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Displaying geographic variability of peri-urban agriculture environmental impacts in the Metropolitan Area of Barcelona: A regionalized life cycle assessment

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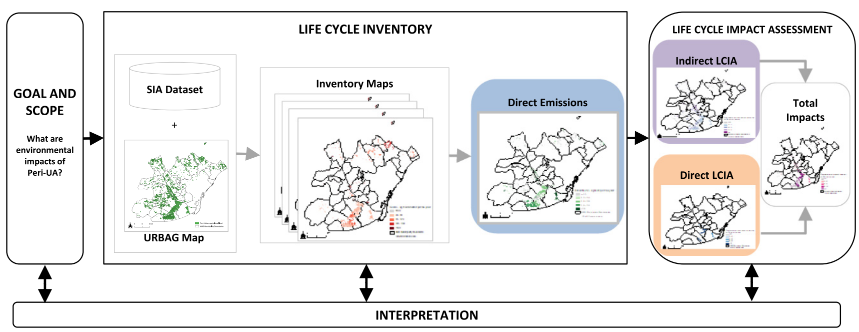
HIGHLIGHTS

- Regionalized LCA is used to assess metabolic flows and impacts of peri-UA.
- High-resolution inventory and eutrophication impact assessment data are computed.
- Geo-variability of peri-UA metabolic flows and impacts is displayed.
- Land uses determine direct/indirect impacts, their drivers and weight on the total.
- This knowledge is essential to guide suitable sustainability strategies for peri-UA.

GRAPHICAL ABSTRACT



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ABSTRACT

Peri urban agriculture (peri-UA) can supply food locally and potentially more sustainably than far-away conventional agricultural systems. It can also introduce significant environmental impacts depending on the local biophysical conditions and resources required to implement it and, on the crops managing practices, which could vary widely among growers. Sophisticated methods to account for such variability while assessing direct (on-site) and indirect (up/down stream) environmental impacts of peri-UA implementation are thus needed. We implemented an attributional, regionalized, cradle-to-gate life cycle assessment (LCA) for which we derive spatially explicit inventories and calculate 14 impacts due to peri-UA using the ReCiPe method. Further, to show the importance of impact assessment regionalization for the environmental assessment of peri-UA, we regionalize eutrophication impacts characterization. We use the Metropolitan Area of Barcelona (AMB) to illustrate these methodological developments. Vegetables and greenhouses, the prevalent peri-UA land uses, had the largest impacts assessed, of all peri-UA land uses. European NPK mineral fertilizer production to cover N demand of these crops drives all impacts. For fruit crops, on-site N emissions drive marine eutrophication impacts and for irrigated herbaceous crops, phosphate runoff drives freshwater eutrophication impacts. Geographic variability of peri-UA metabolic flows and impacts was displayed. Management practices at the plots, which are linked the land use, are responsible for impacts variability. Regionalization of eutrophication impacts highlights the importance of accounting for the biophysical aspects at the geographic scale at which peri-UA takes place, which is a much finer scale than those implemented in current regionalization of impact assessment methods

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in LCA. This study provides a fundamental baseline needed to assess transition scenarios of peri-UA at an appropriate geographic level of analysis and gives essential knowledge to guide appropriate circular and sustainability strategies for the sector.

1. Introduction

Urban agriculture (UA) is becoming increasingly important for cities (Langemeyer et al., 2021; Mohareb et al., 2017). Cities drive major food flows accompanied of transboundary environmental impacts, as, often, most food provision is offered from the production of regions beyond the cities' boundaries (Goldstein et al., 2017; Newell and Ramaswami, 2020). Increasing cities capacity of food production within their boundaries is an alternative for improved self-reliance, for resource and cost-efficient systems harnessing food-energy-water (FEW) nexus synergies (Daigger et al., 2015) and improved resilience capacity during crisis scenarios such as the most recent global pandemic which has put additional stress on global food supply chains and consumers access to food (OECD, 2020). In addition, UA could help increase local resources management (Magid et al., 2006; Rufí-Salís et al., 2020; Saha and Eckelman, 2017), reduce upstream dependencies of resources needed to produce urban food, reduce transboundary impacts of food production (Zumkehr and Campbell, 2015), and enhance global sustainability by for instance addressing environmental externalities of urban land teleconnections (Meyfroidt et al., 2022; Seto et al., 2012) and improving the impacts of the FEW nexus (Mohareb et al., 2017; Villarroel Walker et al., 2014). Yet, important environmental impacts may arise with the broader implementation of peri-UA (Dorr et al., 2021).

To assess whether UA can truly enhance global and local sustainability requires sophisticated analytical methods. Life Cycle Assessment (LCA) has been used extensively to estimate product-systems´ environmental impacts, by accounting for direct impacts, i.e., caused in the production and management of the product-system, as well as for indirect impacts i.e. caused up and down stream by raw materials extraction, transport logistics, and end-of life management of the product (Guinée, 2002). Spatially explicit LCA (also referred to as regionalized LCA) has been identified as a more appropriate method for the assessment of UA, compared to site generic LCA (Dorr et al., 2021). Regionalization in LCA increases geographic representativeness of LCA data, therefore uncertainty is reduced by displaying spatial variability (Patouillard et al., 2018). The inventory can be regionalized and spatialized by adding regional detail to inventory data and by attributing it to a specific location (Patouillard et al., 2018).

In the case of UA, being able to pinpoint the variability due to the technology used, the crop grown, the irrigation, the fertilization practices, the energy, nutrients and water use, etc., as well as the emissions and impacts caused by these in different locations across a city or metropolitan area is essential for urban planning and for the development of urban sustainability strategies (Goldstein et al., 2017). In addition, because substances impact on the environment can be different depending on the location (Hauschild, 2006), it is also vital to regionalize the impact assessment of the LCA. Several steps have been taken towards globally resolved impact assessment methods for regionalized LCA in recent years (Bulle et al., 2019; Verones et al., 2020) where different native resolutions have been used depending on the impact assessment models. For instance, eutrophication in the LC-IMPACT method, uses freshwater ecoregions as native spatial resolution for freshwater eutrophication, and river basins and large marine ecosystems for marine eutrophication (Verones et al., 2020).

Despite the appropriateness of regionalized LCA for assessing the environmental sustainability of UA applications, few spatially explicit LCAs have been performed for UA. One of the main reasons for this gap is the lack of high-resolution inventories e.g. at plot scale when looking at peri-UA or at sub-country level for conventional agriculture (Morais et al., 2017). Also, it is more usual for UA LCAs to look into the growth of specific crops or a technology of interest in order to compare these systems with conventional production (Dorr et al., 2021). More pragmatic issues for few regionalized LCA of peri-UA is the difficult operationalization of regionalized LCA calculations in the classic LCA software (Frischknecht et al., 2019), recently addressed, but not yet broadly used (Mutel et al., 2018; Mutel and Hellweg, 2009).

In this study we have the following overall research question, what are the life cycle environmental impacts of peri-UA production of a metropolitan area? Also, we have three aims, 1) to determine potential direct and indirect environmental impacts of current peri-UA production of a metropolitan area, 2) to establish spatially explicit life cycle inventories and environmental impacts for such production, and 3) to regionalize eutrophication impacts as an example of the importance of impact assessment regionalization for the environmental assessment of UA. To answer the research question and fulfill the aims we perform a regionalized LCA at the individual plot scale level for peri-UA. We focus on peri-UA for several reasons, its potential for urban food production (Mohareb et al., 2017), its proximity to urban demand centers yet avoiding some challenges of UA including high competition for land, its development at scale by professional farmers, and also because extensive periurban farms have been found as one of the most representative UA systems for the global north cities (Orsini et al., 2020), which we will further study. The high-resolution regionalized LCA developed in this study provides a fundamental baseline needed for further exploration of UA development pathways such as circular, organic, or under climate change future scenarios. Finally, we focus on regionalizing eutrophication impacts given the importance of local water quality for peri-UA production and its surroundings.

2. Methods

In this section we describe the goal and scope of the LCA, and the data used in the regionalized foreground (related to peri-UA production) and background (related to upstream and downstream processes needed by peri-UA production). Also, we describe the non-regionalized life cycle impact assessment (LCIA) and the eutrophication impacts regionalization. The overall methodological approach is shown in Appendix A, section 1.

2.1. Goal and scope

The goal of the study is to estimate the attributional regionalized life cycle environmental impacts of peri-UA production within the Metropolitan Area of Barcelona (AMB) (see Section 2.2). The *functional unit* is the total annual peri-urban food production with conventional open-air, soil-based agriculture. The reference flow used is 1 ha of each land use type in 2015 according to the URBAG map (see Section 2.2 for details). We focus on the conventional agricultural production meaning no special techniques such as hydroponics or organic production are considered. The life cycle inventory is regionalized- in other words, it is calculated for each agricultural plot in the AMB, and specifically considers the practices and management used in each one. Life cycle stages included for individual plots are the production of materials for cultivation, the growth of crops, harvesting of products and sub-products, and the treatment of residues. This is thus a cradle-to-gate LCA.

Upstream processes included are (Fig. 1): seeds production, diesel burned in agricultural machinery, manure production, NPK fertilizer production and harvesting machinery production. Downstream processes included are (Fig. 1): treatment of removed residues including four options: 1) return residues to field (which contributes to direct emissions), 2) use residues in animal diets, 3) landfilling residues and 4) open-air on-site residues burning (also contributes to direct emissions). For the fraction of

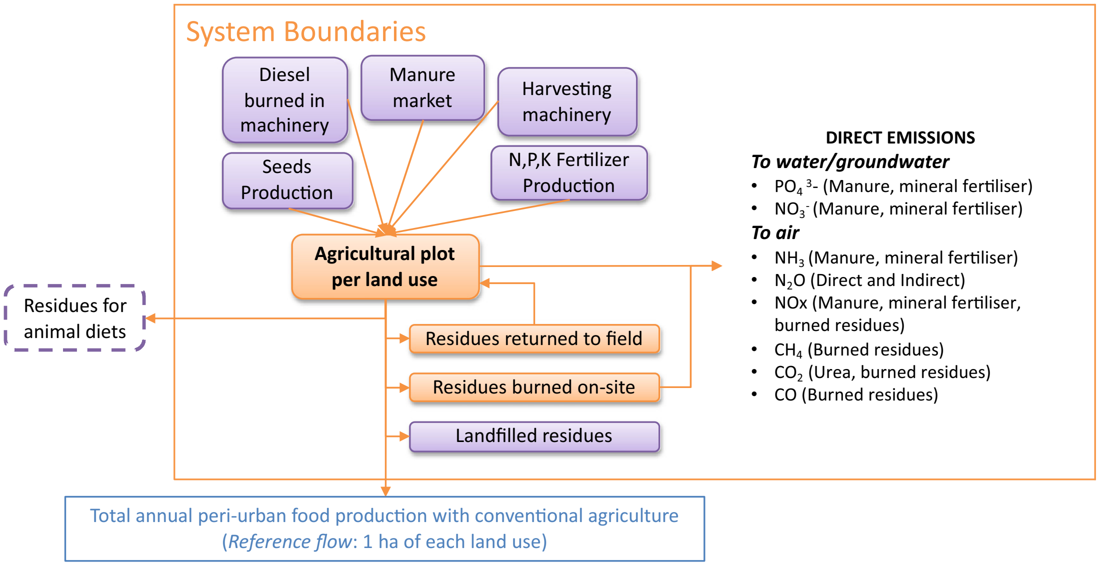

Fig. 1. System boundaries for the total annual peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). The functional unit (shown in blue font) is measured by the reference flow which is 1 ha of each land use according to the URBAG map. Purple boxes represent background processes generating indirect emissions. Orange boxes represent foreground processes generating the direct emissions. The dashed box represents an allocated process.

residues used in animal diets we considered 100 % of the burden is bared by the main product. Thus, this fraction bares no impact as it is reused in the agricultural system. This is equivalent to surplus allocation and this assumption is in line with the background dataset model i.e. the cut-off model of the ecoinvent database v3.8 (Wernet et al., 2016), where there is an incentive to use recyclable products, available burden free.

The Impact Assessment Method selected is the ReCiPe Midpoint (H) V1.13 method (Huijbregts et al., 2016). The 14 impacts included from this method are: terrestrial ecotoxicity (TET), marine ecotoxicity (MET), freshwater ecotoxicity (FET), human toxicity (HT), marine eutrophication (ME), freshwater eutrophication (FE), fossil depletion (FD), climate change (GWP100), agricultural land occupation (ALO), terrestrial acidification (TA), natural land transformation (NLT), particulate matter formation (PMF) and photochemical oxidant formation (POF). For climate change, the IPCC 2013 impact (GWP100_IPCC) is added as an update for the climate change impact which contains AR4 GWPs instead of AR5. In addition, for marine eutrophication and freshwater eutrophication impacts, characterization factors (CFs) have been adapted to reflect the conditions of the AMB (Section 2.4.3 for more details). Thus, marine and freshwater eutrophication foreground impacts are presented for ReCiPe CFs at midpoint level, and for AMB regionalized CFs at midpoint, endpoint and damage levels.

2.2. Study region

The study area is the Metropolitan Area of Barcelona, located in the northeast region of the Iberian Peninsula. It encompasses 36 municipalities of the autonomous community of Catalonia in Spain (Appendix A, Table S.3). Despite the decreasing areas of peri-UA in the AMB during the past 50 years, there are still 5584 ha of cropland according to the most recent Land Cover Map of Catalonia (CREAF, 2015) including crops in abandoned land, land in transformation and fallow land. This map is referred to as the MCSC map and it contains a high-resolution thematic cartography of the main types of land cover for Catalonia (forests, crops, urban areas, etc.). To make more specific inventory calculations, an additional disaggregation of the MCSC agricultural land cover categories was added to the map. This disaggregation was based on the DUN-SIGPAC map, containing cartographic data for individual crops from the Department of Agriculture, Livestock, Fisheries and Food from the Generalitat de Catalunya (DARPA, 2015) and its annual declaration of crops (Declaració agrària, DUN for its acronym in Catalan) made by farmers in Catalonia, which matches the public registry of Geographic Information System of agricultural parcels in Catalonia (Sistema d'informació geogràfica de parcel·les agrícoles, SIGPAC for its acronym in Catalan). Thus, the hybrid map used in this study to determine areas and locations has crop group categories e.g. irrigated herbaceous crops from the MCSC map, and the individual crop categories e.g. Barley from the DUN-SIGPAC map. It is referred to as the URBAG map (.shp file), as it was created in the context of the URBAG project[1] and its creation procedure is detailed in Mendoza Beltran et al. (2022). The URBAG map hybrid land uses are shown in Fig. 2 and its total areas are shown in Appendix A, Table S.2. Total areas and their geographic location coincide with agriculture areas of the MCSC map.

2.3. Life cycle inventory (LCI)

In this section we explain the calculation of the regionalized inventory i.e. foreground data (inputs and outputs of each agricultural plot) and background data (the supply chain of inputs and treatment of waste).

2.3.1. Foreground data: inputs to agricultural plots

To link resource requirements and emissions to each parcel and land type in the URBAG map we start from an inventory dataset of the inputs to agricultural production calculated by the Socioecological Integrated Analysis (SIA) model (Marull et al., 2021; Padró et al., 2020). This dataset contains 18 metabolic flows or parameters at the municipality level and over nine aggregated land cover categories specified in the SIA model (Appendix A, Table S.3). For details on the estimation of each metabolic flow/parameter, the data sources used and the final datasets used here see Appendix A, Sections 4 and 5.

We mapped each SIA model parameter in the URBAG map using a dictionary that links SIA land categories with MCSC land categories (Appendix A, Table S.1). With this dictionary a SIA category can be assigned to each plot in the URBAG map, generating an "extended" URBAG map. Further, following the SIA land category and municipality per plot in the "extended" URBAG map, the corresponding SIA metabolic flow/parameter value was assigned to each plot. This procedure yields 16 maps (.shp file) one for each metabolic flow/parameter, i.e. NPK compounds as N, as P_2O_5 and as

[1] www.urbag.eu.

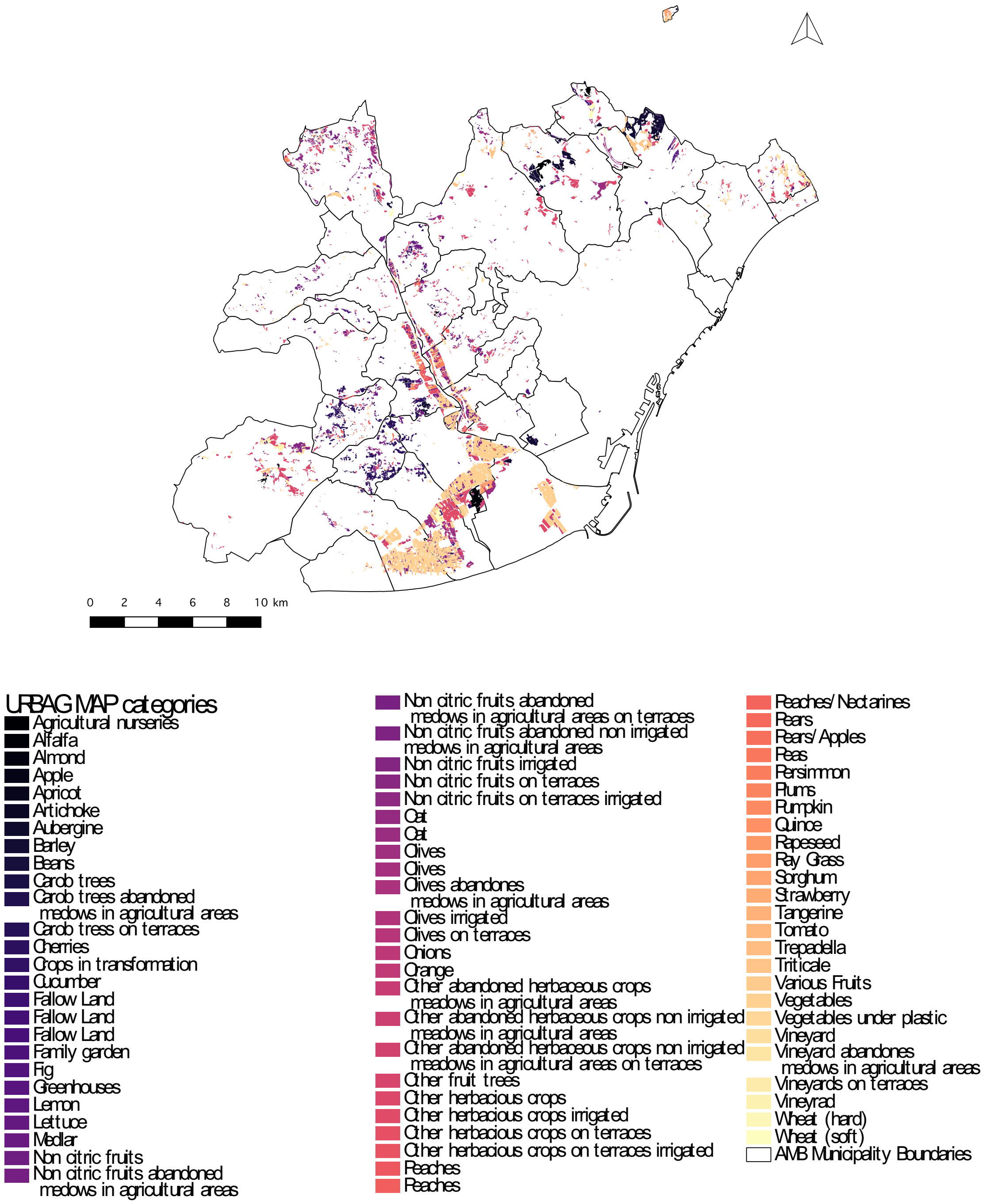

Fig. 2. Location of peri-urban agriculture in the Metropolitan Area of Barcelona (AMB) and different land uses according to the URBAG map. Taken from: Mendoza Beltran et al. (2022). Total areas coincide with the AMB MCSC map (CREAF, 2015) agricultural areas i.e. 5584 ha.

K_2O, urea, ammonium-calcium nitrate, calcium superphosphate, potassium chloride, manure, machinery use, harvesting machinery use, agricultural residues, fraction of residues returned to field, for animal diets, landfilled and burned, and seeds (Appendix A, Section 7, area and produce were not mapped). In addition, data for manure was disaggregated by the animal source based on information from the SIA model, in order to better estimate applied nutrients (N and P) from manure per municipality (Appendix A, Section 4.6). These maps (referred to as inventory maps) provide the basis for quantifying regionalized inputs and outputs per plot (Appendix A, Table S.25). Further, two parameters are added to the inventory: 1) crop-irrigation specific N_2O emission factors (EFs) and 2) N and P removal with the harvest for different crops. For the former, EFs detailed in Mendoza Beltran et al. (2022), based on a meta-analysis and literature review based on Cayuela et al. (2017), are the main source. For crops in transformation and fallow land no N_2O emissions are assumed. Also, whenever the irrigation method (irrigated or rainfed) was unknown for a plot, the larger EF was used, and in the case of some categories e.g. vegetables, the irrigation EF is used as default, as all vegetable plots are irrigated. For any other missing combination of crops and irrigation method for which no EF is available we use the IPCC EF of 1 % of applied N which is estimated to be emitted as direct N_2O (Hergoualc'h et al., 2019). Regarding the N and P removed with the harvest by the different crops, the values applied are provided in Appendix A, Section 8. Values correspond to the nutrients extracted by different parts of the plant depending on the crop e.g. for vineyards it includes the leaves, bunches and shoots; for olives it only includes the olives; for some cereals it includes the grain and straw; for vegetables it refers to the harvested product, etc. (López Bellido et al., 2010; Rural Cat, 2019).

Our current inventories are limited by missing data on water use (out of the scope of this LCA) and pesticides use, which are essential data to understand water depletion and toxicity impacts (included for indirect impacts only). Further completion of these inventories with data at the right resolution would be very relevant to enhance our current conclusions.

2.3.2. Foreground data: outputs and direct emissions of agricultural plots

Direct emissions per plot (.shp file) are calculated using emission factors per substance per activity and the inventory maps previously described (Section 2.3.1). Direct emissions include direct and indirect nitrous oxide (N_2O), ammonia (NH_3), nitrogen oxides (NO_x), methane (CH_4), carbon dioxide (CO_2) and carbon monoxide (CO) to air, nitrate (NO_3^-) to groundwater and phosphate (PO_4^{3-}) runoff to surface water (Appendix A, Table S.25). Emissions' maps are provided in Appendix A, Section 9. Here we described the calculation for each.

The updated IPCC guidelines were followed for the estimation of N_2O emissions from managed soils (Hergoualc'h et al., 2019). Direct N_2O emissions (Appendix A, Fig. S.8) are derived with the Tier 2 crop-irrigation specific EFs (Section 2.3.1) applied to the sum of total N applied per plot i.e. including mineral fertilizer, manure and residues returned to the plot. Indirect N_2O emissions (Appendix A, Fig. S.11) are estimated based on the sum of N_2O produced from atmospheric deposition of volatilized N from managed soils (Appendix A, Fig. S.9) i.e. total ammonia emissions, and N_2O produced from leaching and N runoff, from N additions to managed soils (Appendix A, Fig. S.10) i.e. total nitrate emissions.

Total NH_3 emissions to air (Appendix A, Fig. S.16) are the sum of NH_3 emissions from manure (Appendix A, Fig. S.15) and mineral fertilizer applications (Appendix A, Fig. S.14). Specific Efs from ecoinvent (Nemecek and Kagi, 2007) have been used for urea, ammonium-calcium nitrate and for NPK compounds application. The EF for NH_3 emissions from applied N in manure is derived from the Product Environmental Footprint Category Rules Guidance (PEFCR) from the European Commission (2018). Manure was converted to applied N using conversion factors from Boixadera et al. (2000) per animal source (Appendix A, Section 4.6).

NO_x emissions to air from the sum of total N in manure and in mineral fertilizer applications (Appendix A, Fig. S.17), are estimated using an emission factor from the European Environmental Agency as presented in the World Food LCA database methodological guidelines for LCI of agricultural products (Nemecek et al., 2019).

CO_2 emissions from urea applications (Appendix A, Fig. S.24) are estimated based on the EF from the 2006 IPCC guidelines (De Klein et al., 2006). According to these, the carbon content of urea is 20 % on an atomic weight basis. The factor 0,466 kg N per kg urea is used to derive N content of urea (ecoinvent, 2020).

For the burned on-site fraction of residues, we derive the amount of burned residues in kg dry matter and estimate emissions of CO, CH_4, N_2O and NO_x to air, based on the mean Efs from the IPCC guidelines from 2019 (Ogle et al., 2019) specific for agricultural residues burning, in Table 2.5 of Chapter 2. The maps of these emissions are shown in Appendix A, Fig. S.25, S.26, S.12 and S.18, respectively. For CO_2, net-zero emissions are assumed. For N_2O and NO_x the emissions to air from burning agricultural waste are added to the other sources to calculate the total emissions (Appendix A, Fig. S.13 and S.19, respectively).

In addition, emissions to air from the combustion of diesel in agricultural machinery used at the field are also included. For this purpose, we use two ecoinvent v3.8 datasets: 1) 'diesel burned in agricultural machinery' and 2) 'combine harvest'. Emissions from each process has been split in two, the direct emissions and the indirect emissions. For the direct emissions, we calculate the ReCiPe impacts of these emissions per MJ of diesel and per ha, respectively for each process. For the indirect emissions, we calculate the impacts per MJ and per ha as well. To find direct emissions per plot, the direct emissions fraction per MJ was multiplied with machinery use per plot – based on the hours of machinery use in the SIA dataset, and the per ha fraction was use without modification for plots with harvesting machinery use. Emissions of CO_2, CO, N_2O, NO_x, CH_4 are included in the direct impacts per ha and per MJ, among others. The indirect impacts of machinery use and harvesting machinery use, are described in Section 2.4.2 as part of the background system.

Total NO_3^- emissions to water have been split into two, 1) base loss of nitrate to water (Appendix A, Fig. S.20) and 2) additional nitrate (Appendix A, Fig. S.21) emissions to water (European Commission, 2018). For the former, we used the EF from the IPCC guidelines from 2019 (De Klein et al., 2006) to estimate N losses by leaching/runoff in wet climates and apply it to total N additions (mineral fertilizers, manure and residues returned to field). For the latter, we follow the balance proposed by the PEFCR guidance (European Commission, 2018) in which additional nitrate emissions to water are equal to the N balance shown in Eq. (1).

$$\begin{aligned}&\text{Additional } NO_3^{-} \text{ emissions to water} = \\ &\text{Total N inputs}_{(mineral\ fertiliser+manure+agricultural\ residues)} + N_2 \text{ fixation} - \\ &\Big(\text{N removal with harvest} + NH_3 \text{ emission to air} + N_2\text{O emissions to air} \\ &\quad + NO_x \text{ emission to air} + NO_3^{-} \text{ base loss}\Big)\end{aligned} \tag{1}$$

For N_2 fixation by crops, we have assumed the same amount as for N removed with the harvest for the following crops: alfalfa, chickpeas, beans, carob trees, peas and sainfoin, as suggested in the PEFCR guidance (European Commission, 2018). The rest of crops or groups of crops do not fixate any N via symbiosis with bacteria. Finally, for plots where N inputs are less than N outputs, the balance is adjusted i.e. additional nitrate emissions to groundwater are zero and N remove with the harvest is lowered by the difference between inputs and outputs. Thus, when less N is available the crop will remove less N. In addition, there are some plots in which no N is removed with the harvest after this adjustment of the balance, leading to zero N removed with the harvest, and in some cases, base nitrate emissions should be further reduced in order to achieve a balance between inputs and outputs. Final N balance maps, including inputs and outputs, are shown in Figs. 3 and 5.

For P, phosphate (PO_4^{3-}) runoff to surface waters is estimated and, as no sewage or liquid manure are added to the plots, no phosphate leaching to groundwater is estimated. As with nitrate, phosphate emissions are divided in two: 1) base phosphate runoff to surface waters (Appendix A, Fig. S.22) and 2) additional phosphate runoff to surface waters (Appendix A, Fig. S.23). The first are estimated using the ecoinvent models (Nemecek and Kagi, 2007) following the Swiss Agricultural Life Cycle Analysis (SALCA) Phosphorus models. Phosphate runoff occurs due to the additions of calcium superphosphate, NPK compounds and manure to the crops. Based on these, a correction factor for fertilization with P is estimated. Emissions also depend on the land use and its average quantity of P lost through runoff. For arable land this value is set to 0,175 kg P/ha yr. The product of these two parameters is equal to the phosphate runoff to surface waters. Additional phosphate runoff, is based on the balance of P per plot (Eq. (2)).

$$\begin{aligned}&\text{Additional } PO_4^{3-} \text{ runoff to surface waters} = \\ &\text{P mineral fertilizer input} + \text{P manure input} \\ &\quad - \text{P removed with the harvest} - PO_4^{3-} \text{ base emissions}\end{aligned} \tag{2}$$

Further, when the P balance is positive i.e. there are more inputs than outputs, additional phosphate emissions are equal to this difference and when the balance is negative i.e. there are more outputs than inputs, additional phosphate emissions are equal to zero and P removed with the harvest is reduced with the negative difference. Final P balance maps, including inputs and outputs, are shown in Figs. 4 and 6.

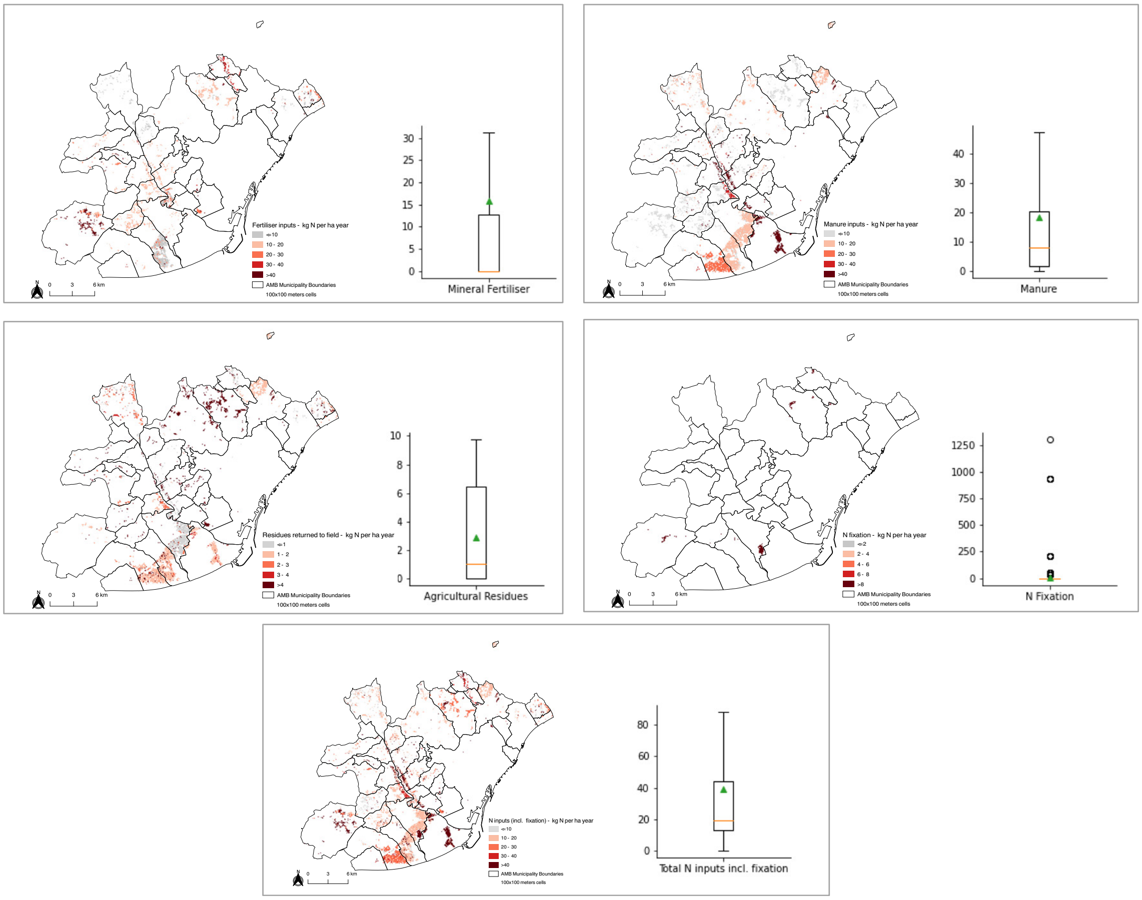

Fig. 3. Maps and boxplots for total N inputs from mineral fertilizer, manure, agricultural residues and N fixation for peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). All figures are in kg N/ha yr⁻. Green triangles in boxplots represent the means.

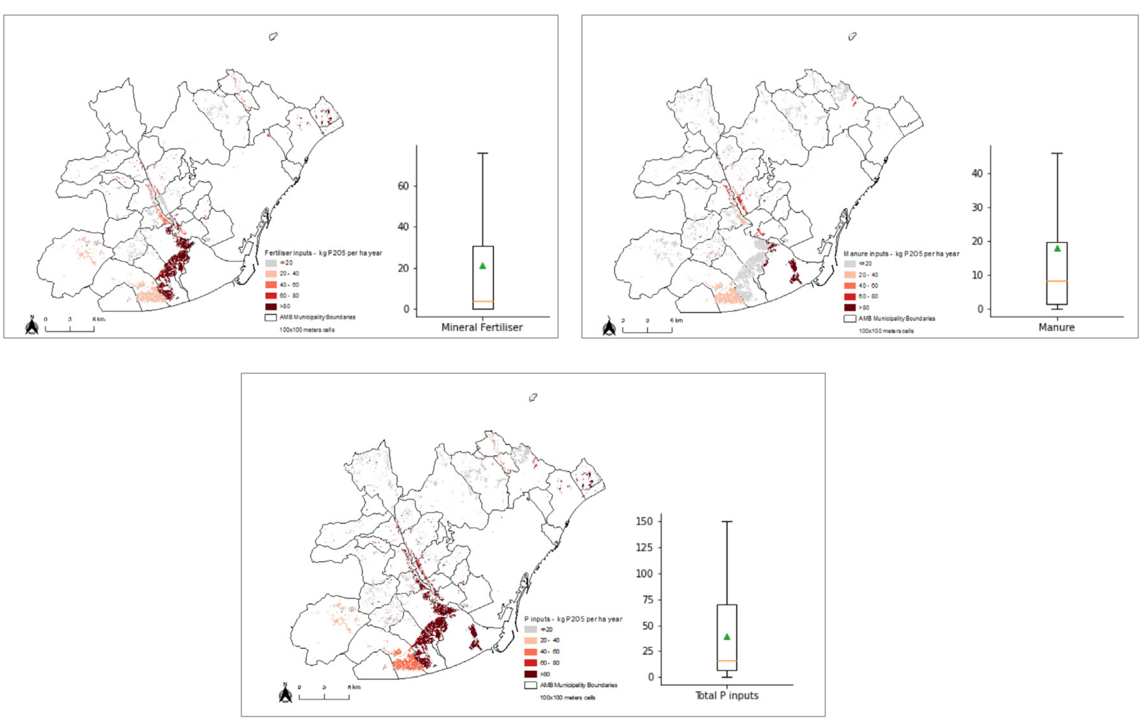

Fig. 4. Maps and boxplots for total P inputs from mineral fertilizer and manure, for peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). All figures are in kg P_2O_5/ha⁻ yr⁻. Green triangles in boxplots represent the means.

2.3.3. Background data: ecoinvent processes

The inventories used to represent the background impacts of inputs included are described in Appendix A, Section 10. These ecoinvent processes cover raw material extraction, product manufacturing and average product transport to market. For the landfilled fraction of residues, the ecoinvent process of municipal solid waste treatment is used. Section 2.4.2 describe the way we have linked each process to the inputs per plot, as this process was done at the impact assessment level. Moreover, background processes have not been further regionalized and are used in the regional scope provided by ecoinvent (ecoinvent, 2022). In the results this means that indirect impacts, although assigned to a plot, they are taking place in the supply chain i.e. in other geographic locations, up and down stream.

2.4. Life cycle impact assessment (LCIA)

2.4.1. Foreground LCIA: direct impacts

Impacts due to direct emissions of Nox, NH_3, N_2O, NO_3^- (base and additional), CO_2, CO, CH_4 and PO_4^{3-} (base and additional) are the following: terrestrial acidification, particulate matter formation, photochemical oxidation formation, climate change, and marine and freshwater eutrophication. To estimate these impacts, the ReCiPe Midpoint (H) V1.13 CFs are used (Appendix A, Section 11).

For marine and freshwater eutrophication, we have further regionalized the CFs at midpoint, endpoint and damage levels given the relevance of these impacts for agricultural production (Section 2.4.3). This means we have recalculated the ReCiPe CFs to account for the specifications of the Llobregat and Besòs river basins, as these are the two main basins where peri-UA takes place within the AMB. For this purpose, another feature was added to the URBAG map indicating the basin in which each plot lies. Strictly speaking there are 6 sub-basins within the AMB i.e. Llobregat, Garraf and Besòs basins and rivers of Maresme, rivers of the Llobregat plain and rivers of the Barcelona plain. We have assigned the CFs for Llobregat basin to Garraf basin, rivers of the Barcelona plain and rivers of the Llobregat Plain, while CFs for Besòs basin are assign to rivers of Maresme, following geographic proximity.

2.4.2. Background LCIA: indirect impacts

The calculation of indirect life cycle impacts i.e. those in the supply chain of each input, follows several steps. First, the total impact per unit of input is calculated using ecoinvent v3.8 and ReCiPe Midpoint (H) V1.13 midpoint CFs. For instance, for urea, we use the European ecoinvent process 'market for urea' which is defined per kg of urea, and we estimate its impacts per kg of urea. Second, we calculate the product of the input per plot and the impact per unit, which is equal to the indirect impact of each input per plot. For example, a plot using 10 kg of urea will have a climate change impact of 1.28 kg CO2eq/kg urea (according to ecoinvent v3.8) $*$ 10 k urea/ha = 12.8 kg CO2eq/ha. All impacts per unit of input are derived using the activity browser (Steubing et al., 2020) of Brightway2 LCA software (Mutel, 2017), and are then read in python to calculate the indirect impacts per plot. Because the indirect impacts depend on the inputs per plot, they are also mapped to each plot across the AMB. This procedure is done for each input using the inventory maps (Appendix A, Section 7) and the ecoinvent processes (Appendix A, Section 10).

2.4.3. Regionalization of marine and freshwater eutrophication life cycle impact assessment

Besides using the ReCiPe Midpoint (H) V1.13 method for the LCIA, CFs for direct eutrophication impacts were regionalized (Table 1). Eutrophication is one of the most serious environmental consequences of agriculture, but unlike global climate change, the effects of nutrient pollution are highly variable depending on local topography, hydrology, water quality, and ecosystem ecology. The ReCiPe method is described in Huijbregts et al. (2016) and considers both freshwater eutrophication (in kg $P_{equivalents}$) and marine eutrophication (in $N_{equivalents}$) at the midpoint level and damages to ecosystems at the endpoint level (in PDF·m^3).

For freshwater eutrophication potential (FEP), midpoint CFs are regionalized based on Helmes et al. (2012) using a gridded model of P residence times in freshwaters (fate factor, FF). Endpoint CFs were calculated by accounting for the local effect factors on freshwater species. The endpoint CFs are extended to damage factors assuming the average freshwater species density of 7.89×10^{-10} species/m^3 (Goedkoop et al., 2009). For marine eutrophication potential (MEP) values are calculated based on the methods of Cosme and Hauschild (2017). In order to calculate endpoint characterization factors, effect factors (Efs) were gathered from Cosme and Hauschild (2016) based on organism vulnerability to hypoxia by taxonomic group. More details and values used for the regionalization of the eutrophication CFs for the Besòs and Llobregat rivers can be found in Appendix A, Section 12. Table 1 shows the CFs for P and N emissions according to ReCiPe and to AMB-specific CFs calculated in this study. For phosphate and nitrate, we have estimated the equivalent of the N and P CFs, by converting each substance to its elemental P and N content.

Direct ME impacts using the regionalized CFs are calculated for direct nitrate (base and additional) emissions and for phosphate runoff to water (base and additional) for the case of FE.

2.5. Total impacts

Total impacts per plot correspond to the sum of foreground and background impacts, as calculated using ReCiPe CFs. For terrestrial acidification, particulate matter formation, photochemical oxidation formation and climate change foreground and background impacts are calculated. For all the other impacts, the total impacts correspond to the background impacts alone, as estimated direct emissions do not influence them. For eutrophication, the foreground, background and total impacts calculated with ReCiPe CFs and foreground impacts using ReCiPe and AMB-specific CFs applied to nitrate and phosphate emissions only are shown. Appendix A, Section 13 shows a summary table for direct, indirect and total impacts calculated for each impact (also covered in.shp files). Further, vector files

Table 1
ReCiPe midpoint and AMB-specific midpoint, endpoint and damage characterization factors for marine and freshwater eutrophication impacts.

Impact category	Env flow		Llobregat basin			Besòs basin		
		ReCiPe	Midpoint	Endpoint	Damage	Midpoint	Endpoint	Damage
		kg Neq/kg	kg Neq/kg	PDF·m^3·yr/kg	species·yr/kg	kg Neq/kg	PDF·m^3·yr/kg	species·yr/kg
Marine eutrophication	Nitrogen to water	1	1,1	726	5,73E-07	1,25	827	6,52E-07
	Nitrate to groundwater	0,23	0,25	167	1,32E-07	0,29	190	1,50E-07

Impact category	Env flow		Llobregat basin			Besòs basin		
		ReCiPe	Midpoint	Endpoint	Damage	Midpoint	Endpoint	Damage
		kg Peq/kg	kg Peq/kg	PDF·m^3·yr/kg	species·yr/kg	kg Peq/kg	PDF·m^3·yr/kg	species·yr/kg
Freshwater eutrophication	Phosphorus to water	1	0,0021	1,5	1,18E-09	0,002	1,44	1,14E-09
	Phosphate to water	0,33	0,0007	0,49	3,90E-10	0,00072	0,47	3,75E-10

for inventory, direct emissions, direct impacts, indirect impacts and total impacts, where rasterized in the QGIS software using a 100 by 100 m pixel size. These are the maps shown in the results and Appendix.

3. Results

3.1. N and P inputs, outputs and balances

The inputs (mineral fertilizer, manure, biomass waste) and outputs (N fixation in crops) in terms of Kg N/ha year are shown as maps and boxplots in Figs. 3 to 6. For N inputs (Fig. 3), manure provides more N to peri-UA in the AMB than mineral fertilizers and agricultural residues. These three N inputs show large variability among plots. For manure, variability comes from the different animal source - which have a different N content, and the amount use per plot. For mineral fertilizer, variability is due to the amount used per plot as well as the type of fertilizer associated to the crop grown. N from agricultural residues correspond to the residues fraction returned to the field. This fraction varies according to the type of crop with herbaceous crops and vegetables having the largest fractions. N fixation is mostly zero for all plots, thus only a few outliers are shown (Fig. 3). Particularly a plot growing beans, six plots growing peas, and 15 plots growing sainfoin display large N fixation values above 200 kg N/ha due to either a high yield or a high rate of N removed with harvest per ha leading to such N fixation. Total N inputs are on average 40 kg N per ha yr. Total mineral fertilizer, manure, agricultural residues and N fixation inputs in the AMB are 53.2, 97.9, 9.9 and 13.3 tons N/yr, respectively. Total N inputs are 174.4 tons N/yr.

The largest N outputs are NH_3 and NO_3^- base emissions (Fig. 5). In the case of some plots, NO_3^- additional emissions are substantial after the balance is made. For these plots there are large N inputs balanced through additional nitrate outputs, however this remains a balancing parameter. N removed with the harvest is on average about 20 kg N per ha. Total NH_3, N_2O, NO_x, NO_3^- base and NO_3^- residual emissions are 23.7, 2.4, 4.9, 38.6, 4.2 kg N/yr, respectively. Total N removed with the harvest is 100.6 kg N/yr. Total N outputs are 174.4 kg N/yr.

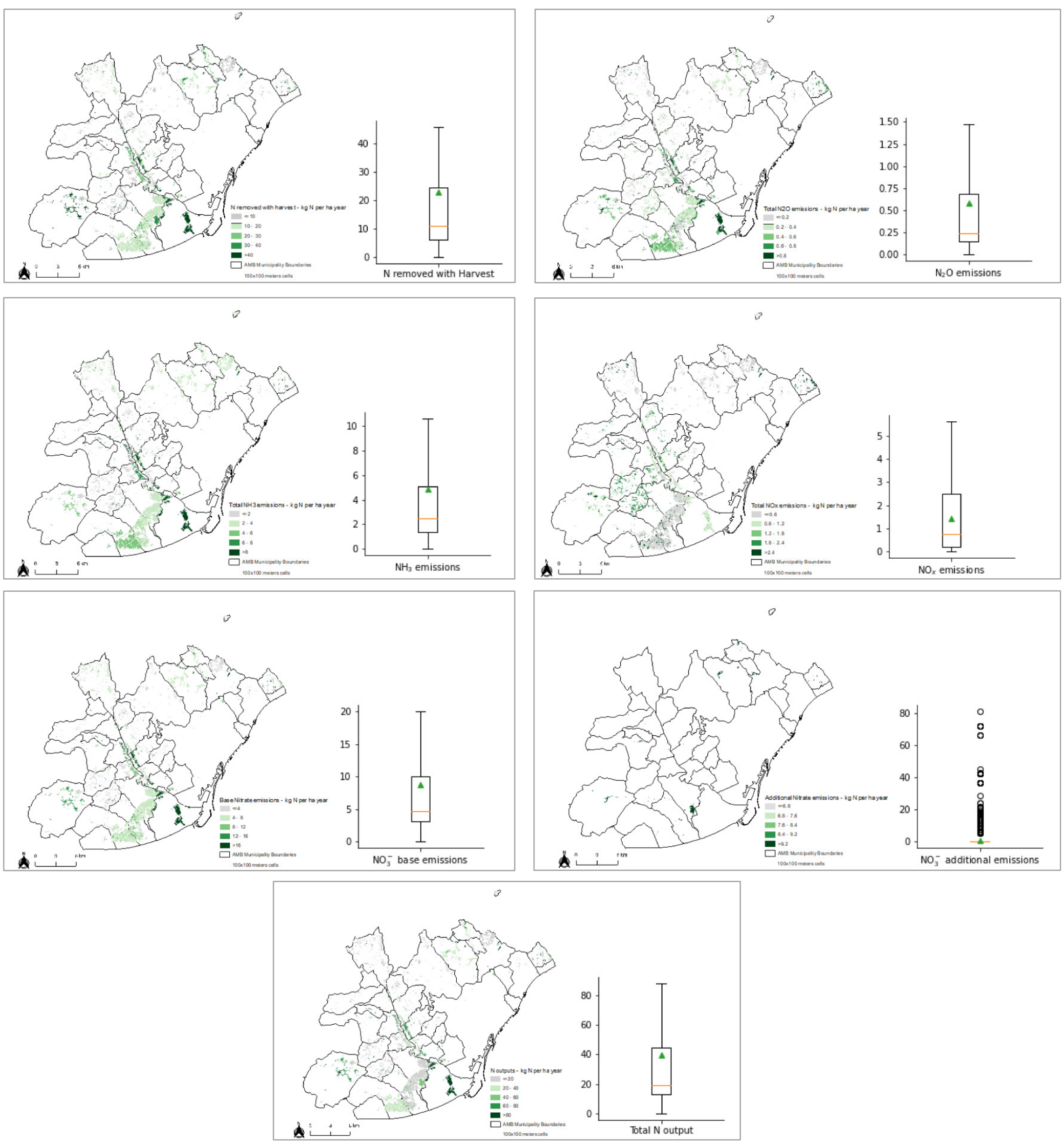

Fig. 5. Maps and boxplots for total N outputs in the form of ammonia, nitrogen oxides, nitrous oxide, nitrate (base and additional) emissions and N removed with the harvest for peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). All figures are in kg N/ha^{-} yr^{-}. Green triangles in boxplots represent the means.

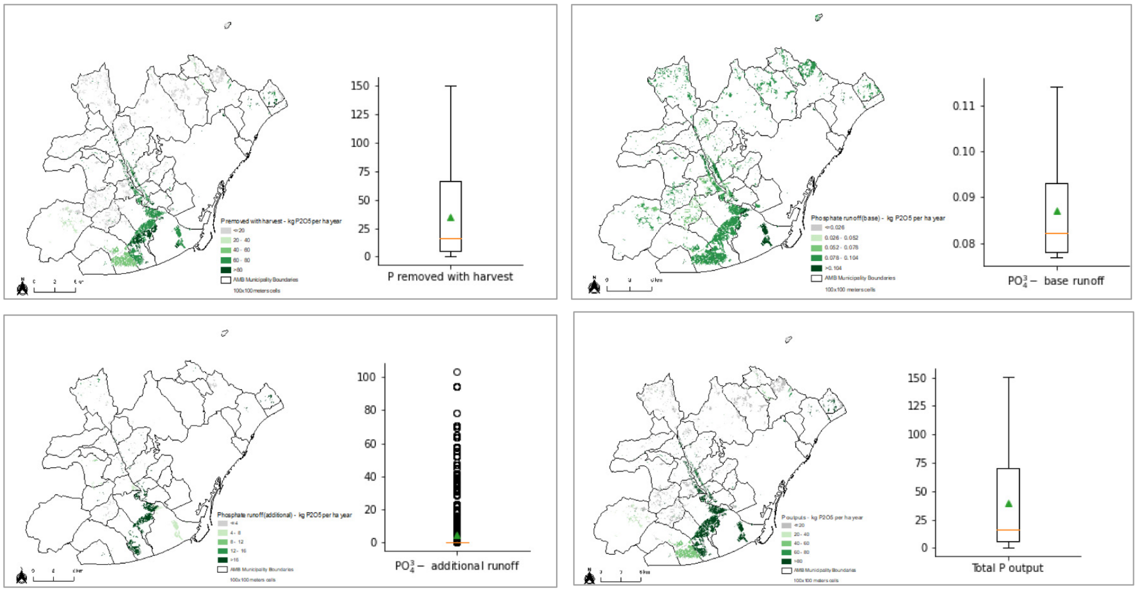

Fig. 6. Maps and boxplots for total P outputs in the form of phosphate runoff (base and additional) and P removed with the harvest for peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). All figures are in kg P_2O_5/ha$^-$ yr$^-$. Green triangles in boxplots represent the means.

For P inputs (Fig. 4), manure and mineral fertilizers provide about the same amount of P to crops on average, yet mineral fertilizers have a large variability resulting in the largest P input for some plots. Variability of P in manure depends on the animal source and amount use, while for P in mineral fertilizers the type of fertilizer and amount use per plot, which in itself depends on the crop grown, determines the variability. Total P inputs

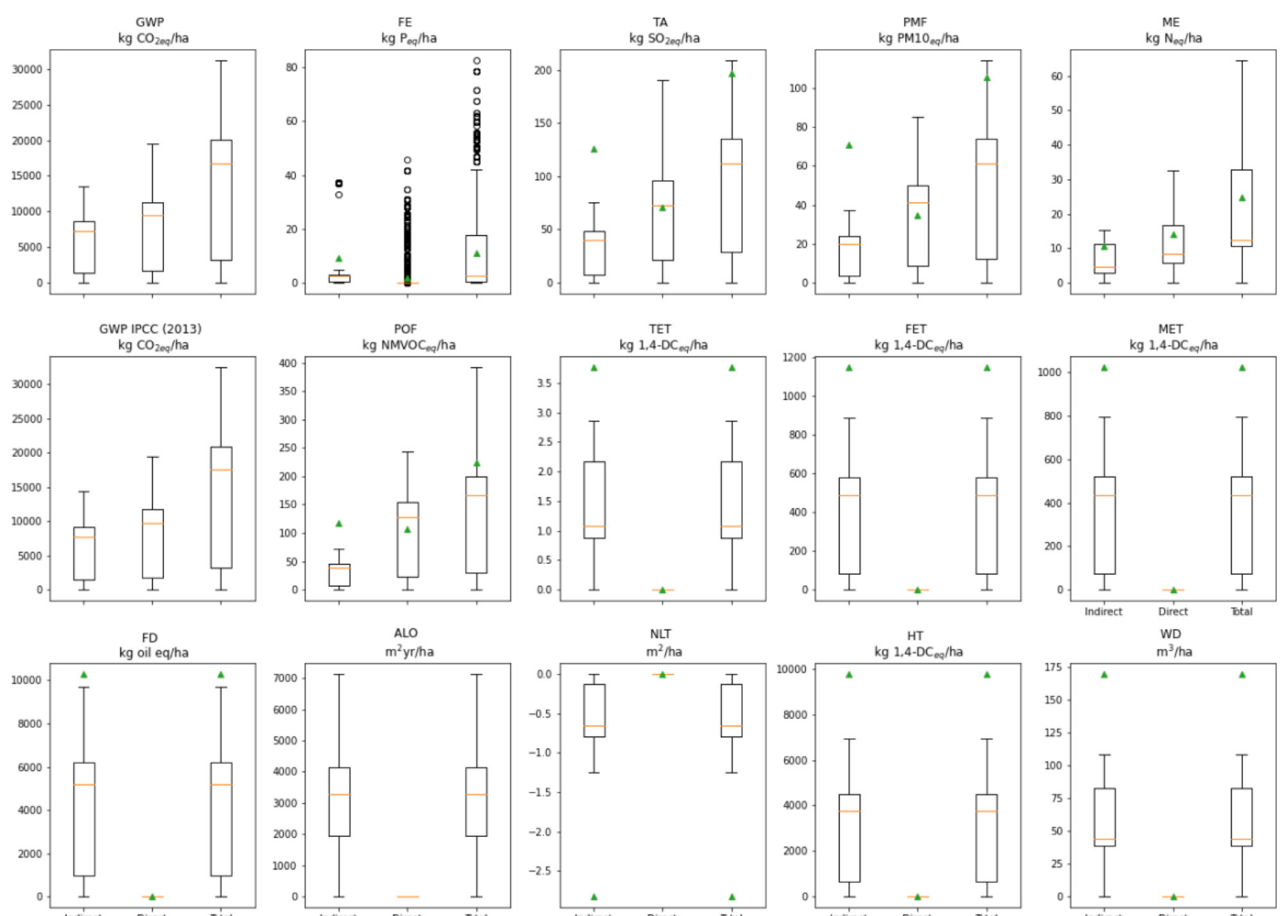

Fig. 7. Boxplots for total, direct and indirect impacts for peri-urban agriculture production in the Metropolitan Area of Barcelona (AMB). Green triangles in boxplots represent the means. In the case of climate change and agricultural land occupation means are outside of the range of the boxplot thus not shown here.

are on average 40 kg P_2O_5 (17,4 kg P) per ha yr. Total mineral fertilizer and manure P inputs in the AMB are 172.5 and 97.8 tons P_2O_5/yr, respectively. Total P inputs are 270.4 tons P_2O_5/yr.

P removed with harvest is the largest output (Fig. 6), followed by the additional phosphate runoff. The latter is a balancing parameter, which appears as relevant for some plots displayed as outliers (Fig. 6). Total PO_3^{4-} base and PO_3^{4-} residual emissions are 0.48 and 35.9 kg P_2O_5/yr, respectively. Total P removed with the harvest is 233.9 kg P_2O_5/yr. Total P outputs are 270.4 kg P_2O_5/yr.

3.2. Direct and indirect impacts from peri-urban agriculture in the AMB

Maps of all (direct, indirect and total) impacts are shown in Appendix A, Section 14. Boxplots for these maps are shown in Fig. 7. For MET, TET, FET, HT, FD, NLT and ALO average indirect impacts are outside of the interquartile range because the distributions are highly skewed. Therefore, mean values are not representative of most plots and should be used carefully for these impacts.

Direct impacts of GWP100, GPW100_IPCC, TA, PMF, ME and POF have highly skewed distributions, which leads to highly skewed total impacts too. Outliers play a significant role in the calculation of average impacts thus, they should be used carefully. In the case of ME, nitrate additional emissions for outlier plots (Fig. 5) lead to highly skewed direct and total impact distributions (Fig. 7). The same occurs for FE (Fig. 7) with phosphate additional emissions (Fig. 6).

Overall, variability is large, especially for direct impacts. For instance, total climate change can vary from 5 to 30 tons of CO_{2eq} per ha yr (0.5 to 3.0 kg CO_{2eq} per m^2 yr) and climate change due to direct emissions varies from 0 to 20 tons of CO_{2eq} per ha yr (0 to 2.0 kg CO_{2eq} per m^2 yr). Yet, total climate change falls within previously reported ranges for peri-UA for instance by Dorr et al. (2021) varying from 0 to 6 kg CO_{2eq} per m^2. Because large part of the variability is expected to come from the different requirements of different land uses and crops, Fig. 7 has been calculated grouping plots per land category (Appendix A, Section 15). These results will be discussed in Section 4.3.

3.3. Regionalized eutrophication impacts

Regionalized ME and FE midpoint, endpoint and damage level impacts' maps are shown in Appendix A, Section 16, while boxplots for these maps are shown in Fig. 8. In the case of ME, non-regionalized CFs lead to underestimated impacts, while for FE they lead to overestimated FE impacts compared to AMB-specific CFs at midpoint level.

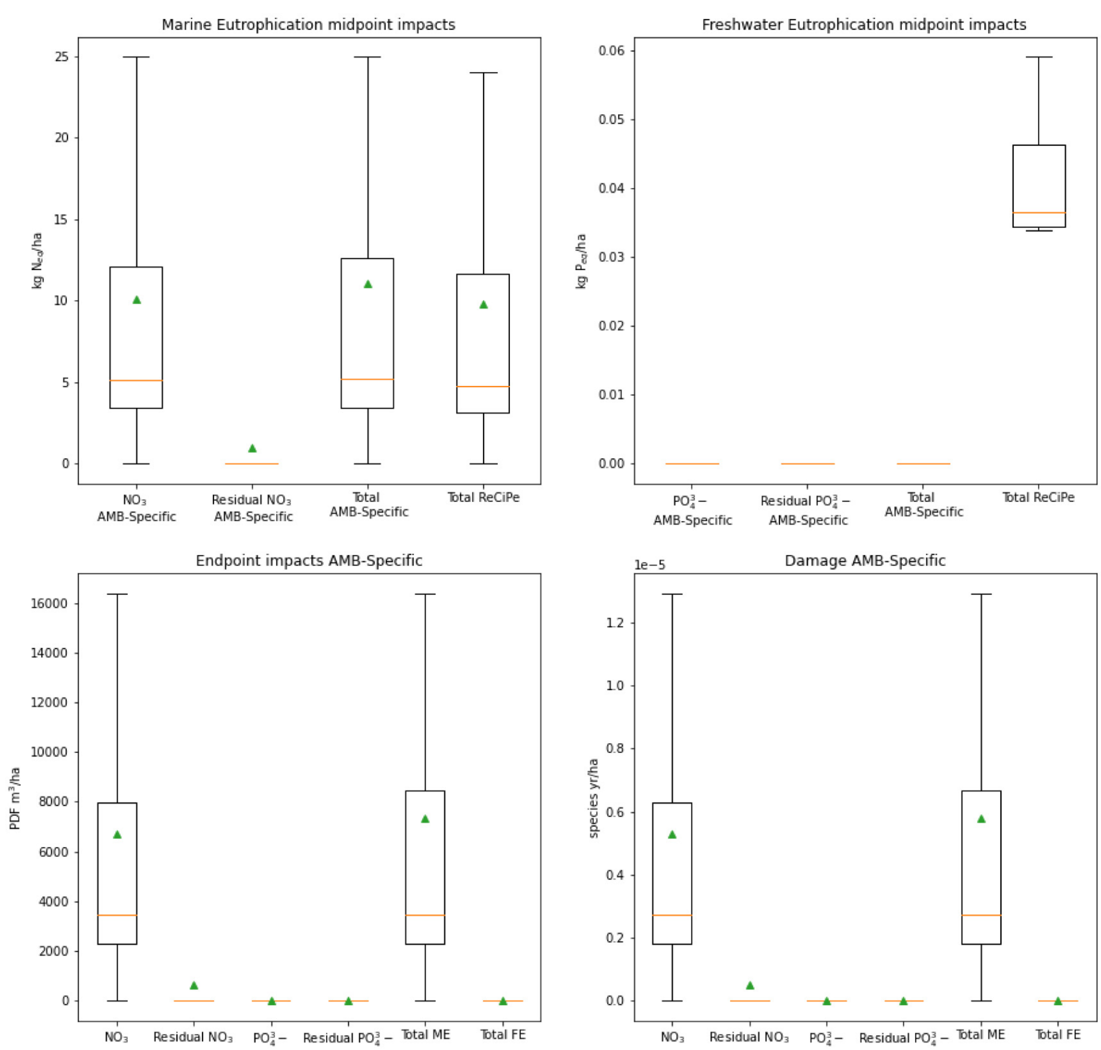

Fig. 8. Regionalized eutrophication impacts at midpoint, endpoint and damage levels for peri-urban agriculture in the Metropolitan Area of Barcelona (AMB) and comparison with non-regionalized results using ReCiPe CFs. For FE, mean values are not shown as the mean for ReCiPe CFs is out of range, at around 2 kg Peq/ha. Green triangles in boxplots represent the means.

ME from peri-UA in the AMB leads to a larger damage than FE in terms of species loss. This damage is driven by nitrate emissions and the agricultural area of the Baix Llobregat is the hotspot where such loss would take place according to the maps.

4. Discussion

4.1. N and P sources play most significant role in peri-UA environmental sustainability

Three sources of N and P nutrients are considered in the inventory to cover the demand of crops: 1) mineral fertilizers – imported from outside of the AMB, 2) manure and 3) agricultural residues – both produced within the AMB. Manure resulted as the largest input of N to crops while mineral fertilizers, from abroad, was the largest source of P, as also found by Marull et al. (2021). In addition, imported mineral N fertilizers production appears as the largest contributor to the total impacts of vegetables and greenhouses – the dominant land uses in the AMB and for fruits, P fertilizers production is the largest contributors to indirect impacts (Appendix A, Section 17). Therefore, mineral fertilizers replacement to avoid upstream environmental damage is crucial for peri-UA sustainability in the AMB. For such purpose, local sources of nutrients could represent an opportunity for closing the cycle of nutrients within the metropolitan area, increasing circularity and self-reliance. In the case of manure, we considered undigested, solid state application. If digestion of manure is included, the impacts of this process should be further assessed and added to the life cycle, including any possible contamination coming from metals, antibiotics, etc. present in the digested manure. This also applies for agricultural residues, for which composting or other processing was not considered. Other organic sources of nutrients, currently less supplied to crops at the AMB and not considered in this LCA, yet available in the region, are wastewater treatment plant (WWTP) sludge and municipal organic waste compost, not included given the high level of uncertainty in secondary data as well as it appears compost is mostly used as a soil enhancer and not as a nutrient supplier and thus may affect other soil dynamics worth investigating further as they seem relevant for peri-UA. Primary data is required to include such nutrient sources, processes and soil dynamics, with a lower degree of uncertainty. Recovered struvite from WWTPs has been shown to provide local-circular nutrients capable of replacing the whole agriculture P demand for Barcelona (Rufí-Salís et al., 2020).

4.2. The importance of nutrients balances completeness and activity-specific EFs for robust of LCA estimates

Understanding the anthropogenic nutrient cycles to determine nutrient balances is fundamental in this study for the estimation of on-site emissions. N- and P-related emissions were estimated with different levels of detail, which determines direct and total impacts. For instance, crop-irrigation-specific Tier 2 EFs were used for the estimation of direct N_2O emissions from mineral fertilizer applications, instead of applying generic Tier 1 EFs that can lead to 15 % higher emissions than Tier 2 (Mendoza Beltran et al., 2022). Similarly, for NH_3 emissions, fertilizer-type and manure specific EFs were applied covering N in the form of urea for different inputs such as mineral urea or manure (Andrade et al., 2021). Base phosphate emissions are based on the P-SALCA model as implemented by ecoinvent which separates the effect of mineral fertilizers, manure and slurry/liquid sewage sludge (not included in the inventory). Using specific EFs for each type of input, land use/crops and type of practices e.g. irrigated or rainfed crops, leads to more representative emissions estimates, however it is a laborious data collection process.

In the case of nitrate leaching, we applied a generic EF from the IPCC guidelines from 2019 (De Klein et al., 2006) to the total N inputs including mineral fertilizers, manure and agricultural residues. This could be further improved by estimating and using crop-practice specific EFs per land use/crops in the area or by using on-site data for irrigation, soil and crop properties, cultivation and harvesting practices, etc., on mechanistic models to estimate these emissions (Andrade et al., 2021) however, this may be a laborious and data intensive process. We further considered all excess N to be emitted as additional nitrate leaching, which manifests in some (outlier) plots (Fig. 5). This assumption should be calibrated as large nitrate emissions dominate the distribution of direct and total impacts of ME for some plots. Similarly for phosphate, most plots have base runoff and few (outlier) have additional runoff (Fig. 6) but those with the latter have much larger direct and total FE impacts (Fig. 7).

N-related emissions considered in the LCI represent most processes of the N cycle (Andrade et al., 2021). Nitrification (relates to N_2O and NO_x), nitrate leaching (NO_3^-), denitrification (relates to N_2O) and volatilization (NH_3) were covered. Moreover, plant uptake and N fixation were covered too, all specific to the type of crop. A parameter not included was atmospheric deposition, given the lack of estimates for the peri-UA areas of the AMB. Studies have looked into atmospheric deposition of other substances, such as heavy metals in UA (Ercilla-Montserrat et al., 2018), as well as N deposition in Mediterranean evergreen forest nearby intensive agriculture areas (Avila et al., 2017). This last study looks into wet and dry N deposition on the canopy of forest. Yet, their estimates were not used as we considered them too uncertain to represent N deposition for all AMB peri-UA plots. Moreover, natural N deposition should not be included according to the PEFCR as N inputs should only be anthropogenic (European Commission, 2018). Further work looking into the effects of air emissions deposition in peri-UA could be explored.

4.3. Variability of LCIA results and weight of direct and indirect impacts is linked to management practices per land use

The results for all impacts for all peri-UA plots in the AMB showed large variability (Fig. 7). This picture changes dramatically when looking at separate land uses (Appendix A, Section 15). Variability for direct, indirect and total impacts is almost ruled out for vegetables, greenhouses, and other fruits irrigated and non-irrigated. Thus, management practices for these categories, which determine inputs and outputs, do not vary much among plots. Therefore, most variability in plots' impacts is associated with such management practices which depends on the land use, as expected.

Further, whether direct or indirect impacts dominate the total impact depends on the land use too (Appendix A, Section 15). For vegetables and greenhouse indirect impacts dominate total impacts, while for other fruits (irrigated and non-irrigated) direct impacts dominate, except in the case of FE and impacts for which no direct emissions are estimated. Vegetables and greenhouses – the dominant land uses in the AMB, have the largest total impacts of all land uses. As mentioned, these are mostly indirect i.e. arise with the production of feedstocks namely NPK compounds production as shown by the contribution analysis (Appendix A, Section 17). Replacing such mineral fertilizers for vegetables and greenhouses would avoid the dominant impacts for the region, while impacts of alternative sources of nutrients would have to be factored in for a complete assessment. Irrigated and non-irrigated fruit crops have the second largest total impacts and these are dominated by emissions at the plots namely N-related emissions and greenhouses estimated. In this case, effective ways to mitigate these emissions would avoid the largest share of impacts. Indirect impacts for other fruit crops are dominated by superphosphate calcium production (Appendix A, Section 17).

Irrigated and non-irrigated herbaceous crops' impacts show more variability than other land uses (Appendix A, Figs. S.46, S.47). This represents more varied practices for these types of plots. Also, depending on the impact direct or indirect impacts dominate. For TA, PMF, ME and POF direct emissions dominate showing the importance of N-related emissions for these plots. For GWP100 direct and indirect impacts contribute more or less equally, on average. FE impacts are rather different for plots with irrigated and non-irrigated herbaceous crops. Here the effect of phosphate emissions is clear in the irrigated plots displaying a highly skewed direct FE impact distribution (Appendix A, Fig. S.46). Measure tackling both direct and indirect impacts, could be effective in this type of plots.

In general, different importance of direct and indirect impacts for different land uses poses challenges for the sustainability of AMB's peri-UA where solutions may come from looking into each land use group management practices instead of applying a "suits-all" type of approach.

4.4. The importance of regionalizing the inventory and eutrophication impacts

The high-resolution regionalization of the inventory for peri-UA in the AMB is a fundamental contribution of this work. The inventory was spatialized and regionalized (Patouillard et al., 2018) by associating inputs to plots as well as by using municipal datasets per land cover, from the SIA model (Marull et al., 2021; Padró et al., 2020). This process tackles geographic uncertainty by associating municipal data to individual plots as an initial step to increase geographic representativeness of the inventory. Also, accounting for the specificities of the production per municipality and land use in the AMB, displayed variability of peri-UA production in the region. In addition, the regionalization of the inventory enables the use of regionalized characterization factors, leading to better estimations of the local impacts caused by peri-UA. We exemplify here such impact assessment regionalization with marine and freshwater eutrophication. Compared to non-regionalized CFs, midpoint eutrophication impacts can be overestimated - for freshwater, and underestimated - for marine water. Also, variability of ME is larger than that of FE, reflecting the larger variability of nitrate emissions in comparison to phosphate emissions. Moreover, averages do not represent large part of the plots and this is a fundamental difference with non-regionalized LCA which is mostly based on averaged data. This particularly holds for FE as the mean FE midpoint impact is around 2 kg Peq ha^{-1}, a value out of the interquartile range for all plots' results, with a median around 0.035 kg Peq ha^{-1}.

For endpoint and damage impacts, the maps show where species loss is more likely to occur, given current areas with peri-UA i.e. mostly around the agricultural area of the baix Llobregat (Appendix A, Figs. S.49, S.50). Here the delta of the Llobregat river brings along a series of biodiversity rich ecosystems such as wetlands, that despite a heavy fragmentation due to for instance transport infrastructures, still contain rich vertebrate and invertebrate fauna, not to mention the richness in plant life (Verdaguer Viana-Cárdenas, 2010). Such results further highlight the importance of inventory and impact assessment regionalization at high resolutions (Mutel et al., 2018). For some impacts, relevant spatial scales of analysis are not at country or global levels but at regional, watershed or subcontinental levels (Verones et al., 2020). For peri-UA, quantifying local damage on human health and terrestrial, freshwater and marine ecosystems from e.g. PMF, POF, acidification, human toxicity and ecotoxicity, should include further regionalization of CFs to represent the biophysical specificities of the region as exemplified here with eutrophication. Such further work should be undertaken with experts on impact assessment methods. Recent work providing soil–freshwater N fate factors at half degree resolution, could be further integrated, to cover possible freshwater eutrophication impacts caused by N (Zhou et al., 2022).

Despite the benefits of the inventory regionalization, the SIA dataset carries modeling uncertainty associated to its calculation. This uncertainty could be reduced by collecting on-site primary data to calibrate the regionalized inventories or it can be characterized with methods such as the multilevel modeling approach (Dai et al., 2020). Also, both inventory and impact assessment regionalization are data collection intensive processes but necessary for a proper assessment of peri-UA environmental impacts happening at local scales. Further regionalization in background datasets, could enable the identification of indirect impacts at the location that they take place e.g. urea production and its impacts could be related to specific producing countries. In our analysis we assigned those impacts to the plot using urea, where the emissions of its production are not taking place in reality.

4.5. A paradigm shift for green infrastructure in the local and global context

The importance of sustainable urban development for peoples wellbeing while preserving the ecosystems and surroundings balance for life support, has more recently returned to governments agendas with the adoption of the 2030 Agenda for Sustainable Development (United Nations, 2015) and its Sustainable Development Goals (SDGs).

Our high-resolution inventories, related to land use, as well as regionalized impacts, help understand the integration of peri-UA food provision within the global and local territory, in the form of metabolic flows and its direct and indirect impacts. Understanding such metabolic-territory (local and global) functional integration of food provision by peri-UA is essential to guide appropriate circular and sustainability strategies for the sector to transition towards a balance preserving life support systems. For example, we identified that replacing foreign sources of nutrients (e.g. mineral fertilizers) is vital for a sustainable peri-UA sector. A follow up question based on this study is the potential of local nutrient sources (e.g. digested manure, WWTP sludge, municipal compost, etc.) to avoid significant upstream impacts and increase circularity. A paradigm shift for green infrastructure in the metropolitan area would rely in such strategies as essential elements to promote an agroecological transition. This agroecological transition should aim at overcoming unsustainability of current peri-UA conventional systems considering human and ecosystems health at in-boundary and transboundary levels (FAO, 2017). Here we provided a fundamental baseline needed to further assess such transition scenarios at an appropriate level of analysis. A further step would quantify a broader set of direct and indirect environmental impacts of transition scenarios for peri-UA, highlighting areas where damage to ecosystems and human health trespass acceptable limits. A natural input for such scenarios is the AMB Land Use Master Plan (PDU for its acronym in Catalan) (Barcelona Regional, 2019) land use scenarios that vary the peri-UA areas as studied here, among relevant scenarios.

5. Conclusions

Peri-UA holds the potential of supplying specific local, sustainable foods, to nearby urban areas. It can also introduce environmental impacts depending on the local biophysical conditions and resources required to implement it. Assessing such impacts require sophisticated methods like regionalized LCA to assess the direct environmental impacts - those caused on-site, and indirect impacts- those caused up and down stream, at different geographic scales. Our case study for peri-UA in the Metropolitan Area of Barcelona showed that metabolic flows i.e. inputs and outputs to and from agricultural plots are determined by the land uses such as vegetables, fruits and herbaceous crops, as specific management practices are used per land use. In this sense, land uses determine direct and indirect impacts of the whole sector for the region. Likewise, depending on the land use, some impacts are driven by direct emissions, while others are driven by indirect impacts. For vegetables and greenhouses (the prevalent land uses in the AMB), foreign NPK compounds production to cover the N demand of crops causes indirect impacts dominating the total impacts, while for fruit crops direct emissions, such as nitrate and phosphate emissions, dominate the total. In addition, P mineral fertilizers to cover P demand of fruit crops is the largest driver of indirect impacts for this land use. In the case of herbaceous crops, whether direct or indirect impacts dominate the total depends on the impact: for TA, PMF, ME and POF direct emissions dominate showing the importance of N-related emissions for these plots. For GWP100, direct and indirect impacts contribute more or less equally, on average. FE impacts are rather different for plots with irrigated and non-irrigated herbaceous crops showing the impact of phosphate runoff for irrigated plots.

Regionalization of life cycle inventories enabled us to display geographic variability of metabolic flows and their direct and indirect impacts. Understanding how such variability originates from the management practices at the plots and how these depend on the land use is essential knowledge to guide appropriate circular and sustainability strategies for the sector. Furthermore, regionalization of the eutrophication impacts, helped highlight the importance of accounting for the biophysical aspects at the geographic scale at which peri-UA takes place i.e. local, which is a much finer scale than currently covered by regionalization of impact assessment methods in LCA, as impacts may be over or under estimated. In the case of the AMB, peri-UA eutrophication impacts could damage biodiversity hot-spot areas of the Baix Llobregat in the delta of the river. It is still essential to assess whether appropriate management of nutrients and the use of biological circular-local sources of nutrients may reduce direct impacts while also tackling the reduction of indirect impacts caused by the foreign production of mineral fertilizers. In this sense, this study provides a fundamental baseline at an appropriate level of analysis for peri-UA systems environmental impacts, which can be further used to study scenarios towards an agroecological transition of peri-UA in the AMB. For instance, peri-UA areas expansion as defined in the AMB Land Use Master Plan, inclusion of circular strategies such as the use of local sources of nutrients, the development of organic production, the assessment of future production under climate change, etc. are possible scenarios to study. Further work to strengthen robustness of results includes inventory calibration with primary data and inclusion of uncertainty of inventory data including emissions, inclusion of nutrient sources not included e.g. municipal compost and WWTP slurry, and compilation of pest control and water use inventories. Finally, more sophisticated methods to calculate nitrate and phosphate emissions accounting for biophysical characteristics, as well as other N and P related emissions not included could be further explored.

CRediT authorship contribution statement

Conceptualization: AMB, RP, MJLRA, JM, GV. Data collection: AMB, RP, MJLRA, MJE, GV. Data analysis: AMB. Visualizations: AMB. Writing, review and editing of manuscript: AMB, RP, MJLRA, JM, MJE, JC, AG, GV. Supervision: GV.

Data availability

Data will be made available on request.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.159519.

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