168 kg of biowaste generated per person/yr in the EU is properly separated, 23 % of it is composted, less than 12 % is applied to agriculture, and the amount applied to UA is inconsequential (Colón et al., 2012; EEA, 2020; Kaza et al., 2018).

A study of the city of Vitoria-Gasteiz in Basque country, Spain, showed that the current waste management practices that promote nutrient circularity recovered only 3 % and 7 % of the potential N and P, respectively, collected from biowaste, which could reach 49 % and 83 % (Zabaleta and Rodic, 2015). Another study determined values of nutrient circularity meeting 0 %, 44 % and 50 % of the N, P, and K crop demands in the *peri*-urban territory of Versailles in Paris, France (Moinard et al., 2021). Comparable to our study, ambitious selective collection scenarios of OMSW for the city of Prague are able to reach a maximum circularity potential of 40 % for N fertilizer consumed in nearby agricultural areas (Guillaume et al., 2023). Similarly, for the city of Porto, Portugal, Weidner et al. (2020) estimated that under a theoretical full recovery scenario, compost from OMSW could meet 47 % of the regional agricultural demand for P.

There have also been some significant advances in nutrient recycling and circularity obtained by diverting nutrients from waste by composting and anaerobic digestion to enhance mineral fertilizer substitution in UA circularity (Arcas-Pilz et al., 2023). Papangelou et al. (2020) show that switching from current food waste incineration to an assumed scenario where all is valorized via anaerobic digestion and compost for use in agriculture in Brussels, Belgium, resulted in substituting for mineral fertilizer at levels close to twenty times more than had been done previously (in terms of P). Similarly, a best-case scenario of adopting composting and reuse of all biodegradable waste in Vienna, Austria, resulted in meeting all the N mineral fertilizer needs of agricultural land in its highly urbanized area (Kaltenegger et al., 2023). This indicates the viability of enhancing nutrient circularity in urban areas for the circularity of the city's food system.

5.2. Direct and indirect benefits of nutrient circularity

We find that the environmental benefits of fertilizing UA by applying compost are significant, as indicated by other studies, such as Tonini et al. (2019) and Trimmer and Guest (2018), even at low degrees of mineral fertilizer substitution, as shown by Fang et al. (2023). Using OMSW compost to replace mineral fertilizer was found to be environmentally preferable in all scenarios for the AMB, mostly due to avoiding the landfilling of nonselectively collected organic waste and refusing digestate, as reported by Colón et al. (2015), Martínez-Blanco et al. (2009), and Quirós et al. (2014). Avoiding the methane emissions from landfills provides the most important environmental benefit of nutrient circularity for the AMB.

Valorization of biogas from the MBT facility type was found to be the second most important environmental benefit. Not only is it advantageous to reduce biomethane emissions to the atmosphere, but biogas is also used to cogenerate electricity, thereby replacing conventional energy sources (Albizzati et al., 2021). A study including Hamburg, Germany, also found that centralized treatment of OMSW with biogas recovery was key for improving the sustainability of MSW management and substituted a high share of fossil fuels in that electricity mix and reduced waste incineration emissions (Sanjuan-Delmás et al., 2021). An important take-home message from our study is that biogas recovery is necessary for nutrient recovery through compost to be beneficial in terms of the environmental impacts. Even though compost characterization records indicate that AMB facility CP2 supplies the best quality compost (type A as per Real Decreto, 506 (2013)), the impact assessment reflected that this was the facility with the highest environmental impacts when supplying compost fertilizer, even when compared to supplying mineral fertilizer, because it lacks biogas emissions treatment. Thus, the amount of nutrient circularity is not sufficient; composting processes need to be optimized in terms of energy and biogas recovery.

The third most significant environmental benefit was a result of replacing mineral fertilizer, mainly by avoiding the energy-intense process of producing N-based mineral fertilizer and the eutrophication emissions associated with mineral N and P fertilizer use-on-land. As seen in Fig. 4 (Section 4), the impacts related to this are between 4–63 % less for the compost scenarios (e.g., SC2 and SC3) compared to using mineral fertilizer only (e.g., SC1), depending on the scenario and category considered. These are the greatest benefits after accounting for the savings associated with avoiding nonselectively collected OMSW and recovering biogas. This is similar to other studies that indicate the benefits of not producing mineral fertilizer and reducing eutrophication impacts are stressed as relevant (Rufí-Salís et al., 2020b, 2020a). Other studies have highlighted the environmental savings from avoiding the transportation of mineral fertilizers (Martínez-Blanco et al., 2013), such as avoiding the impacts from importing P-based fertilizer from Morocco, as shown by Rufí-Salís et al., (2020a).

Given the system boundary definition of this study, additional environmental benefits that are associated with favoring UA through nutrient circularity were not included but are worth mentioning. For instance, contributing to sustainable urban food systems through the provision of local resources reduces the impacts associated with the inefficiency of the long supply chains for imported crops (Mok et al., 2014), illustrating the importance of considering more factors in addition to food miles (Yang and Campbell, 2017).

5.3. Bottlenecks in nutrient circularity through compost

Although using compost as fertilizer is a long-standing practice worldwide, urban areas, which concentrate most of the nutrient-rich waste generated, were found to be far from fully exploiting this resource. Specifically, in the case of the AMB, we have identified significant untapped potential for compost utilization in UA. Our study showed that enough compost is currently being produced to substitute 8 % of the mineral fertilizer currently used by UA, yet it goes unused as such. This underutilization is primarily due to the mismatch between compost quality and composition and farmer needs (Case et al., 2017; Chen et al., 2020).

Overall, the low agronomic quality of OMSW compost produced across the facilities in Catalonia has been associated with a lack of concern by plant managers for commercializing compost as a suitable final product for soil (Huerta et al., 2008), indicating that these facilities prioritize waste treatment instead of providing a valorized product. Local regulations also play a part in this mismatch. In the AMB, applying the compost produced by the MBT type facility is not possible because the compost often has a high percentage of P and cannot be applied to urban agricultural soils that already contain excess P. Local fertilizer regulations limit the concentration of P in soils to 80–150 mg P/kg dry soil, depending on the applicable control (Murillo et al., 2020; Real Decreto 1051, 2022). This was also evidenced by our calculations, which showed that P is the nutrient we can recover the most from OMSW, yet it is least needed in terms of fertilizing needs.

Another constraint to nutrient circularity is the poor quality of urban agricultural soil. Over the past decades, there has been a shift from using biowaste to mineral fertilizers as well as tillage and other agricultural practices, which has deteriorated UA soils and depleted the organic carbon necessary for nutrient metabolism (Garcia-Pausas et al., 2017). The addition of organic or inorganic forms of nutrients to soils with low-quality organic matter has been harmful to the plant-soil system, as organic matter can be further mineralized in these soils, which reduces the available organic carbon and nutrient pools necessary to improve soil biochemical processes (Romanyà et al., 2012; Wander et al., 2007, 1994). Thus, regenerating UA soils and replenishing them with organic matter is an important previous step to promote nutrient circularity. Without healthy soils, crops will not be able to assimilate recovered nutrients. Without addressing these limitations, nutrient circularity in UA will be highly limited, which will hinder the potential synergy between waste and agricultural systems. Stakeholder engagement can help identify and overcome these limitations (Chen et al., 2020) by addressing factors such as poor compost information, lack of incentives, and reformulation of outdated policies that impede compost acceptance, adoption, and thus, nutrient circularity.

6. Conclusions

The objective of this study is to spatially determine the potential benefits and tradeoffs of nutrient circularity through OMSW compost in an urban area. The novelty lies in the application of a geographically explicit MFA, as well as a waste-to-fork LCA based on primary data and georeferenced emission factors, to assess various scenarios of nutrient recovery in line with waste management goals for the Metropolitan Area of Barcelona (AMB). The methods are presented and applied in transparent manner so that similar studies can be applied by other cities. We find that compost produced from half of the selectively collected OMSW could substitute 8 % of the total demand of NPK. Given current waste management infrastructures, the substitution rate could be increased to 21 % by reaching the selective OMSW collection rates and compost production infrastructure goals set by the AMB's waste management program by 2025. Although mineral fertilizer substitution might seem moderate, the environmental benefits of nutrient circularity are high in comparison. The scenarios that included nutrients recovered from OMSW compost resulted in lower impacts and savings of up to 95 % and 1,049 % for 8 % and 21 % levels of mineral fertilizer substitution, respectively, when compared to the application of mineral fertilizer only. The avoided burdens associated with preventing nonselective OMSW treatment and energy recovery from biogas in the compost production stage were the major contributors to reducing environmental impacts. A key message of this study is the need to identify local barriers to nutrient circularity, such as local regulations that limit compost use, the impoverished state of UA soils, and the acceptance of compost as an alternative fertilizer; otherwise, the beneficial synergies between waste and agricultural systems will not be fully exploited. We hope this study informs policymakers about the benefits of nutrient circularity by presenting the entire waste-to-crop life cycle to systemically quantify the benefits of nutrient flow coupling beyond waste management facilities and urban agriculture.

CRediT authorship contribution statement

Juan David Arosemena Polo: Writing – original draft, Visualization, Investigation, Formal analysis, Data curation, Conceptualization. **Susana Toboso-Chavero:** Writing – review & editing, Supervision, Investigation, Data curation, Conceptualization. **Biraj Adhikari:** Writing – original draft, Conceptualization. **Gara Villalba:** Writing – review & editing, Writing – original draft, Supervision, Software, Resources, Project administration, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Gara Villalba reports financial support was provided by European Research Council. Juan Arosemena reports financial support was provided by National Secretariat of Science, Technology and Research of Panama. If there are other authors, they 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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2024.05.009.

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