LETTER

A resource-based phosphorus footprint for urban diets

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Abstract

Large amounts of phosphorus resources, such as mineral fertilizers and manure, are mobilized globally to produce the food consumed in cities. Accounting for the use of these resources can allow cities to plan for interventions that reduce related pressures in their hinterlands, conserve resources, and lead to more circular food systems. In this study we calculate a resource-based phosphorus footprint for the food consumption in Brussels Capital Region and use it to compare different strategies towards increased circularity: waste reuse, waste reduction, dietary changes and shifts to locally produced food. The P footprint of an average inhabitant in Brussels is 7.7 kgP cap yr⁻¹, 10 times higher than the physical P consumption of 0.7 kgP cap yr⁻¹. About 60% of the total P inputs into food production are through manure, and the rest through mineral fertilizers; almost 80% of the inputs occur outside Belgium. Most of these inputs are related to the cultivation of feed for livestock, which is why a shift to vegetarian and vegan diets can reduce the footprint down to 4.8 kgP cap yr⁻¹ and 0.9 kgP cap yr⁻¹. To the contrary, consuming only food produced in Belgium would increase the footprint to 12 kgP cap yr⁻¹, mostly as a result of the high manure use in the north of the country. A reduction in the P footprint signifies an absolute reduce in total resource use that can alleviate pressures in the hinterland and promote a city’s transition towards circularity.

1. Introduction

What food we consume, and how this food is produced, affects the environment in many ways. One of these is by altering the nutrient cycles: excess use of mineral fertilizers and poor management of manure and human excreta have led to aquatic eutrophication at levels beyond the planetary boundaries for nitrogen (N) and phosphorus (P) at the global (Kahiluoto et al 2015, Steffen et al 2015) and European level (EEA and FOEN 2020). At the same time, the excess use of phosphate fertilizers is putting pressure on mineral P resources that are neither infinite nor uniformly distributed in the world. A better management of P resources, including a more efficient agricultural use of P fertilizers, the circularization of P flows and even the reconfiguration of the role of P in the food chain, could lead to more circular and sustainable phosphorus and food systems (Withers et al 2015, 2018) and thus less pressure onto ecosystems and resources.

Cities have a special role to play in the pathway towards a better P management in food systems. Cities concentrate food consumption, drive the demand for food production in their rural hinterlands (Wu et al 2019), and generate P-rich materials such as sewage sludge. Recent studies have been focusing on reusing these material flows, e.g. by reintegrating them into the food system (Harder et al 2020) and creating closed-loop nutrient systems at the urban scale, such as the Fertile City (Wielemaker et al 2019) and Harvest-to-Harvest (Wielemaker et al 2018). Yet, most of the P in urban effluents cannot be reused locally, in such short and tight cycles, because cities usually lack food production sites where these secondary P resources would be valued (Trimmer and Guest 2018, Wu et al 2019). In addition, focusing solely on reuse can lead to rebound effects, where the...
use of secondary (reused) P increases without a parallel decrease in total resource use (Zink and Geyer 2017).

Consumption-based approaches in environmental accounting (footprints) can offer a more complete picture of resource use than city-level studies, by including the sites where the city's food is produced and demand for P is generated (Munksgaard and Pedersen 2001, Lenzen et al 2007). Studies on phosphorus footprints are relatively new in scientific literature (Metson et al 2016, Hu et al 2020) and we can classify them in three broad categories: P emission footprints, P use footprints, and Life Cycle Assessment P (LCA-P) footprints. Emission footprints are studies focusing on P emissions into the environment. Authors either derive factors to estimate nutrient emissions along supply chains, such as in (Leach et al 2012, Metson et al 2020, Oita et al 2020), or use multi-region input-output models and their environmental extensions to assess nutrient emissions (Li et al 2019, Hu et al 2020). P-use footprints quantify the P input flows into the production of food consumed in a given region, like the ones by (Metson et al 2012) at the global scale and by (Nesme et al 2016) at the European scale. Both these studies estimate indirect P inputs for several countries and different types of crops, but only account for synthetic P fertilizer. Finally, LCA-P footprints combine elements from the P-use (Metson et al 2016) and LCA approaches (Grönnman et al 2016, Joensuu et al 2019), to attain a balance between including as much of the global hinterland as possible, while not disregarding flows downstream the city. The goal is to quantify the use of P associated with the provision of food to Brussels for one year, and estimate how much of this use comes from primary (virgin) and secondary (reused) P resources.

We focus on phosphorus embodied in Brussels’ food consumption (figure 1(b)), in the form of mineral fertilizer (Pf, primary) and manure (Pm, secondary) used for producing the food items consumed in Brussels. Following (Nesme et al 2016) we estimate these two input flows (indirect flows) using equations (1) and (2):

\[ P_{f,i,l} = Q_{i,l} \times f_{conv,i} \times \frac{1}{\text{Yield}_{i,l}} \times \text{Fert}_{i,l} \]  

\[ P_{m,i,l} = Q_{i,l} \times f_{conv,i} \times \frac{1}{\text{Yield}_{i,l}} \times \text{Manure}_{i,l} \]

where \( Q_{i,l} \) is the annual consumption of product \( i \) imported to Brussels from region \( l \); \( f_{conv,i} \) is the conversion factor used to convert food products to primary crops; \( \text{Fert}_{i,l} \) (kgP ha\(^{-1}\)) is the P fertilizer application rate in region \( l \) for crop \( i \), and \( \text{Manure}_{i,l} \) (kgP ha\(^{-1}\)) is the manure P application rate in region \( l \). Estimation methods for each term in equations (1) and (2) are documented in the next section.

In addition, we account for the direct P flows, i.e. P actually flowing in and out of the city through food (\( Q_f^P \)), sewage sludge (\( Q_s^P \)) and food waste (\( Q_w^P \)). We do not account for any other P flows in the supply chain between agriculture and consumption, notably excluding the food processing industry. Although the food and feed industry often absorb parts of waste streams generated elsewhere in the chain, they use very little additional P inputs, and so contribute minimally to the total use of P resources. In the case of oats and beef from Finland, for example, this contribution has been 0 and 0.02% respectively (Grönnman et al 2016, Joensuu et al 2019).

For the purposes of this study we define the P footprint (FP\(_P\)) of Brussels’ food consumption as:

\[ \text{FP}_P = P_f + P_m - P^\text{out}_m - Q^P_{w,rc} - Q^P_{s,rc} \]

where \( P_f \) and \( P_m \) are the sums of the \( P_{f,i,l} \) and \( P_{m,i,l} \) over all products \( i \) and producing regions \( l \), \( P^\text{out}_m \) the manure excreted by the animals producing animal

2. Methodology

2.1. System definition

The P footprint approach developed in this study combines elements from the P-use (Metson et al 2012, Nesme et al 2016) and LCA approaches (Grönnman et al 2016, Joensuu et al 2019), to quantify and compare how different interventions affect the P footprint and thus the different parts of Brussels’ hinterland.

(a) quantify the primary and secondary P embodied in the diet of the inhabitants’ of Brussels,

(b) define the extent of Brussels’ hinterland by identifying where this P comes from, and

(c) compare how different interventions affect the P footprint and thus the different parts of Brussels’ hinterland.
products for Brussels, and $Q_{ss,rec}^P$ and $Q_{fw,rec}^P$ the parts of sewage sludge and food waste generated in Brussels that are reused in the food system.

To estimate the input flows, we combine data on fertilizer use and manure generation, with data on food products from the physical part of the multiregional hybrid supply and use tables (MRHSUT) at the subnational level for Belgium. The MRHSUT at the subnational Belgian level is obtained by disaggregating Belgium into Brussels, Flanders and Wallonia, within the hybrid version of EXIOBASE using a regionalisation approach and a balancing procedure (see section 2 in (Towa et al 2020) for more details). The supply table shows the production volumes of 164 goods and services, including food products, and the use table shows the amount of different inputs that have been used to produce these 164 goods and services. More information on the background of supply and use tables can be found in (Eurostat 2008, Miller and Blair 2009). The term ‘multiregional’ refers to a coverage of multiple countries and regions worldwide, namely 42 countries and 5 rest of world regions; ‘subnational’ refers to the 3 Belgian regions: Brussels, Flanders and Wallonia. Such a global database in physical units (ton) with a subnational Belgian specificity allows to trace back the origins of the food items consumed in Brussels. Figure 1(a) offers an overview of the system and methodology, further details of which are presented in the following sections.

2.2. Estimation of P inputs and outputs

2.2.1. Food consumption and primary crops

The starting point of the analysis is the final demand for food products ($\sum_i Q_{i,l}$) in the MRHSU tables. Excluding non-food agricultural product groups, and groups that are not consumed in Brussels, we produced a dataset with 19 food product groups (table 1). In order to be able to estimate nutritional information, including P content, we assigned a representative product to each group. Three of these groups are too diverse to sufficiently be represented by one product: fruit and vegetables, dairy products, and other crops, including coffee, tea, cocoa and spices. We disaggregated these groups into their constituent food items using information from the Belgian Household Budget Survey for 2014 (Statbel 2017), and calculated weighted averages for the nutrient content of each of the three groups. We assumed a 50–50 distribution of beer and wine in the beverages group and that the shares of each food item in their specific groups are the same in 2011 as in 2014.

We converted the amounts of processed food and feed, such as wheat flour and vegetable oil, to
### Table 1. Food product groups included in the analysis, and their characteristics: amounts consumed in 2011, dry matter (DM), energy and P content, and conversion factors used for their conversion to primary crops ($f_{\text{conv}}$). The last two columns provide food consumption estimations for the vegetarian and vegan diet scenarios (nec: not elsewhere classified).

| Food product                     | Representative product | Final consumption households (tDM)$^a$ | DM (%)$^b$ | Energy (kcal/100gr)$^b$ | $P$ (mgP/100gr)$^b$ | $f_{\text{conv}}$ (−)$^c$ | Final consumption — vegetarian (tDM) | Final consumption — vegan (tDM) |
|----------------------------------|------------------------|--------------------------------------|------------|--------------------------|---------------------|-------------------------------|-----------------------------------|---------------------------------|
| Wheat                            | Flour for white bread  | 69 934                               | 85         | 327.0                    | 90.0                | 1.01                          | 94 092                            | 103 511                         |
| Cereal grains nec                | Corn                   | 1048                                 | 89         | 361.0                    | 99.0                | 1.00                          | 1409                             | 1550                            |
| Vegetables; fruit; nuts          | Weighted average       | 56 445                               | 15         | 50.4                     | 33.4                | 1.00                          | 75 943                            | 83 545                          |
| Crops nec                        | Weighted average       | 20 650                               | 55         | 161                      | 57                  | 1.00                          | 27 784                            | 30 565                          |
| Poultry                          | Weighted average       | 37 760                               | 30         | 161                      | 204                 | na                            | 0                                 | 0                               |
| Meat animals nec                 | Pork steak raw         | 52                                   | 28         | 126                      | 210                 | na                            | 0                                 | 0                               |
| Animal products nec              | Honey                  | 41                                   | 82         | 323                      | 4                   | na                            | 0                                 | 0                               |
| Raw milk                         | Milk                   | 228                                  | 13         | 65                       | 89                  | na                            | 0                                 | 0                               |
| Fish and other fishing products  | Fish, lean raw         | 1467                                 | 19         | 76.0                     | 199                 | na                            | 0                                 | 0                               |
| Products of meat cattle          | Beef entrecote         | 51.8                                 | 33         | 177.0                    | 166                 | na                            | 0                                 | 0                               |
| Products of meat pigs            | Pork steak raw         | 9969                                 | 28         | 126                      | 210                 | na                            | 0                                 | 0                               |
| Products of meat poultry         | Poultry                | 1946                                 | 30         | 161                      | 200                 | na                            | 0                                 | 0                               |
| Meat products nec                | Sausage pork-beef raw  | 2211                                 | 36         | 226.0                    | 414                 | na                            | 0                                 | 0                               |
| Vegetable oils and fats          | Oil, salad             | 8620                                 | 99         | 883.0                    | 0                   | 1.87                          | 11 598                            | 12 759                          |
| Dairy products                   | Weighted average       | 14 546                               | 19         | 105                      | 175                 | na                            | 19 571                            | 0                               |
| Processed rice                   | Rice, hulled           | 42                                   | 87         | 347.0                    | 145                 | 1.00                          | 56                                | 62                              |
| Sugar                            | Sugar, granulated      | 40 907                               | 100        | 400.0                    | 0                   | 6.57                          | 55 038                            | 60 547                          |
| Beverages                        | Beer, wine             | 7′977                                | 12         | 57                       | 18                  | 1.37                          | 10 733                            | 11 807                          |
| Fish products                    | Fish, lean raw         | 10                                   | 19         | 76                       | 199                 | na                            | 0                                 | 0                               |

$^a$ (Towa et al 2020).

$^b$ (Nobel 2018).

$^c$ (FAO 2000) and (Scherer and Pfister 2016), in [kg primary crop kg$^{-1}$ derived product].
primary crops using the conversion factor \( f_{\text{conv}} \). Conversion factors account for the mass fraction of the primary crop to the derived product \( \bar{f}_{\text{prim}} \), values from (FAO 2000), as well as its monetary value share \( \bar{f}_{\text{value}} \), to avoid double-counting (Scherer and Pfister 2016). A full list of the conversion and their constituents can be found in the supplementary material (SM2 (available online at stacks.iop.org/ERL/16/075002/mmedia)). At this point we further refined the product resolution in some of the groups, matching them to their country of origin: we assumed for example that sugar crops are sugar beets when exported from European countries and sugar cane when exported from Latin America. A full list of these assumptions is given in SM1. We thus assumed that the primary crops are cultivated in the country exporting them to Brussels, unless this country does not produce any such crop.

2.2.2. Feed requirements of livestock

For animal products, we substitute \( Q_{d,j} \) in (equation (1)) with the equivalent feed crops used by the livestock sectors supplying these animal products, and with the equivalent feed crops and grass ingested by cattle in (equation (2)), since grasslands were assumed to be fertilized only with manure. We converted animal products consumed in Brussels to the equivalent feed intake of the producing animals using the phosphorus use efficiency of each livestock sector \( k \) and animal rearing country or region \( l \) (PUE\(_{k,l}\), equation (4)). For the cattle and milk sectors, we further used a \( f_{\text{gr}} \) factor to differentiate between P ingested through feed and through grazing (equation (5)).

\[
P_{\text{UE}_{k,l}} = \frac{P_{\text{in animal product produced by sector } k}}{P_{\text{digested from animals in sector } k}} \quad \text{(4)}
\]

\[
f_{\text{gr}} = \frac{P_{\text{ingested through grazing}}}{P_{\text{ingested as feed}}} \quad \text{(5)}
\]

Six of the animal-rearing regions \( l \) (Flanders, Wallonia, France, the Netherlands, Germany, and the UK) provide Brussels with 92% of its total consumption of animal-based food products. For these six regions, we disaggregated the total feed intake of each sector into different feed crops \( j \) imported from feed cultivating countries or regions \( m \), using the MRHSU tables. The rest of the animal products are imported from the rest of Europe (4%), Australia (2%), Middle East (1.2%) and rest of the world (0.8%). For these world regions we assumed that all compound feed is wheat produced within the respective regions.

2.2.3. Yields, fertilization rates and manure

Values on crop yields are from official statistics for Belgium (Statbel 2014) and from the FAO database for all other countries and regions (FAOSTAT 2019). Synthetic P fertilizer application rates for Belgium are from official Flemish and Walloon sources (REEW 2018, Departement Landbouw en Visserij 2020); rates for manure use are based on actual livestock numbers and manure management practices (Papangelou and Mathijs 2021). For the other six main supplying countries and regions we used fertilization and manure application rates from national and regional studies (Cooper and Carliell-Marquet 2013, Smit et al 2015, van Dijk et al 2016, Le Noë et al 2017, Le Noe et al 2020, Rothwell et al 2020), supplemented and cross-checked with official statistics (DEFRA 2012, DES-TATIS 2013, Agreste 2017, Eurostat 2020).

For all other countries and world regions we combined data on fertilizer use from the International Fertilizer Association (Heffer 2013) with data on cultivated areas from (FAOSTAT 2020a) to derive P-fertilization rates for (equation (1)). For three of the crops (coffee from Vietnam, cocoa from Brazil and Peru) we use information on fertilization rates for 2009 from (Nesme et al 2016). Finally, we approximated manure application rates using the method proposed in (Sheldrick et al 2003), combining FAOSTAT data on meat production (FAOSTAT 2020b) and P excretion rates (Sheldrick et al 2003). The same approach was used to approximate the P available in manure generated by the livestock supplying Brussels with animal products (\( P_{m_{\text{out}}} \)). Details are given in SM1.

2.2.4. P in urban waste streams

Amounts of household food waste in BCR were estimated based on a yearly per capita production of 89.5 kg cap yr\(^{-1} \) in fresh matter, 70% of which are plant-based food items and 30% animal-based (Zeller et al 2020). Food waste characteristics such as dry matter and phosphorus content were assumed to be the same as the corresponding food groups (details in SM1, table S8). Wastewater and sewage sludge quantities and characteristics were taken from different official sources as documented in (Papangelou et al 2020). In our analysis we assume that 100% of the P in compost, digestate and sewage sludge is potentially reusable, an assumption that leads to overestimated amounts of P reused. However, since P in urban waste streams (wastewater, food waste) is smaller than the input flows by several orders of magnitude, we find the assumption to hardly affect the study’s final results and conclusions. Other streams of urban food waste, notably retail waste, are not included in this study. Retail waste for the USA have been estimated to represented up to 10% of total available food (Xue et al 2017), indicating that our results may be underestimating the actual P footprint of food consumption.
2.3. Scenarios and indicators
In order to compare the effect that different interventions could have on Brussels’ P footprint, we developed five scenarios (table 2). Each one represents a theoretical best case of a strategy for increased circularity. We chose to work with best cases, since we see the comparison as an attempt to set the theoretical boundaries of possibility for phosphorus circularity in Brussels, and to offer an absolute upper limit as a benchmark for monitoring strategies and transitions towards increased circularity. If the scenarios would be to be implemented, they should be refined to account for further factors, e.g. appropriate waste treatment technologies for the reuse scenarios, or the nutritional adequacy of the proposed diets in the vegetarian and vegan scenarios (for a comparison of the protein and P contents of the diets see SM1 table S10).

We focused on consumption-based strategies, e.g. shifts to vegetarian and vegan diets or to locally produced food, and strategies that can be influenced directly by local authorities (waste reuse), rather than supply-side interventions, such as the adoption of agro-ecology principles or precision agriculture techniques. Supply-side interventions could have an important effect on the P footprint, because the type of production system greatly influences nutrient flows in a food system (Le Noë et al 2017, Papangelou and Mathijs 2021). The local scenario, where all food is produced in Belgium, offers a first insight into the influence of the production system to the P footprint. Nonetheless, implementing fully-fledged supply-side scenarios in a study that covers food production globally would require additional analysis that goes beyond its scope. To address this issue, we calculated an exploratory ‘precision agriculture’ scenario as part of the sensitivity analysis, where we assumed a 10% reduction in the fertilization rates, in Belgium and the rest of the world.

3. Results
3.1. P footprint of Brussels’ diet: direct and indirect flows
The per capita P footprint of food consumption in Brussels was 7.7 kgP cap yr⁻¹ in 2011. The total indirect inputs were 11.4 kgP cap yr⁻¹, 4.6 kgP cap yr⁻¹ of which were supplied to crops as fertilizer ($P_f$), and 6.7 kgP cap yr⁻¹, as manure ($P_m$, figure 2(a)). Each inhabitant in Brussels is consuming 0.7 kgP cap yr⁻¹ directly through the food they buy (figure 2(b)), more than 10 times less than the amount used to produce this food. 0.15 kgP cap yr⁻¹ are thrown away, while the rest 0.55 kgP cap yr⁻¹ are digested and eventually end up in the city’s wastewater management system. Two thirds of the 0.7 kgP cap yr⁻¹ are imported in Brussels through animal products, especially domestically produced meat (figure 2(a)).

About 95% of all the P inputs into the system are used by the livestock sector (figure 2(b)). The main reason for this is the low PUE of the production of animal-based food products (~4%), compared to that of plant-based ones (39%). Comparable PUEs of 0.22% and 62% have been reported for the production of beef (Joensuu et al 2019) and oat flakes (Grönman et al 2016) in Finland. Although P inputs into the production of fruit and vegetables are underestimated, their contribution to the total footprint is small enough (0.2%) to only marginally influence the final footprint value (see also SM1, section 2.3).

3.2. Domestic and global P flows
Around 60% (0.44 kgP cap yr⁻¹) of the food and P consumed in Brussels is produced domestically, in either Flanders or Wallonia. Half of this amount (0.22 kgP cap yr⁻¹) is from meat, mostly poultry, imported from Flanders (figure 3(a)). The next three most consumed groups in Brussels are meat from Wallonia (11% of all consumption), fruit and vegetables from Wallonia (9%), and cereals from Flanders (6%). Most of the input P flows into Brussels are thus related to meat production in Flanders, both for mineral fertilizer (2.34 kgP cap yr⁻¹ or 50% of total $P_f$) and for manure (2.13 kgP cap yr⁻¹ and 36%), used within Flanders but also abroad, in the regions where primary feed crops used by the Flemish livestock sector are produced. When we allocate P inputs ($P_f$ and $P_m$) to primary crops rather than food products (figure 3(c)), regions outside Europe, mostly Australia and N. America are the main producing countries, followed by France and Germany. France, Germany and Asia contribute mainly with oil crops used for feed, spending 0.9, 0.7 and 0.45 kgP cap yr⁻¹ of total P inputs to grow crops that are used to feed Brussels. Only 10% and 5% of the P used for either food or feed comes from crops grown domestically in Flanders and Wallonia, and thus the amounts of fertilizer and manure actually applied within the regions are 0.04 and 1.24 kgP cap yr⁻¹ for Flanders, and 0.04 and 0.6 kgP cap yr⁻¹ for Wallonia. Thus, Brussels has a further reaching hinterland than data on imports of food alone can reveal, and P resources throughout the world are mobilized to provide the city’s diet.

3.3. P footprint under different scenarios
Interventions downstream the city only have a small impact on the P footprint, compared to upstream ones (figure 4). Reusing 100% of the P in the city’s waste streams reduces the footprint from 7.7 to 7.1 kgP cap yr⁻¹. If all food waste is avoided, the FP_P is further reduced to 6.8 kgP cap yr⁻¹, thanks to the avoided P inputs into the production of the avoided waste. Interventions that reduce the consumption of animal-based products have a much larger effect on the FP_P. Vegetarian and vegan diets have P footprints of 4.9 kgP cap yr⁻¹ and 0.9 kgP cap yr⁻¹, even if little manure or none at all are produced in these cases.
Table 2. Description of the five scenarios developed and their impact on the calculation of the P footprint (equation (3)).

| Scenario | Description | Factors affected in the FP<sub>p</sub> estimation (equation (3)) |
|----------|-------------|---------------------------------------------------------------|
| Baseline | —           | —                                                             |
| Reuse    | All food waste and sewage sludge are treated and reused into food production | QTY<sub>P,<sub>n</sub><sub>re</sub> = QTY<sub>P,<sub>n</sub><sub>re</sub> <br> QTY<sub>P,<sub>c</sub><sub>re</sub> = QTY<sub>P,<sub>c</sub><sub>re</sub> <br> QTY<sub>P,<sub>n</sub><sub>re</sub> = QTY<sub>P,<sub>n</sub><sub>re</sub> = 0 <br> P<sub>f</sub>, P<sub>m</sub>, P<sub>out</sub><sub>m</sub> ↓ due to the avoided production |
| FWA      | Food Waste Avoidance: all food waste is eliminated | P<sub>f</sub>, P<sub>m</sub> ↓ due to the avoided feed production <br> P<sub>out</sub><sub>m</sub> ↓ due to fewer animals in production |
| Vegetarian | Isocaloric substitution of meat products with dairy and plant-based (table 1) | P<sub>f</sub> ↓ due to the avoided feed production <br> P<sub>m</sub>, P<sub>out</sub><sub>m</sub> = 0 due to no animals in production |
| Vegan    | Isocaloric substitution of all animal products with plant-based ones (table 1) | P<sub>f</sub> ↓ due to the avoided feed production <br> P<sub>m</sub>, P<sub>out</sub><sub>m</sub> change, following the yields, fertilization rates, and PUEs in Flanders and Wallonia |
| Local    | All food products are sourced locally: imported products were allocated equally to Flanders and Wallonia. Products not produced locally, such as coffee, are eliminated from the diet. | |
Figure 3. (a) Indirect P flows through fertilizer ($P_f$) and manure ($P_m$), and direct flows through food items ($Q_p$) per producing country or region in kgP cap yr$^{-1}$; (b) P footprint per food group consumed and (c) P indirect inputs per primary crop grown either for food or feed, in kgP cap yr$^{-1}$; (d) contribution of fertilizer and manure to total indirect flows per producing country or region, and (e) contribution of food and feed crops to total amounts of crops exported to Brussels per producing country or region.
A Papangelou et al.

Figure 4. Comparison of the annual per capita P footprint of food consumption in Brussels, and its components under alternative scenarios. FWA: Food Waste Avoidance. For an overview of the scenarios see table 2.

resource-based phosphorus footprints. Metson and colleagues (2012) estimated the mineral fertilizer P inputs related to food consumption in several countries in the world, and found ‘mineral P footprints’ ranging from 0.45 kgP cap yr$^{-1}$ for Rwanda to 6.09 kgP cap yr$^{-1}$ for the USA and 7.02 kgP cap yr$^{-1}$ for Argentina. For Belgium, they report 5.21 kgP cap$^{-1}$ for 2007, a value close to the 4.6 kgP cap$^{-1}$ we found as inputs through mineral fertilizers for 2011. Nesme et al (2016) reported a sum of direct and indirect fertilizer flows of 2.9 kgP cap yr$^{-1}$ for EU27 and 2009, although their analysis is based on trade of food and fertilizers, and does not include animal-based products. These resource-oriented studies, though, do not include the use and generation of manure and so exclude important amounts of P inputs: in the case of Brussels manure contributes 6.7 kgP cap yr$^{-1}$, almost 60% of the total P inputs. Additionally, the exclusion of manure fails to account for the fact that food systems are not only nutrient consumers, but also nutrient producers. 3.75 kgP cap$^{-1}$ are produced annually by livestock providing Brussels, ~32% of the total P inputs. Although manure is often treated as a waste stream, and sometimes it is, it is also a local, renewable nutrient source that should not be disregarded when analysing the circularity and resource use of food systems.

4.2. Towards a more circular urban food system

Strategies considered to promote circularity in food systems often include the reuse and recycling of nutrients through waste reuse and valorization (closing the loop) or a shift to shorter and more local systems (narrowing the loop). The results of such strategies, however, strongly depend on the local context. The increased FP$P$ of the local diet in our analysis, for example, reflects some of the particularities of food production in Belgium, especially Flanders. The most important of these particularities is the high manure use: on average, 29 kgP ha$^{-1}$ of manure are applied onto Flemish croplands. Germany and the Netherlands are the only regions with comparable figures of 24 (this study) and 21 kgP ha$^{-1}$ (Smit et al 2015), whereas the rates in all other regions are below 10 kgP ha$^{-1}$. Even though the high availability of manure in Flanders also means lower P fertilizer usage than in most countries, the total P inputs into crops produced in Flanders are still higher than most other places in our dataset. Another reason why food produced in Flanders is so P-costly could be traced back to model choices. In our analysis, we assume that all P in feed comes from feed crops disregarding mineral P-additives. Nonetheless, such additives could make up for a substantial share of the animals’ diet in intensive livestock production systems, such as that of the Netherlands (Smit et al 2015), or Flanders (Coppens et al 2016). Accounting for mineral P supplements would give a more accurate, probably lower, figure for the P footprint of food coming for Flanders; it would not, however, change the results dramatically (details in SM1, section 2.1).

Our results further indicate that downstream interventions in the city’s waste management system (closing the loop) are less effective than upstream ones in preserving P resources and fostering P-circularity. Reusing all P generated in the city reduces the P footprint by a shy 8%. Eliminating all food waste reduces the footprint by almost 12%: an improvement, albeit a small one compared to diets shifts. This is because although food waste represents around 25% of all the food purchased in Brussels (Zeller et al 2019), mostly plant-based food items end up to waste. (70% according to (Zeller et al 2020)). As a result, the avoided production has only a small impact on the total FP$P$, that is mostly comprised by P
inputs into animal production. These conclusions are in line with other studies that have addressed environmental concerns related to P embodied in food (Metson et al. 2016), waste management (Hamilton et al. 2015) or urban food (Boyer and Ramaswami 2020). We also find that local food production is more P-costly; however, most of the extra P used in local production is from manure, a secondary, renewable P-resource. This observation is indicative of the potentially important influence that supply-side interventions, not included here, can have on the FP_P. A 10% reduction in the fertilization rates worldwide, following for example the adoption of precision agriculture techniques, would cause a 15% decrease in the FP_P down to 6.4 kgP cap yr⁻¹ (see also figure S4 in SM1). Future work exploring the full potential of such supply-side scenarios would expand our understanding of how a wide range of circularity-oriented strategies influence P use in the food system.

4.3. Limitations and implications of model choices
One of the greatest sources of uncertainty in our results is the variety of data sources they are based upon, especially regarding animal products. This means that: (a) data are often the result of modelling, thus carrying the inherent assumptions of the models that generated them; (b) some data refer to different years than 2011 (e.g. European data from (van Dijk et al. 2016) are for 2005, some modelled data for the livestock sector in Belgium refer to 2014), and to sub-regions instead of the whole country (e.g. German data from (Theobald et al. 2016)), and (c) the analysis of use tables was replaced by a rough approach for animal products coming from the rest of Europe, Australia and Middle East. We have been confirming the quality of the input data by cross-checking values when possible, and by performing reality checks in the intermediary results. What is more, we tested the robustness of our results against some key assumptions and parameters, such as the PUE of the livestock sector, the use of P additives in livestock diets and the allocation method used for crops that give multiple products, by running the model with alternative values for these parameters. Our results are relatively stable for all alternatives tested (details in section 2 in SM1). The highest deviation of ~70% from the original value was observed when assuming higher PUEs for the livestock. Choosing a different method to allocate P inputs to different products derived for the same crop also caused an increase of ~23% to the model, illustrating the significant effect of value-laden modelling choices to the final result.

The adopted approach has the advantage of making the best use of available information on P flows in the agri-food system of Belgium. As Belgium provides Brussels with around 60% of its food, and is the potential recipient of the city’s effluents, we find it important to prioritize accuracy and detail in domestic production. Since the method we used for Flanders and Wallonia is based on existing P flow analyses, we looked for the same type of information to build the dataset for other exporting regions, namely France, the Netherlands, Germany, the UK and the rest of Europe. Procuring data from region- or country-specific studies can capture local differences that are lost in the global data from FAOSTAT, e.g. in manure management or feed mixtures. For example, we found a higher manure-to-fertilizer ratio for all regions we treated individually (figure 3(d)). This partly reflects the high livestock densities in some of these regions, whereas it can also be an indication of a systematic underestimation of manure use in global-scale datasets (Potter et al. 2010, Mekonnen and Hoekstra 2018). Continuing to refine manure accounting in P footprints will provide metrics relevant from a Circular Economy perspective, since manure is the most important secondary nutrient resource in food systems.

5. Conclusion
In this study we developed a resource-based P footprint for an urban diet and used it to quantify the P embodied in the food consumed in Brussels Capital Region. This resource-based P footprint that accounts for indirect P flows and secondary P sources, can complement emissions-based approaches, and offer a tool for assessing food system interventions towards increased circularity and greater resource efficiency. We found that food consumed in Brussels requires as much as 10 times its P content to be produced. Most of the inputs are connected to livestock rearing, which is why a shift to a vegetarian or vegan diet would reduce the P footprint to almost half and a tenth of the current value respectively, while downstream interventions lead to only marginal improvements. Our results indicate that reducing P inputs in the food system through shifts in diets bears great benefits for the transition towards circular food systems in cities, and that accounting for the absolute use of secondary and total resources is an indispensable component of circularity assessments. Further refining the methods to account for manure inputs and outputs, and adding detail with more region-specific data present future challenges towards a more precise resource-based P footprint that can help cities and regions achieve their circularity goals.

Data availability statement
The data that support the findings of this study are available upon reasonable request from the authors.

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