NUTRIENT FLOWS, BUDGETS & LOOPS

UNDERSTANDING, MEASURING AND FOSTERING CIRCULARITY IN URBAN FOOD SYSTEMS

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Summary

Cities are important nodes in a food system, even when they lack any agricultural activity. On the one hand, they concentrate consumption and drive food production; on the other, they produce big quantities of nutrient-rich effluents (organic waste, sewage sludge) that are typically wasted and not returned to the soil. Understanding and fostering the circularity of urban food systems through closing, narrowing, and slowing material and energy loops is therefore important in order to avoid this wastage of resources towards a more sustainable food system.

With this goal in mind, this thesis set off to analyze the food system of Brussels with a focus on circularity, by studying nutrient flows and budgets at different spatial scales. The main objective has been to understand what urban food system circularity can be, how to measure it, and what spatial scale is most appropriate for such an analysis. To answer this questions, I analyzed food and nutrient flows in the food system of Brussels Capital Region and its hinterlands, and tested different sets of circularity metrics.

Starting from the city-region itself, I used a multi-layer Material Flow Analysis to analyze the phosphorus and energy flows within the administrative boundary of the city. In addition, I compared how city-scale circularity strategies simultaneously affect the amounts of phosphorus potentially available for reuse and the net amounts of energy recovered from the system. On a second step, I expanded the system boundary to include Belgium, Brussels' domestic hinterland. Including the hinterland aimed at understanding whether the nutrients produced in Brussels are needed to cover demand in the producing agricultural lands, and how nutrient flows connecting Brussels with these lands would interact and interfere with local nutrient loops. To this end, I used an adapted GRAFS (Generalized Representation of

Agri-Food Systems) approach and a spatially explicit comparison between nutrient crop needs and local nutrient supply through manure and human excreta. Finally, I adopted a footprint approach and a functional rather than geographical system boundary, in order to account for the city's global hinterland. For this last step, I developed a resource-based phosphorus footprint that can be used to quantify direct and indirect phosphorus inputs into the food system, and to identify these parts of the hinterland that cannot be connected in a reciprocal nutrient exchange with the city.

The results indicate that valorizing phosphorus in urban sewage sludge and organic waste streams will not have to come at the expense of energy recovery for Brussels Capital Region. The lack of agricultural activity within the city, however, limits the usefulness of the city-level analysis and requires the inclusion of the hinterland. The agricultural lands in Flanders and Wallonia, Brussels' domestic hinterlands, have their own abundant sources of reused nutrients to absorb. The abundance of nutrients in Flanders, mostly produced by livestock intensively reared and fed by imported crops, dominates the analysis and the circularity assessment. Wallonia, on the other hand, could potentially benefit from absorbing some of the nutrients flowing out of Brussels. Designing a more circular food system for Brussels will have to strike a balance between bringing the hinterland closer, so that the extracted nutrients can be returned to the soil, and dealing with the highly problematic nutrient flows and budgets in these near hinterlands. Ultimately, and unsurprisingly, the most effective strategies towards increased circularity are those that radically redefine the consumption and production systems, e.g. shifting away from diets rich in animal products and from intensive livestock production systems.

Samenvatting

Steden zijn belangrijke knooppunten in een voedselsysteem, zelfs wanneer zij geen landbouwactiviteit hebben. Enerzijds concentreren zij consumptie en zijn ze de drijvende kracht achter voedselproductie; anderzijds produceren zij grote hoeveelheden aan nutriëntrijk afvalwater (organisch afval, zuiveringsslib) die meestal verspild worden en niet naar de bodem terugkeren. Het begrijpen en bevorderen van de circulariteit van stedelijke voedselsystemen door het sluiten, vernauwen en vertragen van materiaal- en energiekringlopen is daarom belangrijk om deze verspilling van hulpbronnen te vermijden en zo te komen tot een duurzamer voedselsysteem.

Met dit doel in het achterhoofd analyseert deze thesis het voedselsysteem van Brussel met een focus op circulariteit, door het bestuderen van nutriëntstromen en -budgetten op verschillende spatiale schalen. De belangrijkste doelstelling was het begrijpen wat de circulariteit van stedelijke voedselsystemen kan omvatten, hoe het te meten en welke spatiale schaal het meest geschikt is voor zulke analyse. Om op deze vragen te antwoorden analyseerde ik de voedsel- en nutriëntenstromen het Brussels Hoofdstedelijk Gewest en haar hinterland en testte ik verschillende maatstaven om circulariteit te meten.

Startend met de stadsregio zelf gebruik ik een meerlagige materiaalstroomanalyse om de fosfor- en energiestromen binnen de administratieve grenzen van de stad te analyseren. Daarenboven vergeleek ik hoe circulariteitsstrategieën op niveau van de stad tegelijkertijd de hoeveelheid fosfor beïnvloeden die potentieel beschikbaar is voor hergebruik alsook de netto hoeveelheid energie die wordt gerecupereerd van het systeem. In een tweede stap breidde ik de systeemgrenzen uit tot België, het binnenlandse hinterland van Brussel. Door het hinterland bij de analyse te betrekken kon onderzocht worden in welke mate nutriënten in Brussel geproduceerd de noden van de

landbouwarealen konden dekken, en hoe nutriëntstromen die Brussel met deze arealen verbinden interageren met lokale nutriëntstromen. Hiertoe paste ik de GRAFS (Generalised Representation of Agri-Food Systems) aanpak aan om een ruimtelijk expliciete vergelijking te maken tussen de nutriëntnoden van gewassen met het lokale nutriëntenaanbod in de vorm van nutriënten in mest en menselijke uitwerpselen. Ten slotte paste ik een voetafdrukmethode toe op een functionele eerder dan een geografisch afgebakend system om rekening te houden met het globale hinterland van de stad. Voor deze laatste stap ontwikkelde ik een input-gebaseerde methode om de fosforvoetafdruk te meten die kan gebruikt worden voor de kwantificatie van directe en indirecte fosfor-inputs in het voedselsysteem, en voor de identificatie van die onderdelen van het hinterland die niet op een wederzijdse manier verbonden via de uitwisseling van nutriënten kunnen worden met de stad.

De resultaten geven dat dat het valoriseren van fosfor in stedelijke zuiveringsslib en organische afvalstromen niet ten koste moet gaan van de recuperatie van energie voor het Brussels Hoofdstedelijk Gewest. Het gebrek aan landbouwactiviteit in de stad beperkt het nut van een analyse op stadsniveau en vereist de inclusie van het hinterland. De landbouwgronden van Vlaanderen en Wallonië, het binnenlandse hinterland van Brussel, hebben hun eigen overvloedige bronnen van nutriënten die kunnen worden hergebruikt. De overvloed aan nutriënten in Vlaanderen, die meestal geproduceerd wordt door een intensief opgekweekte veestapel gevoederd met geïmporteerde gewassen, domineert de analyse en de evaluatie van circulariteit. Wallonië, daarentegen, zou kunnen voordeel halen door een deel van de nutriënten die uit Brussel komen te gebruiken. Het ontwerpen van een meer circulair voedselsysteem voor Brussel zal een balans moeten maken tussen het dichterbij brengen van het hinterland, zodat de nutriënten terug naar de bodem kunnen keren, en het omgaan met de zeer problematische nutriëntstromen- en balansen in dit nabije hinterland. Uiteindelijk, en niet verrassend, zijn de meest effectieve strategieën om tot meer circulariteit te komen die strategieën die consumptie- en productiesystemen radicaal herdefiniëren, bv. het verminderen van het aandeel aan dierlijke producten van intensieve veehouderijsystemen in ons dieet.

Abbreviations & symbols

ABP	Association Bruxelles-Propreté
AD	Anaerobic Digestion
AG	Agriculture
BCR	Brussels Capital Region
BE	Belgium
CE	Circular Economy
DM / DMc	Dry matter / Dry matter content
EC	European Commission
FCS	Food Consumption Survey
FL	Flanders
FM	Fresh matter
FW	Food Waste
FWA	Food Waste Avoidance
FWV	Food Waste Valorization
GRAFS	Generalised Representation of Agrio-Food Systems
GE / GEc	Gross Energy / GE content
GW	Green waste
hh	Household
IBGE	Institut bruxellois de Gestion de l'environnement
K	Potassium
LCA	Life Cycle Assessment
LHV	Lower heating value
LU	Livestock Unit (1 LU = equivalent N excretion milking cow)
MFA	Material Flow Analysis
MRHSUT	Multi-Region Hybrid Supply and Use Tables
MR(H)IO	Multi-Region (Hybrid) Input-Output
MSW(M)	Municipal solid waste(management)
NUE	Nitrogen Use Efficiency
NutUE	Nutrient Use Efficiency

N / Nc	Nitrogen / Nitrogen content
OFMSW	Organic Fraction of Municipal Solid Waste
P / Pc	Phosphorus / Phosphorus content
PE / PrE	Process Energy: electricity, heat and fuel in- and outputs
prof	Professional
PUE	Phosphorus Use Efficiency
QTY	quantity
RoW	Rest of the world
SFA	Substance Flow Analysis
SM	Supplementary Material
SS	Sewage Sludge
SSU	Sewage Sludge Utilization
WA	Wallonia
WAO	Wet Advanced Oxidation
ww	wastewater
WWT(P)	Wastewater Treatment (Plant)

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Chapter 1

Introduction

Waste, like beauty, is in the eye of the beholder.
Carolyn Steele

1.1 Food and the city

1.1.1 A short history of urban metabolism in Europe

Cities, like organisms, have a metabolism: they consume primary materials and energy, use them to grow and prosper, and expel the by-products to the environment as waste. This is the urban metabolism, a metaphor used to describe the city and its relationship to the environment as a social-ecological system (Barles and Knoll, 2019). Because so few of the primary materials can be sourced within the city and so little of the waste can be absorbed within it, the city is mainly a parasitic organism relying on the exploitation of its hinterland to survive and prosper. Port cities have been relying on far away hinterlands for their food supply since ancient times: Athens was importing grain from the Black Sea; Rome from Northern Africa (Steel, 2008). Less lucky cities lacking a direct outlet to the sea, however, had a closer relationship with the agricultural lands that surrounded them: Paris, for example, was importing most of its foodstuffs from places within a radius of 250 km (Barles, 2015; Billen et al., 2012) and was using the by-products of eating, mostly human and horse dung rich in organic matter and nutrients, in suburban gardens and back in the rural hinterland as fertilizer (Barles, 2007). In Brussels, the contents of the cesspits were collected, dried, and transferred back to the land to be used as "urban manure" (Kohlbrenner, 2014).

Eventually European cities started to drift further apart from their hinterlands and to increasingly externalize their metabolism (Barles, 2015). This process started in the beginning of the 18th century, in parallel with the appearance of three mega-trends: globalization, the spatial specialization of agriculture, and the use of resources that needed to be imported, such as fossil fuels (Barles and Knoll, 2019). Newly developed and easy to transport synthetic fertilizers replaced animal and urban manure; cattle were separated from arable land and fed with protein imported from the Americas; urban wastes were piling up and stalling, imposing a threat to public health (Deligne, 2016). As material inputs were coming from further away, and the by-products were increasingly seen as waste, the externalization of the urban metabolism was also reinforcing its linearization. By the mid-twentieth century, the European city was a modern waste-producing factory, thriving on a constant linear throughput of virgin resources that obscured how "the organic inputs and outputs are vitally connected: that they constitute the cycle of life itself" (Steel, 2008, p260).

1.1.2 The problem with a linear metabolism

The impact of a highly externalized and linear urban metabolism on the environment is deep and manifold. By 2050, 80% of all food produced will be used to cover urban food consumption (EMF, 2019). Food consumption is the first most contributing activity to a city's ecological footprint, and the third to its carbon footprint, surpassed only by transport and housing (Goldstein et al., 2017). Intensive, industrialized food production is connected to the excess use of synthetic fertilizers and to hotspots of manure surpluses that lead to nutrients leaching into water bodies, causing eutrophication, and algal blooms (EEA, 2017; EEA and FOEN, 2020). The production of synthetic fertilizers is itself an energy-intensive and wasteful process, and in the case of phosphorus (P) a process that requires the mining and use of phosphate rock, a finite and potentially scarce natural resource (Cordell et al., 2009; Withers et al., 2019). At the same time, the by-products of food consumption, organic waste and excreta rich in nutrients and organic matter, usually end up in landfills and incineration ashes and never return back to where they are needed: in food production.

In Europe, 655 ktons of phosphorus (P) were lost in 2005 through wastewater and organic waste streams, 47% of all the P imported for agricultural use (van Dijk et al., 2016). If reused, these wasted urban streams could not only provide nutrients to crops, but they could further help to replenish organic matter in soils (Trimmer et al., 2019). Nonetheless, current infrastructural, behavioral and legislative lock-ins make their disposal less costly than their treatment and reuse (Trimmer et al., 2020; Withers et al., 2019). Shifting these lock-ins and changing the linear metabolism of cities needs no less than a thorough re-thinking of the food production and consumption system, a paradigm shift towards re-making them circular under the current mega-trends of globalization, urbanization, and environmental degradation.

1.2 Food and circularity

The Circular Economy (CE) emerged in the last decade as a new paradigm that can help to re-think and re-design modern cities, regions, and the production and consumption system, in a way that is regenerative, eliminates waste, makes use of local renewable resources, and celebrates local diversity (EMF, 2013a). The Ellen Macarthur Foundation (EMF) describes the Circular Economy as an economic system based upon five founding principles (EMF, 2013b):

(i) design out waste
(ii) build resilience through diversity
(iii) shift to renewable energy sources
(iv) think in systems, and
(v) think in cascades.

In the discourse of the EMF as well as of similar high-level consulting and policy documents, the CE provides a unique opportunity for “growth within” (EMF, 2015a): the increased reuse and recycling of products and components, together with new business models such as the sharing and service economy promise economic growth, jobs, and value creation while reducing pollution and conserving resources (Accenture, 2014; EMF, 2015a; Ghisellini et al., 2016; Jukka-Pekka Ovaska et al., 2016; WEF, 2014)

1.2.1 Circular Economy and circularity

The CE is thus an umbrella concept covering the whole production and consumption system and providing a vision for the transition to a more sustainable future. In this thesis I follow the CE definition by Geissdoerfer and colleagues (2017, p759), according to whom: "*The CE is a regenerative system in which resource input and waste, emissions, and energy leakage are minimized by slowing, closing, and narrowing material and energy loops.*" The research presented in this thesis addresses only the material and bio-physical aspect of the Circular Economy, what I define as circularity: the process of closing, narrowing, and slowing the material and energy loops. Circularity is, therefore, the means towards a regenerative system, rather than the end goal; it is not the *what* we are trying to achieve, but the *how* we are trying to achieve it. This is an important differentiation and what I have found to be a key driver fueling the debate on the Circular Economy. In the mainstream CE literature (EC, 2020; EMF, 2015a, 2013b), the end goal is the dematerialization of the economy and the decoupling of economic growth from resource use; the CE is presented as the economic system that will achieve exactly that (Giampietro, 2019; Hobson and Lynch, 2016; Korhonen et al., 2018). In contrast, I subscribe to a strong sustainability vision for the ecological transition that requires an absolute reduction in consumption and resource use. In that vision, circularity, i.e. the closing, narrowing and slowing of material and energy loops, is one of the means to achieve the end goal, because it reduces the absolute use of resources, by replacing primary, virgin resources with secondary *re*-sources (Arnsperger and Bourg, 2016). In addition, full integration of waste into the production and consumption system promotes a sufficiency-oriented thinking and a re-appreciation of these locally available re-sources (Arnsperger and Bourg, 2016; Bahers et al., 2017; Taylor Buck and While, 2020): trying to do things with what we have, rather than trying to find ways to do the things we want.

Such a conceptualization of circularity is clearly anchored within a strong sustainability vision: a lower input of virgin resources into the anthroposphere preserves natural capital and balances out the additional secondary resources without a total increase in production, thus avoiding hidden rebound effects (Zink and Geyer, 2017). Besides, it has been shown that lower resource use is a satisfactory predictor of lower environmental impact (Zoran J. N. Steinmann et al., 2017). Further, increasing circularity in European cities and regions, by reusing existing local secondary resources, means that primary resources are left available for regions that need them more; in that sense circularity is also a means towards global equity (Helena Kahiluoto et al., 2015). Finally, fostering circularity does not only mean *closing* the loop, but also *narrowing* it: circularity is a place-bound practice (Arnsperger and Bourg, 2016; Bahers et al., 2017). If we consider material and energy flows to be the threads on the patchwork of social-ecological system interactions, then the shorter the threads keeping the patches together, i.e. the narrower the loop, the closer humans get to their environment, and so the more they develop a sense of care and response-ability to their places (Haraway, 2016). For all these reasons the basic assumption that runs through this thesis is that increased circularity of food systems, as presented here and further refined in the next section, is preferable to a linear system.

1.2.2 What is a circular food system?

To be able to unravel what food system circularity is and can be, let us start by defining the "food system". According to Vaarst and colleagues (2017, p4) a food system is a "*system that involves activities, social and institutional structures, and processes related to the production, distribution, exchange, and consumption of food*". Because this study deals with material flows and is firmly founded on the field of industrial ecology, when I refer to the food system I refer to the interactions between bio-geophysical and human environments and the activities that these interactions determine (Ericksen, 2008), but not necessarily to socio-economic outcomes and other determinants of food security. Further, although most conceptualizations of the food system include all the processes in the food supply chain up to consumption, I always include the process of (organic) waste management in the system definition, because waste is (i) an unavoidable outcome of consumption and (ii) a potential secondary resource (Harder et al., 2020; Trimmer et al., 2020).

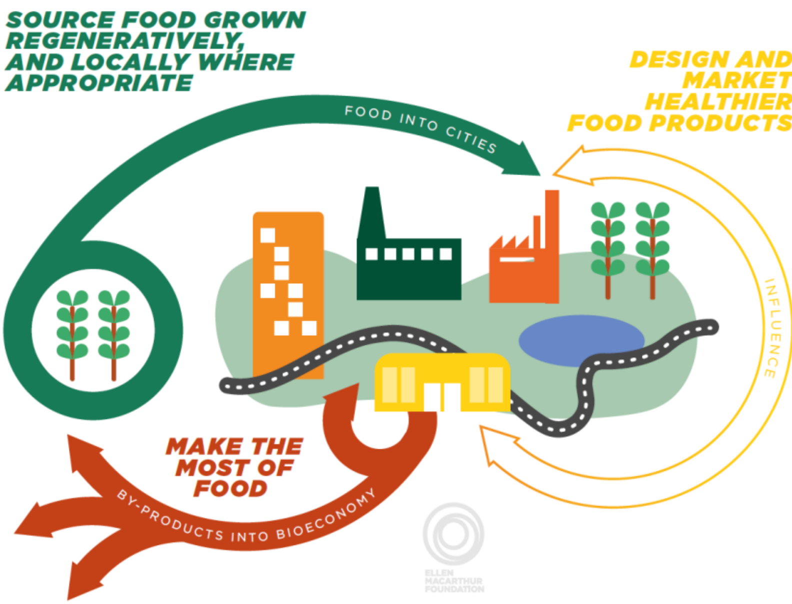


Figure 1.1 Three ways to make urban food more circular [Source: (EMF, 2019)**]**

- *Strategies to achieve food system circularity*

Strategies towards increased circularity in food systems include a variety of approaches that close, slow down, and narrow the system's material and energy loops. Most literature addresses waste management, e.g. food waste avoidance, the recycling (utilization) of organic waste or the nutrient recycling and recovery from wastewater and sewage sludge (Table 1.1). Zooming into urban food systems, the Ellen MacArthur Foundation (2019) offered three overarching ways towards increased circularity, each focused on a different aspect of the system (Figure 1.1): (i) procuring food that is grown in a responsible way, preferably locally (production), (ii) making the most of the system's by-products (waste management), and (iii) shifting diets to healthier food products (consumption). Therefore, although the concept of circularity is often understood as akin to waste management and recycling, strategies towards a more circular food system span the whole food supply chain.

Table 1.1 An overview of strategies towards more circular food systems as presented in key literature [own compilation]

Strategy	Implementation examples	Source
Food Waste Avoidance (FWA)	Avoid / reduce / prevent food waste	(1), (2), (3), (4), (5), (6), (7), (9)
	through digital supply chains	(8)
Food Waste Utilization	Valorization of food industry by products	(1), (2), (6)
	Biorefineries	(1), (10), (11)
	Cascading organic flows	(3), (11)
	Waste-as-resource" business models	(5)
	Reuse of matter (e.g., for feed)	(7)
	Waste separation & return logistics	(11)
Nutrient Recycling & Recovery (NRR)	Recovering nutrients from wastewater	(1), (4), (11)
	Reuse of treated wastewater	(4)
	Nutrient recycling, closed loops of nutrients	(7), (8)
	Cascade utilization of by-products, residues & excreta	(3)
Waste-to-Energy (WtE)	Anaerobic Digestion	(1)
	Biogas Production	(12)
More efficient agriculture	Optimization of input factors in agriculture	(8)
	More efficient agricultural practices	(2)
	Regenerative farming practices	(13)
	Organic AG, permaculture, agro-ecology	(13)
Consumption and diet shift	Promote sustainable diets	(7)
	Consume less animal products	(3)
Local food	(Peri-)urban farming	(2)
	Celebrating local diversity	(13)
	Favouring local resources	(14)

(1) (EMF, 2013a), (2) (EMF, 2015a), (3) (Haas et al., 2015), (4) (EC, 2015), (5) (EEA, 2016a), (6) (Sitra, 2015), (7) (Jurgilevich et al., 2016), (8) (EMF, 2015b), (9) (EMF, 2015a), (10) (Bastein et al., 2013), (11) (EMF, 2017), (12) (Bastein et al., 2013), (13) (Duncan and Pascucci, 2016), (14) (Tedesco et al., 2017)

- *Nutrient cycling and food system circularity*

Beyond such general descriptions of what food system circularity could entail, more concrete implementation strategies and assessment frameworks are still largely missing from the literature (Navare et al., 2020). Anthropogenic systems such as construction, transport, or the production and consumption of durable goods, appear to be more compatible with most of the CE prescriptions, such as reuse and refurbish, prolong lifetimes etc. In food systems, though, CE implementation and assessment cannot be built around the actual products, i.e. the food items. Food items are meant to be consumed and cannot be refurbished, reused, repurposed or remanufactured, unless within very narrow time limits, e.g. in leftover cooking. Circularity of food systems then mainly translates into valorising waste streams from the processes of producing and consuming food (e.g. food waste valorization or nutrient recycling from wastewater), or more general systemic interventions, such as a shift in agricultural practices. As a result, understanding and assessing food system circularity and assessing the effects of different interventions requires the use of a trace element that (i) is able to be traced along the different material streams relevant to the food system (food products, food waste, by-products of human and animal metabolism), and (ii) is central to the utility of the different streams, i.e. answers the question "what makes the food, food?"

Nutrients are such useful trace elements. Food and organic waste streams are resources because of their biomass and nutrient content (Navare et al., 2020; Tseng et al., 2019). Biomass is the carrier of chemical energy that can be released either inside living organisms as metabolic energy, or in Waste-to-Energy (WtE) plants usually as thermal energy. Nutrients are components such as nitrogen and phosphorus needed for the proper functioning of people, animals and crops. From a circularity and secondary resource use perspective, food and organic waste streams can be used as food, as an energy source, or even as a raw material for non-food products, e.g. bio-plastics. Nonetheless, an open-loop recycling system, e.g. using organic waste streams to generate energy or non-food materials, is not only further down the food waste hierarchy (Figure 1.2), but often a false dilemma, too. Anaerobic digestion, for example, can be used for the dual purpose of generating energy *and* producing digestate, a stream rich in nutrient and organic matter that can enrich agricultural soils with organic matter, protect them from erosion, and enhance water and nutrient retention (Trimmer et al., 2019).

These are the reasons why this thesis is primarily focused on nutrient cycles to describe and quantify urban food system circularity. Although the sole focus on nutrients has been criticized as insufficient to capture the full potential of circularity in food systems (Koppelmäki et al., 2020), I see closed-loop recycling, i.e. reusing the nutrients in the food as *nutrients* for animals (food waste to feed) or crops (composted and digested wastes as organic fertilizer), as a preferable solution to open-loop recycling and downcycling. Therefore, I put the emphasis on the *nutrient* rather than the *biomass* aspect of food.

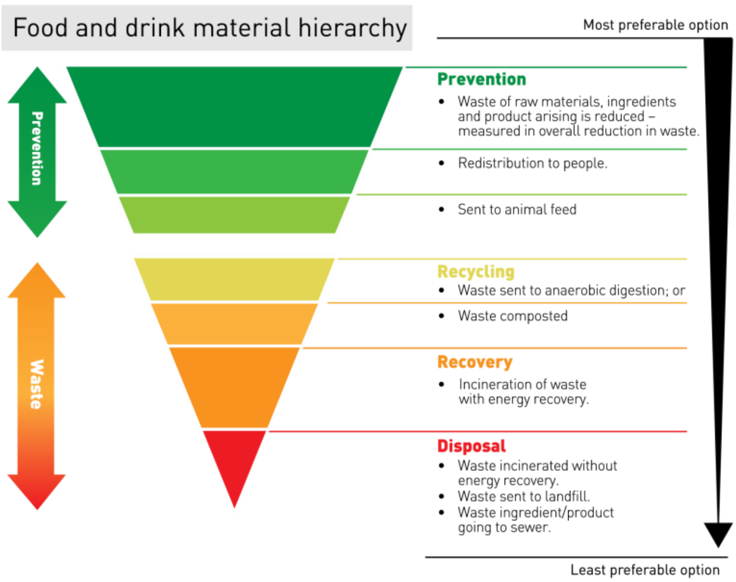


Figure 1.2 The Food Waste Hierarchy [(WRAP, n.d.)]

1.3 Spatial scales & metrics in nutrient cycles research.

1.3.1 Spatial scales in nutrient flow research

Nutrient flow analyses have been widely used to study the food systems of several European and American cities, including Paris (Barles, 2007; Chatzimpiros and Barles, 2013; Esculier et al., 2018), London (Villarroel Walker et al., 2017, 2014), Beijing (Ma et al., 2014), Stockholm (Wu et al., 2016), Linköping (Schmid Neset et al., 2008), and Montreal (Metson and Bennett, 2015; Treadwell et al., 2018). These studies typically use the Material and Substance Flow Analysis methodology (MFA/SFA) (Brunner and Rechberger, 2017), and adopt the administrative boundary of the city as the system boundary. This is often a logical choice, given data availability and the fact that administrative boundaries define the areas within which specific authorities have the jurisdiction to take decisions and implement actions. From a circularity perspective, however, especially a nutrient circularity perspective, agriculture is a central component of the system: it is where the vast majority of the primary resources are used and where the results of circularity interventions will be most evident. Although food production is sometimes included in the system definition of city-scale MFA studies, it is food products ready for consumption that make up the majority of inputs; agriculture is external to the system. This raises the question: if not the administrative boundary, then what system boundary is most appropriate to study nutrient flows in urban food systems? If cities import their food from all around the world, where could the secondary re-sources be returned to, to consider that the loop has closed?

The issue of territory is relatively new and unexplored within the industrial ecology field. As the popularity of the CE concept has been rising, researchers have begun to address questions on the most appropriate scale for analysis and on the role of territory in urban metabolism and material flow studies (Cerceau et al., 2018; Zasada et al., 2019). Further, the field of territorial ecology has been consolidating and flourishing (Bahers and Durand, 2017; Barles, 2017), yielding a wealth of knowledge on the metabolism, imprint and hinterland of Paris, e.g. (Barles, 2009; Chatzimpiros and Barles, 2013; Esculier et al., 2018), as well as other French cities and regions, e.g. (Bahers et al., 2020; Bahers and Giacchè, 2018; Tanguy et al., 2020). There are two broad ways in which researchers in both fields of industrial and territorial ecology have been addressing the question of scale in their quantitative analyses of nutrient flows and budgets:

(i) The **multi-regional approach**, in which the main system boundary is the national one, but its territory is divided into smaller units. The focus of the analysis is the whole agri-food system supplying the country rather than a single city or the city’s consumption. Researchers have used this approach to address how characteristics of different places (e.g., geography, type of production system) affect nutrient flows and to assess how synergies among regions can enhance nutrient recycling at the national scale (Akram et al., 2019; Hanserud et al., 2016; Klinglmair et al., 2015; Le Noë et al., 2017, 2016; Senthilkumar et al., 2012).

(ii) The **imprint approach**, where the focus is clearly on the exchange of flows between a city and its regional and domestic hinterland, and nested spatial scales are often used to capture flow exchanges and the geography of the hinterland e.g. (Bahers et al., 2018; Esculier et al., 2018; Tanguy et al., 2020).

Nevertheless, with the exception of Paris, whose metabolism and food system, as well as domestic hinterland has been studied to a remarkable depth and extent over many years, there is a shortage of empirical studies that systematically address the question of scale in urban food systems, comparing different scales as to their usefulness in investigating urban metabolism in general, and urban food system circularity in particular.

1.3.2 Metrics to evaluate nutrient flows and budgets

To evaluate the results of nutrient flow analyses, researchers have been using several indicators and metrics, depending on the research questions, rationale, and scope of each study. Metrics used include nutrient soil budgets, losses and emissions into the environment, and nutrient use efficiencies at different system levels, but circularity has rarely been the explicit focus. When it comes to circularity metrics, there is a consensus that sets of indicators, rather than one single metric, are most appropriate to capture the complexity of the urban metabolism and of social-ecological systems (Navare et al., 2020; Pauliuk, 2018).

Fernandez-Mena et al. (2020) proposed a set including feed autonomy, bioenergy production, N cycle closing and GHG mitigation potential, and defined a “maximum circularity” scenario where no external inputs were allowed. Koppelmäki and her colleagues (2020) introduced the concept of “nested” circularity, and developed an assessment framework that spans four different scales (farm-region-nation-global) and examines three elements of food system circularity: biomass for food and feed, biomass for energy, and nutrient cycling.

However, to this day there is neither a definition nor a universally accepted set of metrics to evaluate circularity in food systems (Parchomenko et al., 2019; Tseng et al., 2019), despite the interest for them in research and practice (Moraga et al., 2019).

1.4 Synthesis: research questions and approach

1.4.1 Overall research questions

In this thesis I set off to tackle the issue of urban food system circularity through the lens of nutrient cycles. I aim to better understand what urban food system circularity is, how to measure it, and which scales are most appropriate to use when measuring circularity and planning interventions. The overall research questions motivating the research are:

- How can we assess urban food system circularity using nutrient cycles? What metrics to use and at which spatial scales to apply them?
- What insights into circularity can different spatial scales and system levels offer?

1.4.2 Case study: Brussels

To answer these questions, I use Brussels Capital Region and its food system as the case study. Brussels is an interesting case to focus on: it is a vibrant cosmopolitan city, with regional strategies for both a sustainable food system (Ronsmans, 2015) and the circular economy (Anon, 2016). Further, the organic waste management system is currently being re-thought, a process that started with the introduction of the voluntary separation of household organic waste in the whole city in 2017, and continues with the debate over the most appropriate treatment process and final use of the separately collected organic waste (Bortolotti et al., 2018a, 2018b; De Muynck et al., 2018). Additionally, because Brussels Capital Region is one of the three administrative Regions of the federal state of Belgium, there are data available that are not typically collected at the urban level, such as agricultural census and trade data. Finally, there are recent studies available on the city's metabolism (Athanassiadis et al., 2016; EcoRes sprl et al., 2015), as well as on nutrient flows in parts of the domestic hinterland, e.g. Coppens et al. (2016), that serve as the solid foundation to uphold the development of this study.

1.4.3 Approach and outline of the thesis

This thesis unfolds in a step-wise manner. I did not pre-select one single approach to the issue of spatial scale nor one specific set of metrics. Rather, I built a multi-scale approach in successive steps, each of which addresses a different spatial scale, and tests different metrics, as deemed relevant for the particular scale (Table 1.2). I start by focusing on the city (chapter 2) and the phosphorus on the urban effluents that are potentially reusable re-sources. In this chapter, I assess jointly the nutrient (P) and energy flows in the food system of Brussels, in order to identify available quantities of P for reuse, and potential trade-offs between P reuse and energy recovery in the waste management system.

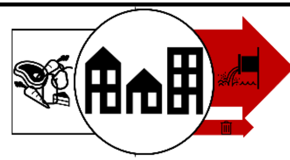
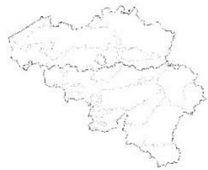
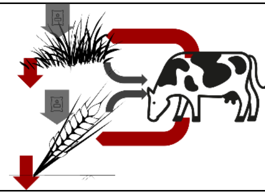
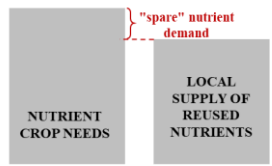
Because there is hardly any agricultural activity within Brussels, the re-sources identified in chapter 2 would have to be exported for reuse to the hinterland. Chapters 3 and 4 thus expand the analysis to include the city’s domestic hinterland: the agricultural lands of Flanders and Wallonia. Is there demand in these places for nutrients coming for Brussels? Are there local urban effluents that could compete with the ones from Brussels? How circular is the agro-food system in Belgium? These are the questions addressed in these chapters.

Finally, in chapter 5, I adopt a footprint approach, and trace the origins of phosphorus consumed in Brussels across the globe, aiming to bring the whole hinterland into the analysis, to understand the global effect of adopting circular strategies in the city, and to uncover the loops that cannot be closed.

At this point it is necessary to clarify the selection of the nutrients studied at each step. The initial ambition was to have a multi-nutrient assessment for all three major agricultural nutrients, nitrogen (N), phosphorus (P), and potassium (K). All three are indeed included in the parts of this thesis where the main focus is on agriculture and soil budgets (chapters 3 and 4). The final overall assessment, however, heavily relies on phosphorus. That was a decision made to operationalize the analysis, when additional layers of complexity had to be addressed, such as the inclusion of energy flows in chapter 2 and the expansion of the system boundary to the whole world in chapter 5. The reasons why phosphorus was selected among the three nutrients and why phosphorus is especially relevant for a food system circularity assessment are discussed in sections 2.1 and 5.1.

Finally, chapter 6 provides a synthesis of results and insights that emerged throughout this study and addresses the overall research questions. That final chapter aims to offer a critical look to the overall approach and outcomes of this work, and summarize the new knowledge on the circularity of urban food systems generated by the research documented in this thesis.

Table 1.2 Overview of the four chapters comprising the main body of the thesis

No	Title	Spatial Scale & Resolution	Research focus	Chapter-specific RQ
Chapter 2	Phosphorus and energy flows through the food system of Brussels Capital Region			How much secondary P is generated in BCR? Would its reuse require additional energy inputs?
Chapter 3	Assessing agro-food system circularity using nutrient flows and budgets			How circular is the agro-food system in Belgium, the domestic hinterland of Brussels?
Chapter 4	The potential of reused nutrients to cover crop needs in dense livestock-dominated regions			Is there any nutrient demand in the domestic hinterland that is not covered locally and could absorb nutrients from Brussels?
Chapter 5	A resource-based P-footprint for urban diets			Where is Brussels' global hinterland? What part of the nutrients consumed in Brussels will never "close the loop"?

Chapter 2

Phosphorus and energy flows through the food system of Brussels Capital Region[1]

2.1 Introduction

Cities have an important role to play in the transition towards a Circular Economy (CE) for food. Food consumption is among the top three drivers of urban environmental footprints (Goldstein et al., 2017) and urban consumption is projected to mount up to 80% of all food produced by 2050 (EMF, 2019). This concentration of consumption in cities suggests the parallel concentration of human excreta and, partially, food waste in them. Subsequently, cities can influence decisions on what and how the agri-food system produces and at the same time drive efforts to avoid, reuse and ultimately phase out waste in the system.

Food cannot grow without supplying it with nutrients, like nitrogen (N), phosphorus (P), and potassium (K). At the same time, the overuse of mineral NPK fertilizers and the disposal of untreated, nutrient-rich urban effluents in aquatic bodies causes the eutrophication of terrestrial and aquatic ecosystems and affects the quality of the agricultural soil

[1] This chapter is based on: **Papangelou A**, Achten W M J and Mathijs E (2020) Phosphorus and energy flows through the food system of Brussels Capital Region *Resour. Conserv. Recycl.* **156** 104687 Online: https://doi.org/10.1016/j.resconrec.2020.104687

(Smil, 2000). While N is available everywhere around us, mineral P fertilizers are made from phosphate rock, which is a finite resource (Cordell et al., 2009). This double status of phosphorus, as a pollutant and a scarce resource, makes phosphorus management a prominent strategy in the context of the circular bio-economy, and so circular food systems (Nesme and Withers, 2016; Withers et al., 2018).

In order to better manage phosphorus we first need to understand how it flows through coupled human-environmental systems, where it comes from, where it leaks worst, and where the biggest potentials for reuse lie. Indeed, lots of research has been dedicated in the past decade to map P flows in cities, regions and countries. Up to this point, detailed P budgets are available for cities in Sweden (Kalmykova et al., 2012; Schmid Neset et al., 2008; Wu et al., 2016), China (Li et al., 2010; Lin et al., 2016; Ma et al., 2014; Qiao et al., 2011) and Canada (Metson and Bennett, 2015; Treadwell et al., 2018). Phosphorus follows similar paths in these cities: it enters through imported food and it either exits with sewage sludge and solid organic waste or accumulates in urban sinks, such as soils and landfills. (Chowdhury et al., 2014; Kalmykova et al., 2012). In some cases, smaller quantities of treated sewage sludge or composted organic waste are recycled in urban agriculture, e.g. (Metson and Bennett, 2015), or in agriculture outside the system boundaries, e.g. (Ma et al., 2014; Wu et al., 2016). The anthropogenic phosphorus cycle, is thus, closely connected to the food cycle, making P an adequate 'trace element' to represent flows related to the food system, e.g. food products, food waste and human excrements.

Besides food, phosphorus management involves and influences the water, energy and waste management sub-systems in a city. Researchers have addressed the nutrient, and thus phosphorus, management from such a nexus perspective, by studying nitrogen or phosphorus along with other resources, such as water (Esculier et al., 2018), energy (Hamilton et al., 2015) or both (Liang et al., 2019; Villarroel Walker et al., 2014; Villarroel Walker and Beck, 2012). These studies are, however, on a regional (Villarroel Walker and Beck, 2012) or national (Hamilton et al., 2015) scale. So far, few studies have taken a multi-resource perspective when studying nutrients at the urban scale (Liang et al., 2019; Ma et al., 2014; Villarroel Walker et al., 2014) and they show trade-offs between different nutrient management scenarios, depending on various objectives and priorities (Villarroel Walker et al., 2014).

Brussels, the capital of Belgium, is currently considering alternatives for treating its inhabitants' kitchen waste that is collected separately in the dedicated orange bag since 2017, on a voluntary basis. The small amounts of kitchen waste collected are currently sent to an Anaerobic Digestion (AD) plant 130 km West of the capital, while Brussels' authorities are investigating solutions for the management of its organic waste (De Muynck et al., 2019). The regional government has been favoring the installation of an AD plant within the city (Bortolotti et al., 2018b). An important motivation behind prioritizing AD seems to be a policy focus on green energy, in a national effort to reach the EU 2020 targets. Nonetheless, Waste-to-Energy (WtE) strategies are further down the food material hierarchy pyramid than prevention and reuse solutions (WRAP, 2016), and can potentially be at odds with other programs, such as the city's Good Food strategy (Ronsmans, 2015) or the regional program for a Circular Economy (Anon, 2016).

Given the scarcity of multi-resource flows analyses at the urban scale and the importance of the interaction between phosphorus and energy from a CE perspective, as the Brussels case exemplifies, the research objectives of this study are:

(i) to identify and quantify the phosphorus flows in Brussels that are potentially available for recovery and reuse and assess the current P circularity of the food system, and

(ii) to evaluate circular solutions towards better resource management in the food system from a multi-resource perspective, accounting for phosphorus and energy simultaneously.

2.2 Materials & Methods

2.2.1 Case Study : Brussels Capital Region (BCR)

Brussels Capital Region (BCR) is one of the three regions that constitute Belgium, along with Flanders in the north and Wallonia in the south. It has a population of 1'163'486 (2014) and a surface area of

161 km^2 (IBSA, 2015). Despite its administrative ‘regional’ status, BCR is not a city-region as meant in planning (Rodríguez-Pose, 2008): it does not encompass the whole metropolitan area of Brussels, nor does it include the peri-urban and rural hinterlands of the city. It has the status of a region for political reasons, but its character is urban, with a population density of >7’000 cap/km^2 (IBSA, 2015). Finally, BCR is not to be confused with the City of Brussels, which is one of the 19 municipalities that together form BCR (Figure 2.1). In the rest of the chapter we use the terms city, Brussels and BCR interchangeably to refer to BCR.

The city’s economy is predominantly based on the service sector. Agricultural production is marginal, with 1’850 ha, ~1% of total area, cultivated in 2014 mainly with cereals, potatoes, and sugar beets (Statbel, 2015a). In economic terms, the contribution of the primary sector to the city’s economy is negligible (IBSA, 2015). Although many industries have their headquarters in Brussels, hardly any actual manufacturing activities take place within the city’s boundaries. The secondary sector in the city mainly consists of small-scale food processing, i.e. bakeries and pastry shops (RDC-Environment, 2014).

Food waste is collected together with mixed waste and is incinerated. Since 2017, the inhabitants of Brussels have also had the option of sorting their food waste, which is then collected separately and sent to an anaerobic digestion plant in Ypres, West Flanders. Green waste is collected separately and composted, either in a facility in the south of Brussels (Bruxelles-Compost) or in composting plants in the surrounding areas. Finally, wastewater is treated in two wastewater treatment plants (WWTP): the South WWTP, established in 2000 and currently under modernization and the North WWTP, in operation since 2007. Brussels has a combined sewer network that serves both the transport of domestic wastewater to the two treatment plants and the runoff drainage. The Northern plant currently receives app. ¾ of the domestic wastewater generated in BCR; the Southern plant treats the rest, together with some industrial effluents (IBGE, 2012; SBGE, 2017).

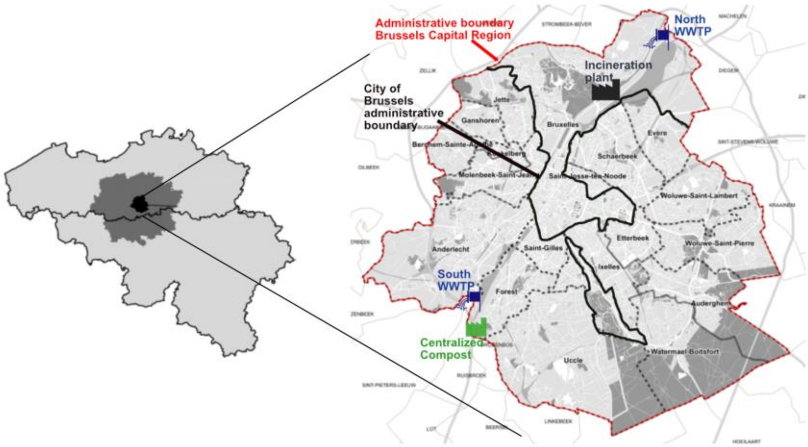


Figure 2.1 Left: map of Belgium (light grey), including the metropolitan area of Brussels (dark grey) and BCR (black). Right: map of BCR, with the locations of the wastewater treatment plants (WWTPs), the incinerator and the composting facility. The administrative regional boundary is indicated in red, the boundary of the City of Brussels in solid black and the rest of the communes in dashed grey lines. [Sources : left: (Athanassiadis et al., 2016), base map right: BruGIS; icons: Factory by Fahmihorizon from the Noun Project; Outfall by Luis Prado from the Noun Project]

2.2.2 System Definition

The system under study is the food system within the administrative boundary of the Brussels Capital Region (Figure 2.1); it corresponds roughly to the activity "*to nourish*" in the city (Baccini and Brunner, 2012). The reference year is 2014, the most recent year for which all data were available. For variables that vary from year to year, like the amounts of wastewater treated or the process energy inputs in the incinerator, we used 5-year averages to ensure that the model is as representative of the current situation as possible.

Within the food system, we can distinguish 4 subsystems: production, trade and consumption, wastewater management, and municipal solid waste (MSW) management. The 'production' sub-system refers primarily to the process Urban Soils, including private and communal gardens, agricultural lands and public green spaces, like parks and forests. We consider the process 'Decentralized Composting' to be part of the ‘production’ sub-system too, since it operates separately from the centralized MSW management system and is closely connected to the urban soil. The food processing industry, mostly bakeries and chocolatiers that sell largely outside of BCR, was excluded due to lack of data, which leads to an underestimation of both inputs and outputs. Table 2.1 list the goods considered in this study and their definitions, as well as the main flows within each good category.

In our study we adopt the multi-layer Substance Flow Analysis framework described in (Hamilton et al., 2015) to quantify the phosphorus and energy flows in Brussels’ food system. For energy, we distinguish between gross energy (GE_f) that represents the chemical energy content of the material flow f, and PE_p that represents the in- or outflows of energy (electricity, fuel, heat) to or from process p. To estimate the GE of the flows we use the calorific content for the different food items, and the lower heating value (LHV) or net calorific content for all other flows. We do not account for transport, due to data gaps; however, in section 2.4.1 we discuss qualitatively some effects of accounting for transport to the energy flows.

2.2.3 Quantification of flows

- *Trade and consumption*

We estimate the total input of food into the system as the sum of the food consumed in Brussels and the food waste generated during consumption and trade (Eq.1). Data on the daily per capita consumption of different food items come from the latest national Food Consumption Survey(Ridder et al., 2016) and their corresponding phosphorus and energy contents from the Belgian database *internubel* (Nubel, 2018) or Dutch database *NEVO* (RIVM, 2016). Based on information on the place of consumption from the Food Consumption Survey (Ost, 2015), we can differentiate between the food consumed at home, at restaurants and cafés (HoReCa), and at school and work canteens (services). We assume that visitors have the same daily food intake as the inhabitants, and that they consume all their meals in a HoReCa establishment. In order to quantify the food consumed by incoming commuters, we assume that they have their lunch, corresponding to 1/3 of the daily intake, at a work canteen. We use the net number of incoming commuters, to account for the food consumed outside Brussels by the outgoing commuters. Details on the calculation are given in the Annex.

Non-food products include detergents and cosmetics and pet food. For the former, we use the value for the specific P detergent consumption in Belgium by (Ott and Rechberger, 2012). Phosphorus in pet food is calculated using official statistics for the number of cats and dogs in Brussels (Statbel, 2018) and literature values for their daily P intakes (Wu et al., 2016).

Food waste generation from households, HoReCa, services, and trade is estimated based on (Vanessa Zeller et al., 2019) and their P and energy content from literature values (Table 2.1 and Table S1). The total imports of food (F1) and non-food products (F2) into trade are approximated as the sum of sold products (F3 – F7) and waste of the food trade sector (F8) from (Vanessa Zeller et al., 2019).

This probably underestimates the values of the import flows and gives an idealized impression of 'Trade' as a tight process. Including trade data, however, would require the combination of national data on international trade with the interregional Input-Output (IO) tables, the translation of IO data in physical quantities and their disaggregation to food product or food group level, and the adoption of assumptions for the shares of food that are re-exported and used for processing. Given the additional uncertainties and complexity that these data treatment steps would introduce to the model, we opted for using the mass balance to calculate the import flows.

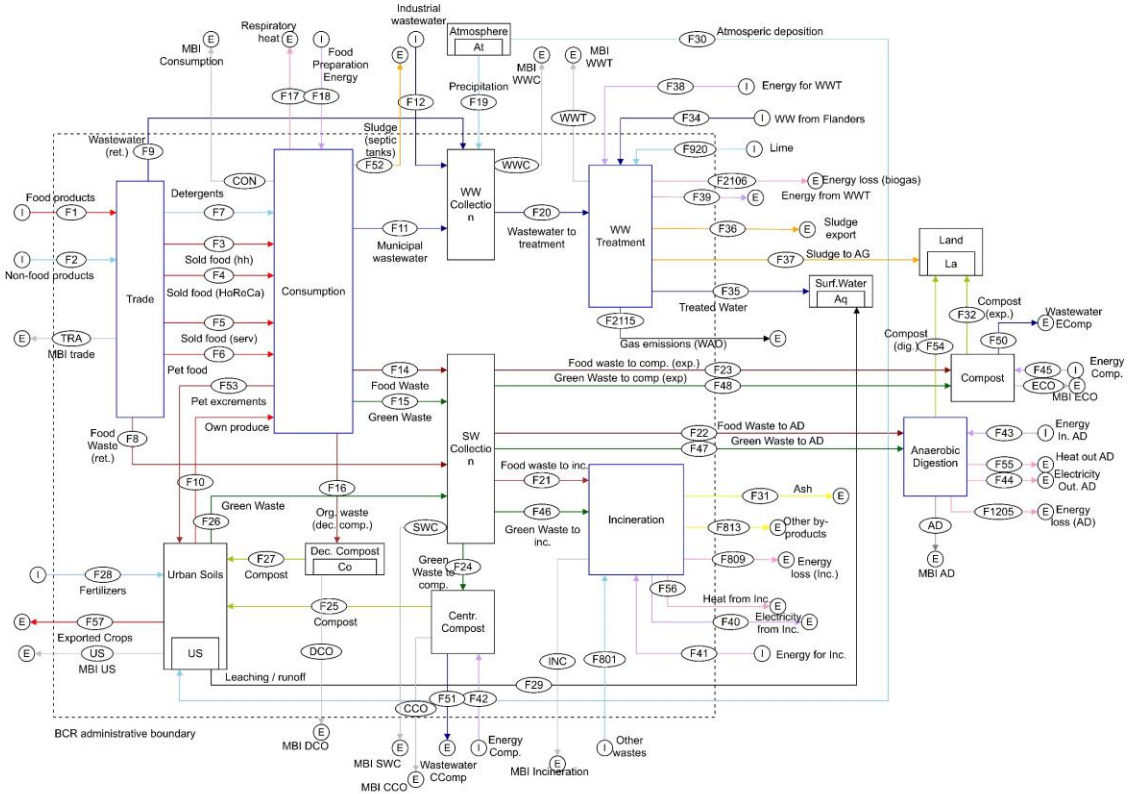
$$Food\ Sold = Food\ Consumed + \ Food\ Waste$$
$$= FC_{inh} + FC_{vis} + FC_{com,net} + (FW_{hh} + FW_{HoR} + FW_{serv})$$

(Eq.1)

FC_i : food consumed by inhabitants (inh), visitors (vis), or net number of commuters (com,net)

FW_j : food waste generated by households (hh), HoReCa (HoR), or services (serv)

Figure 2.2 (next page) System diagram : the food and phosphorus system in Brussels Capital Region. Flows are colored according to the 'good' category they belong to


Atmosphere
At
Atmosperic deposition
F30
Industrial wastewater
Respiratory heat
Food Preparation Energy
MBI Consumption
F17
F18
F12
F19
Precipitation
MBI WWC
MBI WWT
WWT
F38
Energy for WWT
F34
WW from Flanders
F920
Lime
Sludge (septic tanks)
F52
Wastewater (ret.)
F9
CON
WWC
WW Collection
F2106
Energy loss (biogas)
F39
Energy from WWT
Food products
F1
Detergents
F7
F20
Wastewater to treatment
WW Treatment
F36
Sludge export
F2
Non-food products
F3
Sold food (hh)
F4
Sold food (HoReCa)
F11
Municipal wastewater
F37
Sludge to AG
Land
La
Trade
Consumption
F35
Treated Water
Surf.Water
Aq
TRA
MBI trade
F5
Sold food (serv)
F6
Pet food
F2115
Gas emissions (WAD)
Compost (exp.)
F32
Wastewater EComp
Compost (dig.)
F54
F50
F14
Food Waste
F15
Green Waste
F53
Pet excrements
Own produce
Food waste to comp. (exp.)
F23
Green Waste to comp (exp)
F48
Compost
F45
Energy Comp.
ECO
MBI ECO
SW Collection
Food Waste (ret.)
F8
F22
Food Waste to AD
F47
Green Waste to AD
Anaerobic Digestion
F43
Energy In. AD
F55
Heat out AD
F44
Electricity Out. AD
F1205
Energy loss (AD)
AD
MBI AD
F10
F16
Food waste to inc.
F21
Green Waste
F26
Org. waste (dec. comp.)
SWC
F46
Green Waste to inc.
Incineration
F31
Ash
F813
Other by-products
F809
Energy loss (Inc.)
F56
Heat from Inc.
F40
Electricity from Inc.
F41
Energy for Inc.
Dec. Compost
Co
F27
Compost
F28
Fertilizers
Urban Soils
Green Waste to comp.
F24
Centr. Compost
F25
Compost
F57
Exported Crops
US
MBI US
DCO
INC
F801
Leaching / runoff
F29
F51
F42
CCO
BCR administrative boundary
MBI DCO
MBI SWC
Wastewater CComp
Energy Comp.
MBI CCO
MBI Incineration
Other wastes

We assume that 98% of the P intake by humans and 100% of the P intake by pets ends up in their excrements; in addition, we adopt the assumption by Theobald and colleagues (2016) in their analysis of the P flows in Berlin-Bradenburg, that 60% of the pet excrements end up directly in the soil (F53), whereas the rest 40% is disposed of in the household waste

- *Production*

Food production activities in Brussels include both urban agriculture (UA) and conventional crop production. UA represents private and communal gardening and small-scale professional horticultural activities, actual production data for which are scarce and uncertain, e.g. (Boutsen et al., 2018). According to a recent comprehensive study of Brussels' urban metabolism, a maximum 1% of the vegetable consumption in Brussels could be covered by soil-based horticulture within the city, whereas half of the exploitable lands were used for horticulture in 2010 (EcoRes sprl et al., 2015, p. 95). We thus estimate the amounts of food produced in UA and consumed directly by inhabitants (F10, 'Own produce') to represent 0.5% of the total plant-based products consumed. Conventional agricultural production data are taken from official statistics (Statbel, 2015a). We use the average mineral-P application rate of the Walloon region (REEW, 2018a) and official manure export statistics of Flanders (VLM, 2014) to estimate synthetic fertilizer and manure inputs (F28). Details on the calculations are given in theAnnex.

Given the study's focus on nutrient circularity and the tight connection between UA and decentralized compost (household and neighborhood scale) we classify the latter within the 'production' sub-system. Around 420 tonnes of household food waste were composted in private and communal composting sites in 2015 (F16), representing 0.4% of the total food waste production (Bortolotti et al., 2018a).

- *Wastewater Management*

Primary data on the amounts of wastewater treated in the two WWTPs (F20, F34, F35) and their characteristics come from the annual reports of Brussels Society for Water Management (SBGE, n.d.), datasets from Brussels' environmental agency (IBGE, 2018a), and the webpage of Aquiris, the company that operates the North WWTP.

Information and data on the sludge treatment process (F36, F37) were taken from (Chauzy et al., 2010) and assumed to represent the situation in both WWTPs. For the estimation of the process energy (F38, F39) we combined literature values with those provided online by the operator (AQUIRIS, n.d.). The anaerobic digestion of sewage sludge and the energy recovery from biogas were modelled following (Cano et al., 2015). Details on the modelling process, the variables used and their sources, as well as the underlying assumptions can be found in the SI.

- *Municipal Solid Waste Management*

Data on the amounts of food and green waste to the different treatment options (F21 – 24, F46 – 48) come from a recent study by Zeller et al. (2019). Incineration is the main treatment option for MSW in Brussels. Amounts of incineration by-products (F31, F813) were calculated with information from the annual reports of Bruxelles-Proprété, the agency managing the incinerator (Bruxelles-Propreté, n.d.); the same reports, combined with the energy balance of Brussels (IBGE, 2015a), were used to estimate the energy flows (F40, F41, F56, F809). When primary data were missing, the most recent life cycle inventory data available were used (Haupt et al., 2018).

Household green waste collected separately by ABP is composted in the only large-scale composting plant in Brussels (process 'Centralized Compost' – CCO). Compost from CCO is mainly used for landscaping across the city (Robinet, 2019), thus we assume that all of the P content is returned to Urban Soils.

- *Mass Balance Inconsistencies*

Different data sources of variable quality were used for the calculation of the different flows: primary data were available in certain cases, while in others we had to rely either on data referring to the whole country (e.g. food consumption), or another region in the country (e.g. characteristics of compost in Flanders), modeled data from other studies on Brussels (e.g. organic waste streams), design values for some processes (e.g. sludge generation and energy consumption in the North WWTP) or general values from textbooks and peer-reviewed literature (P and energy content for most of the flows). The main data sources used are listed in and Table 2.2 and a comprehensive list can be found in the supplement.

Given the great variability of data origins and quality, we chose to follow the method of Hamilton et al. (2015), and get an idea of uncertainties by calculating the mass balance inconsistencies for each process and keeping them visible in the model. This approach works in this case, since we did not reconcile the data in the model and the input and outputs flows in the different processes were either (i) estimated independently (e.g. Consumption, Wastewater Collection) or (ii) modelled using transfer coefficients and assumptions from actual facility-level data (e.g. WWT, incineration), or general literature (e.g. energy flows in WWT and AD). The only exception to this procedure is the system input flows F1 (Food products) and F2 (non-food products), which are simply the sum of the output flows from the process 'Trade'. We include 'Trade' as a process separate from consumption in the system, in order to account for the food waste generated by it; however, it is mostly a 'proxy process' and its input flows should be interpreted with care.

Table 2.1 Types of goods included in the system, color key for Fig.1 and main data sources for the estimations of their phosphorus (P) and gross energy (GE) content (hh : households, prof : professional, ser : services, AD : anaerobic digestion, WWT : wastewater treatment)

Good	Color	Description	Flows	Sources
Food products	Red	Food products imported and sold within BCR and crop production	F1 : Food products F3 : Sold food to hh F4 : Sold food to HoReCa F5 : Sold food to services F10 : Own produce F57 : Exported Crops	(Le Noë et al., 2017; Nubel, 2018; RIVM, 2016)
Non-food products	Light blue	Detergents, cosmetics, fertilizers and pet food imported and sold within BCR	F2 : Non-food products F6 : Pet food F7 : Detergents & cosmetics F28: Fertilizers	(Ott and Rechberger, 2012; VLM, 2014; Wu et al., 2016)
Organic waste (OW)	Brown	OFMSW, mixed FW and GW, pet excrements to soil	F16 : OW to dec.compost F53 : Pet excrements to soil	(Fisgativa et al., 2016)
Food waste (FW)	Dark red	Inedible parts of food products and wasted food	F14 : Food waste from consumption F8 : Food waste from trade	(Fisgativa et al., 2016; Jensen et al., 2016)

		(gone bad & meals leftovers) from households, HoReCa and canteens (services)	F21 : Food waste to incineration F22 : Food waste to AD F23 : Food waste export to compost	
Green waste (GW)	Green	Garden waste from households and green waste from parks and public spaces	F15 : Household green waste F24 : Green waste to compost F26 : Professional green waste F46 : Green waste to incineration F47 : Green waste to AD F48 : Green waste export to compost	(Boldrin and Christensen, 2010)
Compost	Lime green	Compost from all scales (household, neighborhood, big scale) and all kinds of feed (OW, GW, digestate)	F25 : Compost to gardens & parks F27: Compost to private gardens F32 : Compost (from exp.) F54 : Compost from digestate & GW	(Möller et al., 2018; Vlaco, n.d.)
Wastewater (ww)	Dark blue	Municipal and industrial wastewater flows and treated water discharged in the Senne	F11 : Municipal wastewater to sewers F12 : Industrial wastewater to WWT F20 : Wastewater from BCR to WWT F34 : ww from Flanders to WWT F35 : Treated water to Senne F50 : ww from compost (exp.) F51 : ww from centralized compost	(IBGE, 2018a; SBGE, n.d.)
Sewage sludge (SS)	Orange	Raw sewage sludge (from WWT and septic tanks) and products from sewage sludge treatment (e.g. technosands)	F52 : Septic sludge F36 : Treated sludge from S.WWTP F37 : Treated sludge from N.WWTP	(Chauzy et al., 2010; IBGE, 2018a; SBGE, n.d.)
Incineration by-products	Yellow	Ash (bottom & fly) and rest (metals, slug, salts)	F31 : Bottom and fly ash F813: Other incineration by-products	(Bruxelles-Propreté, n.d.; Haupt et al., 2018)
Emissions (gas & liquid)	Black	P gas or aqueous emissions not accounted for in other flows	F29 : Leaching & runoff F2115 : Gaseous emissions WAO	*P & GE content assumed ~0*

Table 2.2 Process energy flows included in the model, color key for Fig.1 and main data sources for their estimations

Process Energy	Color	Description	Flows	Main sources
Electricity	Violet	Electricity process inputs and outputs	F18 : Food preparation energy	(IBGE, 2017a, 2015a)
			F38 : Energy input WWT	(Aquiris, n.d.)
			F39 : Energy output WWT	(Cano et al., 2015)
			F40 : Electricity output (incineration)	*modelled*
			F41 : Electricity input (incineration)	(IBGE, 2019)
			F42 : Electricity for centralized compost	(Haupt et al., 2018)
			F45 : Energy for centralized compost (e)	(Haupt et al., 2018)
			F43 : Energy input AD	(Haupt et al., 2018)
			F44 : Electricity output AD	*modelled*
Heat	Pink	Heat process inputs and outputs (incl. energy losses)	F17 : Respiratory heat	*balance*
			F2106: Energy loss from biogas (WWT)	Modeled based on (Cano et al., 2015) &
			F56 : Heat output (incineration)	(Haupt et al., 2018)
			F809 :Energy loss from incinerator	
			F55 : Heat output AD	
			F1205: Energy loss from digester	

2.2.4 Scenarios

In order to illustrate the potential for phosphorus reuse and reveal trade-offs between phosphorus availability and energy use, we compare 4 scenarios for the year 2030. The scenarios illustrate theoretical best cases for P management and were chosen taking into account (i) commonly mentioned strategies for circular food systems and (ii) the current plans and discussions on waste management in Brussels (Bortolotti et al., 2018b; IBGE, 2018b). For example, food waste valorization is currently investigated by the city authorities as an alternative to collecting them together with residual waste and incineration (Bortolotti et al., 2018b).

The baseline scenario (Base, Table 2.3) is simply a projection of the current situation into the future, accounting for population and visitors growth. We also apply a 20% reduction in the generation rate of avoidable food waste for all scenarios, since this is the target of the city's new 5-year plan on waste (IBGE, 2018b). We do not account for diet changes in our scenarios, as we do not expect diets to change substantially by 2030, given that Brussels is an already dense and culturally diverse city with a high per capita GDP (IBSA, 2015), which are known to be factors that are influential on the diet. What is more, dietary changes would probably influence P inputs in food production upstream the city, but not the amounts of P consumed and wasted within it (Geneviève S Metson et al., 2016); these upstream effects, although relevant, are outside the scope of the current study.

For the Food Waste Valorization scenario (FWV) we assume a complete diversion of food waste from the incinerator to anaerobic digestion and composting (90% and 10% respectively). In the Sewage Sludge Utilization scenario (SSU) all sewage sludge from the two WWTPs is anaerobically digested and stabilized with lime, making it potentially reusable to agriculture. Lime conditioning is the final treatment step for the sludge from the treatment plants of Liège and Namur, after which it is reused in agriculture (AIDE, 2018; INASEP, 2018). Finally, CO is the combination scenario that accommodates both measures. Table 2.3 summarizes the assumptions for all four scenarios and presents more details on the variables and rationale behind them are given in the SI.

It is worth noting that the scenarios' purpose is neither to represent solutions that can be realistically implemented until 2030, nor to mirror precisely the current plans and wishes of the stakeholders' involved. They are simply a methodological device with the purpose of generating potential upper limits for phosphorus reuse rate or savings. These limits can be used as benchmarks against which progress towards a more circular food system can be measured.

Table 2.3 Scenarios: target year 2030, est. population in BCR : 1'356'000 inh. (FWV : Food Waste Valorization, SSU : Sewage Sludge Utilization, CO : Combination, AG : agriculture, NL : the Netherlands)

	Base	FWV	SSU	CO	Destination
Food waste reduction (avoidable FW)	20%	20%	20%	20%	NA
Share of food waste collected sent to incineration	100%	0%	100%	0%	NA
Share of FW collected digested anaerobically	0%	90%	0%	90%	AG
Share of FW sent to compost	0%	10%	0%	10%	Brussels
Share of sludge reused in AG	0%	0%	100%	100%	AG
Share of sludge exported or to landfill	100%	100%	0%	0%	Landfill /DE

2.2.5 Circularity Indicators

One of the objectives of this work is to assess the circularity of the current food system in BCR and compare it with scenarios that are expected to increase this circularity. There is still neither a widely accepted definition on what circularity is (Kirchherr et al., 2017) nor consensus on how to measure it (EASAC, 2016; Moraga et al., 2019; Pauliuk, 2018), especially when the focus is on food or the bio-economy in general. For the purposes of our study, we chose to compare three different P-based indicators (Table 2.4), depending on whether the phosphorus is potentially reused or reusable: (i) within the administrative boundary of the city ('City Circularity'), (ii) in agriculture both within the city and outside the system boundary, corresponding to the concept of closed-loop recycling ('Food Circularity'), and (iii) anywhere, corresponding to the concept of open-loop recycling ('Weak Circularity'). In the third case, we account for all nutrients that are neither emitted in the environment nor landfilled. The three different circularity indicators are defined as:

- City Circularity : $CC = \frac{P\ reused\ within\ the\ city\ boundary}{total\ P\ input}$ (Eq.2)
- Food Circularity : $FC = \frac{P\ reused\ or\ reusable\ in\ agriculture}{total\ P\ input}$ (Eq.3)
- Weak Circularity : $WC = \frac{P\ reused}{total\ P\ input}$ (Eq.4)

Table 2.4 Flows accounted for in the nominator of the three different circularity indicators used in this study.

Flow		City	Food	Weak
F25	Brussels Compost	X	-	X
F27	Decentralised compost	X	X	X
F2119	Treated sludge from N. WWTP to AG	-	X	X
F2218	Treated sludge from S. WWTP to AG	-	X	X
F54	Digestate from AD of org. waste	-	X	X
F31	Incineration ashes	-	-	X
	Technosands and exported SS	-	-	-

These three simple metrics capture the extent of P-reuse in two different geographical scales (city and hinterland) and under three management scopes (city self-sufficiency, food system perspective and open-loop recycling). However, they are focused on reuse only: any upstream interventions, like the adoption of more efficient and less wasteful farming techniques, will have to happen outside the city's, and thus this study's, boundary. The sole focus on recycling has been justly criticized as too narrow-scoped to be used alone as a CE metric, e.g. (Parchomenko et al., 2019; Pauliuk, 2018), and we recognize that further research including the agricultural hinterlands in the system boundary will improve the metrics proposed here.

2.3 Results

2.3.1 Phosphorus flows in 2014

The phosphorus flows in BCR have a linear pattern (Figure 2.3). Almost 700 tonnes of P entered Brussels with the imported food products in 2014 (69% of the total input), and the rest came from detergents (99 tP/yr), pet food (96 tP/yr), wastewater imports from Flanders (106 tP/a), industrial wastewater (15 tP/a) and fertilizers (16 tP/a). Most of the P inputs end up in the city's sewage system and are finally either released in the environment with the treated effluent (139 tP/yr) or captured in the sewage sludge. Sludge from the South WWTP is exported for incineration (107 tP/yr), whereas that from the North WWTP is processed into technosands and then used as cover material in landfills (455 tP/yr).

Phosphorus in the solid organic waste streams (food and green waste) represents around 17% of the total input. Almost all food waste (147 tP/yr) and around 40% of the green waste (11 tP/yr) are collected in the white bags and incinerated together with residual waste (Figure 2.3c and d). Another 12 tP/yr from green waste are sent to the composting facility of the city and around 8 tP/yr are exported through food and green waste to Flanders, to be treated anaerobically or composted. 160 tP/yr, or ~90% of all the P in the organic waste collected ends up in the incineration ashes and from there to the road construction industry in the Netherlands.

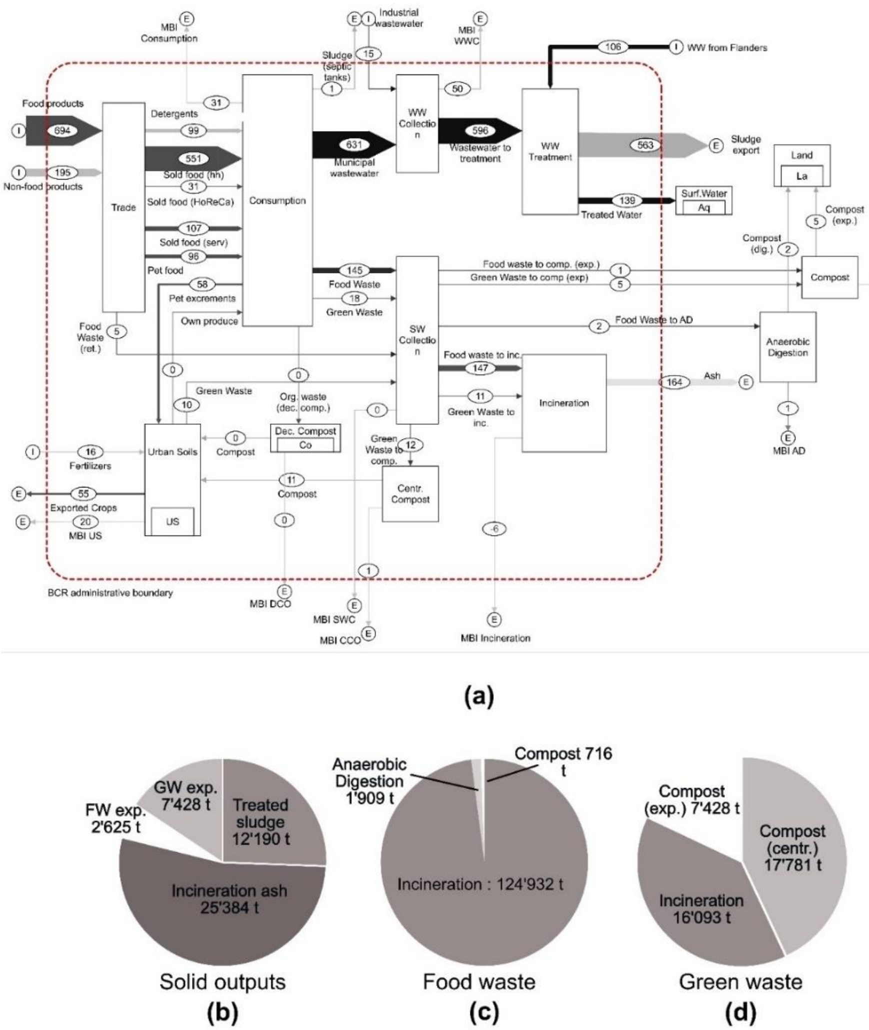


Figure 2.3 Phosphorus flows in BCR for 2014 in tonnes P/year (a), distribution of total amounts of solid outputs (b) [in tonnes/year], and distribution of food (c), and green waste (d) according to treatment type [in tonnes/year]. Deviations from the numbers in the text are due to rounding. (MBI : Mass Balance Inconsistency, WW : Wastewater, WWC : Wastewater Collection, US : Urban Soils, SW : Solid Waste, DCO : Decentralized Compost, CCO : Centralized Compost, ECO : Exported Compost, FW : Food Waste, GW: Green Waste).

2.3.2 P and energy flows under different scenarios

The baseline scenario is simply an extrapolation of the situation in 2014. As such, more than three quarters of the incoming phosphorus enter the city with food (Figure 2.4a) and is either discharged through treated wastewater in the environment (12%), used in landfills through sewage sludge (67%), or turned into incineration ashes (15%). Imported food dominates the energy flows, too (Figure 2.4c): 5’300 TJ/yr enter the city through imported food, most of which are taken up by human metabolism. 1’200 TJ/yr of process energy (PE) are used for the storage and preparation of food at the household level. The energy content of the organic waste entering the MSW management system is around 1’000 TJ/yr for the baseline scenario. This energy is recovered as electricity (100 TJ_e/yr) and heat (309 TJ/yr) in the incinerator.

In the FWV scenario, all food waste is assumed to be valorized through anaerobic digestion and further use of the digestate in agriculture. Through the composted digestate, 114 tP/yr can be potentially reintroduced into food production, from 6 tP/yr for the baseline (Figure 2.5a). Besides, the energetic valorization of the biogas produced during fermentation results in 116 TJ_e/yr of gross generated electricity, an increase of +16%, compared to the baseline (100 TJ_e/yr). If we account for the net electricity output, however, the MSWM contributes almost three times more electricity, compared to the baseline (114 TJ_e/yr versus 38 TJ_e/yr, Figure 2.5b), thanks to the lower energy input requirements of an AD plant, compared to the incinerator. Due to the diversion of the organic waste from the incinerator, the amount of heat generated is drastically smaller for FWV.

The SSU scenario has the most drastic effect on the amount of P potentially reintroduced into agriculture (Figure 2.5a). When all the sewage sludge is conditioned and used in agriculture, instead of turned into technosands and landfilled, 674 tP/yr or almost 60% of the phosphorus input can be circulated back into the food system. This scenario has no effect on the energy flows, since the MSWM system is the same as for the baseline. The combined effects of FWV and SSU are illustrated with the combined scenario, CO. In that scenario, more than 800 tP/a can be circulated, while generating 116 TJ_e/yr.

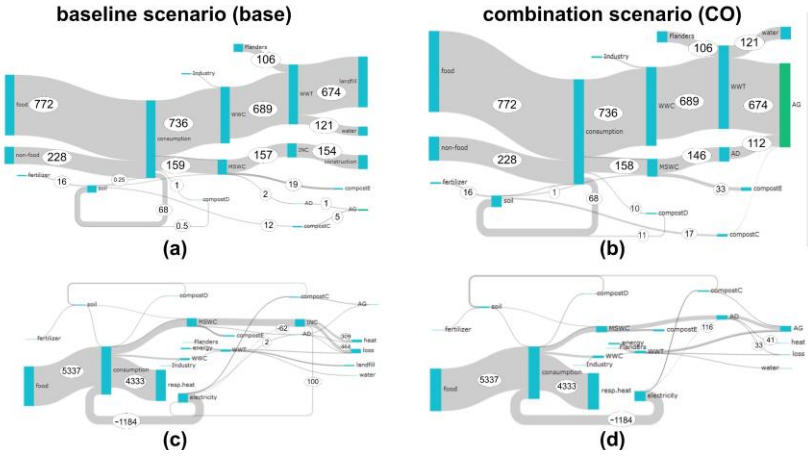


Figure 2.4 Sankey diagrams for the phosphorus (first row, in tonnes / year) and energy (second row, in TJ / year) flows in BCR for the baseline (left), and the combination (right) scenarios.

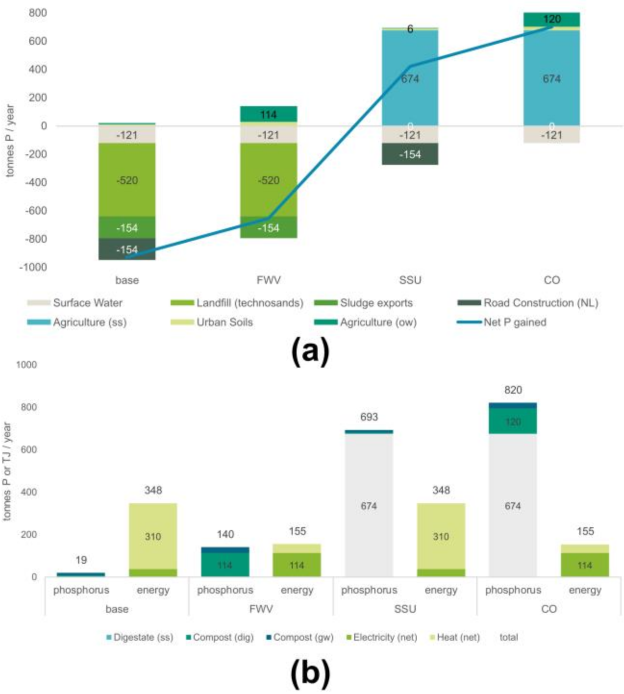


Figure 2.5 (a) Amounts of phosphorus outputs per destination and net P gained ($P_{reusable} - P_{lost}$, in tP/yr) and (b) amounts of phosphorus (in tP/yr) potentially available for reuse and of net energy (in TJ/yr) recovered from the municipal waste management system for the different scenarios.

2.3.3 Levels of phosphorus circularity per scenario

The degree of circularity of the system varies depending on the scenario and type of circularity (Figure 2.6). When all the potential alternative uses of the P-rich effluents are taken into account, a score of 14% 'weak circularity' is achieved for the baseline scenario. In this case, the reuse corresponds to the phosphorus amounts ending up in the road construction industry through the incineration ashes. On the other hand, the degree of phosphorus 'city circularity' that Brussels can achieve is persistently low, with a maximum of 2% of the P inputs reused within the city's boundaries for the FWV scenario. 'Food circularity' ranges from 0 for the baseline scenario to more than 70% for the theoretical case when all sewage sludge is used in agriculture and all food waste is diverted from the incinerator to an anaerobic digestion unit.

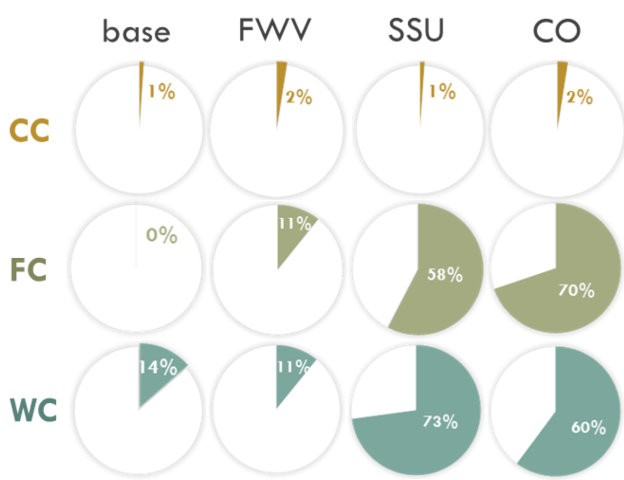


Figure 2.6 Phosphorus circularity indicators for each scenario: City Circularity (CC), Food Circularity (FC) and Weak Circularity (WC). For an explanation on the indicators and the acronyms of the scenarios, see text.

2.4 Discussion

Since the phosphorus ‘city circularity’ stays negligible across the scenarios, P-containing products will probably need to be transported for reuse outside of the city. For this reason, most of this section is dedicated to discussing parameters that can affect whether and where these potentially reusable P flows will be reused, even outside our system boundary. We address two aspects: (i) the characteristics of the outflows as alternative products to synthetic fertilizer, and (ii) necessary conditions for their reuse outside the city boundary. We further put our results into context and we close an evaluation of our methodology and model.

2.4.1 Characteristics of output flows regarding their agricultural reuse

The P-containing outputs vary according to scenario (Figure 2.5a). For the baseline, two are the main outputs: 520 tP/yr of technosands, used as covering material to landfills, and incineration ashes exported to the Dutch road construction sector (154 tP/yr). For FWV and SSU, two additional products occur: composted digestate (114 tP/yr for the FWV scenario) and treated sewage sludge (674 tP/yr for the SSU and CO scenarios), both of which are reusable in agriculture. Green waste compost is produced within the region in all scenarios, though in much smaller quantities than the other products: 12 tP/yr for the baseline, and SSU scenarios and 17 tP/yr for FWV and CO. The last three (treated sewage sludge, composted digestate and green waste compost) are the products that we consider reusable in agriculture and that we take into account to calculate the ‘food circularity’ of the system, based on their P-content.

Several parameters will affect whether these streams will indeed be reused or not; their demand and the legal framework governing their management are two crucial ones that are covered in the following section of this chapter. Table 2.5 summarizes some properties of these products that we expect to affect the desirability and profitability of each scenario: (i) the P concentration of the flow; (ii) the plant availability of P in each product; (iii) their market price; and (iv) the net energy output of the treatment processes that generate each of the products.

Treated sludge (digested and dried) is by far the product with the highest P concentration. The concentration depends on the performance of the drying: in our case we assumed a total solids (TS) content of 90% in dry sludge, according to the current performance of the filter press in the N.WWTP (Chauzy et al., 2010). Even with a more moderate assumption of 35% solid content, the sludge remains the most concentrated stream. What is more, plants can take up P easier from digestate than from compost (Table 2.5) and digestate is currently sold at a higher price than compost. Sewage sludge digestate is therefore the most valuable of the P effluents from Brussels. The major disadvantage of sewage sludge as a secondary P source is its potentially high concentration of heavy metals and pharmaceuticals. Heavy metals in urban wastewater can be due to atmospheric deposition, street runoff, but mainly the presence of industrial effluents in the sewer system (Angelidis and Gibbs, 1989; Sperling, 2007). Since hardly any industries operate within BCR and only the South WWTP receives some industrial wastewater[2], it is unlikely that heavy metal concentrations in the sludge will surpass the legal thresholds for agricultural use adopted by the region in 1993 (Bruxelles-Capitale, 1993), which is still valid until this day. According to recent measurements (Brion, 2019) the heavy metal concentrations of the mineralized sludge (technosands) are close to the threshold values for lead and nickel and well above thresholds for zinc and copper. The fresh, dried sludge from the South plant performs better, with all metals being well below the legal thresholds. Since the mineralization process in the North WWTP reduces the mass of the sludge drastically by oxidizing all organic matter, we can expect that the concentrations of fresh, dried sludge from the North WWTP would be lower and probably closer to the ones from the South WWTP (Brion, 2019, personal communication). At the moment, there is not much information on the concentrations of micro-pollutants, like pharmaceuticals and hormones.

[2] 30% of its total according to (IBGE, 2017b), corresponding to 7.5% of the total water treated in the region.

Table 2.5 Characteristics of compost and digestate (gw: green waste, dig. : digestate, ss : sewage sludge). For the estimation of the fuel needed for transport we assume transport with lorries at 2 MJ/tkm.

	Compost gw	Compost dig.	Digestate ss	Source
Concentration [kgP/tn]	1.2	2.0	21.4	This study
Plant availability [%]	50%	50%	90%	(1), (2)
Price [€/tn]	8	8	16	(3), (4)
Fuel for 50 km transport [L/kgP]	2.3	1.4	0.1	this study and (5)
Fuel for 50 km transport [kWh / kgP]	22.5	13.9	1.0	
Electricity generation from process [kWh_{net} / kgP]	-14	462	274	This study

(1) (Hamilton et al., 2017), (2) (Haupt et al., 2018) (3) (Bruxelles-Compost, 2014), (4) (Vlaco, 2016a), (5) (EEA, 2016b)

2.4.2 Use of urban effluents in and around BCR

BCR has ~1’850 ha of arable land that produce mostly winter wheat, sugar beets and potatoes, destined for the processing industry outside the city. Next to the conventional agricultural activity, some of the vegetables, mushrooms, and fruits that Brussels inhabitants eat are cultivated in community and private gardens and in a few professional horticulture urban farms. The region supports these initiatives through its ‘Good Food’ strategy (Ronsmans, 2015) and the number of gardens and micro-farms has been rising in the past four years. The role of vegetables grown within the city, however, remains marginal in covering the demand: local professional farms cover hardly 0.1% of the current demand (Boutsen et al., 2018). In our model we adopted the assumption that a maximum 1% of the demand for fruits and vegetable can be provided from within the region (EcoRes sprl et al., 2015, p. 95). Still, the role of locally produced food remains negligible in the P cycle: around 0.5 tP/a come from locally grown fruits and vegetables, a tiny fraction of the 770 tP/a imported through food.

The inability of the agricultural production within BCR to absorb the P in the city’s waste streams means that the P-rich urban effluents will need to be exported and valorized in the neighboring provinces, or even further away in Belgium and abroad. Exports will become easier after the new European Fertilizers Directive, which accounts for organic fertilizers, will come into force in 2022 (EC, 2018a). At the same time, the Flemish and Walloon regions generate their own P-rich flows, that will probably be given priority over imported ones. The livestock sector in Flanders especially, is a big manure producer and exporter, with strict regulations in the region regarding the nutrient application rates to the soil (Kristel Vandenbroek, 2017). Consequently, a detailed evaluation of the nutrient flows and needs in the agri-food system around Brussels is indispensable in order to locate possible sites for the reuse of secondary P.

The focus of this chapter is phosphorus, and so much of the discussion refers to the fertilization value of treated organic waste. Compost and digestate are, however, products different from mineral fertilizer and manure: although they release nutrients in the soil, their primary purpose is often soil conditioning. So, even if we can say that the >650 tP/yr available in the system's outputs can theoretically cover the whole P-demand for the two Brabant regions (assuming 100% substitution) or around 10% of the whole demand for Belgium, future analyses must be more nuanced. It is, for example, expected that small organic urban and peri-urban farms will show a higher interest in using compost produced in the city than big conventional farms. What is more, products from a city's organic waste management system can be valorized for reasons other than food production, e.g. landscaping and ornamental horticulture. Such uses are excluded from the discussion in this study, since the focus is on the food system and 'closed-loop' recycling of urban organic waste; nonetheless, these alternative uses can be substantial components of a more sustainable phosphorus management in urban areas.

2.4.3 Phosphorus versus energy

An underlying hypothesis behind this study has been that, if Brussels makes more phosphorus available for reuse through the implementation of Circular Economy inspired strategies, energy expenditure will rise, due to additional treatment steps in the municipal solid waste and urban water management systems. In Figure 2.5b we can see that the total net energy output in each scenario is indeed either lower or the same as for the baseline. The main reason for this reduction is losing part of the heat generated in the incinerator by diverting the waste to an anaerobic digester (FWV). The heat from the incinerator is currently used on site for heating the different facilities and is partially converted to electricity (IBGE, 2018c). However, in the case where we only account for electricity and not heat, the FWV scenario by far outperforms all others. This indicates that implementing a common food waste valorization strategy (digestion of food waste) increases both the amount of potentially reusable P available, and the amount of net electricity recovered. On the other hand, the SSU scenario has an important effect on the amounts of P potentially available for reuse, but none at all on the net energy recovered, compared with the baseline (Figure 2.5b). This is because anaerobic digestion with energy recovery from biogas is already part of the baseline scenario. Besides, all energy generated in the WWTPs is consumed on-site.

As a consequence, we see a synergetic effect between P recoverable and net electricity recovered, albeit a weak one. Further, it is worth putting the numbers into perspective, especially if the results are used to inform policy: the total net electricity production for the scenarios ranges from 38 – 114 TJ_e/yr or 10 – 32 GWh_e/yr. Even the high ends of these ranges represent a tiny fraction of both the total energy consumption within BCR and the residential energy consumption, which in 2016 were more than 20’000 GWh/yr and 7’600 GWh/yr respectively (IBGE, 2018c). Most of the residential energy consumption comes from natural gas and only 17% (1’313 GWh_e/yr) from electricity. This means that even for the ideal scenario where all food waste in BCR would be collected separately and valorized through anaerobic digestion, the electricity generated would cover a theoretical 3% of the total electricity demand of the households in Brussels, or 0.2% of the total energy demand in the region. For comparison, the national target for Belgium is to achieve 13% of the gross final energy consumption coming from renewables by 2020; the share of BCR in achieving this target was set to 0.073 Mtep (~850 GWh).

2.4.4 Methodological considerations

In order to study the food system of Brussels Capital Region, we used the Region’s administrative boundary as the system boundary. The focus has thus been on downstream solutions that are under the direct control of the Region's authorities, i.e. wastewater and solid organic waste management. The agricultural hinterland that produces the food consumed in Brussels was not included in the system. This means that we have not accounted for inputs from mineral fertilizers and manure, which are typically the largest phosphorus flows in a food system. In addition, strategies like food waste avoidance, which would result in large phosphorus savings upstream and thus larger savings in total, have not been included in the scenario analysis. This calls for a careful interpretation of our scenario results, and for future research to address this shortcoming by including food production in the system boundary.

As is usually the case for MFA studies, data are coming from different sources of varying quality. A range of methods was used to estimate the flows, from actual statistics and facility-specific data (e.g. mixed household waste, wastewater quantities and characteristics) to modelling based on generic literature data (e.g. sludge treatment). Despite this, the Mass Balance Inconsistencies typically represent a small fraction of total input in a process. An exception is the MBI for Urban Soils (Figure 2.3). The process "Urban Soils" was modeled quite rough, so it is possible that most of the P accounted for in the MBI will either accumulate in the soil, or contribute to the growth of plants other than the agricultural crops. A more detailed accounting of the fate of P in the urban soil will allow to better understand the capacity of the soil to take up more P and under what conditions.

Lack of readily available data was also the reason why the food industry is not included in the system. In fact, Zeller and colleagues (2019) estimate that up to 25'000 tons of organic waste were generated in Brussels in 2014. Assuming average P and energy contents of 0.15% (Coppens et al., 2016) and 13 MJ/kg (Bernstad and la Cour Jansen, 2011), industrial food waste could have contributed an extra 40 tP/yr and 340 TJ/yr to the MSWM sub-system. This is more than 20% of the phosphorus we account for currently, and indicates that food industry is potentially an important part of the system. However, import data for BCR are tricky to collect and interpret, as illustrated in detail in (EcoRes sprl et al., 2015), so that the estimation of the inputs would introduce additional uncertainties. For this reason, we left food processing out of the system.

Finally, the scenarios are ideal best cases, a methodological device meant to provide information on maximum theoretical amounts of P potentially reusable. These values can be used, for example, as targets or benchmarks against which progress towards better P-management and a more circular food system can be measured. Realistic and implementable scenarios for the organic waste system in Brussels have been generated within the Opération Phosphore project and can be found in (Bortolotti et al., 2019, 2018b; De Muynck et al., 2019).

2.5 Conclusions and outlook

In this chapter we analyzed the phosphorus and energy flows in the food system of Brussels Capital Region and assessed the impacts of 3 alternative, theoretical scenarios on these flows: food waste valorization, sewage sludge utilization and a combination of the two. Brussels has a nutrient metabolism typical of a city: almost all P that enters through imported food ends up in the sewage sludge and from there to landfill. Diverting these amounts of phosphorus from the landfill to agriculture has the potential to greatly increase the circularity of the city's food system. The impact of the alternative scenarios on the energy flows is not as clean-cut as for P. Yet, energy recovery from organic waste can only marginally contribute to covering the city's energy demand.

Our study contributes to the literature on urban phosphorus flows and urban food systems, by providing a case study for the city of Brussels, linking food to phosphorus and energy flows in a multi-resource approach and exemplifying ways to create and measure food system circularity in an urban context. Our results indicate that the options for closing loops in the food system of Brussels are limited within the city-region's boundary. The P-rich urban effluents have to re-enter the food system somewhere outside the borders and this means that the connection of the city with its local hinterland needs to be strong, in a mutual beneficial relationship. Studying this connection on different scales (local-national-global) can offer a more concrete and substantial view on how circular urban food systems can be designed and implemented.

The information on the amounts of resources flowing in-, out-, and within the city, is an indispensable first step to design policy interventions. Such interventions will also depend, however, on the views and aspirations of different stakeholders, the legal framework, the prioritizing of different goals etc. This holds even more true for Brussels and Belgium, where governance is fragmented and the regulatory landscape complex. Understanding these factors can thus offer a whole different level of insight into the system and the need for integrating such knowledge into urban metabolism research is widely recognized.

2.6 Annex I (Table S1) : Model Equations

Description of all flows, equations for their calculation and sources of variables. The second column gives the section of the SI 2 that includes details on the modelling choices and graphs of all sub-systems. (P) Only P flow calculated; (E) only energy flow calculated; (P,E) only P and energy flows calculated; (PrE) : Process Energy. For other abbreviations refer to the index.

Sym.	Flow Description	Equation	Source mat./ DMc	Source P	Source GE / PE
F28	Fertilizers	(Agricultural area in BCR x application rate P) + Manure imported from Flanders	(1), (2)	(2), (3)	(4)
F1	Food products	Calculated through the mass balance for TRADE (F3+F4+F5+F1001+F1102)			
F2	Non-food products	Pet food (F6) + Detergents (F7)			
F18	Food Preparation Energy (PrE)	Population x specific energy consumption for food preparation in BCR			(5) – (7)
F12	Industrial wastewater	Amount of industrial wastewater in S.WWTP x Pc/ DMc / GEc of wastewater	(8), (9)	(9), (10)	(11)
F3	Sold food (hh)	Food consumed at home *(see text for details)* + food waste from households (F201+F202)	(12), (13)	(14), (15)	(14), (15)
F4	Sold food (HoReCa)	Food consumed at restaurants *(see text for details)* + food waste from HoReCa (F203+F204)	(12), (13)	(14), (15)	(14), (15)
F5	Sold food (serv)	Food consumed at school/work *(see text for details)* + food waste from services (F205+F206)	(12), (13)	(14), (15)	(14), (15)
F6	Pet food(P)	Population x ratio dogs/cats per capita x daily ingestion of P per dog / cat	(16)	(17)	
F7	Detergent(P)	Population x specific P dishwasher detergent consumption in Belgium		(18)	
F9	Wastewater (ret.)	*Not an actual flow, included in the ww from Consumption*			
F8	Food Waste (ret.)	Amount of "*Food waste from food commerce"* from (Zeller et al. 2019) x Pc/ DMc / GEc	(19)	(20)	(21), (22)
F27	Compost	Amount of incoming organic waste (F16) x yield [compost:waste] x Pc / DMc / GEc	(23)	(24)	(4)
F1001	Food Waste (ret, av.)	Food waste trade (F8) x ratio avoidable fw in commerce	(25)	(20)	(21), (22)
F1002	Food Waste (ret, unav.)	Food waste trade (F8) x ratio unavoidable fw in commerce	(25)	(20)	(21), (22)
F57	Exported Crops	Area cultivated for crop i x yield of crop i x Pc / DMc / GEc *(see text for details)*	(1), (26)	(27)	(28)
F29	Leaching / runoff	Precipitation x non-built area x leaching co-efficient (~0) x Pc / DMc / GEc	(29), (30)		

F10	Own produce	0.5% of current consumption of plant-based products, 1% in 2030; assumption based on (EcoRes sprl et al., 2015)	(31)		
F26	Green Waste	Amount of *"professional green waste"* from (Zeller et al. 2019) x Pc / DMc / GEc	(19)	(32)	(32)
F30	Atmospheric deposition	Precipitation x non-built area x Pc / DMc / GEc *(~0)*	(29), (30)		
F19	Precipitation	Precipitation x (built + non-registered area) x Pc / DMc / GEc *(~0)*	(29), (30)		
F17	Respiratory heat	Energy in food consumed – energy in excretions			
F52	Sludge (septic tanks)	Population x specific production of septic sludge	(33)		
F16	Org. waste (dec. comp.)	Population x specific FW generation x ratio in decentralised compost x Pc / DMc / GEcs	(34)	(20)	(21), (22)
F53	Pet excrements$^{(P)}$	P digested by animals (F6) x ratio P excreted/P digested (100%) x ratio of pet excrements to soil	(35)		
F11	Municipal wastewater	Population x specific wastewater generation in BCR x DMc / GEc P excreted (98% of P consumed) + P in detergents	(9)		(36), table 3-16
F14	Food Waste (Con.)	Sum of all food waste from households, services and HoReCa (avoidable and unavoidable)			
F15	Green Waste (Con.)	Household green waste			
F202	Food Waste (hh, unav.)	(Population x specific FW generation x ratio unavoidable hh FW – F16) x Pc / DMc / GEc			
F201	Food Waste (hh, av.)	(Population x specific FW generation x ratio avoidable hh FW – F16) x Pc / DMc / GEc	(25), (34), (37), (38)	(20)	(21), (22)
F203	Food Waste (HoR, av.)	Food waste from HoReCa in BCR x ratio avoidable HoReCa FW x Pc / DMc / GEc	(19), (25)	(20)	(21), (22)
F204	Food Waste (HoR, unav.)	Food waste from HoReCa in BCR x ratio unavoidable HoReCa FW x Pc / DMc / GEc	(19), (25)	(20)	(21), (22)
F205	Food Waste (serv, av.)	Food waste from services in BCR x ratio avoidable services FW x Pc / DMc / GEc	(19), (25)	(20)	(21), (22)
F206	Food Waste (serv, unav.)	Food waste from services in BCR x ratio unavoidable services FW x Pc / DMc / GEc	(19), (25)	(20)	(21), (22)
F20	Wastewater to treatment	Amount of ww from BCR to each WWTP x Pc / DMc / GEc : F901 + F902	(9), (33)	(9), (10)	(11)
F2106	Energy loss (biogas) (PrE)	Energy losses from biogas in the two WWTPs : F915 + F914			
F2115	Gas emissions (WAO)	Mass Balance in WAO process F2102 – F2105			

F36	Sludge export	Total (treated) sludge to landfill or incinerator (in BE or abroad) : F2120 + F2219			
F37	Sludge to AG	Total sludge to AG : F2119 + F2218			
F39	Energy from WWT (PrE)	Energy Balance			
F35	Treated Water	Total treated water from both WWTPs : F905 + F906			
F901	Wastewater (N)	Amount of ww from BCR to N.WWTP x Pc / DMc / GEc	(9), (33)	(9), (10)	(11)
F902	Wastewater (S)	Amount of ww from BCR to S.WWTP x Pc / DMc / GEc	(9), (33)	(9), (10)	(11)
F903	WW from Fl. (N)	16% of inflow to N.WWTP x Pc / DMc / GEc of wastewater *(same for all scenarios)*	(33)	(9), (10)	(11)
F904	WW from Fl. (S)	12% of inflow in S.WWTP x Pc / DMc / GEc of wastewater *(same for all scenarios)*	(33)	(9), (10)	(11)
F2118	Lime (N)	Amount of sludge treated for AG reuse in N.WWTP x lime dosage (*Pc and GEc of lime ~0*)	(39)		
F2216	Lime (S)	Amount of sludge treated for AG reuse in S.WWTP x lime dosage (*Pc and GEc of lime ~0*)	(39)		
F908	Energy for WWT (N) [PrE]	Total ww treated in N.WWTP (F901+F903) x specific energy consumption N.WWTP			(40)
F911	Energy for WWT (S) [PrE]	Total ww treated in S.WWTP (F902+F904) x specific energy consumption N.WWTP			
F909	Energy for Sludge tr. (N) [PrE]	Energy Balance			
F912	Energy for Sludge tr. (S) [PrE]	Energy Balance			
F905	Treated water (N)	Amount equals incoming water (F901+F903) x Pc / DMc / GEc	(9), (33)	(9), (10)	(11)
F907	Sludge (N)	Solids : ww in N.WWTP x TSSc(ww,N) x solids removal efficiency N.WWTP (x GEc) Wet sludge : (solids / solids content of raw sewage sludge) x sludge density Phosphorus : ww in N.WWTP x P removal efficiency N.WWTP	(9), (10), (36)	(10)	(41)
F906	Treated water (S)	Amount equals incoming water (F902+F904) x Pc / DMc / GEc			
F916	Sludge (S)	As sludge in the N.WWTP (F907)			
F2101	Sludge (thick.)$^{(P,E)}$	Assumption : 100% of incoming solids and P (F907) pass in thickened sludge (x GEc)			(41)
F2113	Water from centrifuge	(Volume of sludge in the centrifuge – volume of sludge out) x water density (assum.: Pc=0)			
F2102	Digested Sludge (WAO)	Thickened sludge (F2101) x (1 - solids removal eff. AD) x **share of digested sludge to tech.**	(42)		

F2116	Digested sludge (AG)	Thickened sludge (F2101) x (1 - solids removal eff. AD) x **share of digested sludge to AG**	(42)	
F2103	Biogas(E)	Sludge in AD (F2101) x solids rem. eff. AD x constant for energy (approximation after (Cano et al., 2015))		(11), (42)
F2105	Oxidized sludge	Solids : Digested sludge (F2102) x (1 – solids removal eff. WAO) (x GEc) Wet sludge: (solids/TSc of oxid.sludge) x density of oxid.sludge (assum: ~density raw sludge)	(42)	(41), (42)
F2117	Sludge for AG	Solids : equal to (4102,4104) (xGEc) ; wet sl.: solids / TSc of pressed sludge; P: unchanged	(42)	
F2114	Water from press	Mass Balance		
F2120	Tr. Sludge exp. (N)	Solids : equal toF2105 (xGEc) ; wet sl.: solids / TSc of pressed sludge; P: unchanged	(42)	
F2119	Tr. Sludge to AG (N)	Dewatered sludge to AG + lime (assum: Pc and Wc of lime $\rightarrow$0) : F2117 + F2118		
F915	Energy losses AD (N) [PrE]	Energy in biogas (F2103) x share of energy loss from biogas		(11)
F2107	Electricity biogas (N) [PrE]	Energy in biogas (F2103) x share of electricity from biogas		(11)
F2108	Heat Biogas (N) [PrE]	Energy in biogas (F2103) x share of heat from biogas		(11)
F910	Energy from AD (N) [PrE]	Energy Balance		
F2110	Energy (centrif.) [PrE]	Total ww treated in N.WWTP (F901+F903) x specific energy consumption of centrifuge		(43)
F2109	Energy for AD (N) [PrE]	Total ww treated in N.WWTP (F901+F903) x specific energy consumption of AD		(43)
F2111	Energy for WAO [PrE]	Total ww treated in N.WWTP (F901+F903) x specific energy consumption of WAO		(39)
F2112	Energy for filter press [PrE]	Total ww treated in N.WWTP (F901+F903) x specific energy consumption of filt. press		(43)
F2104	Sludge Liquor (N)	(F2113 + F2114) (xGEc)		(41)
F2201	Thickened sludge (S)	Solids & P: 100% of incoming sludge (F916) (xGEc); wet sl.: solids/TSc centrifuged sludge;	(42)	
F2213	Water from thickener (S)	Mass Balance in thickener		
F2202	Digested sludge (S)	Solids in thickened sludge (F2201) x (1 – solids removal efficiency AD) / TSc dig. sl. * density digested sludge ; P : 100% of incoming sludge;		
F2203	Biogas (S)(E)	Solids in thickened sludge (F2201) x solids removal efficiency AD x constant for energy		(11)

F2219	Tr. Sludge exp. (S)	Solids & P : (F2202) x **share of sludge exp.** (x GEc) ; wet sl.: solids / TSc of pressed sludge	(42)		(11)
F2217	Sludge for AG (S)	Solids & P : (F2202) x **share of sludge to AG** (x GEc); wet sl.: solids/TSc of pressed sludge	(42)		(11)
F2214	Water from dewatering (S)	Mass Balance			
F2218	Tr. Sludge to AG (S)	Dewatered sludge to AG + lime (assum: Pc and Wc of lime →0) : F2217 + F2216			
F914	Energy losses AD (S) [PrE]	Energy in biogas (F2203) x losses share in biogas utilization			(11)
F2207	Electricity biogas (S) [PrE]	Energy in biogas (F2203) x biogas energy to electricity co-efficient			(11)
F2208	Heat from biogas (S) [PrE]	Energy in biogas (F2203) x biogas energy to heat co-efficient			(11)
F913	Energy from AD (S) [PrE]	Energy Balance			
F2210	Energy for thickener (S) [PrE]	Total ww treated in S.WWTP [(401,406)+(402,406)] x specific energy consumption of centrifuge			(43)
F2209	Energy for AD (S) [PrE]	Total ww treated in S.WWTP [(401,406)+(402,406)] x specific energy consumption of AD			(43)
F2212	Energy for dewatering (S) [PrE]	Total ww treated in S.WWTP [(401,406)+(402,406)] x specific energy consumption of press			(43)
F2204	Sludge Liquor (S)	(F2213 + F2214) x GEc			(41)
F34	WW from Flanders	Sum of ww from Flanders to N. and S. WWTPs : F903 + F904			
F38	Energy for WWT [PrE]	Energy balance for the two WWTPs			
F920	Lime	Sum of lime consumed in the N. and S. WWTPs : F2118 + F2216			
F21	Food waste to inc.	FW,hh (F201 + F202) x **ratio of FW,hh to incineration** + FW,prof. (F203 + F204 + F205 + F206) x **ratio of FW,prof to incineration** (x Pc / DMc / GEc)	(19)	(20)	(21), (22)
F46	Green Waste to inc.	GW,hh (F15) x **ratio of GW,hh to incineration** + GW,prof. (F26) x **ratio of GW,prof to incineration** (x Pc / DMc / GEc)	(19)	(32)	(32)
F22	Food Waste to AD	FW,hh x **ratio of FW,hh to AD** + FW,prof. x **ratio of FW,prof to AD** (x Pc / DMc / GEc)	(19)	(20)	(21), (22)
F47	Green Waste to AD	GW,hh (F15) x **ratio of GW,hh to AD** + GW,prof. (F26) x **ratio of GW,prof to AD** (x Pc / DMc / GEc)	(19)	(32)	(32)

F23	Food waste to comp. (exp.)	FW,hh x **ratio of FW,hh to compost (e)**+ FW,prof. x **ratio of FW,prof to compost (e)** (x Pc / DMc / GEc)	(19)	(20)	(21), (22)
F48	Green Waste to comp (exp)	GW,hh (F15) x **ratio of GW,hh to compost (e)** + GW,prof. (F26) x **ratio of GW,prof to compost (e)** (x Pc / DMc / GEc)	(19)	(32)	(32)
F24	Green Waste to comp.	GW,prof. (F26) x **ratio of GW,prof to compost (c)** (x Pc / DMc / GEc)	(19)	(32)	(32)
F801	Other wastes	Total amount of waste in the incinerator – food and green waste in the incinerator	(37)		
F31	Ash	Input waste (F802) x waste-to-ash ratio (x Pc/DMc/GEc)	*see text*	(44)	Assum. ~0
F40	Electricity from Inc. [PrE]	If electricity generated (F804) < auto-consumption (F807) → 0, else: F804 – F807			
F41	Energy for Inc. [PrE]	If electricity generated (F804) > auto-consumption (F807) → 0, else: F807 – F804			
F809	Energy loss (Inc.) [PrE]	Equals to losses from incinerator (F808)			
F56	Heat from Inc. [PrE]	Equals to heat recovered from incinerator (F805)			
F813	Other by-products	Sum : F810 + F811 + F812			
F802	Waste to Inc. (all)	Sum of organic material inputs into the incinerator : F41 + F46			
F804	Electricity inc. [PrE]	Total energy input ($F802_{EN}$ + $F807_{EN}$ + $F814_{EN}$) x efficiency of turbine (electr.)			(44)
F805	Heat inc. [PrE]	Total energy input ($F802_{EN}$ + $F807_{EN}$ + $F814_{EN}$) x heat co-efficient			(44)
F808	Heat losses [PrE]	Energy balance around incinerator			
F810	Metals	Input waste (F802) x waste-to-metals ratio (assum: Pc=GEc ~0)	*see text*		
F811	Slug	Input waste (F802) x waste-to-slug ratio (assum: Pc=GEc ~0)	*see text*		
F812	Salts	Input waste (F802) x waste-to-salts ratio (assum: Pc=GEc ~0)	*see text*		
F807	Auto- consumption Inc. [PrE]	Total material input x specific electricity auto-consumption of incinerator			(45), (46)
F814	Fuel in incineration [PrE]	Total material input x specific fuel consumption of incinerator			(45), (46)
F54	Compost (dig.)	Digestate+GW (F33 + F47) to AD x yield [compost:waste] x Pc / DMc / GEc	(23)	(24)	(4)
F44	Electricity Out. AD [PrE]	If electricity generated (F1204) < electricity needs (F1206 + F1208)→0, else: F1204 – (F1206 + F1208)			
F1205	Energy loss (AD) [PrE]	Equals to losses in biogas (F1202)			

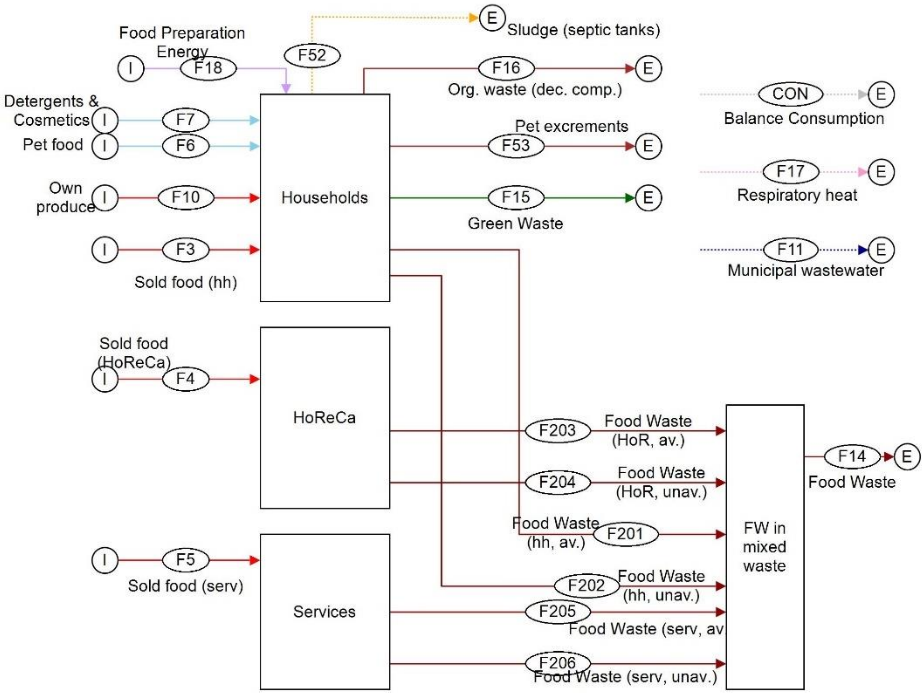
F55	Heat out AD [PrE]	If heat recovered (F1203) < heat demand (F1207) → 0, else: F1203 – F1207			
F1201	Biogas bw[E]	FW to AD (F22) x yield [biogas:org.waste] x GEc (Pc=0)	(44)		(44)
F33	Solid digestate	FW to AD (F22) x yield [raw digestate:waste] x Pc / DMc / GEc	(47)	(48)	(4)
F49	Liquid digestate	*Assumption : only raw (solid) digestate produced*			
F1202	Energy loss bw [PrE]	Energy in biogas (F1201) x losses share in biogas utilization			(11)
F1203	Heat from AD bw [PrE]	Energy in biogas (F1201) x biogas energy to heat co-efficient			(11)
F1204	Electricity from AD bw [PrE]	Energy in biogas (F1201) x biogas energy to electricity co-efficient			(11)
F1206	Electricity for AD bw [PrE]	FW to AD (F22) x specific electricity demand for AD of org.waste			(44)
F1207	Heat for AD bw [PrE]	FW to AD (F22) x specific heat demand for AD of org.waste			(44)
F1208	Energy for compost [PrE]	Digestate + GW (F33+F47) x specific energy demand of composting			(44)
F43	Energy In.AD [PrE]	Energy Balance			
F32	Compost (exp.)	Input food & green waste (F23+F48) x yield [compost:waste] x Pc / DMc / GEc	(23)	(24)	(4)
F50	Wastewater EComp	Input food & green waste (F23+F48) x compost ww co-efficient x Pc / DMc / GEc (~0)	(49)	-	-
F45	Energy Comp. [PrE]	Input food & green waste (F23+F48) x specific energy demand of composting			(44)
F51	Wastewater CComp	Input green waste (F24) x compost ww co-efficient x Pc / DMc / GEc (~0)	(49)		
F25	Compost	Input green waste (F24) x yield [compost:waste] x Pc / DMc / GEc	(23)	(50)	(4)
F42	Energy Comp. [PrE]	Input green waste (F24) x specific energy demand of composting			(44)
AD	Balance AD	Mass & Energy Im-balance around process AD			
CCO	Balance CCO	Mass & Energy Im-balance around process Centralised Compost			
CON	Balance Consumption	Mass & Energy Im-balance around process Consumption			
DCO	Balance DCO	Mass & Energy Im-balance around process Decentralised Compost			
ECO	Balance ECO	Mass & Energy Im-balance around process Exported Compost			
INC	Balance Incineration	Mass & Energy Im-balance around process Incineration			
SWC	Balance SWC	Mass & Energy Im-balance around process Solid Waste Collection			

TRA	Balance trade	Mass & Energy Im-balance around process Trade
US	Balance US	Mass & Energy Im-balance around process Urban Soils
WWC	Balance WWC	Mass & Energy Im-balance around process Wastewater Collection
WWT	Balance WWT	Mass & Energy Im-balance around process Wastewater Treatment

(1) (Statbel, 2015a); (2) (VLM, 2014); (3) (REEW, 2019a) ; (4) (Phyllis2, n.d.) ; (5) (IBSA, 2015) ; (6) (IBGE, 2017a) ; (7) (IBGE, 2015a) ; (8) (IBGE, 2017b) ; (9) (SBGE, n.d.) ; (10) (IBGE, 2018a) ; (11) (Cano et al., 2015) ; (12) (Brocatus et al., 2016) ; (13) (Ost, 2015) ; (14) (Nubel, 2018) ; (15) (RIVM, 2016) ; (16) (Statbel, 2018) ; (17) (Wu et al., 2016) ; (18) (Ott and Rechberger, 2012) ; (19) (Vanessa Zeller et al., 2019) ; (20) (Fisgativa et al., 2016) ; (21) (Zeller, 2019) ; (22) (Jensen et al., 2016) ; (23) (Bruxelles-Compost, n.d.) ; (24) (Möller et al., 2018) ; (25) (Roels and Van Gijseghem, 2017) ; (26) (Statbel, 2014) ; (27) (Le Noë et al., 2017) ; (28) (Wirsenius, 2000) ; (29) (IBGE, 2015b) ; (30) (IBSA, 2017) ; (31) (EcoRes sprl et al., 2015) ; (32) (Boldrin and Christensen, 2010) ; (33) (IBGE, 2012) ; (34) (Bortolotti et al., 2018a) ; (35) (Theobald et al., 2016) ; (36) (Tchobanoglous et al., 2014) ; (37) (Bruxelles-Propreté, 2016) ; (38) (Bruxelles-Propreté, n.d.) ; (39) (Houillon and Jolliet, 2005) ; (40) (Aquiris, n.d.) ; (41) (Shizas and Bagley, 2004) ; (42) (Chauzy et al., 2010) ; (43) (Longo et al., 2016) ; (44) (Haupt et al., 2018) ; (45) (Bruxelles-Propreté, 2014) ; (46) (IBGE, 2019) ; (47) (Vlaco, 2012) ; (48) (Vlaco, 2016b) ; (49) (Bortolotti et al., 2018c) ; (50) (Vlaco, n.d.)

2.7 Annex II : Model Description

2.7.1 Consumption and trade



We estimate the sold food to households, HoReCa and services (school and office canteens) as the sum of the food consumed in them and the food waste they generate:

$$Food\ Sold_i = Food\ Consumed_i + \ Food\ Waste_i\ ,$$
$$i = hh, HoReCa, serv$$

We first compile the data for the amounts of food, phosphorus and energy consumed per capita, and then extrapolate them for all inhabitants, visitors and incoming commuters.

- *Food consumed*

Food consumption in BCR was estimated with data from the Belgian Food Consumption Survey (FCS) 2014-15 (Ridder et al., 2016). The report does not include separate values for the consumption in Brussels, so we used the national values. The results are reported in two classification systems: the *Globo*, used in the earlier Belgian Food Consumption Surveys, and the *FoodEx2*, which is the one recommended by EFSA and the one that most countries are using. In our study we used the results classified according to FoodEx2, which includes 18 food groups and 62 subgroups (Table S1). We accounted for all food sub-groups for which ‘usual consumption’ data were available, except for two: the special children’s food, since it is obviously only reported for the age group 3-9 and the food for special diets, the consumption of which was negligible.

Table S1 Food Groups, sub-groups and representative products included in the calculation of the per capita food consumption

Food group	Food sub-group	Representative product	Food group	Food sub-group	Representative product
grains	Cereals and derived products	rice, parboiled, cooked	Eggs	Eggs, raw	egg
	Bread and similar products	bread, white		Processed eggs	boiled egg
	Pasta and similar products	Pasta, extra, cooked	Dairy	Milk and cream	cow milk
	Bakery products	croissant		Milk and cream, fermented	cow yoghurt
	Breakfast cereals	breakfast cereals		Milk, powder and concentr.	conc. Milk
vegetables	Leafy vegetables	lettuce		Cheese	gouda
	Brassica vegetables	broccoli		Milk desserts	ice cream
	Stem vegetables	celery	Sugar and confectionery	Sugar and sweeteners	white sugar
	Bulb vegetables	onion		Sweeteners	fructose
	Legume vegetables	green pea		Confectionery and chocolate	chocolate, dark
	Fruit vegetabales	tomato		Sugary desserts	sorbet
	Tube vegetables	carrot	Oils & fats	oils	olive oil
	Herbs and edible flowers	basil		fats	butter
	Conserved vegetables	canned tomatoes	Juices	Fruits & veg. juices	orange juice

roots & tubes	Starchy roots & tubes	potato, old, boiled, peeled	Water & bev.	Sugary beverages	cola
Legumes & nuts	Nuts & grains	walnut	Alcohol	Beer	beer, pils
	Processed legumes & nuts	peanut butter		Wine	red wine
Fruits	Fruits	apple		Spirits & liquors	whisky
	Processed fruits	jam	Coffee & tea	Coffee, tea, cacao	coffee, instant
Meat	Mammals and poultry	beef (roastbeef, roasted)		Warm beverages	coffee (prepared)
	Cold cuts	ham, smoked raw	Composite dishes	Prepared dishes	pizza margherita
	Sausages	pork sausage (raw)		Soups & salads	vegetable soup (powder)
	Meat specialties	pate		Chips	potatoe crisps
Fish	Fish	cod, cooked	Condiments	Seasoning	salt
	Processed fish	smoked salmon		Bouillon	beef bouillon cubes
				Sauces	mayonnaise

- *Food composition*

The phosphorus (P), dry matter (DM) and gross energy content (GE) of the different food groups were derived from *internubel*, the Belgian nutritional database (Nubel, 2018). The database gives the composition per food item, so that one food item was chosen to represent each sub-group (Table S1), usually the first one to appear in the list of examples given in the FCS report (Brocatus et al., 2016). In a few cases where information on the product was not available on *internubel*, *NEVO*, the respective Dutch database was used instead (RIVM, 2016). The results, aggregated to food groups are given in Table S2.

Table S2 Average daily per capita consumption of food products, dry matter, phosphorus and energy in Belgium.

Food groups	Consumption [g/cap.d]	Dry matter [g/cap.d]	Phoshorus [mgP/cap.d]	Energy [kJ/cap.d]
Grains	230.7	143.3	207.3	2564.8
Vegetables	133.4	9.6	48.9	113.8
Roots & tubes	43.3	8.2	17.3	115.1
Legumes & nuts	5.8	5.6	23.4	162.1
Fruits	113.3	26.8	12.1	380.3
Meat	98.8	35.8	229.7	683.9
Fish	17.8	5.1	53.7	95.3
Dairy	183.5	44.3	370.9	1063.9
Eggs	9.6	2.5	34.4	61.7
Sugar & confectionary	94.7	36.4	34.5	715.5
Oils & fats	17.8	16.0	2.6	596.0
Juices	64.9	7.1	12.3	107.1
Water and non-alcoholic beverages	1063.2	22.3	44.6	392.8
Alcohol	377.9	16.4	24.7	302.2
Coffee & tea	296.6	5.5	0.0	49.3
Composite dishes	55.3	23.6	72.0	445.5
Condiments	29.1	24.2	9.9	869.6
Others	0.0	0.0	0.0	0.0
TOTAL	**2835.6**	**432.9**	**1198.5**	**8718.9**

- *Food consumed outside home*

By inhabitants: The Food Consumption Survey includes information on the locations where meals are consumed (Ost, 2015). Meals reportedly consumed at “Restaurant” are accounted for in the HoReCa sector; meals consumed at “School/ work” are accounted for in the services sector (Table S3). All the rest are included in the meals assumed to be eaten at home, or more precisely, to be shopped for by the ‘household’ sector. We assume the results for the whole country to be representative for Brussels.

Table S3 Share of meals consumed according to different locations in Belgium [Source: (Ost, 2015)]. The last row gives the final figures used for food consumed by inhabitants at the household, services (school/work) and HoReCa sectors.

Meal	Home	School /work	Restaurant	At family / friends	On the go	Other
Breakfast	89.0%	5.8%	0.5%	1.8%	2.0%	0.9%
Lunch	54.6%	29.1%	5.8%	5.2%	3.9%	1.4%
Dinner	86.3%	1.7%	4.0%	6.6%	0.9%	0.6%
AVG all meals	**84.4%(1)**	**12.2%**	**3.4%**			

(1) Including also "at family/friends", "on the go", and "other"

By visitors: We assume that the food intake of visitors is the same as for the inhabitants and that each visitor spending a night in Brussels consumes the equivalent of a daily intake in a HoReCa establishment. Multiplying the daily intake by the number of nights spent in Brussels, we arrive at the total amount of food consumed by visitors.

By commuters: Several thousands of people commute every day to Brussels. In 2014, more than 360'000 people were commuting to Brussels while living in either Flanders or Wallonia, while only around 70'000 workers were making the reverse travel. To estimate the amount of food consumed by commuters, we assume that all commuters have lunch at work (services). In practical terms, this means that 1/3 of the daily food consumption of the net number of commuters (incoming minus outgoing) is consumed in the services sector of BCR. To convert the daily consumption to annual, we use the average number of working days in Belgium (230).

Adding the food waste generated by each sector to the food consumed, we arrive to the food sold to households, HoReCa and services (Eq. A-C):

Food sold to households :

$$F3 = FC_{inh} \cdot a_{fc,hh} + FW_{hh} \qquad \text{(Eq.A)}$$

Food sold to HoReCa :

$$F4 = FC_{inh} \cdot a_{fc,HoReCa} + FC_{vis} + FW_{HoReCa} \text{(Eq.B)}$$

Food sold to services :

$$F5 = FC_{inh} \cdot \alpha_{fc,services} + FC_{com} + FW_{services} \text{(Eq.C)}$$

FC_i : Food Consumed by inhabitants (inh), visitors (vis) and commuters (com)

$a_{fc,j}$: share of food consumed by inhabitants at home (hh), HoReCa and services

FW_j : Food Waste generated by households (hh), HoReCa and services

Table S4 Shares and absolute amounts of food consumed in households, restaurants/ cafés/ hotels (HoReCa), and work and school canteens (Services) in Brussels for inhabitants, commuters and visitors (2014). The number of inhabitants, net daily commuters and annual nights spent by visitors are given in brackets.

	Inhabitants (1'163'486)		Commuters (288'816)		Visitors (6'290'243)		TOTAL
	% of consumption	[tn/yr]	% of consumption	[tn/yr]	% of consumption	[tn/yr]	[tn/yr]
Households	84.4%	458,563	-	0	-	0	**458,563**
HoReCa	3.4%	18,473	-	0	100%	8,459	**26,932**
Services	12.2%	66,285	100%	28,329	-	0	**94,614**
TOTAL	**100%**	**543,321**		**28,329**		**8,459**	**580,109**

- *Non-food products & wastewater*

Detergents: Wind (2007) estimates the phosphorus amount in ADD (automatic dishwashing detergents) for Belgium and Luxembourg to be 910 t/a. This value corresponds to a per capita annual consumption of **0.085 kg/cap.a**. Coppens and colleagues (2016) use this same value for their phosphorus SFA of Flanders, citing the European phosphorus balance compiled by Ott and Rechberger (2012). Lacking more refined data, we use this value for Brussels as well.

Pets: In 2014, 7% and 18% of the households in BCR owned at least one dog or cat respectively, with an average of 1.19 dogs per dog-owning household (0.09 for all households) and 1.47 cats per cat-owning household (0.26 for all households) (Statbel, 2018). The total population of pets in BCR, thus, sums up to 48'640 dogs and 140'514 cats. The phosphorus intake of pets was taken from (Wu et al., 2016) who used assumptions for the P intake of cats (0.27 kg/cat.yr) and dogs (1.2 kg/dog.yr) based on (Baker et al., 2007). The animals are assumed to excrete 100% of their nutrient intake (Wu et al., 2016) and we assume that 60% of the excrements tend up directly in urban soils (Theobald et al., 2016). The rest 40% is directed to the MSWM system.

- *Household food waste*

In order to be able to extrapolate the amounts of waste generated into the future, we first calculate the per capita food waste generation. We start from the total amount of mixed MSW collected by Bruxelles-Proprété (the ‘white bags’), 66% of which is household waste (Bruxelles-Propreté, 2016) and 50% of which is organic (Bortolotti et al., 2018a). Other sources, e.g. (Zeller et al., 2018) and (IBGE, 2015a), use similar estimations for the share of organic to total waste. Assuming further that all organic waste is food (kitchen) waste, the per capita food waste generation in Brussels is :

$$q_{fw} = \frac{Q_{MSW}(2014) \cdot (share\ hh\ to\ total\ MSW) \cdot (share\ organic\ to\ total\ MSW)}{Population_{BCR}(2014)}$$

$$q_{fw} = \frac{316'687tn \cdot 66\% \cdot 50\%}{1'163'486} \cdot \left(\frac{10^3 kg}{tn}\right) = \frac{104'506'710'000}{1'163'486}$$

$$q_{fw} = 89.48 \frac{kg}{cap \cdot yr}$$

The value agrees well with the app. 90 kg/cap.d given in (Vanessa Zeller et al., 2019). 423 t of household food waste were composted in private and communal composting sites in 2015 (F16), representing 0.4% of the total food waste production (Bortolotti et al., 2018a).

- *Household green waste*

We use the value of 26’449 tn/yr from (Vanessa Zeller et al., 2019) for the year 2014. This value agrees with the corresponding estimation by Opération Phosphore for 2010 (24’383 tn/yr) (Bortolotti et al., 2018a). The per capita green waste generation in BCR is thus:

$$q_{gw} = \frac{26'449}{1'163'486} \frac{\left[\frac{tn}{yr}\right]}{cap} \Rightarrow q_{gw} = 22.73 \frac{kg}{cap \cdot yr}$$

- *Professional food waste*

Professional food waste refers to food waste from the “Administration and Education” and “Health” sectors, together making the FW_{serv} and from HoReCa (FW_{HoReCa}) and we use the quantities estimated by (Vanessa Zeller et al., 2019) for 2014.

For the scenarios for 2030 we assume no changes in the absolute amounts of professional organic waste, unless explicitly dictated by the scenario, which is the case for the two FWA ones. This assumption is the safest one to make, due to the lack of data on the evolution of professional waste. Bruxelles-Propreté has been reporting the collected quantities of household and professional waste separately since 2014 only. The trend for these 3 years, for which the amounts of professional organic waste collected by ABP are available, is a decreasing one. So, although we cannot make any safe assumptions on the overall trend from just 3 data points, we have an indication that a stable rate of professional organic waste generation is potentially realistic.

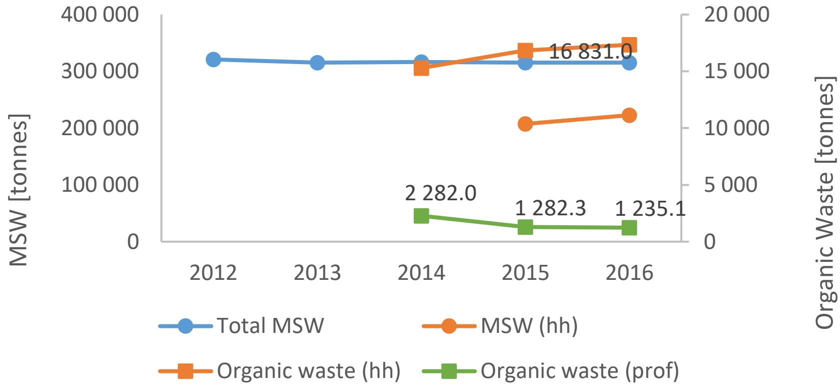


Figure S 1 Annual amounts of municipal solid waste (MSW – white sacs) and organic waste (garden and green waste) collected in Brussels between the years 2012-2016. [own elaboration with data from IBGE 2012]

- *Professional green waste*

The only economic sector that generates green waste in BCR is the one of "Administration and Education" (Vanessa Zeller et al., 2019). Most of these green wastes come from parks and green spaces (Recydata, 2014). Therefore, we model all professional green waste flowing directly from Urban Soils to Solid Waste Collection.

For the scenarios, we assume no changes in the area and management practices of parks and green spaces for BCR. Therefore, the same amount of professional green waste is generated in 2030 as in 2014. In addition, no green waste is generated by HoReCa and trade (same as for present).

Table S5 Shares and absolute amounts of food consumed in households, restaurants/ cafés/ hotels (HoReCa), and work and school canteens (Services) in Brussels for inhabitants, commuters and visitors (2014). The number of inhabitants, net daily commuters and annual nights spent by visitors are given in brackets.

Sector	Food Losses	Residues
Households	45%	55%
Catering (services)	95%	5%
Hospitality sector (HoReCa)	28%	72%
Food retail	67%	33%
Food processing	*10%*	*90%*
Fisheries	*50%*	*50%*
Agriculture	*74%*	*26%*
Auctions	*96%*	*4%*
Total chain	**26%**	**74%**

- *Avoidable food waste*

In order to estimate the shares of avoidable food waste, we use a recent report on food waste in Flanders (Roels and Van Gijseghem, 2017). The authors of the report distinguish between "residue" (the inedible for our culture part of food products) and "food loss" (edible parts of food products that could have been eaten). We adopt these classification and use from now on the terms "avoidable food waste" and "food loss" interchangeably. The report contains the more detailed currently available data for Belgium, to our knowledge; however, many of the results are based on estimations and assumptions. Assuming further that the data for Flanders represents the situation in Brussels well enough introduces additional uncertainty to the data. As a result, these numbers (Table S5) are simply indications and should be interpreted with great attention.

- *P and Energy content of organic wastes*

Lacking measured data on the composition of waste in BCR, we used literature values from (Fisgativa et al., 2016). The average P content of household food waste is 0.5% of dry matter (DM) with a DM content of 24%; the value agrees with (Coppens et al., 2016), who report a P content of 0.11% for organic waste in Flanders (VGF – vegetable garden and fruit):

$$25\%\ {}^{DM}/_{FM} \cdot 0.5\%\ {}^{P}/_{DM} = 0.12\%\ {}^{P}/_{FM} \approx 0.11\%\ {}^{P}/_{FM}$$

Gross energy: There are some studies that give values on the energy potential of food and other domestic organic waste. These values typically range from 10-18 MJ/kgDM (HHV). (Bernstad and la Cour Jansen, 2011) report a value of 18 MJ/kg DM for vegetable food waste with a dry matter content of 23%, close to what we assumed as dry matter content of the different food waste streams for BCR. Other sources report lower HHV (app. 10-12 MJ/kg) and higher content of dry substance (app.45-50%). This is, for example the case, for numerous samples of domestic organic waste included in the database Phyllis2 (Phyllis2, n.d.). Bruxelles-Energie and Bruxelles-Environnement are using a net calorific value of 2.41 kWh/kg (Manon Urbain 2019, personal communication) of organic waste to estimate the amount of renewable energy generated in the incinerator.

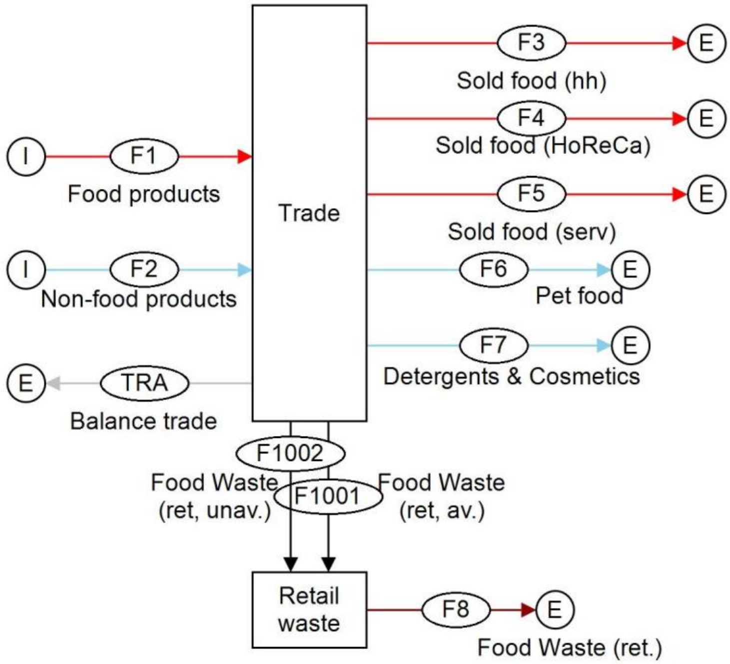
In the end we estimated the energy content following the method used by the researcher currently conducting an LCA study of food waste management in Brussels (V. Zeller et al., 2019), as shown in Table S6.

Table S6 Estimation of the LHV of the food waste entering the incineration as a fraction of the MSW (white bags)

	Share in food waste[(1)] [%]	LHV of fractions[(2)] [MJ/kgFM]	LHV of food waste [MJ/kgFM]
Vegetable food waste	70%	4.21	6.1
Animal food waste	30%	10.55	

(1) (V. Zeller et al., 2019) (2) (Jensen et al., 2016)

2.7.2 Trade



"The total imports of food (F1) and non-food products (F2) into trade are calculated as the sum of sold products (F3 – F7) and waste of the food trade sector (F8) from (Vanessa Zeller et al., 2019). Ratios of avoidable and unavoidable food waste are given in Table S5.

This probably underestimates the values of the import flows and gives an idealized impression of 'Trade' as a tight process. Including trade data, however, would require the combination of national data on international trade with the interregional Input-Output (IO) tables, the translation of IO data in physical quantities and their disaggregation to food product or food group level, and the adoption of assumptions for the shares of food that are re-exported and used for processing. Given the additional uncertainties and complexity that these data treatment steps would introduce to the model, we opted for using the mass balance to calculate the import flows."

- *Process Energy*

We interpolated values on energy consumption for cooking in BCR (F18), given in the regional energy balances of 2013 and 2015 (Table

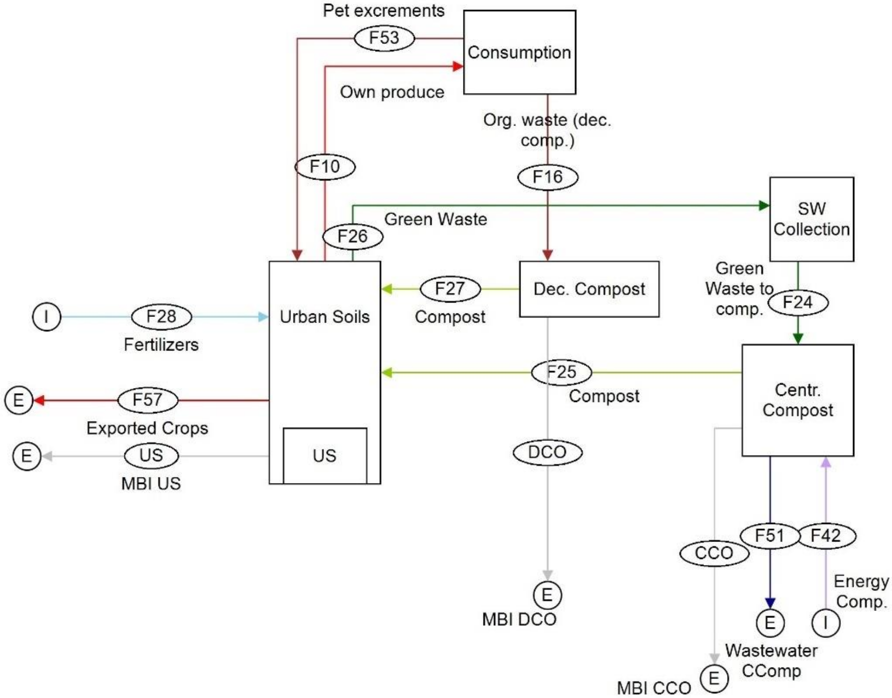
S7). Respiratory heat (F17) is calculated as the difference between the energy content of food items consumed and the energy content of excretions (wastewater).

Table S7 Estimation of the process energy in consumption (energy for food preparation).

Sector	2013(1)	2015(2)	2014
Energy consumption for cooking in BCR [GWh/yr]	309	253	281*
Total residential energy consumption in BCR [GWh/yr]	7’312	8’786	
Population			1’163’486
Per capita energy consumption for cooking [kWh/cap.yr]			241.5
Per capita energy consumption for cooking [MJ/cap.yr]			869.5

(1) (IBGE, 2015a)*; (2) (IBGE, 2017a); * Interpolated value*

2.7.3 Production – Urban Soils



- *Urban agriculture*

Own produce (Flow F10) from private and collective gardens, as well as professional urban agriculture (UA) practice is assumed to be 0.5% of the total plant-based food consumed in BCR for the baseline scenario and 1% for all future scenarios. These assumptions are based on the estimation by the authors of a recent comprehensive study of Brussels' urban metabolism (EcoRes sprl et al., 2015), that if vegetables were cultivated on all open cropland in the region, they would cover around 1% of the total demand. Besides, urban gardening is receiving much attention and support in Brussels and new more productive ways of cultivating the city are also being explored, e.g. rooftops and (bio)intensive cultivation in very small surfaces (Boutsen et al., 2018).

- *Conventional crop production in BCR*

BCR had a total of 1'852 ha of cropland in 2014, which were used for the production of grains and industrial crops (Table S8). To estimate the mineral P inputs in these agricultural soils we assume the same application rate per ha of agricultural land as for Wallonia (14.9 kgP_2O_5/ha) (REEW, 2019a). The total amount of P imported as mineral fertilizer is then app. 12 tnP / yr (Eq.D). Another 3.6 tnP/yr entered the region in the form of manure imported from Flanders (VLM, 2014).

$$P_{fert} = 1'852ha \cdot 14.9 \frac{kgP_2O_5}{ha} \cdot 0.4364 \frac{kgP}{kgP_2O_5} = 12.04 \frac{tnP}{yr}$$

(Eq.D)

For a rough estimation of the P content of the exported crops (F57), we combine agricultural data from StatBel on cultivated area per crop and crop yields (Table S8), with figures on P content of various crops from (Le Noë et al., 2017,SI). Vegetable production is missing from Table S8 because Statbel does not provide information on yields for vegetables. Even so, cereals contain one order of magnitude more P per mass unit and the crops in Table S8 represent 81% of the total cropland in BCR. Therefore, we assume that the total of **54.8 tnP/yr** is a good enough approximation of the exported P through crops.

We assume that P inputs into the soil through rain (F30) are negligible, following the model for the P flows in Flanders (Coppens et al., 2016).

- *Decentralised Compost*

423 t of household food waste (0.4% of total) were composted in private or communal composting sites in 2014. For small scale compost, we assume a yield of 50% compost/incoming waste as for the centralised one. Literature values from (Möller et al., 2018) were used to estimate dry matter and P content. An energy content of 15.75 MJ/kgDM (LHV) was assumed for all composts in the model (Phyllis2, n.d.).

- *Leaching and runoff*

Despite its density and urban character, BCR is still a relatively green and sparse city compared with other European capitals: 27% of its area is covered with green spaces and forest and 40% of the residential buildings have a garden (IBGE, 2017b, p. 63).

Regarding the quality of surface waters, P_{tot} is almost in its entirety originating from domestic and industrial activities and reaches the water bodies through the sewers, especially during wet weather when reservoirs for rainwater storage overflow (IBGE, 2017b, p. 96). Elevated concentrations of phosphorus and orthophosphates have been observed in Woluwe, the origins of which, though, are unknown (IBGE, 2017b, p. 111). Bruxelles-Environnement plans to initiate a study to investigate this issue. Groundwater P-concentrations are below the limit value of 2.185 mg/L (IBGE, 2017b, p. 228); nitrates, on the other hand, are a significant source of pollution for groundwater in Brussels, due to fertilizers use in gardens and green spaces (IBGE, 2017b). Due to the low solubility of phosphates, leaching of P to groundwater is generally negligible; the main losses from soils are due to erosion and runoff (Smil, 2000). Erosion rates are mostly relevant for bare and arable lands; much less so for grasslands and several orders of magnitude lower for forests: (Cerdan et al., 2010) reports mean erosion rates of 17, 6.3, 0.29 and 0.003 t/ha·yr respectively. Given that (i) BCR has a combined sewer network system and (ii) most of the unbuilt surfaces in the region are forests (11% of total and 35% of unbuilt surface) and parks (8% and 25% respectively), while the arable lands only occupy 4% of the total area (11% of unbuilt) (IBSA, 2018), we assume that P-runoff is included in the wastewater and direct runoff from unbuilt areas is negligible. Wastewater Managemen

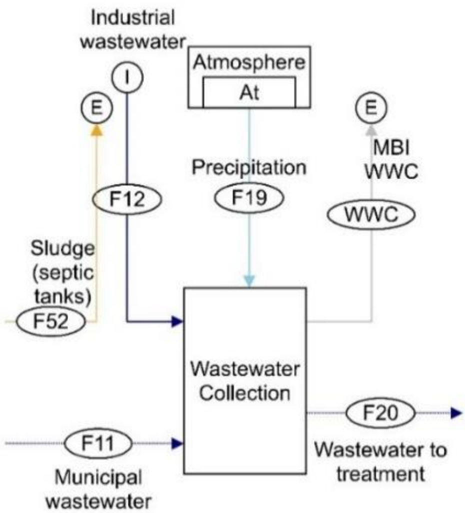
Table S8 Cultivated surface, yield, DM and P content of the main crops cultivated in Brussels Capital Region (Pc : P content, DMc : Dry Matter content, GEc : Gross Energy content

Crop	**Area[1] [ha]**	**Yield[2] [100kg /ha]**	**Prod. [(*)] [100kg /yr]**	**Pc [3] [kgP/ 100kg]**	**P pr[(*)] [tnP /yr]**	**DMc[4] [%]**	**DMpr[(*)] [tn / yr]**	**GEc[4] [kJ /gDM]**	**GE[(*)] [TJ / yr]**
Cereals	964		8, 738		28.1	88%	769	18.4	14.1
Wheat (winter)	608	91.5	5,564	0.33	18.4				
Wheat (spring)	13	63.8	81	0.33	0.3				
Spelt	18	81.7	150	0.33	0.5				
Barley (winter)	174	87.0	1,515	0.33	5.0				
Barley (spring)	16	56.3	90	0.33	0.3				
Oats	10	56.0	57	0.33	0.2				
Corn	72	129.6	936	0.25	2.3				
Triticale	35	74.0	258	0.33	0.9				
Other cereals	17	49.4	86	0.33	0.3				
Other Crops (most important)									
Sugar beets	180	856.6	15,445	0.041	6.3	24%	371	17.0	6.3
Rapeseed	41	44.4	181	0.400	0.7	91%	16	23.4	0.4
Potatoes	135	522.9	7,059	0.025	1.8	21%	148	17.3	2.6
Maize forage	189	525.7	9,932	0.180	17.9	30%	298	17.5	5.2
TOTAL	**1,509**		**41,338**		**54.8**		**1,602**		**28.6**

[(1)] (Statbel, 2015a), [(2)] (Statbel, 2014), [(3)] *(Le Noë et al., 2017)*, [(4)]*(Wirsenius, 2000,Appendix I)* [(*)] *calculated value*

2.7.4 Wastewater collection

We estimate the per capita wastewater generation by deriving the municipal part of the wastewater treated in the two WWTPs (Table S9) and dividing it with the number of inhabitants in BCR in 2014. We then use this value to estimate future wastewater flows for 2030. This flow (F11) includes domestic wastewater, as well as wastewater from services, HoReCa, and trade (F9).



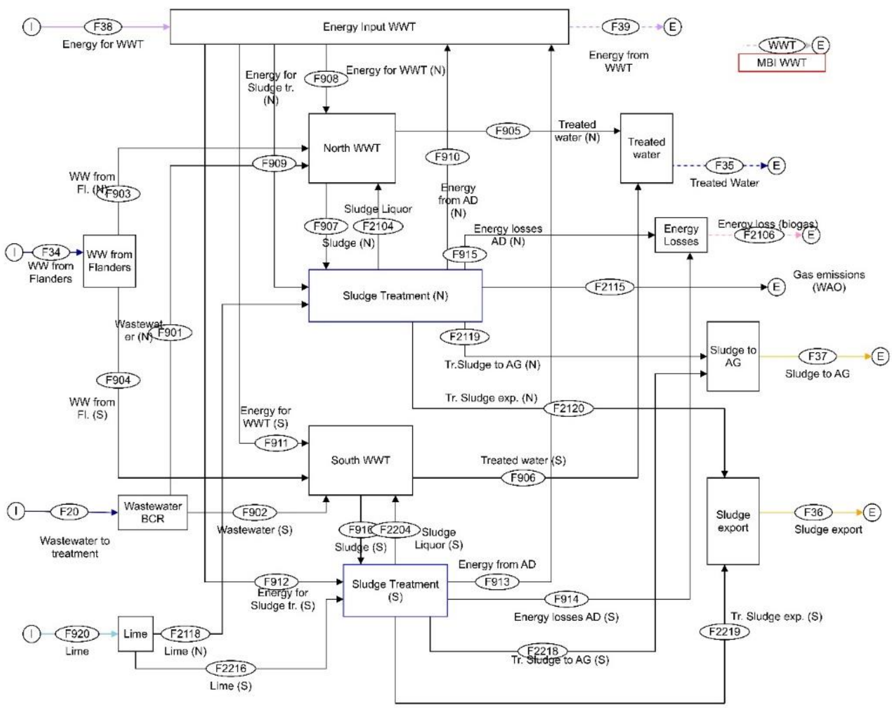
Instead of using a literature P content value for F11, we assume that 98% of P digested by humans is excreted and that all P from detergents ends up in the wastewater (Eq.D). The solids content of F11 is estimated using a 'typical' value for TS in a medium strength municipal wastewater (Tchobanoglous et al., 2014, table 3-16). We chose the value for the 'medium strength' wastewater because its TSS content is 210 mg/L, on the same order like the one of the incoming wastewater to the WWTPs of Brussels (app. 250 mg/L). There are still some septic tanks in BCR, producing app. 4'000 tn/yr of septic sludge (F52), according to the latest available data (IBGE, 2012). 30% of the incoming water in the South WWTP is of industrial origin (IBGE, 2017b). Since there is no further information on the exact origins and characteristics of this industrial wastewater, and data in Table S10 refer to the total wastewater in the plant, we represent industrial wastewater (F12) as an input flow in the *'Wastewater Collection'* process. Finally, as for atmospheric deposition and following (Coppens et al., 2016), we assume a negligible P-content of the rain; therefore, precipitation (F19) does not contribute to the P load of the raw wastewater.

$$F11_P = 98\% \cdot \left(FC_{inh,P} + FC_{vis,P} + FC_{com,P}\right) + F7_P \quad \text{(Eq.D)}$$

Table S9 Shares and absolute amounts of food consumed in households, restaurants/ cafés/ hotels (HoReCa), and work and school canteens (Services) in Brussels for inhabitants, commuters and visitors (2014). The number of inhabitants, net daily

	North WWTP	South WWTP	Source
Treated ww in 2014 [m^3/yr]	104’425’916	23’920’699	See Table S10 S10
Share of ww coming from Flanders [%]	16%	12%	(IBGE, 2012)
Share of industrial ww [%]	0%	30%	(IBGE, 2017b)
Municipal wastewater from BCR only [m^3/yr]	*87’717’769*	*13’874’005*	*calculated*

2.7.5 Wastewater amounts, characteristics and treatment



Brussels has a combined sewer system that channels the municipal wastewater, along with the urban runoff, to two WWTPs: the smaller and older South WWTP , built close to the point where Senne enters the city, and the modern North WWTP, close to the where the Senne cross the city’s northern border. To calculate the amounts and characteristics of the different flows in the Wastewater Management sub-system, we used two main sources of information (Table S10)

(i) the annual reports of SBGE for the years 2011 to 2017, which provide information on the annual amounts of wastewater treated and the amounts of COD, TSS, P and N eliminated, and

(ii) the dataset from Bruxelles-Environnement, that provides information on the volumes of water to and from the two WWTPs, characteristics of the effluent (concentrations of COD, TSS, P and N) and removal efficiencies for the same parameters (IBGE, 2018a).

Table S10 Annual amounts of treated wastewater, eliminated TSS, COD and P and removal efficiencies of TSS and P for the two WWTPs in BCR. Values in italics are the ones used for the analysis of the current situation (2014).

YEAR	$Q_{ww,tr}$(1)	TSS_{elim}(1)	COD_{elim}(1)	P_{elim}(1)	$C_{TSS,out}$(2)	$C_{P,out}$(2)
	[m^3/a]	[tn/a]	[tn/a]	[tn/a]	[mg/L]	[mg/L]
North WWTP						
2011	106,814,782	26,103	42,410	423	96.3%	85.8%
2012	115,499,490	26,065	42,976	444	95.6%	82.3%
2013	104,541,982	24,693	42,970	452	94.9%	83.3%
2014	103,732,869	26,503	43,305	447	*96.2%*	*84.8%*
2015	105,002,898	24,752	43,902	443		
2016	112,000,981	25,038	43,254	441		
2017	105,493,570	25,100	44,500	448		
AVG 2013-2015	*104,425,916*	*25,316*	*43,392*	*447*		
South WWTP						
2011	23,697,625	8,953	13,527	121	76.0%	50%
2012	26,631,559	7,867	10,773	111	74.0%	58%
2013	25,720,730	10,756	14,631	160	78.5%	67%
2014	22,229,733	5,697	9,585	105	*73.0%*	*48%*

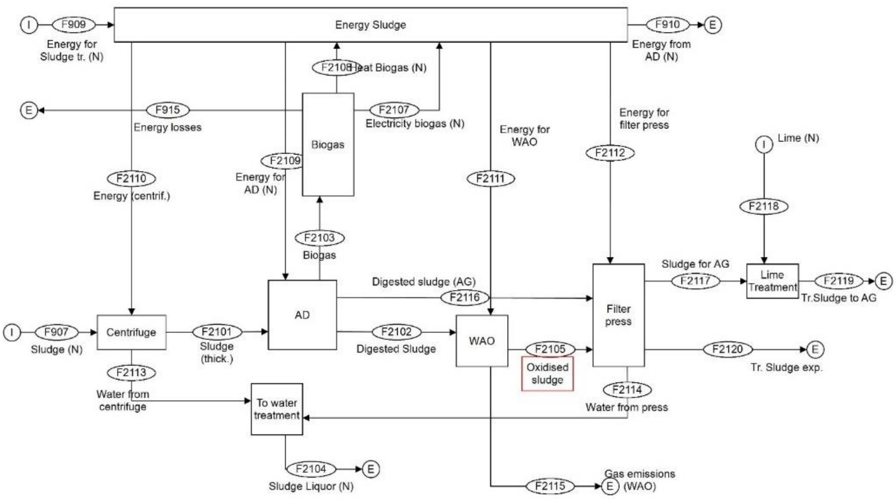
2015	23,811,633	5,134	8,641	110
2016	25,009,942	6,327	7,327	102
2017	22,281,758	4,445	8,570	123
AVG 2013-2015	*23,920,699*	*7,196*	*10,952*	*125*

(1) *Values taken from the annual reports of SBGE 2011-2017, last accessed 18.08.2018* [*http://www.sbge.be/fr/infos_documents_fr.html*](http://www.sbge.be/fr/infos_documents_fr.html)

(2) *Values taken from Bruxelles-Environnement, accessed 04.09.2017* [*https://environnement.brussels/etat-de-lenvironnement/rapport-2011-2014/eau-et-environnement-aquatique/epuration-des-eaux-usees*](https://environnement.brussels/etat-de-lenvironnement/rapport-2011-2014/eau-et-environnement-aquatique/epuration-des-eaux-usees)

- *Sludge amounts, characteristics and treatment*

Sludge treatment in the North WWTP



Sludge treatment was modeled based on information from the webpage of Aquiris (AQUIRIS, n.d.), information from (Chauzy et al., 2010) and literature values. We first established the solids balance in three steps:

(i) The amount of solids entering the sludge treatment (F907) equals the amount of TSS eliminated during water treatment

(ii) We used a 100% solids retention in the drying and thickening processes, and solids elimination efficiencies of

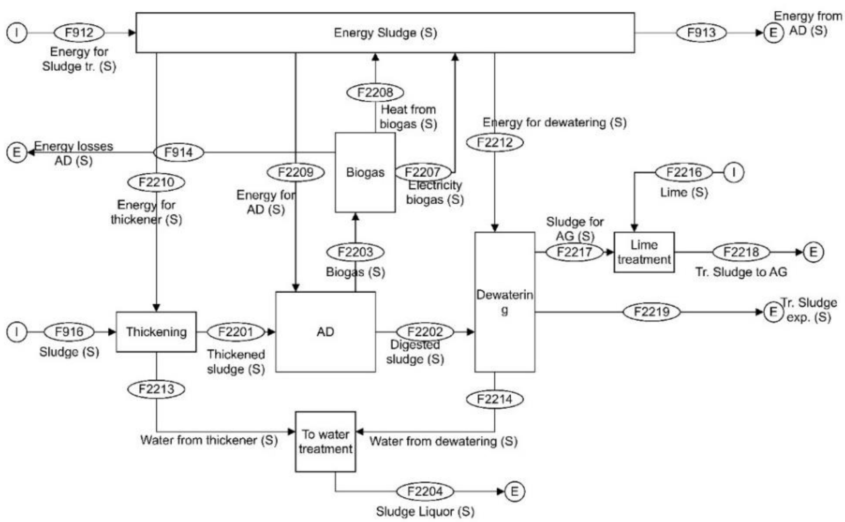
45% and 48% in the Anaerobic Digestion (AD) and the Wet Advanced Oxidation (WAO) processes respectively (Chauzy et al., 2010).

(iii) We cross-checked the results from step (ii) with the figures from Aquiris on the amounts of solids exiting the North WWTP through mineralized sludge.

Solids content of the sludge after the different treatment steps was also taken from Chauzy (2010). The sludge is ultimately exported from the region (F36), either as mineralized sludge ("technosand") towards landfilling, or as digested sludge towards an incineration plant in Flanders (Brion, 2019).

We assume that all P is bound to the solids, and that it is conserved through all treatment steps. This leads to an overestimation of the amount of P exiting the plant with the technosand, since some of it is in organic form and will thus be oxidized in the WAO and transformed into gas emissions. Still, since both gas emissions and technosands are pure exports of the system, we find that this assumption does not distort the system-wide balance.

Sludge treatment in the South WWTP



The primary and secondary sludge from the South WWTP is thickened and dehydrated, and ultimately sent to Germany for safe incineration. Due to lack of more refined data, we assume the same efficiencies and solids contents as for N. WWTP. The plant is currently undergoing a refurbishment of both the water and sludge treatment trains. The new infrastructure in the South WWTP will include (SBGE, 2017):

- New primary sedimentation units (commissioned in 2016)
- New secondary treatment: AS with anoxic zone, followed by membrane filtration (expected to be finished in Feb.2019).
- Digesters for the treatment of the sewage sludge (expected by 2020).

To model the situation in 2030, we assume that the removal efficiencies of nutrients and solids in the new South WWTP will be at the same level of those of the North WWTP, although this is a conservative assumption (N.WWTP has no membrane filtration unit after the secondary treatment). In addition, we assume that the sludge treatment will include thickener (centrifuge) + AD + dewatering (filter press): the same set-up as in the N.WWTP minus the WAO unit. Thus the diagram above represents the future situation at the South WWTP. For 2014, the sludge goes directly from thickening to dewatering.

- *Energy content of wastewater and sludge*

To calculate the energy content of the wastewater and sludge, we use the values reported in the study by Shizas and colleagues (2004) as shown in Table S11. Although already rather old, this study is one of the few that report combustion energy values for wastewater and sludge as a function of their solids content. We thus use an energy content of **3.2 MJ/kgTS** (reported also on (Cano et al., 2015)) for wastewater and **12.4 MJ/kgDM** for all sludges.

The values by Shizas for the GE content of sludge agree with newer studies that provide such data: (Cano et al., 2015) report 17.50 MJ/kgTS and (Syed-hassan et al., 2017) 16.05 MJ/kg DS. The database Phyllis2 includes several samples of sewage sludge, with most ranging around 10-12 MJ/kg DS and several up to 16-18 MJ/kg DS.

Table S11 Energy of combustion and characteristics of wastewater and sewage sludge [Source: (Shizas and Bagley, 2004)]

Parameter	Raw Wastewater	Primary sludge	Secondary sludge	Anaerobically digested sludge
Energy of combustion at constant volume - ΔUc,s [kJ/g dry]	3.2 ± 0.1	15.9 ± 0.2	12.4 ± 0.1	12.7 ± 0.3
Relative standard deviation [%]	3.2	1.1	1.0	2.0
TS [mg/L]	1’980 ± 50	30’500 ± 475	3’160 ± 80	39’100 ± 1’700
VS [mg/L]	242 ± 15	20’500 ± 700	1’900 ± 70	19’900 ± 950
VS:TS ratio	0.12	0.67	0.60	0.51
COD [mg/L]	431 ± 8	43’600 ± 1’260	3’250 ± 100	42’500 ± 1’500
TOC [mg/L]	73.2 ± 1.5	6’480 ± 60	850 ± 10	5’480 ± 30
TKN [mg/L]	11.3 ± 0.2	1’140 ± 90	181 ± 7	1’460 ± 60

- *Process Energy*

Energy consumption of the different treatment steps in the WWTPs

The North WWTP produces 20% of its own electricity demand: 10% from the biogas generated in sludge treatment and 10% from the hydraulic energy harvested through a turbine at the outlet of the sedimentation basin (IBGE, 2017b, p. 70). According to Aquiris (n.d.) the North WWTP in Brussels consumed 182 MWh daily (0.64 kWh/m^3 treated water), while producing 32 MWh/d, partly from the turbines that recuperate the energy of the water at the treatment phase and partially from the utilization of biogas produced during the AD of sludge. The latter contributes with 25 MWh/d of electricity and another 25 MWh of heat to the energy needs of the plant. No values on the energy demand of the South WWTP are reported, so we assume the same energy consumption for the South plant, too. The value (0.64 kWh/m^3) agrees with respective ones from literature: the study by (Cano et al., 2015) reports a range of 0.30-0.51 kWh/m^3 for WWTPs in Europe and 0.78 kWh/m^3 for the US; the review by (2015) mentions an average energy consumption of conventional WWT processes of 0.5 kWh/m^3.

We estimate the total energy needed for sludge treatment using values on the specific energy demand for sludge centrifugation ($1.8 \cdot 10^{-2}$ kWh/m^3 treated water), anaerobic digestion ($3.2 \cdot 10^{-2}$ kWh/m^3 treated water)and dewatering in a filter press ($1.0 \cdot 10^{-2}$ kWh/m^3 treated water) by (Longo et al., 2016). Data on the energy requirements for the wet oxidation process is very scarce on the literature. The only value found was from a LCA study from 2005 and corresponds to 797 kWh/tn DM disposed (Houillon and Jolliet, 2005). The same value was also used in the most recent LCA study on advanced sludge treatment (Tarpani and Azapagic, 2018) to this point. We use the same value, after converting it to kWh/m^3 of treated water, to agree with the other energy values. For the conversion, we use the information from 2014: 5'475 t of technosands were disposed and app. 104'500'000 m^3 of wastewater were treated at the North WWTP.

Energy generation from anaerobic digestion of sludge:
We estimate the amount of energy generated during the AD process using (Eq.E) from (Cano et al., 2015). We refer the reader to the main text for the derivation of the equation and a discussion on the implicit assumptions it entails.

$$E = 3.77 \cdot \eta_{AD} \cdot c \;\; [kWh/m^3 \; sludge] \quad \text{(Eq.E)}$$

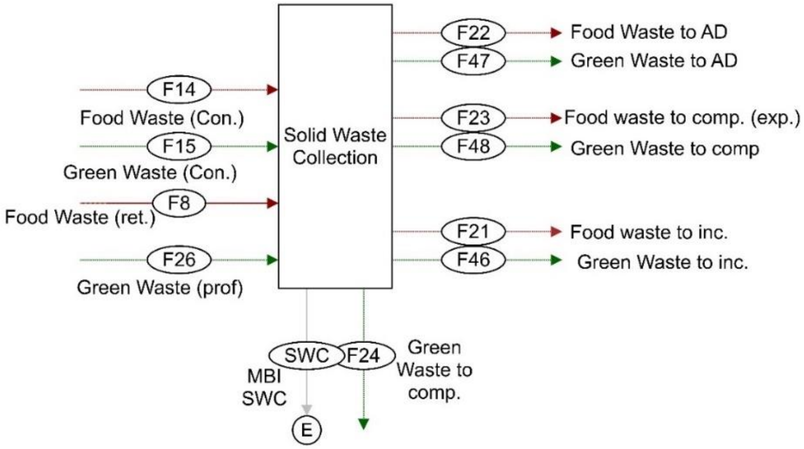
c = sludge concentration in $kgTS/m^3$

η_{AD} = 0.45 for fresh sludge and 0.63 for pretreated sludge

According to the same study, 15% of this energy will be lost, while the rest can be transformed into electricity and heat at rates of 55% and 30% respectively. We use these values to estimate the process energy flows F2107, F2108, F915, F2207, F2208, and F914.

2.7.6 Municipal Solid Waste Management

- *Solid waste collection*

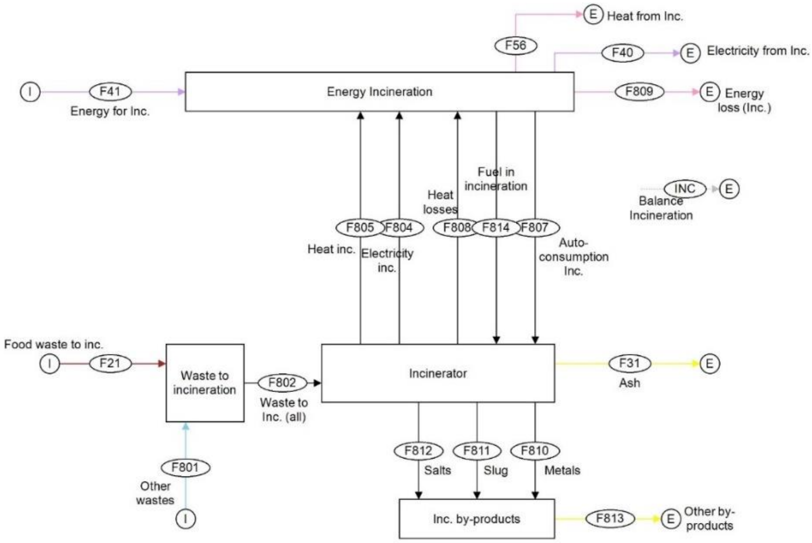


We allocated the streams of food and green waste to the different treatment options (Table S12) using information from (Vanessa Zeller et al., 2019).

Table S12 Shares of food and green waste to the different treatment options for 2014 [Source: (Vanessa Zeller et al., 2019), except for [(1)](De Muynck et al., 2018)]

Input flow	Treatment destination	Model variable	Share [%]
Household food waste (FW,hh): part of F14	Incineration	**a_fw.hh.inc**	99.6%
	Anaerobic Digestion (outside BCR)	**a_fw.hh.AD**	0%
	Composting (outside BCR)	**a_fw.hh.comp**	0%
	Decentralised composting (F16)[(1)]	**a_fw.hh.fcomp**	0.4%
Professional food waste (FW,prof): part of F14 + F8	Incineration	**a_fw.pr.inc**	89%
	Anaerobic Digestion (outside BCR)	**a_fw.pr.AD**	8%
	Composting (outside BCR)	**a_fw.pr.comp**	3%
Household green waste (GW,hh) : F15	Incineration	**a_gw.hh.inc**	44%
	Composting (BCR)	**a_gw.hh.comb**	56%
	Anaerobic Digestion	**a_gw.hh.AD**	0%
Professional green waste (GW,prof) : F26	Incineration	**a_gw.pr.inc**	30%
	Composting (BCR)	**a_gw.pr.comb**	20%
	Composting (outside BCR)	**a_gw.pr.comp**	50%
	Anaerobic Digestion	**a_gw.pr.AD**	0%

- *Incineration*



Material and phosphorus flows

Until this day, almost all household organic wastes and almost 90% of professional organic wastes are collected in the white bags (residual waste) and incinerated with the mixed waste (Vanessa Zeller et al., 2019). The incinerator has a capacity of 535 kt/yr (IBGE, 2002) and in 2014 it received app. 432 kt of waste (Bruxelles-Propreté, 2014). When modelling the incineration process, we account only for the organic inputs (food and green waste); F801 ('Other wastes') represents the rest of the incoming waste, thus F801 is set to zero across all scenarios. This presumes a strictly linear relationship between the amount of waste incinerated and the net energy recovered from the incinerator, so that the diversion of organic waste (in the case of the FWV scenario, for example) from the incinerator will not affect its functioning substantially. In reality, diverting such big amounts of humidity from the incinerator will lead to overheating and probably a loss of capacity.

All by-products of the incineration process are exported and valorised outside Belgium (IBGE, 2018d):

- Bottom ash (66'934 t in 2014) are used for road construction in the Netherlands

- Fly ash (7’350 t in 2014) are used as filling material in salt mines in Germany.
- Metal scraps (6’516 t in 2014) are recovered, treated and used in the steel industry.
- Salts (2’427 t in 2014) recycled by a soda producer (“producteur de soude”) in France.

To estimate the by-product generation also for the different scenarios, we calculated ratios for the production of each by-product in $kg_{by\text{-}product}/kg_{incoming\ waste}$ based on information provided in the annual reports of Bruxelles-Propreté (Table S13). We convert material to phosphorus flows using a phosphorus content of 1.48% P_2O_5 per weight for incineration ashes from biowaste (Haupt et al., 2018) and assuming a zero phosphorus content for the rest of the by-products. Using these assumptions, our model gives >95% diversion of the incoming P to the ashes; the rest are accounted for in the MBI for the incineration.

Table S13 Annual quantities of input waste into and by-products out of the incinerator and calculated shares of the by-products as percentages of the total inputs. The values used in the model are the ones for 2014, in italics. [Source: (Bruxelles-Propreté, 2017, 2016, 2015, 2014)]

Absolute amounts [t/yr]	**2012**	**2013**	**2014**	**2015**	**2016**
Total inputs to incinerator	469,806	449,514	431,742	474,076	499,501
Bottom ash	79,627	73,763	66,934	77,426	83,322
Fly ash	7,482	7,210	7,350	7,507	8,354
Metal scraps	6,733	6,508	6,516	7,122	6,840
Residue	501	604	540	657	697
Salts	2,984	2,784	2,427	2,740	3,061
As a share of inputs [%]					
Bottom ash	16.9%	16.4%	*15.5%*	16.3%	16.7%
Fly ash	1.6%	1.6%	*1.7%*	1.6%	1.7%
Metal scraps	1.4%	1.4%	*1.5%*	1.5%	1.4%
Residue	0.1%	0.1%	0.1%	0.1%	0.1%
Salts	0.6%	0.6%	*0.6%*	0.6%	0.6%

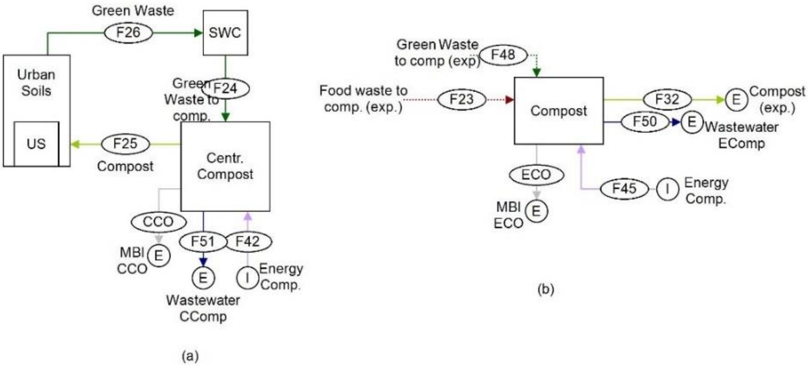
Energy flows and process energy

We assume a negligible energy content of the incineration by-products, so that all incoming chemical energy of the waste is transformed into process energy in the incinerator. For the process energy, we use data from Bruxelles-Propreté and from the regional energy balance to calculate the process energy inputs into the process (Table S14): electricity auto-consumption (F807) and the consumption of natural gas (F814). We use the average values for the period 2012-2016, since the values vary from year to year. The energy released during incineration is transformed into electricity in a steam turbine, with an efficiency of 15% (Haupt et al., 2018). For the future scenarios, we also include heat recovery with an efficiency of 34% (Haupt et al., 2018). A heat recovery system was installed in 2016.

Table S14 Energy consumption, as natural gas and electricity auto-consumption, and electricity generation from the incinerator. [Source: (Bruxelles-Propreté, 2017, 2016, 2015, 2014) except for [(1)](IBGE, 2019)]

Absolute amounts [GWh/yr]	2012	2013	2014	2015	2016	Average 2012-6
Total inputs to incinerator [t/yr]	469,806	449,514	431,742	474,076	499,501	
Total electricity generated	262	251	245	268	279	
Electricity auto-consumption	37	35	32	36	40	
Electricity to network	226	216	213	231	240	
Natural gas consumption[(1)]	58	64	63	64	54	
Values per t of incoming waste [kWh/t_{waste}]						
Electricity auto-consumption	78.8	77.9	74.1	75.9	79.3	*77.2*
Natural gas consumption	124.1	141.4	146.8	135.1	108.8	*131.2*

- *Centralized Composting in BCR*



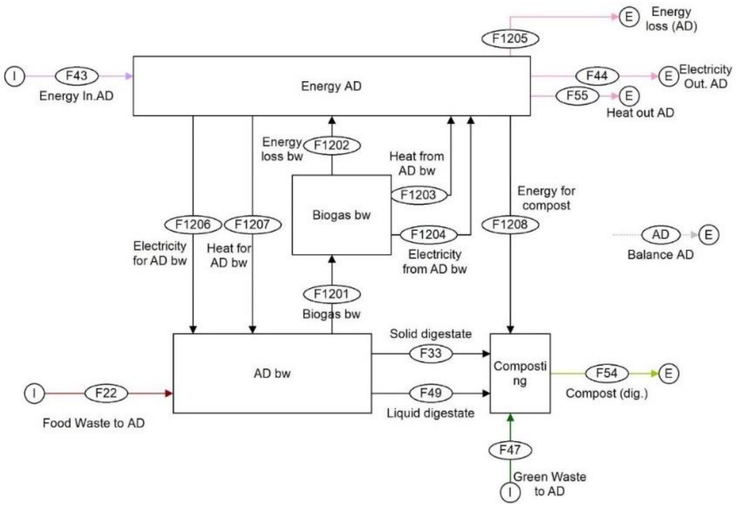
Separately collected garden waste in BCR are composted within the region. The responsible organization is Bruxelles-Compost, a subsidiary of Bruxelles-Propreté. Bruxelles-Compost produces annually between 9’000 and10’000 tonnes of compost from double the amount of garden waste (Bruxelles-Compost, n.d.). In 2016, Bruxelles-Compost treated 17’839 tn of green waste (IBGE, 2018d). The composition of the compost is not reported, so we use the characteristics of the garden waste compost from Vlaco, the umbrella-organization responsible for compost in Flanders (Table S15). We further assume: (i) a negligible P and energy content of any leachate from the composting piles (F50, F51); (ii) an energy demand of 11.8 kWh/$t_{biowaste}$ (Haupt et al., 2018) and (iii) an energy content of 15.75 MJ/kgDM (LHV) (Phyllis2, n.d.). We use the same values and assumptions to model the composting outside of the region.

Table S 15 Average composition of Vlaco compost [Source: (Vlaco, n.d.)]

Parameter	**VGF Compost**	**Green Compost**	**“Operating” co-efficient**
Dry matter [kg/tn]	700	600	
Organic matter [kg/tn]	250	200	
Total nitrogen [kgN/tn]	12	7	10-15%
Total phosphorus [kgP_2O_5/tn]	6	2.8	50%
Total phosphorus [kgP/tn][(1)]	2.62	1.22	
Total potassium [kgK_2O/tn]	10	6	80%
C/N ratio	12	17	

(1) converted from value for P_2O_5

- *Anaerobic Digestion*



No anaerobic digestion plant exists currently within the administrative boundary of BCR. Household organic waste collected separately after 2017 is sent to Flanders (AD plant in Ypres), along with a small fraction of professional organic waste (Vanessa Zeller et al., 2019). The AD of organic waste is therefore outside our system boundary for the current situation; however, for the scenarios we assume food waste valorization to be inside within the system.

In the study by Haupt and colleagues (2018) digestate is modelled as two streams, one liquid (0.30 $kg/kg_{org.waste}$) and one solid fraction (0.32 $kg/kg_{org.waste}$), based on (Dinkel et al., 2012). For our modelling we assume only one fraction (raw digestate) both for simplicity and because most the digestate was anyway sold in its raw form, at least until 2010 (Vlaco, 2012, p. 4). For the raw digestate we assume a yield of 0.9 $kg_{digestate}/kg_{waste}$ (Vlaco, 2012), a dry matter and phosphorus content of 8.4% and 0.13% of fresh weight respectively (Vlaco, 2016b) and an energy content of 8.55 MJ/kgDM (Phyllis2, n.d.).

We modelled the energy flows using information from (Haupt et al., 2018): the specific electricity and heat demand for the AD process are 2.14 kWh_e and 242 MJ per tonne biowaste, the biogas yield is $0.1m^3/kg_{biowaste}$ and the energy content of the biogas 21 MJ/Nm^3. The allocation of the energy in biogas to losses, electricity and heat is done as for the biogas from the sludge treatment, with the coefficients from

(Cano et al., 2015). Finally, the composting of the digestate together with green waste is modelled as described in the previous section.

2.8 Annex III : Scenarios

We developed and assessed 4 scenarios, based on possible strategies towards more circular food systems that are applicable to Brussels. These strategies are Food Waste Valorization (FWV), Nutrient Recovery and Recycling, which we represent with our Sewage Sludge Utilization (SSU) scenario, and a combination of the two (CO). The scenarios refer to year 2030, because this is the time base for some of the city's targets, e.g. the food waste reduction one. In addition, it is a time in the near future, for which we can extrapolate data from the current situation with enough confidence.

2.8.1 Baseline

For a fair comparison, we compare the 4 scenarios with the baseline for 2030, rather than with the current situation for 2014. The baseline scenario is a projection of the current situation into the future, accounting for population and visitors growth. In addition, we apply a 20% reduction in the avoidable food waste (household and professional) to all scenarios, as this is the region's target for 2030 (IBGE, 2018b). The per capita green waste and wastewater generation are the same as for 2014. The MSWM system in the city remains the same as for today, while for wastewater treatment we account for the modernization of the South WWTP. Further assumptions regarding wastewater management used for the modeling of scenarios include:

- The same absolute amounts of wastewater from Flanders will be treated in the two WWTPs in the year 2030. Thus, we assume that the current expansion works at the South WWTP and any idle capacity in the North WWTP will be used explicitly to accommodate the population growth.
- Amount of industrial wastewater is the same as for the reference year 2014. Implicit assumption that any new industrial activity will install own plants or send to neighboring plants.

- The South WWTP will reach the performance of the North WWTP after its refurbishment.
- For the scenarios, we assume the same volume reduction in the final step (drying-filter press) whether the sludge is first oxidized or comes directly from the digester. This probably overestimates the volume reduction in the non-oxidized sludge, because its solid content is mainly organic rather than mineral.

2.8.2 Food Waste Valorization (FWV)

For the Food Waste Valorization scenario (FWV), all food waste is diverted from incineration and valorized within BCR either through anaerobic digestion and composting of the digestate (90%) or through composting (10%). Green waste currently incinerated is diverted to compost within the city. We assume a strictly linear relationship between the amount of waste incinerated and the net energy recovered from the incinerator, so that the diversion of organic waste from the incinerator will not affect its functioning substantially. In reality, diverting such big amounts of humidity from the incinerator will lead to its overheating and probably loss of capacity.

Table S 16 Shares of food and green waste to the different treatment options for the baseline (sources as in Table S5 of the Model Description) and assumptions for FWV scenario.

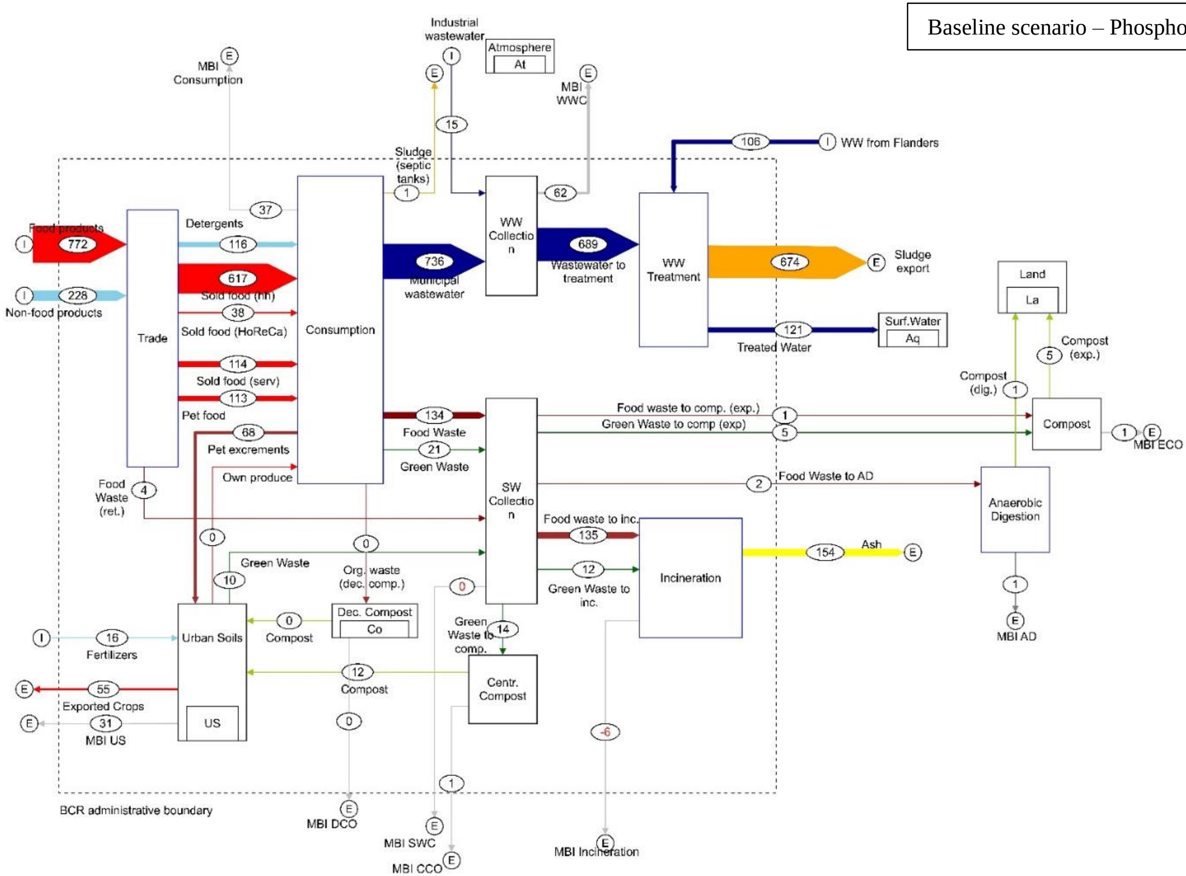
Input flow	Treatment	Base	FWV
Household food waste	Incineration	99.6%	0%
	Anaerobic digestion (outside BCR for baseline, in for FWV)	0%	90 %
	Composting (outside BCR)	0%	0%
	Decentralised Composting	0.4%	10%
Professional food waste	Incineration	89%	0%
	Anaerobic digestion (outside BCR for baseline, in for FWV)	8%	100%
	Composting (outside BCR)	3%	0%
Household green waste	Incineration	44%	0%
	Composting (BCR)	56%	56%
	Anaerobic digestion	0%	44%
Professional green waste	Incineration	30%	0%
	Composting (BCR)	20%	70%
	Composting (outside BCR)	50%	0%
	Anaerobic digestion	0%	30%

2.8.3 Sewage Sludge Utilization (SSU)

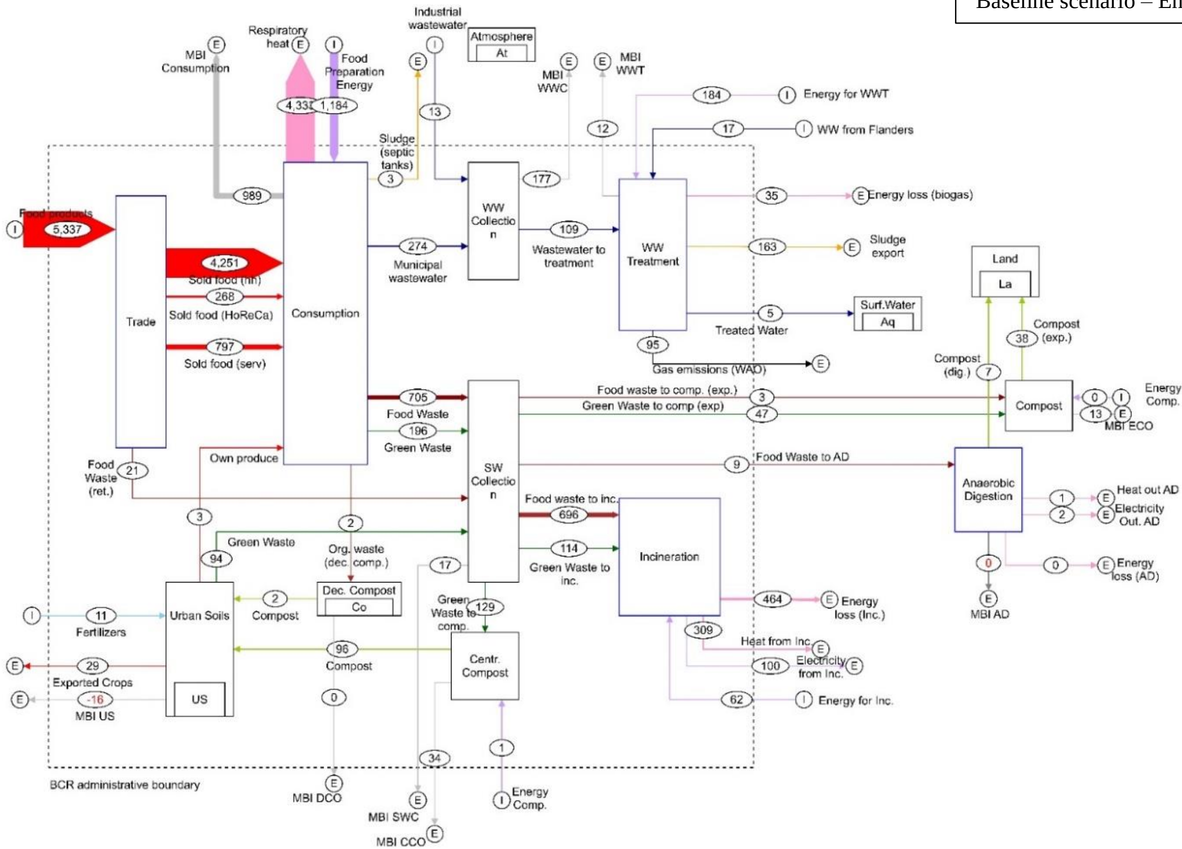
The last scenario, Sewage Sludge Utilization (SSU), represents the ideal case where all sewage sludge is treated in a way that makes it available for agricultural use. We assume this to be a two-step process: first the thickened sludge is anaerobically digested with energy recovery from biogas. Sewage sludge digestion already takes place in the N.WWTP and is part of the refurbishment of the S.WWTP, currently under way. In the SSU scenario, digested sludge is conditioned with lime and dried. We chose lime conditioning as the extra treatment step for the hygenization of the sludge because it is what happens to the urban sewage sludge aimed to agricultural lands in the Walloon cities of Liège and Namur (AIDE, 2018; INASEP, 2018). Lime is dosed at an average rate of 41 kgCaO/t sludge valorized (AIDE, 2018). Since we model based on dry matter, and the quantities of wet sludge are less certain, we use the value of 400 kgCaO/tDM for the dosage of lime (Houillon and Jolliet, 2005). We assume the energy demand for lime stabilization to be negligible and the energy for WAO to be independent of the volume of sludge treated since, according to (Chauzy et al., 2010), after start-up the process is self-sustaining. The agricultural production in Brussels is too small to absorb all sludge produced; most of it will have to be exported, generating additional fuel demand for its transport.

2.8.4 Sankey diagrams for all scenarios

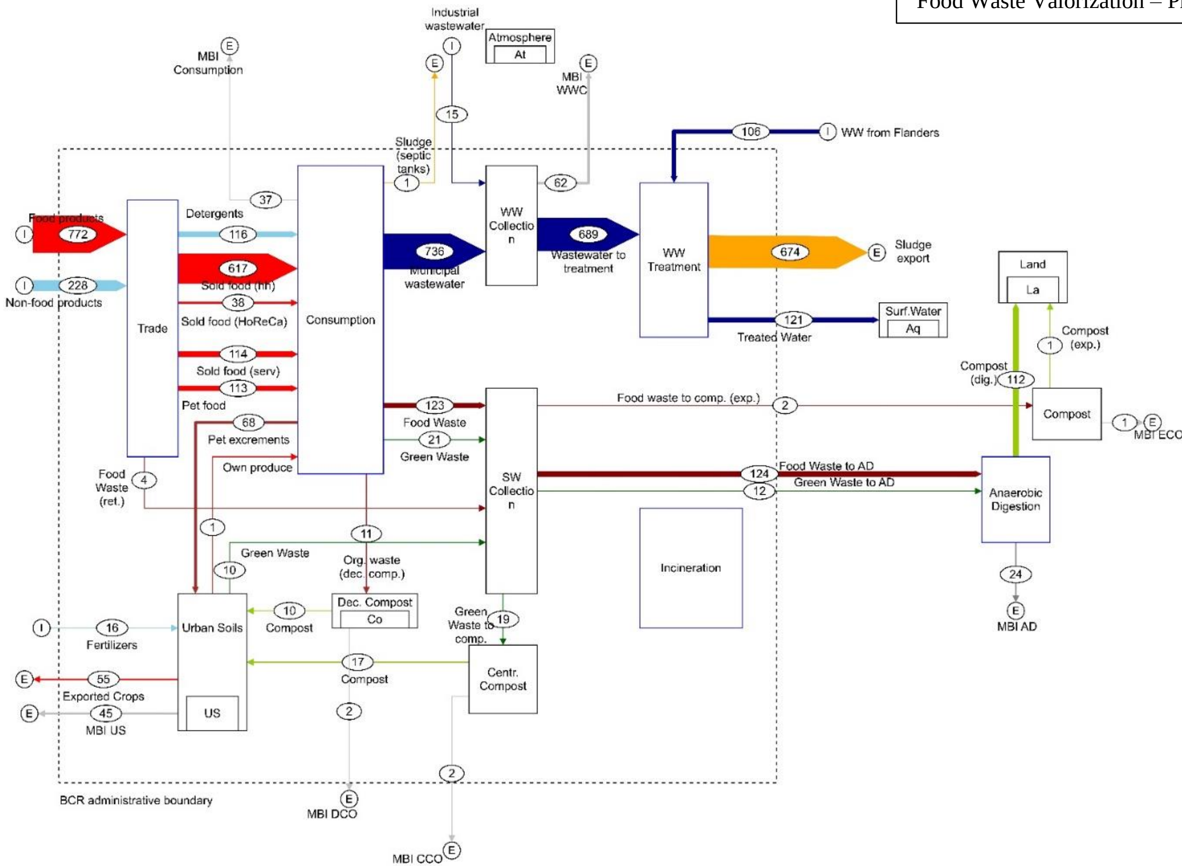
Baseline scenario – Phosphorus flows [tP/yr]



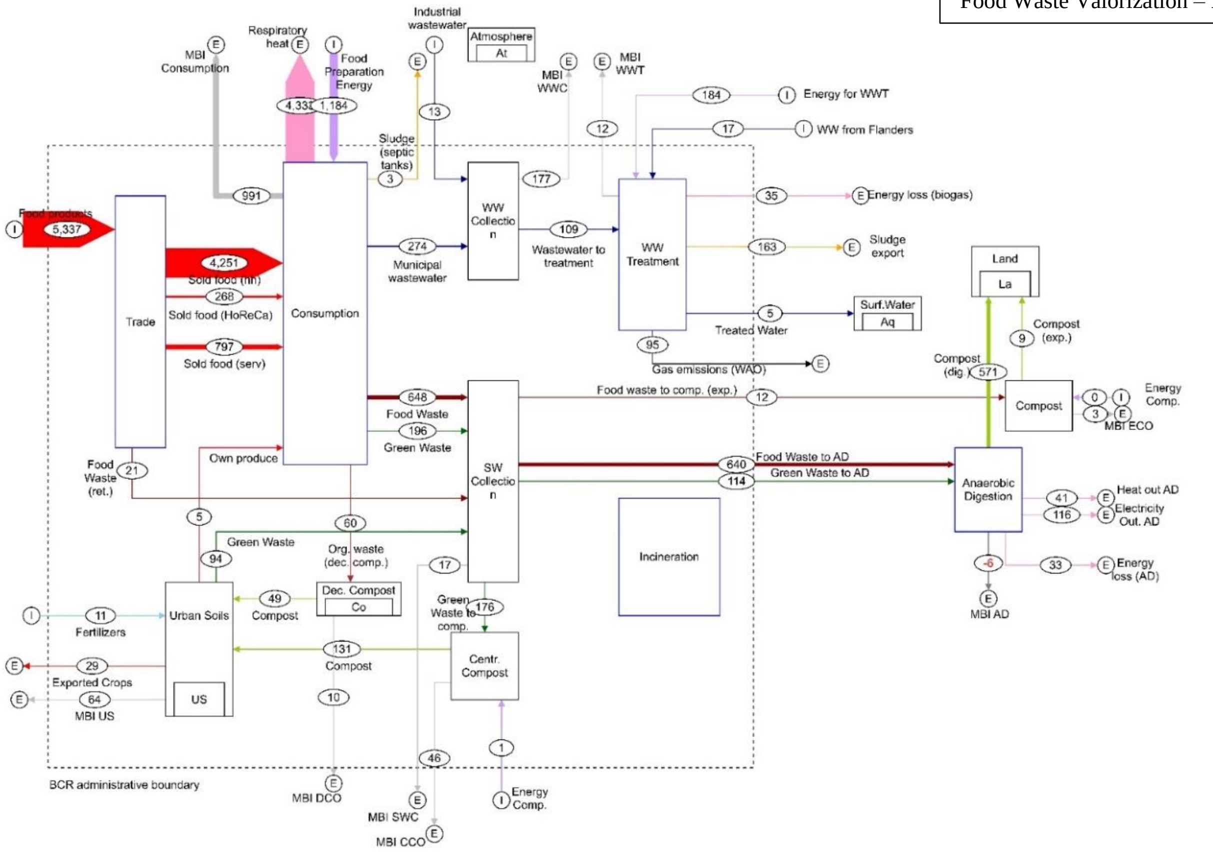
Baseline scenario – Energy flows [TJ/yr



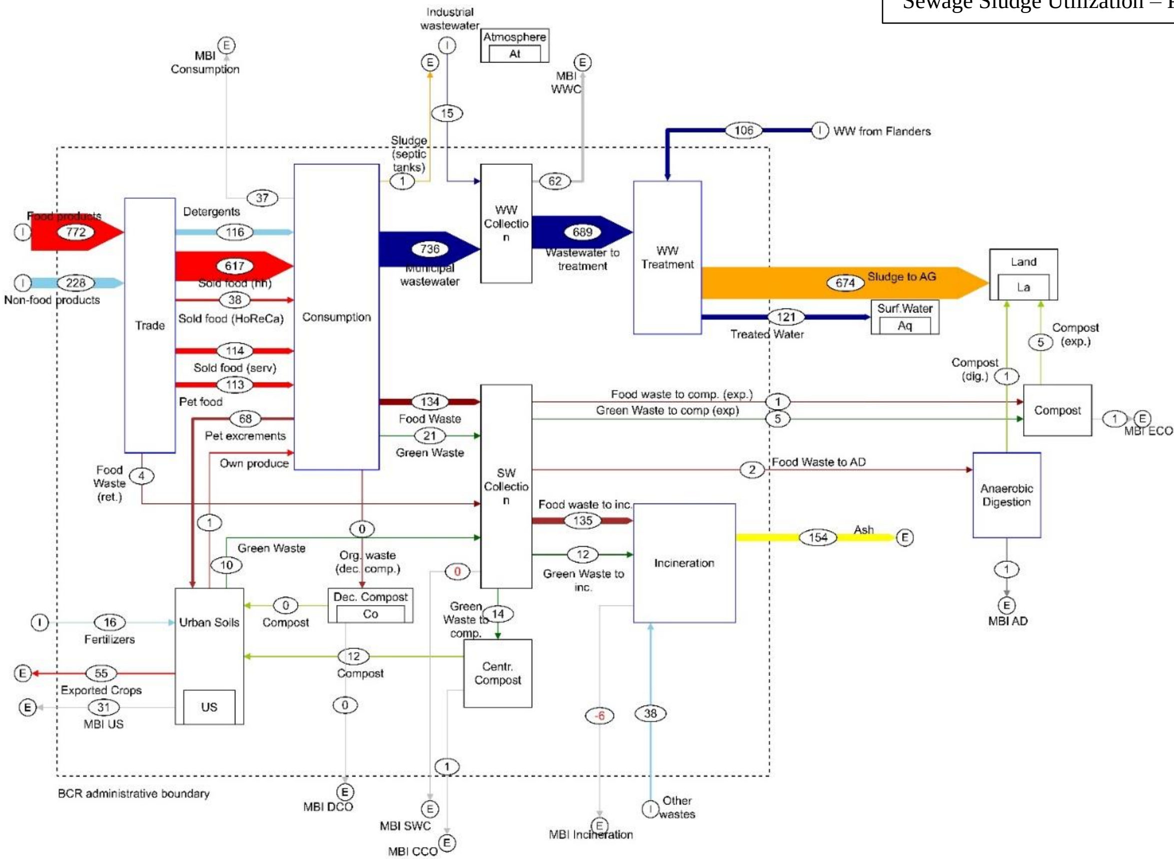
Food Waste Valorization – Phosphorus [tP/ya]



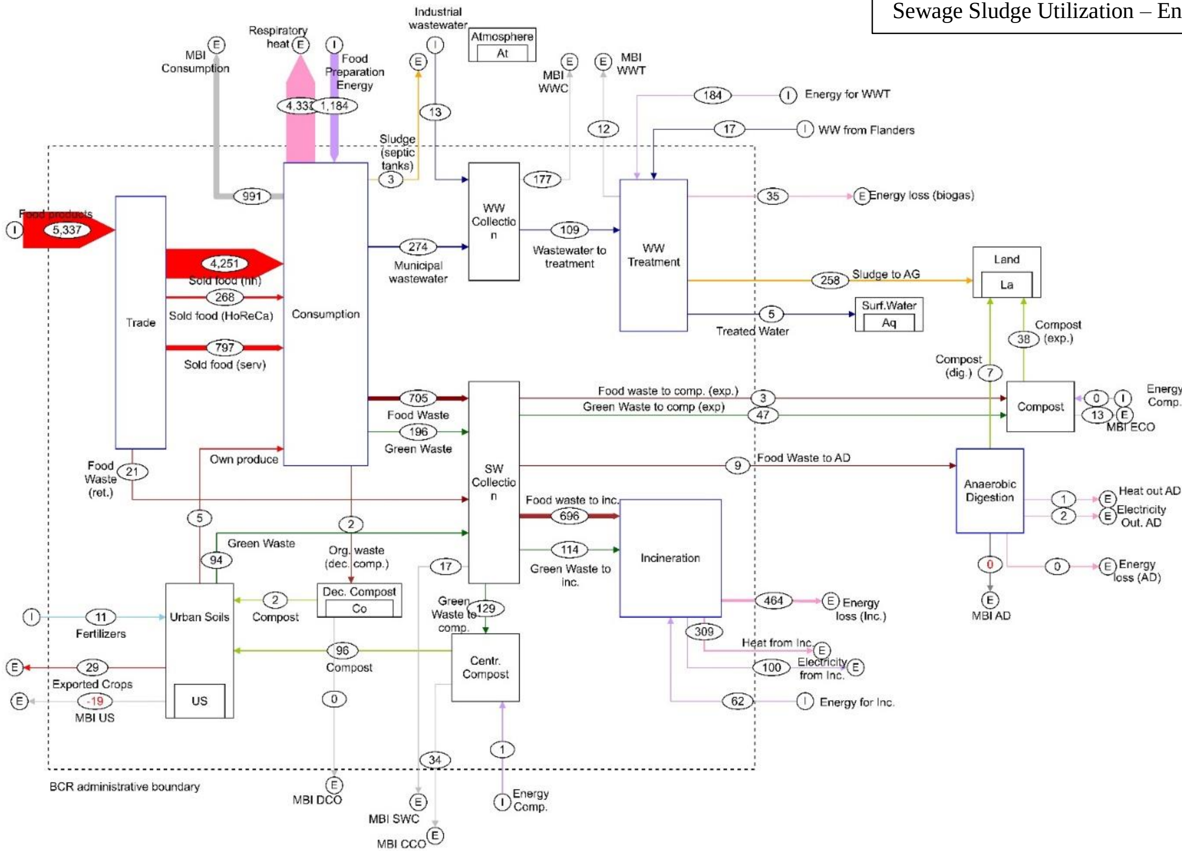
Food Waste Valorization – Energy [TJ/yr]



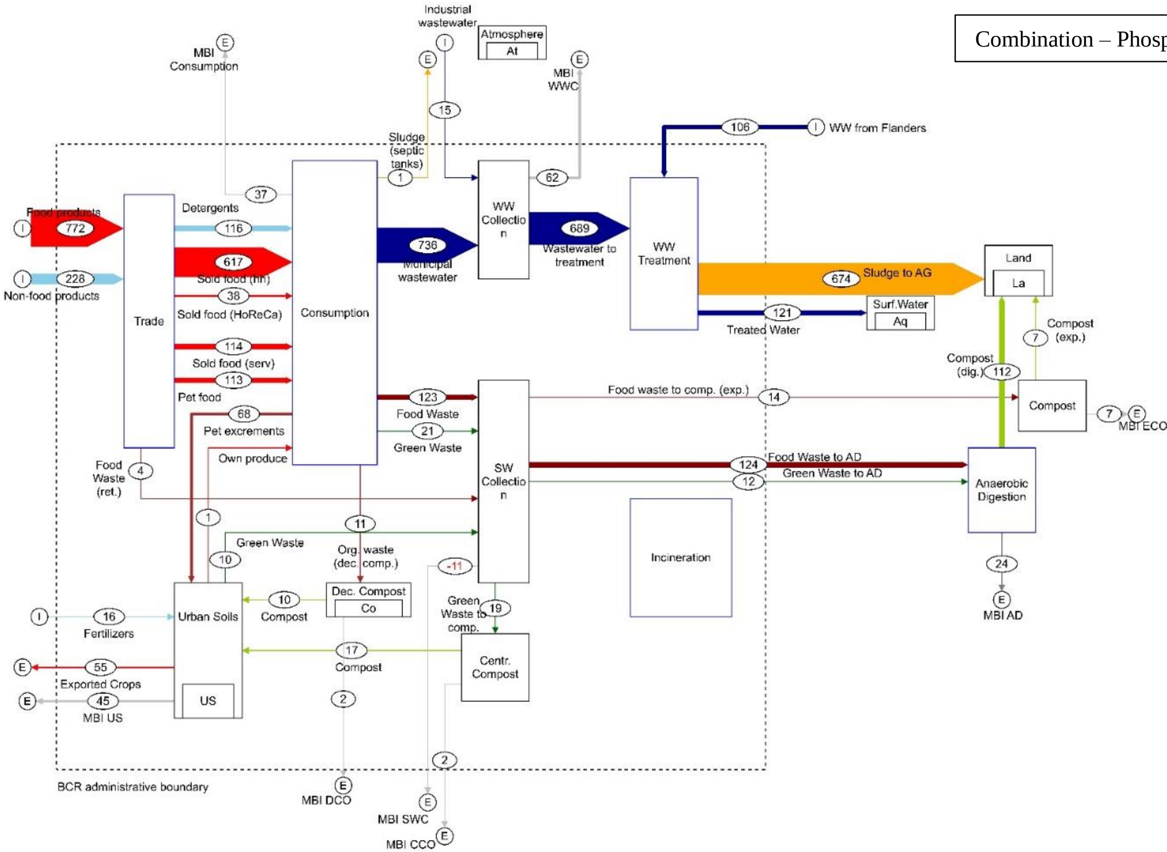
Sewage Sludge Utilization – Phosphorus [tP/ya]



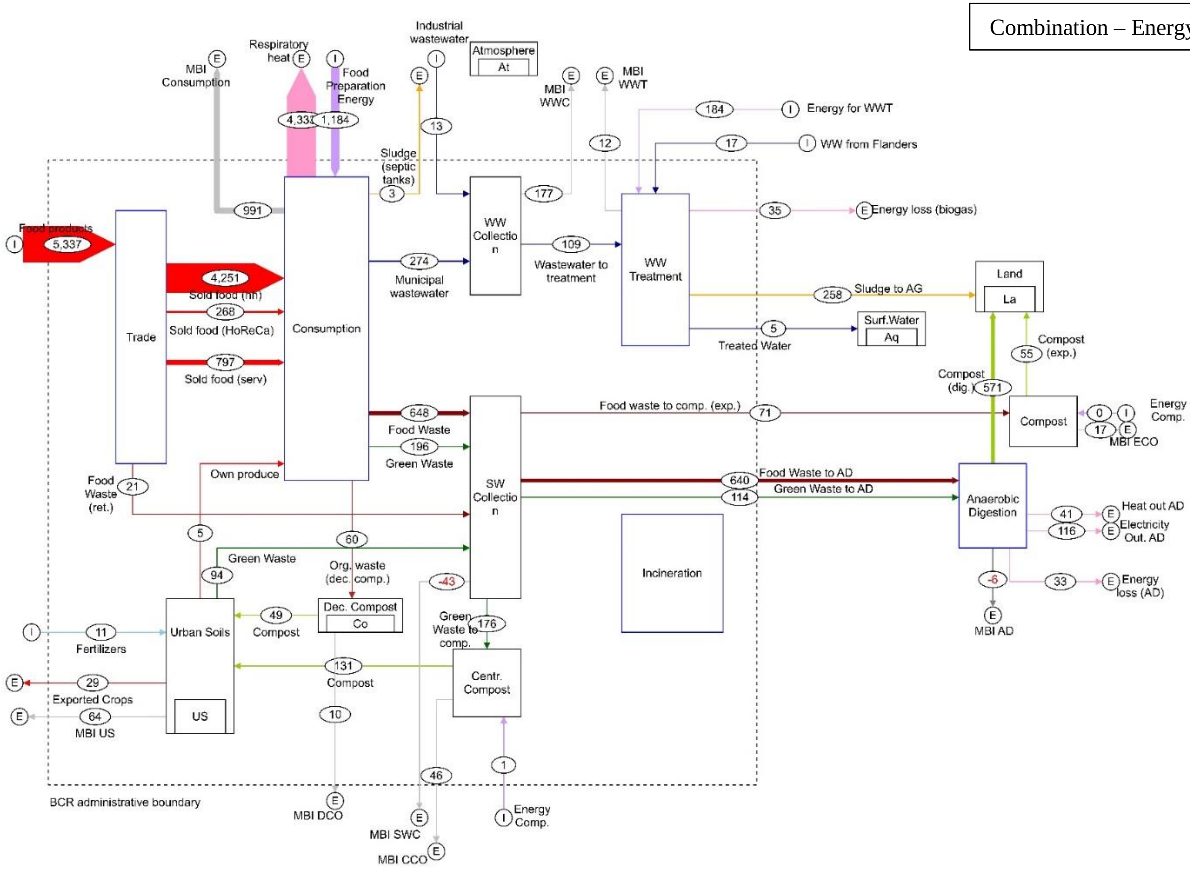
Sewage Sludge Utilization – Energy [TJ/yr]



Combination – Phosphorus [tP/ya]



Combination – Energy [TJ/yr]



Chapter 3

Assessing agro-food system circularity using nutrient flows and budgets[3]

3.1 Introduction

Nutrients, such as nitrogen (N), phosphorus (P) and potassium (K) are vital inputs to agriculture, necessary for food production and food security (Godfray et al., 2010). At the same time, intensive agriculture and excess fertilizer use have greatly disturbed the natural cycles of these nutrients, becoming one of the main reasons for the eutrophication of coastal and freshwater bodies (Elser and Bennett, 2011; Galloway et al., 2004). Fertilizer production puts additional burdens to the environment, because it is energy costly and requires raw materials, notably phosphate rock, that are non-renewable and potentially scarce (Cordell et al., 2009; Withers et al., 2019). Secondary nutrients, reclaimed from nutrient-rich streams such as manure and treated urban effluents, can help reduce the need for synthetic fertilizer and diverge these effluents from getting discharged into the environment (Mayer et al., 2016; Withers et al., 2014). Nutrient reuse and recycling is indeed one of the main pathways towards more circular food systems (EMF, 2019; Tseng et al., 2019). A growing body of research has been mapping and quantifying nutrient flows and budgets

[3] This chapter is based on: **Papangelou A** and Mathijs E (2021) Assessing agro-food system circularity using nutrient flows and budgets. *J. Environ. Manage.* 288, 112383. https://doi.org/10.1016/j.jenvman.2021.112383

in human-environmental systems, and assessing the reuse potential of these flows (Chowdhury et al., 2014; Fernandez-Mena et al., 2016; Withers et al., 2019; Zhang et al., 2020).

Most nutrient flows analyses are conducted for a single spatial scale, whether farm, regional, or national (Fernandez-Mena et al., 2016; van der Wiel et al., 2019). Each approach is suited for addressing different research questions (Chowdhury et al., 2014; van der Wiel et al., 2019). The country scale is often used to address aspects relevant to policy-making, such as establishing and monitoring targets for waste reuse or exploring the implementation of different treatment technologies. Studies at lower spatial resolutions than the country allow to explicitly address three concerns on nutrient management and circularity: (i) environmental concerns related, for example, to different soil and water conditions; (ii) agronomic concerns due to the different production systems that depend on geographical and economic contexts; and (iii) organizational concerns related to managing waste flows and transportation. For example, manure, sewage sludge and other organic waste streams are bulky materials that are difficult and uneconomical to transport over long distances. The need to import nutrients to agriculture depends on the type of crops, soil conditions and availability of alternatives such as manure (Akram et al., 2019; Parchomenko and Borsky, 2018).

Restoring nutrient and food circularity thus depends on geography (Julia Le Noë et al., 2018; van der Wiel et al., 2019). Senthilkumar et al. (2012) were among the first to compile 21 different soil phosphorus (P) budgets for each of the 21 regions in France. They noted how sharply the budgets differed from one region to another, because of the regions' different specializations in production systems, and different biophysical and economic realities. Klinglmair and his colleagues (2015) mapped P flows in Denmark at two geographical scales, regional and whole country, and showed that the P loads differ from region to region, according to the soil type and dominant product (crop or animals). More recently, researchers from Norway (Hanserud et al., 2016) and Sweden (Akram et al., 2019) focused their multi-regional flow analyses on the need for manure transport due to the geographical segregation of livestock-bearing regions, that generate nutrient surplus, and producing regions, that concentrate nutrient demand.

The most comprehensive multi-regional nutrient flow and budget analyses to this day have been the studies on the French agro-food system using the Generalised Representation of Agro-Food Systems approach (GRAFS) (Billen et al., 2014). In one of the most recent applications, Le Noë and colleagues (2018; 2017) established a typology that connects patterns of nutrient flows and budgets to the type of production system, especially the intensity of production and whether animal or crop production dominate the landscape(Le Noë et al., 2017) (Le Noë et al., 2017). They then used their results to calculate a series of environmental indicators, for example nutrient inputs, soil surpluses and nutrient use efficiency.

Absolute metrics such as total inputs or losses, or relative ones such as nutrient use efficiency (NutUE), are being routinely used to evaluate results of nutrient flow analyses (Chowdhury and Zhang, 2021; Le Noë et al., 2017; Tanzer et al., 2018) where the focus is often on improving resource use efficiency, minimizing losses and emissions, and exploring the reuse potential of secondary flows. Yet food system circularity goes beyond these metrics. Fostering food system circularity does not only require the reuse of secondary flows and the minimization of emissions by means of, for example, improvements in use efficiency; it ultimately aims at the absolute reduction in the demand for primary resources and as such it involves both a higher share of secondary inputs and an absolute reduction in total inputs. To this date, however, there is no broadly accepted framework or set of indicators that explicitly address food system circularity (Moraga et al., 2019; Navare et al., 2020; Pauliuk, 2018).

In this study we analyze the nutrient flows in Belgium using the GRAFS approach as the methodological basis, and use the results of this analysis to assess the circularity of the agro-food system. Belgium is a small country intensively cultivated and densely populated by people and animals. From an agronomic point of view, Belgium can be divided into three main regions (Figure 3.1): a region of intensive agriculture in the North, dominated by dairy cattle, pigs and horticulture; a region of extensive crop production in the loamy soils of central Belgium; and a hilly, grassland-dominated landscape in the South (Van Hecke et al., 2000). In addition, Belgium is an important food producer and exporter in the EU, with an open agricultural economy largely based on trade. On the one hand, our study enriches the literature with a new detailed case study of a nutrient flow analysis at the subnational level, where studies, especially outside France, are still scarce. On the other hand, it can provide insights into, and a base of comparison with regions with similar agro-food systems, such as the Netherlands, Denmark, N. Ireland, or the French region of Bretagne. Further, we propose and test a set of circularity-related indicators that go beyond merely estimating the use efficiency or the total inputs at the system level. Our research objectives are:

(i) to describe the Belgian agro-food system from a biogeochemical perspective at different sub-national spatial scales, in order to understand how geography and the type of production system affect the nutrient flows and budgets, and

(ii) to assess the system's circularity.

While doing so, we also

(iii) propose a set of indicators to assess agro-food system circularity, and discuss the application of these indicators at different spatial scales and system levels.

3.2 Methodology

3.2.1 Nutrient flows and budgets

We used the GRAFS Excel tool (Le Noë et al., 2017, SI) described in detail in (Le Noë, 2018, ch.2) as the basis of our nutrient flow analysis, after expanding it and adapting it where needed to better reflect the reality of the Belgian food system. In this section we summarize this process, highlighting the deviations from the original approach. Further details and all primary data used can be found in the supplementary material (SM).

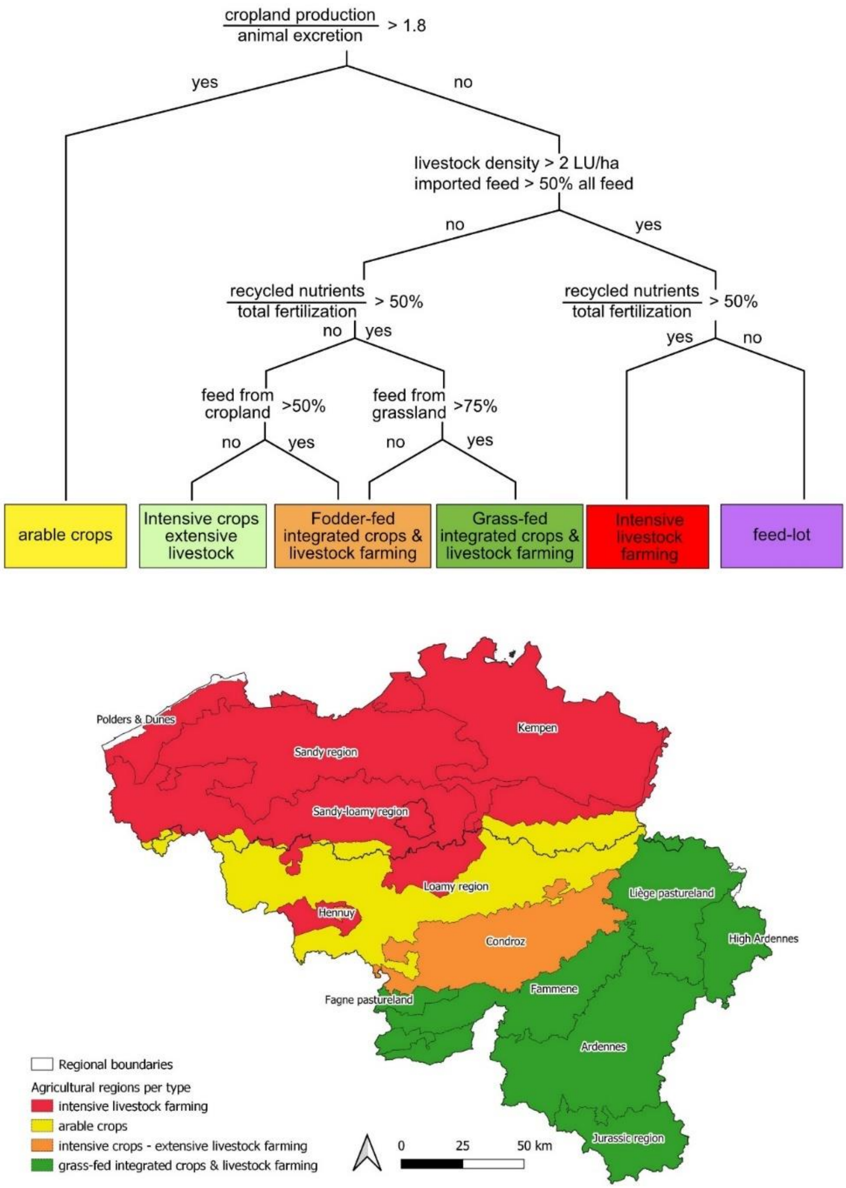


Figure 3.1 Typology of Belgian agricultural systems and characterization of the Belgian agricultural regions accordingly, based on nitrogen flows. The criteria and types of production systems are taken directly from (Le Noë, 2018) while most of the threshold values in the decision tree were adjusted to better reflect the reality of the Belgian agriculture, such as the high overall livestock density and the high share of recycled nutrients in the input.

- *System definition & processes*

We define the Belgian agro-food system from a biogeochemical perspective as the group of processes within the national boundary of Belgium, and the nutrient flows into and out of these processes, that are intended to provide food to the local population. The original GRAFS methodology includes four processes: cropland, grassland, the livestock sector, and consumption (population). To these processes we added the ‘proxy processes’ of food and feed industry because they are pivotal in the Belgian agro-food system. We also introduced waste management as a separate process, due to our focus on circularity and secondary nutrient resources. Our system ultimately consists of seven processes that were grouped into three sub-systems (Figure 3.2): (i) agriculture, including cropland and grassland, (ii) food production, including cropland, grassland, livestock, food industry and feed industry, and (iii) waste management, including the sole process of waste management. These sub-systems were used for the circularity assessment at different system levels.

- *Flows*

We studied the nutrient flows between these seven processes for nitrogen (N), phosphorus (P) and potassium (K), three of the main nutrients from an agronomic and resource reuse perspective. Because our focus is on the food system, we only accounted for nutrients embedded in food (e.g., harvested crops, food products, food waste) and by-products of food production and consumption (e.g., manure, human excreta, inedible animal products). We did not calculate flows such as phosphorus from cosmetic and households products separately, but since they are mixed with the food-related nutrients in household waste streams, we assumed that they are included in the mass balance around consumption (Figure 3.2, flow 4,0). Table 3.1 presents the full list of processes and flows and summarizes the approach or equation adopted and the data sources used for their calculation. Further details can be found in theAnnex.

- *Spatial and temporal scope*

We studied the Belgian agro-food system in two different scales. Firstly, we worked on the scale of the agricultural regions, a division of the Belgian territory into fourteen regions with similar landscape and soil characteristics (Figure 3.1). Because this scale is the most relevant from an agronomic perspective, we used it to analyse nutrient flows in cropland and grassland, and to estimate nutrient soil budgets. Secondly, we worked at the scale of regions and at the whole country scale. We focused on two of the three Belgian NUTS2 regions, Flanders and Wallonia, the third being Brussels Capital Region with virtual no agricultural activity. The regional level is the most relevant from a whole food system perspective, since each region independently operates their production systems and regulates issues of agronomic and environmental concern, e.g. waste reduction targets or allowable nutrient inputs to agriculture.

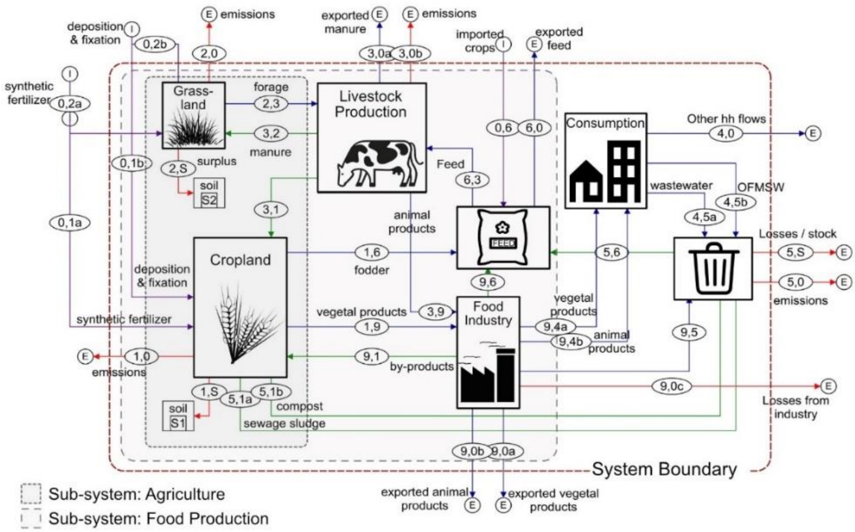


Figure 3.2 System Diagram including processes, flows, and boundaries for the whole system, as well as the sub-systems "Agriculture" and "Food production". (OFMSW: Organic Fraction of Municipal Solid Waste, hh: household)

The reference year for the analysis is 2014, the most recent year for which all data are available. We consider 2014 to sufficiently represent the present situation, as most key data have not been fluctuating significantly in recent years (Figure 3.3). To further eliminate the influence of the choice of a specific year to the final result, and wherever that was possible, we used 5-year averages around 2014 instead of single year values for a few variables that tend to fluctuate, e.g. fertilizer inputs. Choosing a snapshot of the present situation as the temporal scope enabled us to add detail to the model and to use reliable data. It also means, however, that we disregard dynamics and temporal aspects, such as the accumulation of legacy nutrients in the soil and the evolution of agronomic practices over long periods of time. A future expansion of this study with long time-series for key data could complement it with a richer understanding of the system, as it could reveal: (i) dynamic relationships between the system's components; (ii) factors that drive the nutrient flows; (iii) environmental implications of nutrient balances that unfold over longer periods of time; and (iv) annual fluctuations in the input variables that influence the end result.

- *Proxy processes and assumptions*

Food and feed industry are two additional processes to the original GRAFS model that we introduced as 'proxy' processes in our study, We did so to (i) better handle and represent net import and export flows in a system as open as the Belgian food system, and (ii) explicitly account for the reuse of organic waste in the feed industry, an important flow in the Flemish waste management sector. We call them 'proxy processes' because they are meant as a device to handle the imports needed to sustain the local population of humans and animals, or to export surplus food and feed, and so they only account for net imports and exports. In reality, and especially in Flanders, big quantities of food are imported, processed together with local crops, and re-exported, while the by-products and waste flows stay in the regions to be managed locally. Nonetheless, accounting for net imports and exports only is in line with the scope of our study, where the primary focus is on agricultural production and the reuse of secondary nutrients, and the main function of the system is to provide food to the local population.

To estimate the net imports and exports, and to allocate the crop production to the food and feed industry, we used mass balances and a series of assumptions. We first assumed that all vegetables, fruits and industrial crops are grown for human consumption, and thus directed to the food industry. Surpluses are exported. Cereals are grown primarily for human consumption too, but surpluses are directed to the feed industry. We estimated the nutrient intake of livestock by summing the nutrients in animal excretions and in animal products (Le

Noë, 2018). The intake was assumed to be covered stepwise by three sources: (i) grassland production; (ii) local forage production and cereal surpluses and (iii) imported forage and nutrient supplements.

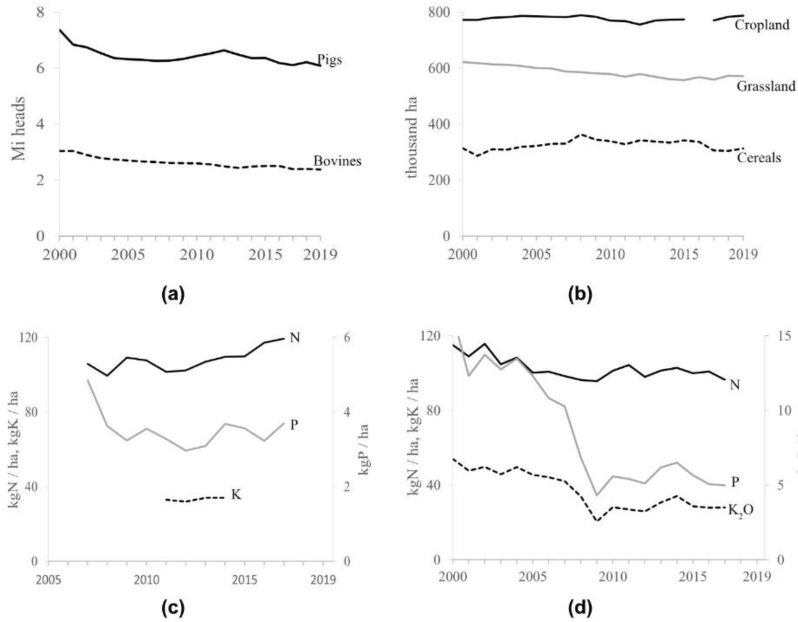


Figure 3.3 Time series for some key agricultural figures in Belgium: numbers of pigs and cattle (a), total cultivated area, permanent grassland area and agricultural land cultivated with cereals (b), amounts of mineral fertilizers applied in Flanders (c) and Wallonia (d). [Sources: Statbel, (Departement Landbouw en Visserij, 2020; REEW, 2019b)]

Table 3.1 List of processes and flows included in the model, along with equations and data sources used for the estimation of each flow (FL: Flanders, WA: Wallonia, DM: Dry Matter, AD: Anaerobic Digestion, OFMSW: Organic Fraction of Municipal Solid Waste, MSWM: Municipal Solid Waste Management, hh: household, ww: wastewater).

Process	Flow no.	Flow Description	Equation / estimation method	Data sources
Crop-land (1)	0,1a	synth. fertilizers to cropland	Crop area (FL) or cropland area (WA) x fertilization rates from official statistics & reports	[1], [2]
	0,1b	atm. deposition & N-fixation	Cropland area x deposition or N-fixation rates (N: official regional statistics, P&K : literature)	[3] – [6]
	1,9	vegetal products	Cultivated area x yield x nutrient content for: industrial crops + vegetables + cereals for human consumption	[7] – [15]
	1,6	fodder	Cultivated area x yield x nutrient content for: all fodder produced + cereals not for food	[7] – [15]
	1,S	surplus to soil	Balance: fertilizers (0,1a) + manure (3,1) + deposition/fixation (0,1b) + wastes (5,1) – production (1,9+1,6) – emissions (1,0)	
	1,0	N emissions	Cropland area x volatilization loss rate from fertilizers	[4], [9]
Grassland (2)	0,2a	synth. fertilizers to grassland	Grassland area x fertilization rates from official statistics & reports	[1], [2]
	0,2b	atm. deposition & N-fixation	Grassland area x deposition or N-fixation rates (N: official regional statistics, P&K : literature)	[3] – [6]
	2,3	forage	Grassland area x yield (8tDM/a) x nutrient content of grass	[7], [16], [17], [13]
	2,S	surplus to soil	Balance: fertilizers (0,2a) + manure (3,2) + deposition/ fixation (0,2b) – production (2,3) – emissions (2,0)	
	2,0	N emissions	Grassland area x volatilization loss rate from fertilizers	[4], [9]
Livestock (3)	3,1	manure to cropland	FL: Cropland area x manure application rates WA: Cropland area x no of animals x animal excretion rates – storage & handling losses – excretion directly on grassland	[18], [19], [22] [7], [10], [19] – [21]
	3,2	manure to grassland	FL: Grassland area x manure application rates + excretion directly on grassland WA: Grassland area x no of animals x animal excretion rates – storage & handling losses + excretion directly on grassland	[18], [19], [22], [23] [7], [10], [19] – [21]
	3,9	animal products	Animal production x nutrient content – inedible parts	[26], [10], [11], [12]
	3,0a	manure exported	No of animals x animal excretion rates – storage & handling losses – manure on cropland &grassland (3,1+3,2)	[7], [18] – [23]
	3,0b	N emissions	Emissions from housing &storage of manure	[19], [24], [25]
Feed industry (6)	6,3	feed	Total ingestion of livestock (edible & inedible animal products + excretion) – intake from grassland (2,3)	[26], [10] – [12]

	0,6	imported feed crops	Feed balance: Total ingestion – fodder (1,6) – forage (2,3) – by-products (9,6+5,6)	
	6,0	exported feed	Mass balance: Fodder (1,6) + by-products (9,6+5,6) + imports (0,6) – feed (6,3)	
Food industry (9)	9,6	food ind. by-products to feed	Food industry by-products to feed (from regional reports) x nutrient content of different by-product streams	(27) – (29), (11), (13)
	9,1	by-products to agriculture	Food industry by-products to agriculture (from regional reports) x nutrient content of straw	(27), (28), (13)
	9,5	food ind. organic waste to AD	Food industry by-products to waste treatment (from regional reports) x nutrient content of organic waste	(27), (28), (30)
	9,0a	exported vegetal products	Mass balance: Vegetal crops (1,9) – vegetal products (9,4a) – by-products to AG (9,1) – 98% of by-products to feed (9,6)	(29)
	9,0b	exported animal products	Mass balance: Animal products (3,9) – local consumption (9,4b) – inedible fraction of animal production	(26), (10) – (12), (29)
	9,0c	losses	Inedible fraction of animal production – waste to AD (9,5) – 2% of by-products to feed (9,6)	
	9,4a	vegetal products	Consumption + 85% of hh food waste x nutrient content	(27), (30) – (34)
	9,4b	animal products	Consumption + 15% of hh food waste x nutrient content	(27), (30) – (34)
Consumption (4)	4,5a	Municipal ww	Population x per capita nutrient load in municipal wastewater	(35), (36)
	4,5b	OFMSW	Generation of household food waste x nutrient content	(27), (32), (37), (30)
	4,0	Other hh flows	Mass Balance: (9,4a) + (9,4b) – (4,5a) – (4,5b)	
Waste Management (5)	5,1a	Sewage sludge to cropland	Nutrients in ww x removal efficiency x share of treated sewage sludge to agriculture	(35), (37), (38), (30)
	5,1b	Compost/ digestate	OFMSW x share composted/digested x process yield x nutrient content of compost/ digestate	(27), (30), (32), (37), (39), (40)
	5,6	OFMSW to feed	OFMSW x share to feed industry x nutrient content	(27), (32), (30), (33)
	5,S	ash, landfilled waste	OFMSW/ sludge x share to incineration/ landfill x nutrient content	(27), (32), (37), (41)
	5,0	emissions	Mass Balance: (9,5) + (4,5) – (5,1) – (5,6) – (5,S)	

(1)(Papangelou and Mathijs, 2021a), (2)(REEW, 2019b), (3)(VMM Milieurapport Vlaanderen, 2012), (4)(REEW, 2018a), (5)(Sardans and Peñuelas, 2015), (6)(Tipping et al., 2014), (7)(Statbel, 2015a, 2014), (8)(FAOSTAT, 2019), (9)(Lenders et al., 2012), (10)(Le Noë et al., 2017), (11)(Nubel, 2018), (12)(USDA, n.d.), (13)(INRA et al., n.d.), (14)(Lederer et al., 2015), (15)(Bellarby et al., 2018), (16)(Crémer, 2015), (17)(ILVO, 2019), (18)(VLM, 2014), (19)(VLM, 2015a), (20)(Maes, 2019), (21)(Thibaut, 2016), (22)(Braekman et al., 2014), (23)(VLM, 2018), (24)(Godden and Luxen, n.d.), (25)(CGDD, 2013), (26)(Statbel, 2017a), (27)(Braekevelt and Vanaken, 2017), (28)(ICEDD and BioWallonie, 2017), (29)(BEMEFA, 2016), (30)(Coppens et al., 2016), (31)(Ridder et al., 2016), (32)(SPW, 2018), (33)(Fisgativa et al., 2016), (34)(Papangelou et al., 2020), (35)(Aquafin, 2014), (36)(Larsen and Maurer, 2011), (37)(REEW, 2018b), (38)(Gouvernement Wallon, 2005), (39)(Vlaco, 2016b, 2012, n.d.), (40)(IDELUX, 2019), (41)(Haupt et al., 2018)

3.2.2 Indicators for the circularity assessment

In the introduction of this thesis (section 1.2), circularity was defined as the process of closing, narrowing, and slowing the material and energy loops, towards a reduction in absolute resource use. In a recent review study on the circularity of biological cycles, Navare et al. (2020) identified four main categories of circularity strategies relevant to the cycling of bio-based materials: reducing the input of resources, minimizing wastes and losses, increasing the inputs of recycled materials, and maximizing the value, utility, and durability of products. Whereas the last category is better suited for durable, non-food bio-based products, the other three more generally apply to the system rather than the product level and are applicable to food systems too.

Translating the circularity definition and circularity strategies into indicators, we came up with a set of five metrics. The first overarching metric is the Total Input (In), mirroring the end goal of absolute reduction in resource use. Total Input is normalized to the population in each region and expressed in kgP/cap·a, so that it can be used to compare the different regions. Four relative indicators expressed in % complement Total Input for a comprehensive circularity assessment: the Nutrient Use Efficiency (NutUE), quantifying the relative resource input; the secondary-to-total input ratio (Sec), addressing the need for increased input of recycled resources, and the narrowing of the loop; the recycling rate (RR), corresponding to the closing of the loop; and the emissions and losses (Loss) addressing the minimization of waste and losses. A higher secondary-to-total input is considered to represent the narrowing of the loop, i.e. making it more local, because secondary nutrient sources such as treated organic waste streams typically come from the same farm or region, given that transporting them further away is often difficult and expensive. The slowing of the loop (reducing consumption, increased sufficiency) is not adequately addressed in this set of indicators, because it requires a consumption-based approach. Nonetheless, we try to address it through the definition of the nutrient use efficiency, by varying what the useful outcome or the products of the system might be (see the discussion in section 3.3.3).

We focused the circularity assessment on P for two reasons. First, P is not volatile in the way N is, so it is always bound to biomass. This makes P both more practical to work with, and a better ‘trace indicator’ for food flows. Second, P is especially fitting to the Circular Economy concept, as it can be both a pollutant and a scarce resource (Nesme and Withers, 2016). We estimated values for the indicators at the whole system level and, if applicable, also for the three sub-systems: agriculture (including cropland and grassland), food production (including cropland and grassland, the livestock sector and the food and feed industry), and the waste management system, including the management of organic municipal solid waste (MSW) and domestic wastewater. Table 3.2 offers an overview of the indicators.

Table 3.2 Definition of indicators used to assess circularity.

Indicators	Abbreviation	Definition	Unit	Favourable trend
Total Input	In	Sum of all inputs into the (sub-)system	kgP/cap·a	↘
Nutrient (Phosphorus) Use Efficiency	PUE	$\frac{products}{total\ input}$	%	↗
Secondary-to-total Input	Sec	$\frac{secondary\ input}{total\ input}$	%	↗
Recycling Rate	RR	$\frac{reused\ flows}{total\ waste\ generated}$	%	↗
Losses	Loss	$\frac{emissions\ \&\ losses}{total\ input}$	%	↘

3.2.3 Sensitivity Analysis

In order to test the robustness of the study outcomes against variations in input variables, and to screen which of these variables have a significant effect on the outcome and which not, we performed a simple local sensitivity analysis (Cariboni et al., 2007; Pianosi et al., 2016). We tested how some key model outcomes (soil balance, nutrient use efficiency at different levels for N and P) deviate from their original value when the values of selected input variables increase one at a time by a small, arbitrary value, in this case 10%. The sensitivity analysis gives an indication of which model variables have the greatest influence on the final outcome, and so where efforts to improve the model and the system should be concentrated. We chose the key input variables for the sensitivity analysis considering the following criteria:

i. The variable influences directly one of the largest flows (manure, crop production, forage production, imports).
ii. The variable value is at least partly based on assumptions or on scientific literature. This excluded variables whose values have very low uncertainty, such as data from the official agricultural census.

3.3 Results & Discussion

3.3.1 Nutrient flows in the agro-food system at regional & national level

Feed flows dominate the agro-food system in Flanders, especially compound feed from the industry to the livestock sector (Figure 3.4, left): N and P flows in compound feed mount up to 200 ktN/yr and 45 ktP/yr. To sustain this level of production, the feed industry in Flanders receives inputs from local cropland (52 ktN/yr and 9.4 ktP/yr, or 25% and 21% of all N and P input to feed industry) and the food industry in the form of by-products (43 ktN/yr and 6.6 kt P/yr, or 21% and 15% of input), but mainly from imported products and components. Imports account for more than half of the N (110 ktN/yr) and three quarters of the P (29 ktP/yr) of the total inputs into the Flemish feed industry. Another 43 ktN/yr and 5.5 ktP/yr enter the livestock sector as forage grazed directly on grasslands. 32% of the total N ingested by animals (78 ktN/yr), 42% of the P (21 ktP/yr) and 4% of the K (8.3 ktK/yr) are converted into animal products (carcass, milk, eggs, etc.) and transferred to the food industry for processing. The largest proportion of the ingested nutrients is transferred to animal manure and either applied to cropland and grassland, or exported from the region. Manure is the single largest input of nutrients into agricultural land, carrying 49%, 78% and 68% of the total N, P, and K into cropland (58 ktN/yr, 12 ktP/yr, and 54 ktK/yr) and 76%, 99% and 97% of the respective inputs into grassland (52 ktN/yr, 9 ktP/yr, and 52 ktK/yr).

In contrast to Flanders, a smaller livestock sector in Wallonia allows for a tighter connection between livestock and the local grassland. The dominating N (Figure 3.4b) and P (Figure 3.4d) flows are grazed forage (75 ktN/yr and 9.4 ktP/yr) and the manure returned to grassland (54 ktN/yr and 8 ktP/yr), either as direct excretion during grazing or applied on temporary grasslands. A smaller livestock sector means lower system losses, too. The gaseous N losses from animal housing and manure storage in Wallonia (17 ktN/yr) are almost 1/3 of the respective losses in Flanders (45 ktN/yr). P losses from the food industry consist mainly of the inedible parts of animals and are also much lower in Wallonia (2.7 ktP/yr) than in Flanders (16 ktP/yr). K deviates slightly from N and P in Wallonia, with around 28% of the K intake of animals coming from imported sources (22 ktK/yr of a total of ~80 ktK/yr). Most K excreted from animals ends up onto grassland (55 ktK/yr) and eventually accumulates in the soil, while a K deficit is observed for cropland (Figure 3.4f).

3.3.2 Nutrient soil balances in Belgian agriculture

If we zoom in the soil balances resulting from the flow analysis (flows 1,S and 2,S), we observe that they show a North-South trend for all three nutrients (Figure 3.5), a trend that mirrors the classification of the regions according to their production system. The highest N and P soil surpluses are observed in Flanders, especially in Polders (108 kgN/ha and 3.8 kgP/ha), the sandy region (94 kgN/ha and 4.8 kgP/ha) and Kempen (83 kgN/ha and 4.8 kgP/ha). The sand-loamy region, that includes parts of West Flanders, the pig-raising province of Flanders, comes closely behind with 92 kgN/ha and 2.4 kgP/ha. These P surpluses are comparable to findings for regions with similar production systems. P budgets were estimated to be 8.7 kgP/ha in Northern Ireland (Rothwell et al., 2020) and 9.6 kgP/ha and 4 kgP/ha in the Dutch cropland and grassland (Smit et al., 2015). In France, surpluses are even higher, reaching 14.6-16.1 kgP/ha in regions with intensive livestock farming (Le Noë et al., 2017). Interestingly, Coppens and colleagues (2016) found a P deficit of -1.45 kgP/ha (1.14 ktP/yr) for the Flemish region in 2009, while we observe a surplus of ~5.5 ktP/yr. The difference is almost in its entirety due to the lower

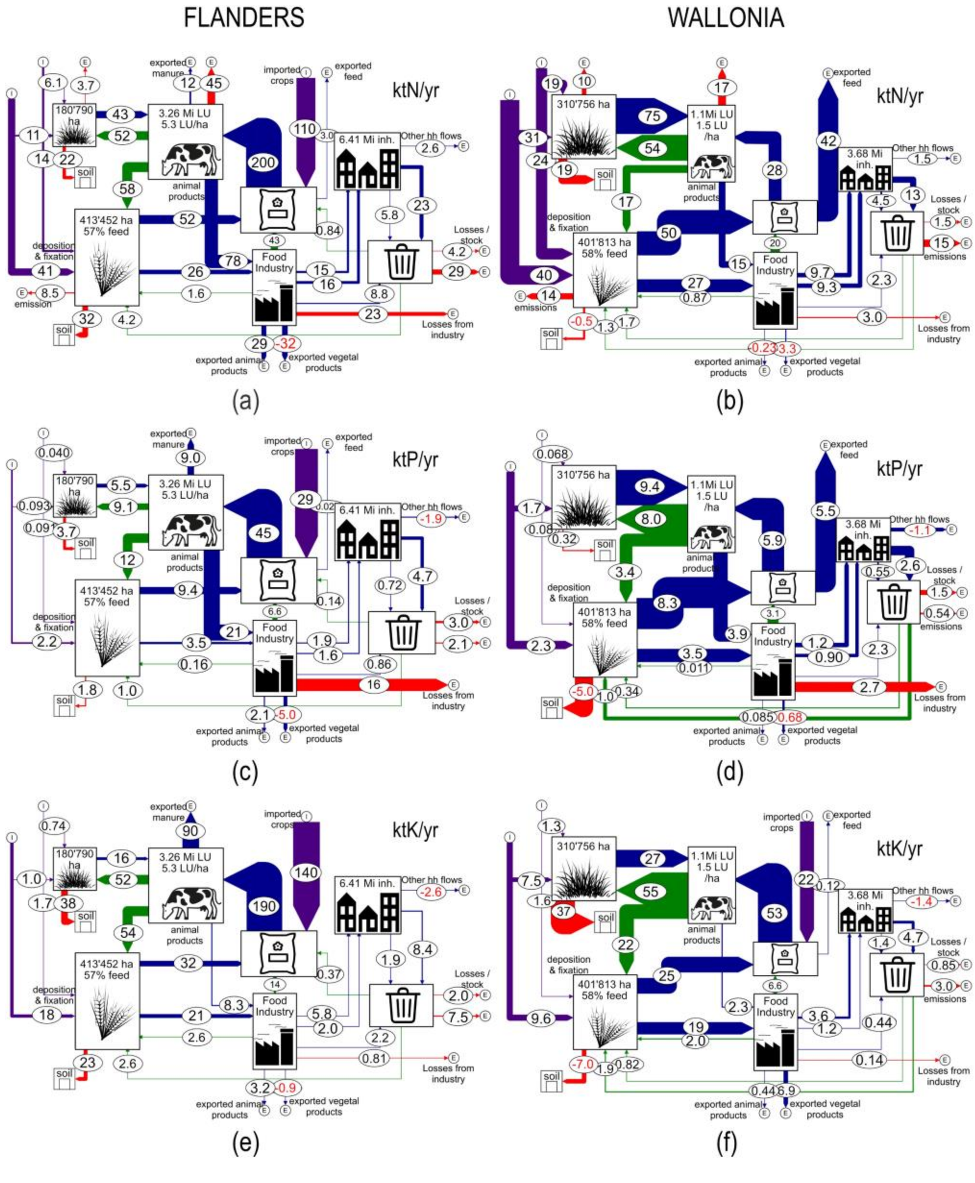


Figure 3.4 Flows of N (a, b), P (c, d) and K (e, f) in the agro-food system of Flanders (left) and Wallonia (right) for the reference year 2014. All flows in kt/yr. The thickness of the arrows are proportionate to their values within each diagram, but not across all of them (LU: Livestock Units, Mi: millions, hh: household).

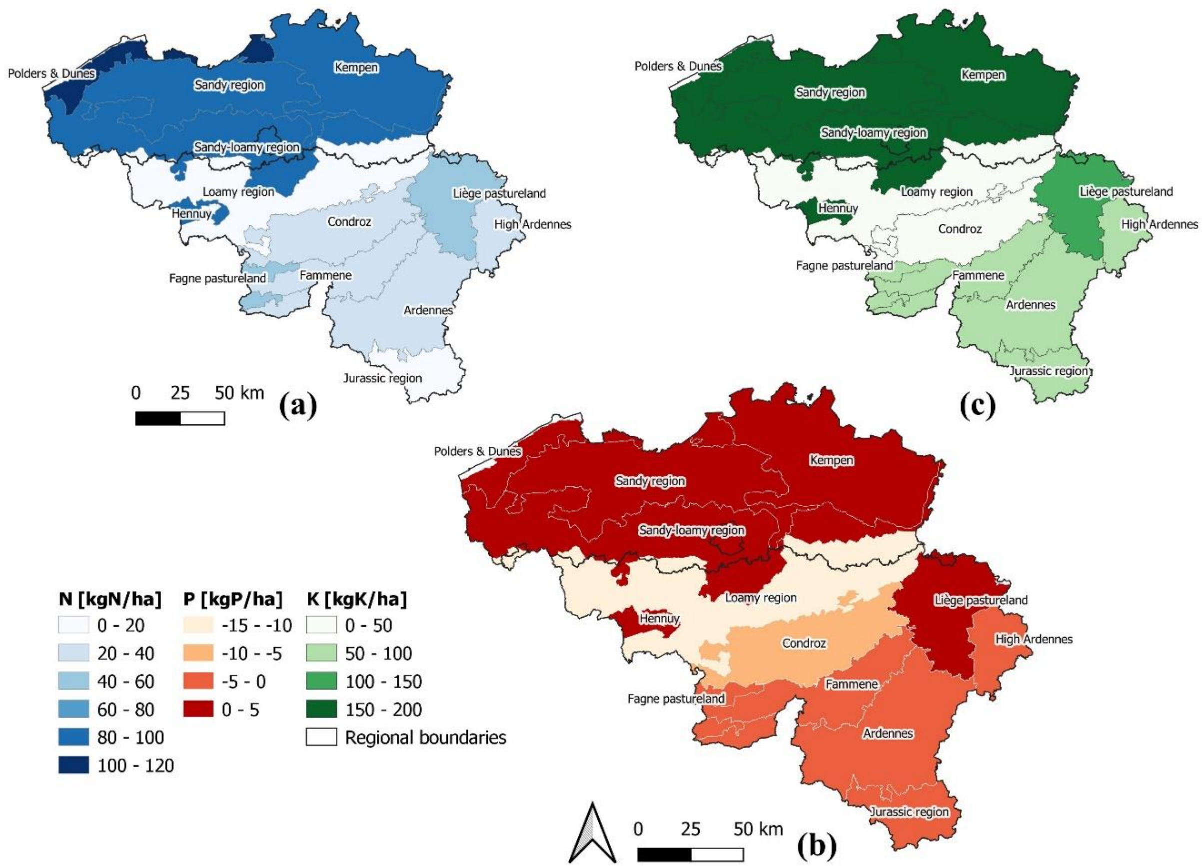
extraction rates of P through crop production we find compared to the Coppens study (19 and 23.5 ktP/yr respectively). The different extraction rates can be explained with fluctuations in the yields from year to year, or the choice of data sources used for the crops' P content, a choice that significantly influences the end result.

The southern agricultural regions, on the other hand, are characterized by low N surpluses and P deficits (Figure 3.5a,b). The Liège pastureland is the only exception, with surpluses of 58 kgN/ha and 3.3 kgP/ha. This is due to the large grassland area and high livestock density in the region, characteristics that imply lower crops nutrient content and high manure inputs from grazing animals. The hilly grassland-covered regions in the south of Wallonia have low N surpluses of around 20-30 kgN/ha, and moderate P deficits of -1 kgP/ha to -4 kgP/ha. The regions in the central, crop-producing part of the country, have the lowest N surpluses and highest P deficits. Most characteristically, the loamy region has a N surplus of 19 kgN/ha and a P deficit of -13 kgP/ha. These stark contrasts in the agricultural regions within Wallonia as compared to Flanders are the result of a larger variety in landscapes and production systems across the region. They also highlight a potential for great improvement in our model. We are currently using a uniform fertilizer application rate in the whole of Wallonia (REEW, 2019b). Updating the model with data specific to agricultural regions and crops, would offer a more accurate, and perhaps different, picture of the nutrient balances in Walloon agriculture.

Despite the negative soil P balances in Wallonia, roughly 22 ktN and 2 ktP were emitted annually from agricultural lands into surface and groundwater bodies in the period 2011-15 (REEW, 2018c). Indeed, 57% of the area in Wallonia is considered vulnerable to groundwater pollution by nitrates, while high ammoniac and orthophosphate concentrations are observed in 20% and 24% of monitoring sites, especially in the north of the Region (REEW, 2019c, 2019d). The negative P soil balances in Wallonia can be an indication of the efforts to reverse this trend, by lowering nutrient inputs and allowing the plants to utilize the nutrients already accumulated in the soil. To the contrary, stricter measures are needed in Flanders where nutrient surpluses from agriculture can further compromise the quality of water bodies, in which nitrate and orthophosphate concentrations are often beyond the accepted threshold values (VMM, 2020).

The K balances across the country resemble those of N, with K accumulating in all agricultural regions, despite a deficit in the croplands of the loamy region and the region of Condroz (Figure 3.5c). These regions have the lowest surpluses over the whole agricultural land (13 kgK/ha in the loamy region and 37 kgK/ha in Condroz), while surpluses range from ~60-100 kgK/ha Wallonia and above 150 kgK/ha in Flanders. K surpluses reach >60% of the total input in most of Flanders and between 40-50% in southern Wallonia. This means that only a small percentage of the K applied through mineral fertilizers and manure is used up by plants, while most of it is accumulating in the soil, especially in grasslands. Soil K surpluses are typically neither of agronomic nor of environmental concern, since plants need abundant K sources to thrive, and most soils can retain large amounts of the nutrient without excessive leaching. Leaching only becomes a concern for sandy soils, like the ones in Polders and the Flemish sandy region. Even then, however, K emissions into water bodies do not pose an environmental threat, like N and P emissions do. The agronomic and environmental concerns related to N and P are the reasons why manure application is regulated based on these two nutrients, rather than K. Nonetheless, taking K into consideration too, can prevent adverse effects of K oversupply to plants and help avoid the adverse effect of potash mining, such as the need to dispose of the large amount of excess salts generated.

Figure 3.5 (next page) Total annual soil balances per agricultural region for nitrogen (a), phosphorus (b) and potassium (c) in kg/ha.



Polders & Dunes
Sandy region
Kempen
Sandy-loamy region
Loamy region
Liège pastureland
Hennuy
Condroz
High Ardennes
Fagne pastureland
Fammene
Ardennes
Jurassic region
0 25 50 km
(a)
(c)
(b)
N [kgN/ha]
0 - 20
20 - 40
40 - 60
60 - 80
80 - 100
100 - 120
P [kgP/ha]
-15 - -10
-10 - -5
-5 - 0
0 - 5
K [kgK/ha]
0 - 50
50 - 100
100 - 150
150 - 200
Regional boundaries

3.3.3 Circularity assessment of the agro-food system

- *Sub-system: Agriculture*

The abundance of manure and the ubiquity of its use on agricultural land make us anticipate a circular and efficient agricultural system in Belgium. Indeed, agriculture (including cropland and grassland) scores better in Phosphorus Use Efficiency (PUE), Secondary-to-total input and Losses, compared to the other two sub-systems (food production and waste management), as well as compared to the whole system (Figure 3.6). The overall PUE for Belgian agriculture is 96%: 74% in Flanders and 125% in Wallonia (Figure 3.6). The value for Wallonia is greater than 100% due to the overall negative balance in soil, meaning that some of the P in harvested crops comes from the soil stock (and thus from within the system) and so it is not accounted for in "total input". Secondary-to-total input is also high in Belgium (84%) because of the high rates of manure application in both regions. Similarly high PUE values have been observed in Flanders (86%, Coppens et al., 2016), the Netherlands (65%-84%, Smit et al., 2015) and Northern Ireland (67%, Rothwell et al., 2020). The absolute total inputs in agriculture are 3.8 kgP/cap·a and 4.7 kgP/cap·a for Flanders and Wallonia.

- *Sub-system: Food production*

The Walloon food production sub-system scores better than the Flemish one in all indicators. Total input in Wallonia is more than three times smaller than in Flanders (1.7 kgP/ cap·a versus 5.6 kgP/ cap·a), and much more efficiently utilized (PUE of 123% versus 15%). This is because the livestock sector dominating Flanders is less efficient in nutrient use, a fact that has been confirmed in all similar, intensive livestock-rearing regions (Coppens et al., 2016; Le Noë et al., 2017; Rothwell et al., 2020; Senthilkumar et al., 2012; Smit et al., 2015). In addition, Belgium's food and feed industry are concentrated in Flanders, and depend heavily on imports. The Flemish food and feed industry are considered highly circular: feed industry has virtually no nutrient waste, while 6.6 ktP/yr of by-products from the food industry (1.03 kgP/ cap·a, >¼ of the total input in food industry, Figure 3.6c) are reused for feed production. Nevertheless, these flows are still low compared to (i) the import flows and (ii) the share of P in animal by-products that is not reused domestically. As a result, these two trends dominate in Flanders, and subsequently in Belgium, and cause the low PUE and secondary-to-total input values.

- *Sub-system: Waste Management*

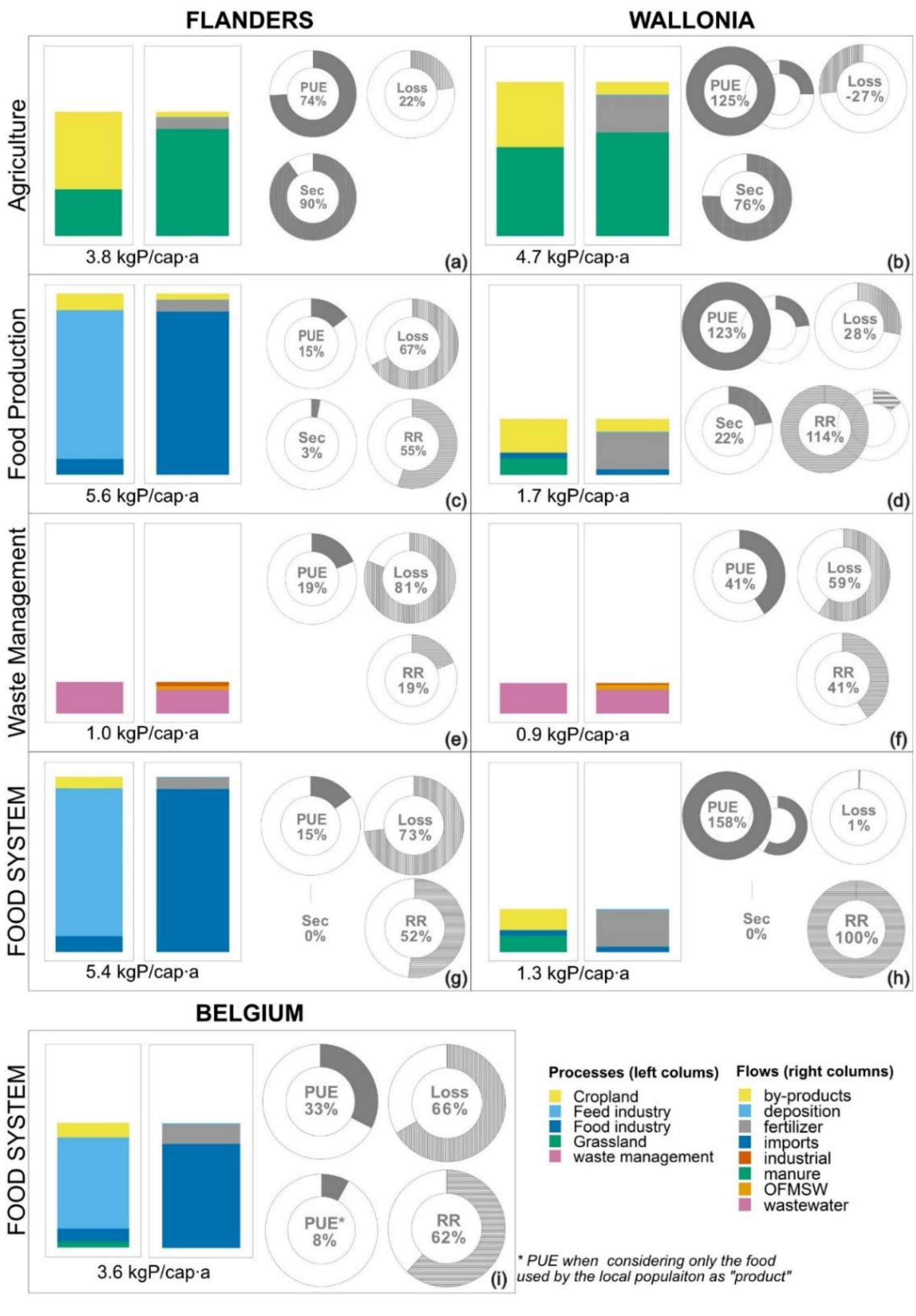
Wallonia outperforms Flanders regarding Waste Management (WM) too. The two regions have comparable per capita waste generated, however, the Walloon WM sub-system had both higher recycling rates (41% versus 19% in Flanders) and lower losses (49% versus 81%). These results may sound counter-intuitive, because Flanders has a more advanced management system for the organic fraction of municipal solid waste (OFMSW): organic waste have been being collected separately for years and 74% of the total household food waste are valorized as either animal feed or compost (Roels and Van Gijseghem, 2017). In Wallonia, less than 20% of the organic waste generated are separately collected and valorized through composting (SPW, 2018). At the same time, the reuse of wastewater treatment products such as treated sewage sludge is strictly prohibited in Flanders, so all phosphorus from human excreta ends up in incineration ashes and landfills. In Wallonia, around 50% of the sewage sludge from municipal WWTPs is conditioned with lime and applied in agriculture . It is thanks to the reuse of sewage sludge that the WM recycling rates and PUEs are higher in Wallonia.

Nutrients from human excreta is a source of nutrients currently overlooked in Belgium, especially in Flanders, despite the fact that human excreta contains more than five times as much P as all household food waste (Figure 3.6e, f). These amounts may still be small compared to manure; however, they are comparable to the P inputs in agriculture through mineral fertilizers. Their valorization in agriculture can therefore increase both the performance of the WM management system and provide a recycled, organic, and local nutrient source to agriculture.

- *Factors and choices affecting the circularity assessment*

The choice of the geographical and functional system boundary is maybe the most decisive factor when evaluating the five indicators. Some indicators do not even apply for some sub-systems, for example the secondary-to-total input for waste management. The secondary-to-total input represents the share of the input that comes from secondary (reused) rather than primary resources. A high secondary-to-total input ratio means that more reused than virgin resources are entering the system, whether they originate within the system or not. In this way, the secondary-to-total ratio adds information that the widely used recycling rate alone cannot. Nonetheless, assessing the indicator for the whole system level (Figure 3.6g, h, i) hides information on internal nutrient reuse, as all secondary flows are internal to the system and none crosses the system boundaries, resulting in a zero value. The value could have been slightly larger if we had information on the type of imports entering the feed industry: they could partly consist of agricultural or industry by-products.

Besides the use of a set of indicators, rather than a single metric, a transparent definition of the system's function is necessary to capture the complexity of the food system and the circularity concept. The PUE of the whole system is 33% (15% for Flanders and 158% for Wallonia), an efficiency comparable to reported values from literature, e.g. a 38% food system efficiency in Northern Ireland (Rothwell et al., 2020), and a 24% efficiency of food production in Flanders (Coppens et al., 2016). This definition of system-level PUE assumes that all food produced in the system is a useful product, whether directed to domestic consumption or exported. If we adhere to the definition of our system's function that is *to nourish* the people living within its boundary, the PUE drops to 8%, since the only useful product of the system in that case is food for domestic consumption. Under a similar assumption for the system's function, we estimate that the PUE for Northern Ireland and Flanders would fall to 8% and 10% respectively. Such a definition of the system's function points towards a sufficiency vision, where the food system in a specific place is primarily geared towards providing food for the local population. In that respect, PUE of this system could also be seen as a metric partially quantifying the slowing of the loop, e.g. producing only what is needed.



FLANDERS
WALLONIA
Agriculture
PUE 74%
Loss 22%
Sec 90%
3.8 kgP/cap·a
(a)
PUE 125%
Loss -27%
Sec 76%
4.7 kgP/cap·a
(b)
Food Production
PUE 15%
Loss 67%
Sec 3%
RR 55%
5.6 kgP/cap·a
(c)
PUE 123%
Loss 28%
Sec 22%
RR 114%
1.7 kgP/cap·a
(d)
Waste Management
PUE 19%
Loss 81%
RR 19%
1.0 kgP/cap·a
(e)
PUE 41%
Loss 59%
RR 41%
0.9 kgP/cap·a
(f)
FOOD SYSTEM
PUE 15%
Loss 73%
Sec 0%
RR 52%
5.4 kgP/cap·a
(g)
PUE 158%
Loss 1%
Sec 0%
RR 100%
1.3 kgP/cap·a
(h)
BELGIUM
FOOD SYSTEM
PUE 33%
Loss 66%
PUE* 8%
RR 62%
3.6 kgP/cap·a
(i)
Processes (left colums)
Cropland
Feed industry
Food industry
Grassland
waste management
Flows (right columns)
by-products
deposition
fertilizer
imports
industrial
manure
OFMSW
wastewater
* PUE when considering only the food used by the local populaiton as "product"

Figure 3.6 (previous page) P circularity indicators score for Flanders (left) and Wallonia (right) per sub-system, and for the whole system in Belgium (i). The columns within each tile give the total input into the (sub-)system in kgP/cap·a, once grouped according to the process that receives the input, and once according to the different input flows (for colour keys see legend in bottom right). The height of the columns are scaled across all tiles. (PUE: Phosphorus Use Effciency, Sec: Secondary-to-total input, RR: Recycling Rate, OFMSW: Organic Fraction Municipal Solid Waste)

3.4 Sensitivity Analysis

Some of the tested variables have an important effect on the end result and their values should be selected with care. The nutrient excretion rates of animals is such a variable: a 10% increase in the assumed values for cattle causes a 150% increase in the soil P surplus from 0.7 kgP/ha to 1.7 kgP/ha, while a further increase for all types of animals causes a 163% increase (Table 3.3). The animal excretion rates are technically a set of 11 (cattle) and 32 (all animals) variables, that we varied simultaneously. This partly explains the disproportionate big impact on the end results compared to other variables. Nonetheless, we find that varying the full set of variables simultaneously is the most suitable approach, since these variables are always provided as a set.

The nutrient content of crops is another set of influential variables. Increasing the grass P content by 10% causes a -190% change in the soil P balance from 0.7 kgP/ha to -0.6 kgP/ha. Varying the P content of maize has a lower but still significant effect, causing a change of -78%. In this study we used a 0.38% P content (dry matter) for the whole grassland production (permanent and temporary), as given in the official Flemish soil balance (Lenders et al., 2012), and an average yield of 8 tDM/ha. Many Substance Flow Analysis (SFA) studies use the same sets of crop nutrient contents and yields, for example data from FAOSTAT or the US Food Composition Database. Based on the results of our sensitivity analysis, we recommend the use of local data on the types of grass grown, as well as their yields and nutrient contents.

Two main trends emerge in the sensitivity analysis (Table 3.3). First, the soil balances tend to be more sensitive than the indicators; second, the results for P tend to be more sensitive than the ones for N.

The soil P balance is especially volatile, with small changes in key variables having a tremendous effect on the end result. It is important to note, however, that this is partly due to the low absolute values of the P balance. With a base value so close to zero (0.7 kgP/ha), we should expect changes to appear very high in relevant terms. Besides, the high sensitivity of the P soil balance reveals the need to establish this balance for a longer period of time rather for a single year, because time series could smooth out small fluctuations of P inputs and outputs that affect the balance. In conclusion, we find that working on P flows is like preparing a Mediterranean dish: exactly because only few ingredients are involved, the quality of each one is crucial to the outcome.

Table 3.3 Sensitivity analysis results [%]; effect of a 10% change in 8 selected variables to key results: soil surplus of N and P, and nutrient use efficiencies for P (PUE) and N (NUE) for the whole system (syst) and the food production sub-system (prod).Values x are red when |x|>75%, orange when 10% <|x|<75%, and green when |x|<10%.

variable	P soil balance	N soil balance	PUE (prod)	PUE (syst)	NUE (prod)	NUE (syst)
Excretion rate (cattle)	150	12	-10	-3	-9	-2
Excretion rate (all)	163	13	-14	-7	-12	-5
Excretion share outdoors	27	2	0	0	1	0
Imports	0	0	14	-7	5	-3
Fertilization cropland	49	12	-1	-1	-2	-2
Fertilization grassland	20	6	0	0	-1	-1
Grass nutrient cont.	-190	-20	10	1	8	1
Maize nutrient cont.	-78	-5	3	1	2	1

3.5 Conclusion

In this chapter we described the Belgian agro-food system through the nutrient flows between its different compartments, evaluated the soil balances for N, P, and K at the scale of agricultural regions, and used this information to evaluate a set of circularity metrics, such as nutrient use efficiency, secondary-to-total input and recycling rate. We performed the different parts of the analysis in three spatial scales: national, regional (Flanders and Wallonia) and sub-regional (fourteen agricultural regions). Working on different scales allowed us to make use of the most appropriate data available, and to gain a detailed and multi-level understanding of the food system. We confirmed that the nutrient flows in Flanders follow patterns similar to those of regions with intensive livestock production: high imported quantities for feed, high losses, and mostly positive soil nutrient balances. Wallonia, on the other hand, has generally lower soil N surpluses and P deficits, indicating a potential opportunity for better nutrient management at the national level. Agriculture in both regions shows high nutrient use efficiencies and high secondary-to-total input rates. Wallonia outperforms Flanders with regards to losses and nutrient recycling rates, thanks to a less intensive livestock sector, a smaller food industry on its territory and the practice of reusing sewage sludge in agriculture, which returns important amounts of nutrients back to the crops. Our results confirm the low nutrient use efficiencies of intensive, livestock-dominated agro-food systems and their contribution to nutrient surpluses in agricultural soils. A lower dependency on imported flows, stricter regulations on allowable agricultural inputs and the valorization of nutrients in human excreta would all improve the system's circularity. Ultimately, though, there is only so much the regulation of individual flows can do. The concept of circularity implies a tight connectedness of the food system to the landscape and the local population, so that nutrients flow in short and tight loops within the different compartments. A thorough rethinking of the agro-food system may be the only way towards increased circularity.

3.6 Annex

3.6.1 Details on methodology

- *Crop production – inputs to agriculture*

Data on fertilization are from the Walloon and Flemish environmental reports and soil balances (Table S17). We used average values for the period 2011-2017 to represent a 'standard' year, since the annual values fluctuate slightly around the year 2014. We used information from (REEW, 2018a) to estimate mineral fertilizer application rates for Wallonia. One application rate per nutrient was used for the whole of Wallonia, irrespectively of whether it refers to cropland or grassland. For Flanders, we estimated actual application rates per crop based on information from (Lenders and Deuninck, 2016) and aggregated them at the agricultural region level, as well as for the whole of Flanders (Papangelou and Mathijs, 2021a). Values for N atmospheric deposition and symbiotic fixation were taken from the Flemish soil balance study (VMM Milieurapport Vlaanderen, 2012) and the annual Walloon "State of the Environment" report (REEW, 2018a). The same sources were used for the estimation of N losses through volatilization, and N and P losses through leaching and runoff. Lacking any data for Belgium, European and global averages from the literature were used for the atmospheric deposition on P and K (Sardans and Peñuelas, 2015; Tipping et al., 2014).

Table S 17 Data on nutrient balances for Flanders and Wallonia

	Flanders(1)	**Wallonia(2)**
Synthetic N fertilizer application to agricultural land [kgN/ha UAA]	109.5(9)	100.6(3)
Manure N application to agricultural land [kgN/ha UAA]	152.7(5)	97.7(3)
N losses from volatilisation of synthetic fertilizer [kgN/ha UAA]	20.6(4)	33.6
N losses from leaching and runoff [kgN/ha UAA]	-	28.0
N-fixation [kgN/ha UAA]	6.8	30.8
N atmospheric deposition [kgN/ha UAA]	26.9	29.4
Synthetic P fertilizer application to agricultural land [kgP/ha UAA]	3.5(9)	5.6(3)
Manure P application to agricultural land [kgP/ha UAA]	30.2(5)	*19.2(6)*
P losses from leaching and runoff [kgP / ha UAA]	2.1(7)	3.03(8)
P atmospheric deposition	0.22(12)	0.22(12)
Synthetic K fertilizer application to agricultural land [kgK/ha UAA]	33.3(10)	24.0(3)
K atmospheric deposition (global average) [kgK / ha·yr]	4.1(11)	4.1(11)

(1) from (VMM Milieurapport Vlaanderen, 2012) (2) Average values for 2011-7 from (REEW, 2018a), unless indicated otherwise, (3) Average 2011-7 from (REEW, 2019e), (4) Ammonia losses (5) (VLM, 2015a) (6) Estimated value based on the assumption that the ratio N:P is the same for manure in Wallonia as in Flanders (7) The negative balance, (8) Average 2011-5 from (REEW, 2018c), (9) Average 2011-7 from (Departement Landbouw en Visserij, 2020), (10) Average 2011-2014 from (Lenders and Deuninck, 2016), (11) (Sardans and Peñuelas, 2015) (22) (Tipping et al., 2014)

Table S 18 Application rates for N, P, and K from synthetic fertilizers in the Flemish provinces

province	cropland			grassland		
	N [kgN/ha]	P [kgP/ha]	K [kgK/ha]	N [kgN/ha]	P [kgP/ha]	K [kgK/ha]
Antwerp	87.1	6.29	34.8	71.8	0.62	6.74
East Flanders	90.8	4.35	41.6	70.5	0.61	6.62
Flemish Brabant	111.0	4.80	37.6	44.4	0.38	4.17
Limburg	90.3	4.52	35.5	43.3	0.38	4.06
West Flanders	110.6	5.80	54.4	54.6	0.47	5.13

- *Crop production - Outputs through harvest*

We calculated the amounts of crops and grass produced in each region using official agricultural data on cultivated surface area for each crop and their respective yields (Statbel, 2015a, 2014). Gaps in the Belgian yields dataset were filled gaps with values from FAOSTAT (FAOSTAT, 2019), and an assumed average grassland yield of 8 tDM/yr, based on information from (Crémer, 2015; ILVO, 2019). We then converted the harvested quantities into nutrient amounts by multiplying them with the crops' nutrient content. Values for the N and P content of the different crops were taken from the most recent soil balance report for Flanders (Lenders et al., 2012) and from the original GRAFS dataset (Le Noë et al., 2017). We compiled the data on crops' K content using the Belgian food composition database (Nubel, 2018), the American Food Data Central (USDA, n.d.), Feedipedia (INRA et al., n.d.) and data from previous studies on K (Bellarby et al., 2018; Lederer et al., 2015).

- *Livestock – manure production & management in Flanders*

The basis of the estimation is the number of animals per province and municipality from the agricultural census data (Statbel, 2015a, 2015b). The statistical tables from Statbel do not include numbers of sheep, goats and horses. We use the ones given in the most recent report on trends in the Walloon agriculture (SPW, 2016). This report (Annex I.16) lists numbers of sheep, goats and horses for Belgium, the Flemish and Walloon regions, and for the Walloon provinces for May 2013. The numbers of sheep, goats, horses and rabbits for Flanders for 2014 are from the annual *Mestrapport* (VLM, 2015a). The national populations for these animals are estimated as the sum of the Walloon and Flemish ones. Since there is no data on rabbits in Wallonia, we assumed that rabbits are only bred in Flanders.

Amounts of N and P excreted per animal are from the Flemish Mestrapport (see Table S19; the values agree with the French ones given in (Le Noë et al., 2017). Lacking a local data source on amounts of K excreted, we used average data for the years 2000-2008 from a Dutch study (CBS, 2012) in a first approximation. We then compared the resulting N:K ratios excreted with Flemish data on N:K ratios in raw manure. The resulting N:K ratio for bovine excretion rates was close to that of raw manure (0.78 and 0.7 respectively), but the ones for pigs and poultry were deviating significantly from raw manure (1.29 and 1.0 for pig, and 3.06 and 1.6 for poultry). We thus decided to use the N:K ratios in raw manure to determine the K excreted from ruminants, pigs and poultry in Flanders. Lacking a N:K ratio for horse and rabbit manure we used the Dutch values. The resulting K excretion rates are shown in the Table S19.

The handling and storage losses are also estimated per animal with information from the Mestrapport (VLM, 2015a) and the amounts produced during grazing, and thus deposited directly onto grassland, are estimated to be roughly 25% of the total bovine manure produced (VLM, 2018). Detailed data on livestock heads per housing type are not available at the level of agricultural regions; in order to apply this information to Wallonia and to lower spatial scales (agricultural regions, municipalities), we adopt the following assumptions:

- All housing of **bovines** are of the type where liquid manure in between 10% and 90%.
- All **broilers** are assumed to be housed in low-emission housing (as is the case for 90% of broilers in Flanders).
- For all other **poultry**, we assume a uniform amongst the different types of housing and use the average per head emissions of them.
- For **pigs up to 100 kg** we assume traditional housing with slurry (83% of pigs in Flanders).
- For all **other pigs** we assume a uniform distribution between low emissions housing, traditional housing with slurry and traditional housing with solid manure, and thus use the average of the three.

The resulting N-loss coefficients are illustrated in the last column of Table S19.

Information on actual manure application is available for Flanders in the annual reports of VLM (Mestbank). The amounts of N and P from manure used on agricultural soils per province in 2013 are shown in Table S18. We are using the numbers from 2013, since it is the last year for which data at the province level are provided in the report, to estimate the application rates for organic N and P per province (last two columns in Table S20).

The manure reports do not contain information or applied amounts of K. We thus estimate these amounts using K:N ratios for different types of manure (Table S21). We opt for K:N over K:P ratios, because the former seem to be more stable and consistent across manure types and studies. We further chose the ratios for raw manure, since most of the manure applied within Flanders is raw manure.

Table S19 Excretion rates of livestock and N-losses in 2014 in kg/head·yr.: data for Flanders from (VLM, 2015a), N in Wallonia from (Maes, 2019), K using N:K ratios in raw manure from Table S21. LU from Eurostat, N loss from (VLM, 2019a)**. P for Wallonia estimated by multiplying the N:P ratio of Flanders with the N column for Wallonia.**

	Flanders			Wallonia				
animal category	**N**	**P**	**K**	**N**	**P**	**K**	**LU**	**N loss**
Milking cows	115.58	16.28	162.78	90	12.67	100.00	1	15%
Suckler cows	65	12.219	91.55	66	12.41	73.33	1	15%
Heifer for milking herd renewal >2 yrs	77	12.874	108.45	66	11.03	73.33	0.8	15%
Heifer for slaughter over 2 yrs old	77	12.874	108.45	66	11.03	73.33	0.8	15%
Males of butcher type over 2 yrs old	77	12.874	108.45	66	11.03	73.33	1	15%
Heifer for milking herd renewal, 1-2 yrs	58	8.379	81.69	48	6.93	53.33	0.7	15%
Heifer for slaughter, 1-2 yrs old	58	8.379	81.69	48	6.93	53.33	0.7	15%
Males of butcher type, 1-2 yrs old	58	8.379	81.69	40	5.78	44.44	0.7	15%
Veal calves	10.5	1.571	14.79	10	1.5	11.11	0.4	15%
Other females under 1 yr	33	4.364	46.48	28	3.7	31.11	0.4	15%
Other males under 1 yr	22.3	3.055	31.41	25	3.42	27.78	0.4	15%
Goats <1yr	4.36	0.751	4.36	6.6	1.14	9.43	0.1	1.39
Goats >1yr	10.5	1.807	10.50	6.6	1.14	9.43	0.1	3.31
Sheep <1yr	4.36	0.751	4.36	6.6	1.14	9.43	0.1	1.06
Sheep >1yr	10.5	1.807	10.50	6.6	1.14	9.43	0.1	1.75
piglets	2.18	0.668	2.18	1.9	0.58	1.27	0.027	0.6

young pigs between 20 and 50 kg	13	2.326	13.00	7.8	1.4	5.20	0.3	5.86
Sows over 50 kg	24	6.328	24.00	15	3.95	10.00	0.5	5.86
Boars over 50 kg	24	6.328	24.00	15	3.95	10.00	0.5	5.58
fattening pigs between 50-80 kg	13	2.326	13.00	7.8	1.4	5.20	0.5	5.86
fattening pigs between 80-110 kg	13	2.326	13.00	7.8	1.4	5.20	0.5	5.86
fattening pigs >110 kg	24	6.328	24.00	7.8	2.06	5.20	0.5	3.91
Laying hens for hatching eggs	0.81	0.196	0.52	0.6	0.15	0.35	0.014	0.112
Laying hens for consumption eggs	0.81	0.196	0.52	0.6	0.15	0.35	0.014	0.219
young hens	0.61	0.113	0.39	0.27	0.05	0.16	0.014	0.400
chickens	0.61	0.113	0.39	0.27	0.05	0.16	0.007	0.135
Horses	65	13.092	66.317	65	13.09	66.317	0.8	10.46
Ponies	50	9.164	36.852	50	9.16	36.852	0.8	7.47

Table S 20 Maximum allowable and actual applied amount of manure for Flemish provinces in kgN/year and kgP_2O_5/year. [Source: (VLM, 2014) and the corresponding reports from 2015 and 2016]. AR: Application Rate

Province	max allowed N [kgN/year]	actually applied N	max allowed P [kg P_2O_5/year]	actually applied P_2O_5	N AR [kgN/ha]	P AR [kgP/ha]
Antwerp	19’967’906	17’037’274	8’974’688	7’147’416	188.4	35.0
Limburg	16’928’402	11’027’650	7’845’722	4’960’599	370.2	26.4
East Flanders	28’049’395	22’989’795	12’857’047	10’249’777	69.9	31.2
Flem. Brabant	15’400’709	10’172’524	7’444’604	4’816’857	130.5	25.2
West Flanders	36’282’685	30’759’145	16’359’422	13’980’941	115.6	31.2
TOTAL	116’629’096	91’986’388	53’481’484	41’155’590	**152.7**	**30.2**
TOTAL 2014	104’610’514	94’100’000	45’312’743	42’700’000		
TOTAL 2015	117’609’987	92’400’000	50’818’534	41’000’000		
Dif. 2013-14		2.3%		0.4%		

Table S 21 Composition of different unprocessed manures for Flanders (FL - Braekman et al., 2014) and Wallonia (WA - Thibaut, 2016) and comparison with literature values for the UK (UK - AHDB, 2017) and the Netherlands (NL - Remmelink G et al., 2019). Values in italics are calculated.

manure type		DM [%]	N [kg/t]	P_2O_5 [kg/t]	*P [kg/t]*	K [kg/t]	*N:P*	*N:K*
bovine manure, raw	FL	8.5	4.4	1.6	*0.7*	6.2	*6.3*	*0.7*
pig manure, raw	FL	9.0	7.2	4.2	*1.8*	7.2	*3.9*	*1.0*
goat manure, raw	FL	6.0	4.2	3.0	*1.3*	4.3	*3.2*	*1.0*
poultry manure, dried	FL	54.0	25.4	23.0	*10.0*	16.2	*2.5*	*1.6*
bovine manure, fresh	WA	18.7	5.3	2.6	*1.1*	5.7	*4.7*	*0.9*
bovine slurry, fresh	WA	7.3	3.5	1.4	*0.6*	3.2	*5.7*	*1.1*
pig manure, fresh	WA	25.7	7.8	5.1	*2.2*	5.6	*3.5*	*1.3*
pig slurry, fresh	WA	7.4	6.1	3.1	*1.4*	3.7	*4.5*	*1.6*
poultry manure	WA	47.7	22.8	14.1	*6.2*	13.7	*3.7*	*1.7*
cattle manure	UK	25.0	6	3.2	*1.4*	7.8	*4.3*	*0.8*
pig manure	UK	25.0	7	6	*2.6*	6.6	*2.7*	*1.1*
poultry manure	UK	20.0	9.4	8	*3.5*	7.1	*2.7*	*1.3*
cattle manure, slurry	NL	2.5	4.0	0.2	*0.09*	6.6	*45.8*	*0.6*
cattle manure, solid	NL	26.7	7.7	4.3	*1.9*	7.3	*4.1*	*1.1*
pig manure, slurry	NL	2.0	6.5	0.9	*0.39*	3.7	*16.5*	*1.7*
pig manure, solid	NL	26	7.9	7.9	*3.5*	7.1	*2.3*	*1.1*
poultry manure, dried	NL	56.2	28.4	23	*10*	15.9	*2.8*	*1.8*

- *Livestock – manure production & management in Wallonia*

Amounts of N excreted per animal are from (Maes, 2019), P and K excretion rates are derived from N rates, applying N:K and N:P ratios from (Thibaut, 2016). We assume that half of the total manure produced in Wallonia is produced in houses and can thus be managed, while the other half is produced directly on the pastures, based on a figure given in Table S22. These numbers agree with the Flemish data, where the total N-losses in the pork and poultry sector are 35% and 27%, as well as with the estimation of ~26% N housing losses in Belgium (Oenema et al., 2007).

Table S 22 Shares of N, P and K in slurry per agricultural region in Wallonia [Source: (Godden and Luxen, n.d.)]

Region	Slurry Ratio			Ratio used [% slurry]
	N [%]	P [%]	K [%]	
Sablo-limoneuse	25-30	30-40	20-25	30
Limoneuse	25-30	30-40	20-25	30
Herbagère	70-80	70-80	70-80	75
Campine hennuyère	25-30	30-40	20-25	30
Condroz	20-25	20-25	15-20	20
Fagne	20-25	20-25	20-25	22.5
Famenne	20-25	20-25	20-25	22.5
Haute Ardenne	80-90	80-90	80-90	85
Ardenne	15-20	20-25	15-20	20
Jurassique	15-20	20-25	15-20	20

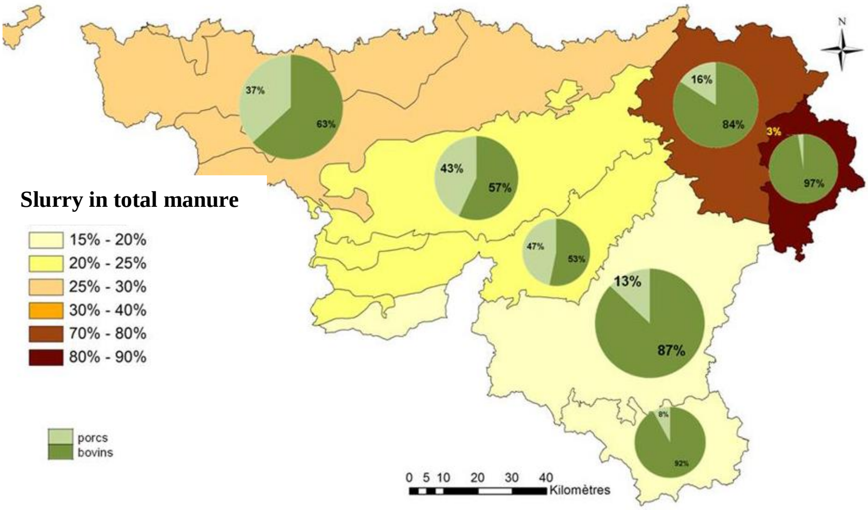


Figure S 2 Localization of the slurry production in Wallonia and contribution of bovines and pigs [Source:(Godden and Luxen, n.d., Figure 2a)

Table S 23 Slurry/total manure per province

Province	Ratio used [% slurry]
Brabant Walloon	30.0
Hainaut	28.4
Liège	61.0
Luxembourg	21.8
Namur	21.9

Table S 24 Factors used to estimate N-losses during manure handling, storage and application for Wallonia [Source: (CGDD, 2013)]

Stage at manure management	Manure type	NH_3-N losses [%]	NO_2-N losses [%]
Volatilization from litter manure indoors	cattle	26	4
	pork	24	3.4
	poultry	35	4
Volatilization from slurry indoors	cattle	27	3
	pork	27	2
	poultry	35	3
Volatilization from excretion outdoors	cattle	10	1
	pork	*10*	*2*
	poultry	*60*	*2*

As in the original study of Le Noë (2017), we assume that all manure produced in Wallonia is returned to agricultural land. The same assumption is made for the estimation of the organic N input, in the report for the State of the Environment in Wallonia (Maes, 2019). The amounts of organic N applied in Wallonia are based on N excretion rates as given at the Walloon Code for the sustainable nitrogen use in agriculture from 2014 (Gouvernement Wallon, 2014) given in Table S19. The values reported in Annex III of the statute are net production rates, excluding storage losses. This means that these values would represent the sum of flows 2 and 3 in the simplified diagram of Figure S3. The original GRAFS methodology, as well as the numbers for Flanders in Table S19, correspond to flow 1. The field of manure management in the GRAFS workbook was thus adjusted to calculate the N emissions from the net amounts of nitrogen to land.

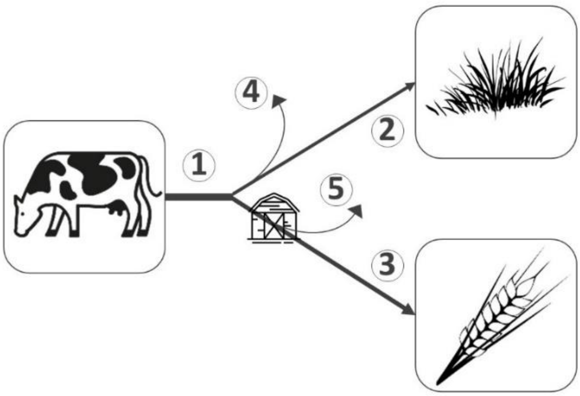


Figure S 3 Simplified diagram of nutrient flows from animal excretion to land application

- *Production of meat and animal products*

We use the production values of meat and animal-derived products from the Supply Balances for the whole country (Statbel, 2017a). From these datasets, we use the gross production for the year 2014 (Table S25), or for the closest one, when 2014 is not available. The production of eggs is given in 1'000 pieces; to convert to mass we use a weight of 50 g per egg given in (Nubel, 2018).Since these numbers are given only at the national level, we allocate the production to each province according to the corresponding numbers of animals as shown in the last column of Table S25.

We exclude horse meat because there is no data on the provincial scale; since the quantities of produced horse meat are much lower than that of beef, pork and poultry, we assume that their exclusion will not affect the results significantly. We add the production of rabbit meat (using the carcass partitioning and nutrient content coefficients as for small ruminants), since it is a significant part of the meat production in Flanders, especially in the province of Antwerp.

Table S 25 Gross production of animal products in Belgium for 2014 (or the year closest to it with available data) and base for the allocation of the production to the provinces [Source: (Statbel, 2017a)]

	Gross production BE [kton/yr]	**Base for allocation to provinces & AG regions**
Bovine meat	245.3	Number of non-milking cows
Pork meat	1’128.9	Total number of pigs in province
Sheep & goat meat	0.7	Total number of sheep in province
Poultry	341.8	Number of broiler chicken in province
Eggs	137.7	Number of laying hens in province
Cow milk	1’275.5	Number of milking cows in province
Sheep & goat milk	12.1	Total number of sheep in province

To calculate the different parts of the animal (edible meat, bones, grease, blood etc.) we use the same data as in (Le Noë et al., 2017), for both the weights of each part and their N and P content, and we cross-checked the data with information from (Alexander et al., 2016) and (FAO, 2000). No detailed information on the K-content of each organ is available to the best of our knowledge. More than 90% of the K in animals’ bodies is found in the intracellular fluid (Pond et al., 1995), we thus assume that K is uniformly distributed in the animal and use the meat K-content from internubel to approximate it. The resulting total concentration in live bovines agree with the estimation of 3 gK/kgBW by (INRA, 2018).

Table S 26 Assumptions used to calculate the amounts of potassium in the edible and inedible fractions of animals [Sources: edible fractions from (Alexander et al., 2016); carcass weight from (FAO, 2000); K-content of meat from (Nubel, 2018)].

	edible fr. [%LW]	carcass [%LW]	K-content [mgK/100gr]	internubel category
Bovine meat	40	54	340	Beef, roastbeef, raw
Pork meat	55	79	360	Pork steak, raw
Sheep & goat meat	55	50	*340*	*assumption, as for beef*
Poultry	70	72	359	Chicken with skin, raw

- *Consumption*

Food consumption is based in data from Belgian Food Consumption Survey (Ridder et al., 2016) shown in Table S27, while food flowing into households additionally includes food waste. In order to be able to add the wasted amounts separately to the vegetal and animal products consumed, we use the distribution of food waste for Flanders (Roels and Van Gijseghem, 2017, p. 74). In Figure S4 we see that meat and dairy make up of 5% and 7%; we add another 3%, to account for animal products in parts of the desserts, snacks, and processed food wasted. We thus use a distribution of 85/15 for vegetal/animal origin products in household food waste.

We use the national averages for the per capita ingested amounts of food from the national Food Consumption Survey (Table S27) and information on their nutrient content from the databases internubel and NEVO (Papangelou et al., 2020). The N-content is estimated from the protein content given in the food composition databases, multiplied by a 16% N-content in proteins (FAO, 2003). The aggregated values used for the GRAFS model are given in Table S28.

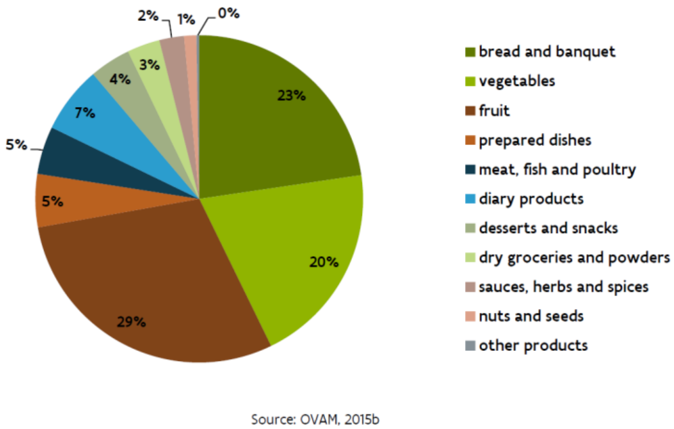


Figure S 4 Distribution of household food waste (in residual waste) per product category for Flanders [Source: (Roels and Van Gijseghem, 2017)]

Table S 27 Average per capita ingestion of food, protein and nutrient amounts for Belgium

Food groups	Consumption [g/cap.d]	Protein [g/cap.d]	Nitrogen [mgN/cap.d]	Phoshorus [mgP/cap.d]	Potassium [mgK/cap.d]
Grains	230.7	17.6	2825.1	207.3	291.9
Vegetables	133.4	1.7	67.9	48.9	356.4
Roots & tubes	43.3	0.7	110.8	17.3	171.8
Legumes & nuts	5.8	1.1	181.2	23.4	135.4
Fruits	113.3	0.3	52.2	12.1	138.2
Meat	98.8	26.1	4169.6	229.7	453.5
Fish	17.8	4.1	691.0	53.7	64.8
Dairy	183.5	14.1	2251.7	370.9	262.3
Eggs	9.6	1.2	189.0	34.4	13.6
Sugar & confectionary	94.7	1.1	168.9	61.7	145.1
Oils & fats	17.8	0.1	9.1	2.4	2.4
Juices	64.9	0.6	103.8	12.3	119.4
Water and non-alcoholic beverages	1063.2	0.0	0.0	44.6	0.0
Alcohol	377.9	0.4	60.1	24.7	82.6
Coffee & tea	296.6	0.4	65.0	0.0	232.3
Composite dishes	55.3	3.9	628.6	72.0	130.1
Condiments	29.1	0.5	72.9	9.9	4.0
Others	0.0	0.0	0.0	0.0	0.0
TOTAL	**2’835.6**	**73.7**	**11’814.5**	**1’216.5**	**2’603.5**

Table S 28 Aggregation of annual per capita consumption into food groups

	Consumption [kg/cap.a]	Protein [kg/cap.a]	Nitrogen [kgN/cap.a]	Phosphorus [kgP/cap.a]	Potassium [kgK/cap.a]
food of vegetal origin	259.8	9.8	1.57	0.16	0.50
food of animal origin	110.7	15.3	2.42	0.23	0.27
dairy and eggs	*74.7*	*5.8*	*0.89*	*0.15*	0.10
edible meat	*36.1*	*9.5*	*1.52*	*0.08*	0.17
fish	6.48	1.49	0.25	0.02	0.02
beverages	658.0	0.5	0.08	0.03	0.16
Total (w/o beverages)	**1035.0**	**26.9**	**4.32**	**0.44**	**0.95**

- *Wastewater and MSW management*

Solid Waste Management in Flanders

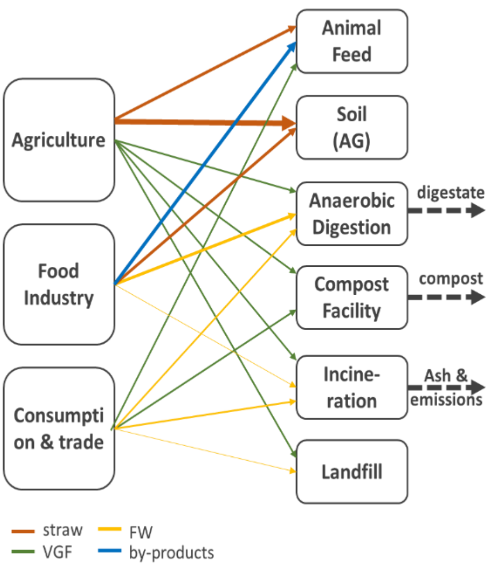


Figure S 5 Types of materials of organic waste in Flanders and their destinations. VGF: vegetable, garden, fruit waste; FW: food waste; AG: agriculture. Compiled based on information from (Braekevelt and Vanaken, 2017).

To estimate the amounts of nutrients in the OFMSW in Flanders, we used data on the amounts of food residue and their destinations for 2015 from OVAM (Braekevelt and Vanaken, 2017), aggregated into three sectors: agriculture, food industry and consumption & trade. We further hypothesized on the nature of the different streams of food residues, depending on their destination and their origin (Figure S5). Residues from the agricultural sector were assumed to be straws when used as feed or reintroduced into the soil, and food waste of vegetal origin (VGF) when ending up in the waste management system. By-products of the food industry are used in the manufacturing of animal feed; the rest of the streams were assumed to be food items (to AD and incineration) and straw-like material (to AG). Consumption and trade generate VGF and kitchen waste (food items).

Food industry is the biggest producer of food residues (Table S29) and most of it are used for the manufacturing of animal feeds. We used information from BEMEFA (2016) to identify the specific by-products that make up this stream and information from feedipedia (INRA et al., n.d.) for their nutrient content The composition of VGF and kitchen waste is from (Coppens et al., 2016) and the one from straw from feedipedia.. We converted input waste to output products (compost, digestate, incineration losses and ashes), based on the information listed in Table S30. For the flow of by-products from food to feed industry, the composition of the flow was combined with nutrient content data from feedipedia. The full dataset is shown in Table S31.

Table S 29 Amounts of food residues in the Flemish agri-food chain in 2015 and their valorization. Incineration includes also other energetic uses of waste and landfill includes also streams for which the final destination is unknown. [Source: (Braekevelt and Vanaken, 2017)]

Sector	Total Food Residue [t/yr]	Animal Feed	Soil	AD	Compost	Incineration	Landfill
Agriculture	464’579	12	69	4	4	1	10
Food Industry	**2’359’847**	55	11	26	-	7	-
Households	**660’681**	20	-	14	30	34	3
Total Chain	**3’485’157**	**43**	**17**	**21**	**6**	**6**	

Table S 30 Amounts of food residues in the Flemish agri-food chain in 2015 and their valorization. Incineration includes also other energetic uses of waste and landfill includes also streams for which the final destination is unknown. [Source: (Braekevelt and Vanaken, 2017)]

Product	Yield [% of input]	Dry Matter [%]	Organic matter [%]	N content [% FM]	P content [% FM]	K content [% FM]
Compost (VFG waste)(1)	50	70	25	1.2	0.26	0.83
Compost (GW)(1)	50	60	20	0.7	0.12	0.50
Digestate (raw)(2)	90(3)	8.4	5	0.45	0.13	0.42
Digestate (dried)(2)		82.5	50	1.76	1.25	2.32
Incineration ashes(4)	18	*100*	*0*	*0*	0.65	2.00
Compost (GW, WA)(5)	44(6)	60	30	1.4	0.22	0.66

(1) (Vlaco, n.d.), (2) (Vlaco, 2016b), (3) (Vlaco, 2012), (4) (Haupt et al., 2018) (5) (IDELUX, 2019), (6) (SPW, 2018),

Table S 31 Composition of the food waste streams in Flanders: food industry by-products, agricultural waste and household organic waste. Source for the nutrient compositions: (INRA et al., n.d.), except for the VGF and kitchen waste (Coppens et al., 2016).

Type of product	Product	Share in total by-products used in feed industry(1)	N content [% FM]	P content [% FM]	K content [% FM]
By-products of oil seeds	Linseed meal	48	4.9	0.8	1.1
Cereal products	Wheat	28	1.8	0.3	0.4
By-products from sugar manufacturing	Beet pulp molasses	11	1.7	0.0	3.9
Bakery & pasta products	Bread	3	1.3	0.01	0.14
By-products from biofuel production	Maize grain	2	1.3	0.3	0.3
Oils & fats	Rapeseed oil	2	0.0	0.0	0.0
High fibre products	Hay	2	0.7	0.1	0.0
Animal by-products	MBM	2	8.4	4.7	0.4
Brewery by-products	Brewers grain	2	3.8	0.5	0.3
AVERAGE		**100**	**3.4**	**0.6**	**1.1**
Straws (AG waste)			0.60	0.10	1.0
VGF waste(2)			0.63	0.11	0.3
Kitchen waste(2)			1.30	0.15	0.2

(1) (BEMEFA, 2016), (2) (Coppens et al., 2016) for N and P, (Fisgativa et al., 2016) for K

Solid Waste Management in Wallonia

Amounts on household organic waste generation in Wallonia (Table S32) were taken from the Regional Plan for Waste and Resources of 2018 (SPW, 2018). More than 40% of the mixed MSW is organic waste, of which 21% kitchen residues (unavoidable), 16% wasted food (avoidable) and 4% green waste (SPW 2018, initial source is (RDC Environnement, 2010)). The goal of the region is to reduce organic waste by 33% or 9 kg/cap.yr by 2025, compared to 2013 (SPW, 2018).

Table S 32 Generation of mixed residual and organic waste from Walloon households [Source: (REEW, 2018b; SPW, 2018)]

Source	Total MSW [kg/cap.yr]	Residual MSW [kg/cap.yr]	OFMSW [%]	Organic waste [kg/cap.yr]	Green waste [kg/cap.yr]
Plan Déchets-Ressources 2018(1)	528.9	148.1	41.4%	65.3(2)	62.85(3)

(1) The OFMSW includes diapers (10 kg/cap.yr) and non-recycled paper (11.5 kg/cap.yr). Values for 2013 (SPW, 2018)
(2) 12.63 kg/cap.yr separately collected (extra). Doesn't include nappies (10kg/cap.yr) and non-recyclable carton (11.5kg/cap.yr)
(3) 6.3 kg/cap.yr in the residual waste, the rest separately collected

Green waste in Wallonia is composted in 8 centralized composting plants (SPW, 2018), and the produced compost is valorised in agriculture (66%) and in landscaping (6%). A further 9% is sold directly to individuals and less than 0.5% to the service sector. The remaining 19% is reintroduced in the WM system, either to be used in the composting process or to be incinerated.

Data on the quantities and management of organic waste from the food industry for the year 2015 are from the industry survey (ICEDD, 2017). Nutrient contents for all materials are the same as for Flanders (Table S30).

Wastewater generation and management

The amounts of N and P in the wastewater and sewage sludge in Flanders can be estimated using data on the theoretical loads and the removal efficiencies for the pollutants from Aquafin (2014) (Table S33). Lacking local data, we used the literature value of 3.6 gK/cap·d for the K load (Larsen and Maurer, 2011). 64% of the P in the incoming wastewater leaves the WWTP with raw sludge, whereas another 30% is in the stream that is digested (Coppens et al., 2016). We assume that all P from the incoming wastewater ends up in the sludge. To estimate the nitrogen concentrations of the sludge streams, we use N:P ratios derived from (Coppens et al., 2016). The ratios are 1.26 and 1.42 for digested and raw sludge. In Flanders, 80% of the treated sludge are incinerated or landfilled and 20% end up in an 'unknown' destination (Coppens et al., 2016). In Wallonia only 49% are incinerated while the rest 50% is reused in agriculture (REEW, 2019f).

Table S 33 Characteristics of wastewater and sewage sludge in Flanders. The total loads are calculated based on a population of 6.41Mi people in Flanders in 2014 and a 81% connection rate to treatment plants (Aquafin, 2014).

Product	COD	TSS	N	P
Theoretical load [g/cap.d]	135	90	10.0	2
Removal efficiency at the WWT [%]	91%	96%	83%	84%
Total produced in Flanders [t/yr]			23'397	4'679
Total in ww to treatment plants [t/yr]			18'856	3'717

Chapter 4

The potential of reused nutrients to cover crop needs in dense livestock dominated regions[4]

4.1 Introduction

Nutrients, such as nitrogen (N), phosphorus (P) and potassium (K) are necessary inputs into the cultivation of food and feed crops. The excess use of synthetic fertilizers, however, has been accumulating in soils and led to increased nutrient runoff and leaching from agricultural lands into water bodies. Increased nutrient concentration in these water bodies causes eutrophication, algal blooms and hypoxia (Smil, 2002; Yuan et al., 2018). Besides, the production of synthetic fertilizer is an energy-intensive and wasteful process (Withers et al., 2014), and one that at times relies on finite raw materials, such as phosphate rock for the production of phosphate fertilizers (Cordell et al., 2009; Yuan et al., 2018).

Replacing synthetic fertilizers with secondary nutrient resources, such as manure and human excreta, could alleviate some of the drawbacks of excess fertilizer use. Manure and human excreta are renewable sources of nutrients that can be sourced locally in all places

[4] This chapter is based on the following article currently under review: **Papangelou A.,** Mathijs E. (2021) The potential of reused nutrients to cover crop needs in dense livestock-dominated regions. *under review*

where animals and humans live. Despite the commonly held view that nutrients from organic fertilizers are not as readily available to plants as those in mineral fertilizers (Wu et al., 2019), recent research shows that P in manure and treated sewage sludge is more recyclable than P in mineral fertilizer (Kahiluoto et al., 2015). Additionally, secondary nutrient sources can improve soil fertility and structure, thanks to their organic matter content (Withers et al., 2014).

The reuse potential of the secondary nutrient sources is bound to place and the specific local conditions. Manure use is often limited by the need for transportation (Schneider et al., 2019), as livestock and crop production are separated in today's specialized agricultural systems (Bateman et al., 2011; Kahiluoto et al., 2015; Powers et al., 2019). Studies have found that manure and human excreta could cover the crop needs for nutrients country-wide in Sweden (Akram et al., 2019), and Norway (Hamilton et al., 2017; Hanserud et al., 2016), and "feed the corn belt" in the USA (Metson et al., 2016), if only supply and demand for these nutrients were more uniformly distributed geographically (Withers et al., 2014). Human excreta reuse is also limited by the need for transportation to the food production sites (Powers et al., 2019; Withers et al., 2014). However, as technologies for resource recovery from human excreta advance, the major barrier that emerges to the widespread reuse of nutrients from urban effluents is the lock-in in the current urban water management paradigm in infrastructure, legislation, and public opinion (Schneider et al., 2019; Withers et al., 2014). The prominence of the circular economy is already shifting some of these lock-ins. Rethinking human excreta as part of the food system (Harder et al., 2020) and as human-derived resources (Trimmer et al., 2020) could be key in the challenge of feeding a growing urban population with less impact (Withers et al., 2019).

Reusing nutrients from human excreta has thus been attracting renewed attention, as a way to make cities more circular, more productive and more food self-sufficient (Wielemaker et al., 2019, 2018). As the land available for food production within the cities, however, is limited, these nutrients will have to be transported to their peri-urban and rural hinterlands, where they are needed by the crops grown there (Trimmer and Guest, 2018). In large countries with distinct sites of crop and livestock production, like Sweden and Norway, a gap exists in supply and demand for nutrients, that can be covered by human excreta (Akram et al., 2019; Hanserud et al., 2016). What about small, densely populated and intensely cultivated countries like the Netherlands and Belgium? Withers et al. (2019) report that Western European countries with these characteristics have the highest soil P surpluses in Europe. In these areas, human excreta may compete with manure for access to agricultural land (Wielemaker et al., 2020), because manure is abundant and transport distances small. However, there is no study to the best of our knowledge, that jointly assesses the crop nutrient needs and the nutrient supply through secondary, local, renewable sources (manure, human excreta) in such regions.

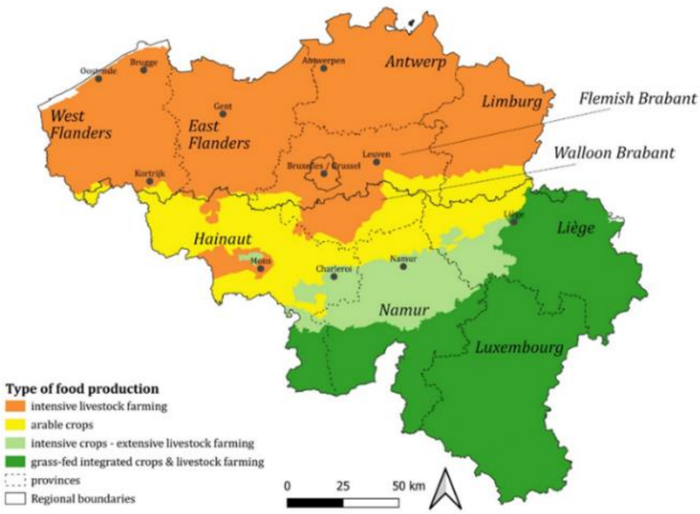


Figure 4.1 Map of Belgium with main cities (dots), provinces (dotted lines) and their names (in italics) and the dominant food production typology

In this chapter we carry out such a joint assessment of nutrient supply and demand at sub-national (regional and provincial, Figure 4.1) scales for the federal state of Belgium. Belgium is made up of three regions: Flanders in the north, Wallonia in the south and the city-region of Brussels Capital Region (BCR) in the center. The country is characterized by high urbanization, an open economy, and intense livestock production, especially in Flanders where most of the pork and poultry production is based (Statbel, 2019). Due to this intensive agricultural activity, and its location downstream several important European rivers, nutrients have been accumulating in Belgium's soils and water bodies (REEW, 2019a, 2019b; VMM Milieurapport Vlaanderen, 2012). The regions have been facing these problems in various ways. Flanders, for example, has been implementing a strict manure management plan (VLM, 2019), and Wallonia has closed its borders to the import of materials characterized as waste. Nonetheless, as new priorities arise, including ambitions towards a transition to a circular paradigm across the country , and new legislation takes force, such as the new European guidelines for organic fertilizers (EC, 2018), past approaches will need to be revised.

The aim of this study is to contribute towards a rethinking of nutrient management across the whole country with increased circularity as the primary focus. To do so, we quantify (i) the crop needs (demand) for the three majors nutrients, nitrogen (N), phosphorus (P), and potassium (K) across the country, as well as (ii) the nutrient supply through secondary, local, and renewable resources, i.e. manure and human excreta. Then, (iii) we compare the supply and demand of the nutrients, assessing the potential of manure and urban effluents to cover crop needs and (iv) we discuss possible ways towards that direction at different scales. We find that despite the abundance of manure, there is still potential for human excreta to re-enter the food and farming systems, especially in the south and around urban centers.

4.2 Methodology

We estimated the annual supply, demand and actual application rates for the three main, from an agronomic perspective, nutrients: nitrogen (N), phosphorus (P) and potassium (K). We used 2014 as the reference year, as the most recent year for, or around which, all data are available. We assumed 2014 to represent the current situation well enough (Papangelou and Mathijs, 2021b). We define supply as the amount of nutrients in local, secondary resources, namely manure and human excreta. We excluded organic waste streams, because they typically represent only a fraction of the nutrients in human excreta (Papangelou et al., 2020; Wu et al., 2016). We estimated the demand for nutrients as the theoretical crop needs in N, P, and K, using local fertilizing recommendations. We finally estimated the actual nutrient application through synthetic fertilizers and manure to check where current practices lie between satisfying crop needs and using up local secondary resources. We estimated nutrient supply, demand and application for Flanders and Wallonia, whereas we only accounted for supply through human excreta for Brussels Capital Region, as the number of animals and extent of agricultural land in Brussels is negligible compared to the two other regions (Statbel, 2015a). To capture local differences to the extent possible, while keeping the results relevant for policy making, we worked on the municipality level and aggregated the results to higher administrative levels (province, region, country), where needed for interpretation. We used QGIS to create the maps and process geographical information (QGIS.org, 2021) and R for the rest of the analysis (R Core Team, 2020). Table 4.1 provides an overview of the main calculation steps and sources used, which are elaborated in the following sections.

Table 4.1 Overview of the different data sources and approaches used to estimate the supply, demand and actual application of N and P in the agricultural lands of Flanders and Wallonia.

	Flanders			Wallonia		
	manure	synthetic fertilizer	human excreta	manure	synthetic fertilizer	human excreta
Supply	No of animals[1] x excretion rates per animal[2]		Population x daily nutrient intakes[3]	No of animals[1] x excretion rates per animal[2]	-	Population x daily nutrient intakes[3]
Demand	Fertilization recommendations per crop & agricultural region from the Belgian Soil Service[4]			N : Walloon recommendations based on the Nitrate Directive[5] P : Flemish recommendations[8] for soil type III		
Application	Annual Manure Report (N, P), assumption for N:K in manure	Flemish Agricultural Statistics[6]	-	Walloon State of the Environment Report[7]; assumptions for N:K and N:P	Walloon State of the Environment Report[7]	-

[1](Statbel, 2015b), [2](Papangelou and Mathijs, 2021b), [3](Nubel, 2018; Ridder, 2016), [4](Tits et al., 2016), [5](Gouvernement Wallon, 2014), [6](Departement Landbouw en Visserij, 2020), [7](REEW, 2018a), [8](VLM, 2015b)

4.2.1 Supply: nutrients available in manure and human excreta

We estimated the amounts of nutrients in manure by multiplying numbers of animals from official statistics (Statbel, 2015b) with their excretion rates (Papangelou and Mathijs, 2021b). We accounted for nutrient losses during storage and handling of manure using factors on N-losses from the Flemish manure monitoring reports (VLM, 2015a), but otherwise assumed that all nutrients excreted are potentially available to crops. For human excreta, we used previous estimations of nutrient intakes (Papangelou et al., 2020), based on the amounts of different foods consumed (Ridder, 2016) and their nutrient content (Nubel, 2018). Each inhabitant of Belgium consumed on average 11.8 g/cap·d of N, 1.2 g/cap·d of P, and 2.6 g/cap·d of K, 99% of which, were assumed to end up in their excreta (Esculier et al., 2018). In reality, considerable losses will occur, especially losses of nitrogen gas during the treatment of wastewater or excreta. Therefore, the actual amount of nitrogen that will be available to crops will be lower than what we define here as supply.

4.2.2 Demand: nutrient crop needs

For Flanders, we used the most recent fertilization recommendations of the Belgian Soil Survey (Tits et al., 2016). The recommendations are given per crop and agricultural region, and are based on the Survey's periodic sampling campaigns that monitor several soil properties, including nutrient concentrations. Agricultural regions are a categorization of the Belgian land into regions with similar soil properties and agricultural patterns. We disaggregated data from the agricultural region to the municipality level in QGIS (QGIS.org, 2021), merging the map of the Belgian agricultural regions (Departement Landbouw en Visserij, 2013) once with the Flemish map of the administrative boundaries (Informatie Vlaanderen, 2016) and then with the respective Walloon one (SPW, 2017).

No similar fertilizing recommendations exist to our knowledge in Wallonia. Instead, we used the maximum allowable values based on the Nitrate Directive for N (Gouvernement Wallon, 2005, Annex XV), and the maximum recommended values in Flanders for P (VLM, 2015b). The latter provide maximum allowable P-rates based on a classification of the parcels in 4 categories, reflecting the sensitivity of the soil to nutrient accumulation and leaching. We used the recommendation for category III (Table S36 inAnnex), which is the guideline for Flanders in 2015. We further checked the difference in the results, if we had assumed soil type II or IV (see Results). We have opted for these rougher, policy-based recommendations for Wallonia, rather than the ones from the Soil Service (Tits et al., 2016), to avoid introducing too many uncertain assumptions on the soil characteristics in each agricultural region.

4.2.3 Application: actual nutrient application rates

Actual manure application rates at the regional and provincial level are based on official data for Flanders (VLM, 2015a, 2014) and Wallonia (REEW, 2018a), and assumptions for K:N ratios in manure based on analyses of manure nutrient content. We further used regional statistics and reports on fertilization per crop for Flanders (Departement Landbouw en Visserij, 2020; Lenders and Deuninck, 2016), whereas no equivalent data are available for Wallonia, so we used one uniform fertilization rate per nutrient for the whole region (seeAnnex), using the 7-year averages around 2014 from (REEW, 2018a).

4.2.4 Data uncertainties and disparities

All data in this study are sourced from official statistics, studies, and reports. The analysis consists mostly of combining these data, and (dis-)aggregating them into different levels of detail, e.g. per crop or for the total agricultural land, and spatial scales, e.g. municipality, province, regional scale. The analysis sometimes required the adoption of assumptions to fill data gaps: we assumed, for example that similar crops (grain and fodder maize, sugar and fodder beets) have the same fertilization needs, or that tomatoes represent well enough all vegetables grown in greenhouses. All such assumptions were based on actual data, often from higher spatial scales or neighboring regions, so, even though they come with inherent uncertainty, we do not expect them to jeopardize the reliability of our results.

It is further worth noting that there is an imbalance in the analysis between Flanders and Wallonia, regarding the detail in application rates and crop needs. Application rates in Wallonia were only available as one average value per nutrient for the whole region, while Flemish values were given per crop. In addition, although detailed guidelines are available to calculate phosphorus needs in Wallonia at the plot level (Genot et al., 2011), no guidelines for the macroscale exist, like the ones for Flanders (Tits et al., 2016). We thus decided to adopt the guidelines that the Flemish Land Agency issues annually and that contain recommended application rates for P for four different types of soil (VLM, 2015b). We assumed soil type III for the whole Wallonia, because that was the guideline for the default assumption in Flanders, and tested the effect of assuming a different soil type on the final balance for Wallonia (see Figure S6 inAnnex). Because no similar alternative was available for K, we skipped the estimation of K needs in Wallonia altogether.

4.3 Results

4.3.1 Supply, demand and application at the national and regional level

The crop needs in nutrients at the country level are smaller than the local supply for phosphorus and potassium, but greater for nitrogen (Figure 4.2). The crops requirements in 2014 were 239 kgN/ha, while the supply through manure and human excreta only 167 kgN/ha. The difference is mostly due to the gap between supply and demand in Wallonia: the N demand in the region mounts up to 268 kgN/ha, while supply only reaches 103 kgN/ha, with roughly 21 kgN/ha being of human origin and the rest of cattle manure. In contrast, nitrogen demand in Flanders (204 kgN/ha) is smaller than the supply (244 kgN/ha), thanks to a smaller grassland area and the abundance of pig manure. Phosphorus shows an opposite trend at the national scale: 26 kgP/ha were needed and 34 kgP/ha supplied in the whole country in 2014. Whereas Wallonia is in deficit for P, supply in Flanders (52 kgP/ha) is almost 2.5 times higher than demand (22 kgP/ha), defining the country’s positive budget. Similarly, potassium supply in

Flanders by far exceeds crop needs (323 kgK/ha and 84 kgK/ha respectively). Finally, actual application rates of nutrients in Belgium are higher than the crop needs by 6% and 18% for N and P (grey bar in Figure 4.2). 58% of the nitrogen and 84% of the phosphorus applied onto the land are through manure and the rest through synthetic fertilizers.

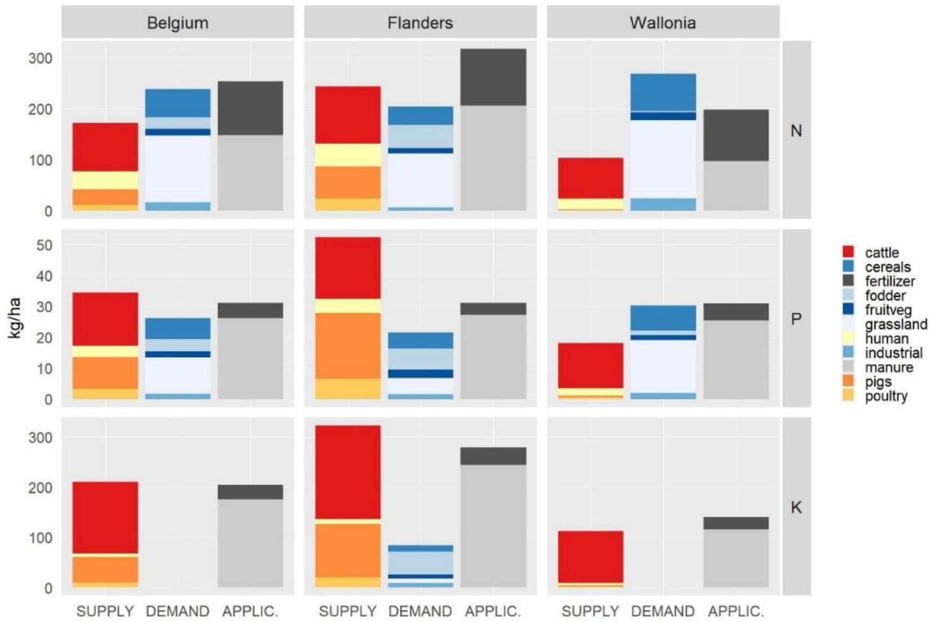


Figure 4.2 Total supply, demand and actual application of N, P, and K in the whole of Belgium (BE), and the two regions of Flanders (FL) and Wallonia (WA) – all values in kg/ha for the reference year 2014.

4.3.2 Supply and demand at the provincial and municipality level

We can distinguish two main trends regarding nutrients supply and demand in the country: the first is the clear distinction between the nutrient deficient Walloon region in the South versus the oversupplied Flanders in the North. The second trend is a greater affluence in P, relatively to crop needs, as compared to N. The same trends persist if we disaggregate the results further at the level of provinces. Nitrogen demand is greater than the local supply in all Walloon provinces and only slightly smaller than supply in the Flemish ones, except for the Flemish Brabant in the center of the country (Figure 4.3a). Phosphorus demand also lags behind supply in Wallonia and exceeds it in Flanders, yet the difference is smaller than the one for nitrogen (Figure 4.3b): P supply exceeds demand by 129%, on average, in Flemish provinces and the difference reaches up to 212% and 251% in West Flanders and Antwerp. The average difference for N is only 15% for Flanders.

The gap in the supplied and needed nutrients are a direct outcome of the type of dominant agricultural activity in each province: Antwerp and West Flanders, for example, accommodate most of the region's intensive livestock production and thus concentrate high amounts of P-rich manure. The south, on the other hand, is more scarcely populated by people and animals and dominated by grasslands that need higher amounts of nutrients. At the same time, human excreta contributes on average 23% and 13% to the total supply of local secondary nitrogen and phosphorus across the country (yellow slices in pies, Figure 4.3), making it a secondary nutrient resource comparable to manure, and one that merits special attention towards closing the food and nutrient loops.

The importance of human excreta as a secondary nutrient source is also visible in Figure 4.4: the highest balances, i.e. differences between supply and demand, are observed in municipalities that lie in and around cities. Urban centers show some of the highest surpluses, e.g. the cities of Antwerp (21254 kgN/ha, 233 kgP/ha, 550 kgK/ha), Liège (941 kgN/ha, 102 kgP/ha) and Charleroi (992 kgN/ha, 105 kgP/ha). Other hotspots are the municipalities in the pig-rearing area in West Flanders and the cattle-rearing area in the border with the Netherlands, where balances range from 200-500 kgN/ha, 80-160 kgP/ha, and 580-900 kgK/ha. In contrast, the regions most lacking are the pastures in the South and South-East Wallonia, as well as parts of the cereal-producing zone in the center of the country, where nutrient deficits range roughly between -250 to -200 kgN/ha, and -20 to -15 kgP/ha.

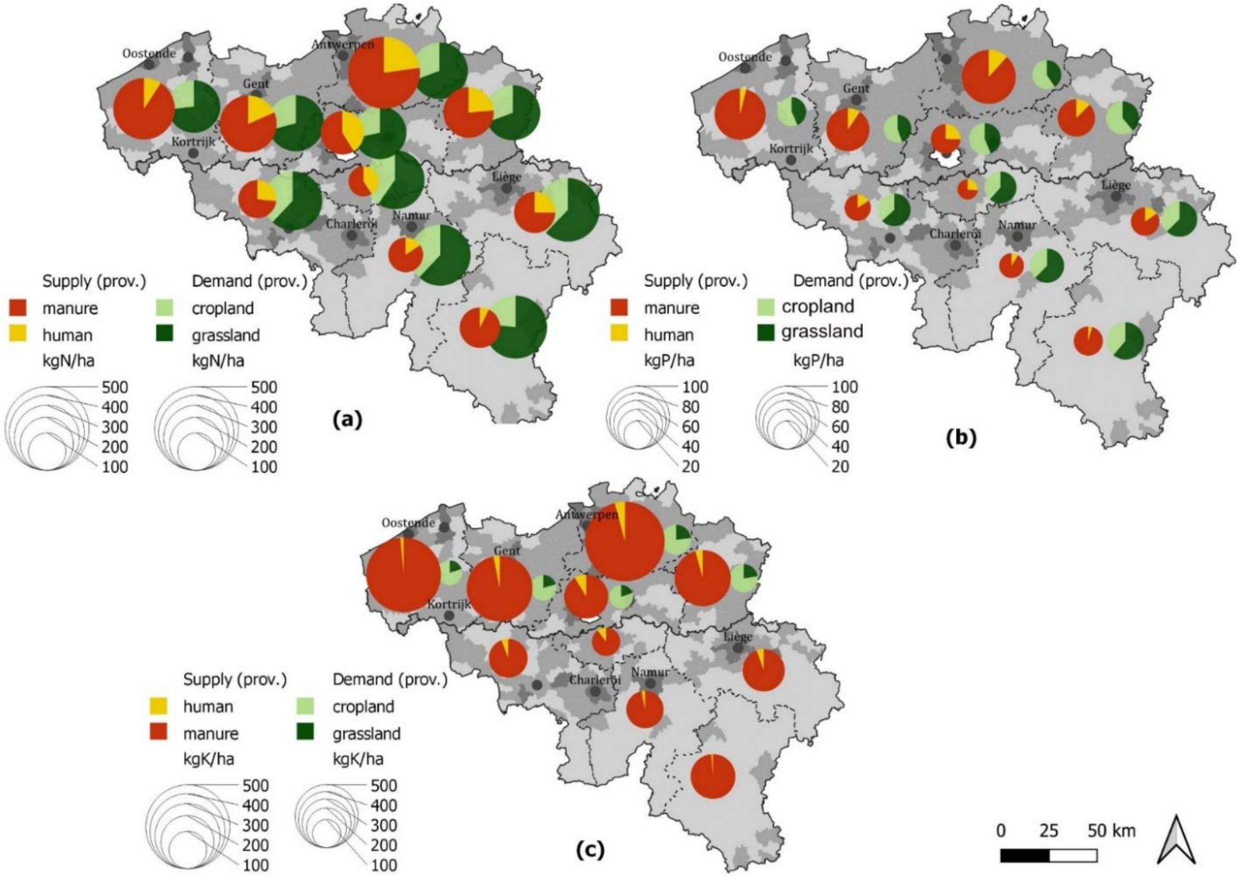
4.4 Discussion

4.4.1 Mismatch of supply and demand

A mismatch between the availability of secondary nutrient resources, such as manure and human excreta, and the demand for these nutrients, i.e. the crop requirements in nutrient inputs, has been observed in several countries. In Sweden , for example, municipalities have average deficits of -19 kgN/ha and -2 kgP/ha (Akram et al., 2019), meaning that at the national level all crop needs could theoretically be covered by secondary nutrients. Similarly, up to 71% of the 113.7 ktP needed by crops in England could be covered by manure only (Bateman et al., 2011), and the total P demand of 1'933 ktP in the American Corn belt could be met by just 37% of the P excreted by animals and humans (Geneviève S. Metson et al., 2016). All these studies point out how secondary nutrients are underutilized, due to the need to transport them from the hotspots where they are generated to the places where they are needed.

A similar mismatch in the supply and demand for nutrients is observed at the municipality and regional level in Belgium, due to the division of the land and specialization of agriculture that has created the gap between the places where nutrients are produced (cities, intensive livestock rearing zones) with the places where they are mostly needed (in rural areas and pasturelands). Since this gap has a clear North-South division, excess nutrients from the North, especially P, could be transported and used to cover the crops' needs in the South. This arrangement would improve the overall national nutrient balance, it is, however impossible under the current legislation.

Figure 4.3 (next page) Supply and demand of N (a), P (b), and K (c) in kg/ha per province in Belgium. The pies show the distribution of the supply into manure and human excreta and of demand into cropland and grassland. The size of the pie diagrams are proportionate in each map, but not across all three. The base layer of grey shades shows the degree of urbanization (SPF, n.d.)



Oostende
Gent
Antwerpen
Kortrijk
Liège
Namur
Charleroi
Supply (prov.)
manure
human
kgN/ha
500
400
300
200
100
Demand (prov.)
cropland
grassland
kgN/ha
500
400
300
200
100
(a)
Supply (prov.)
manure
human
kgP/ha
100
80
60
40
20
Demand (prov.)
cropland
grassland
kgP/ha
100
80
60
40
20
(b)
Supply (prov.)
human
manure
kgK/ha
500
400
300
200
100
Demand (prov.)
cropland
grassland
kgK/ha
500
400
300
200
100
(c)
0 25 50 km

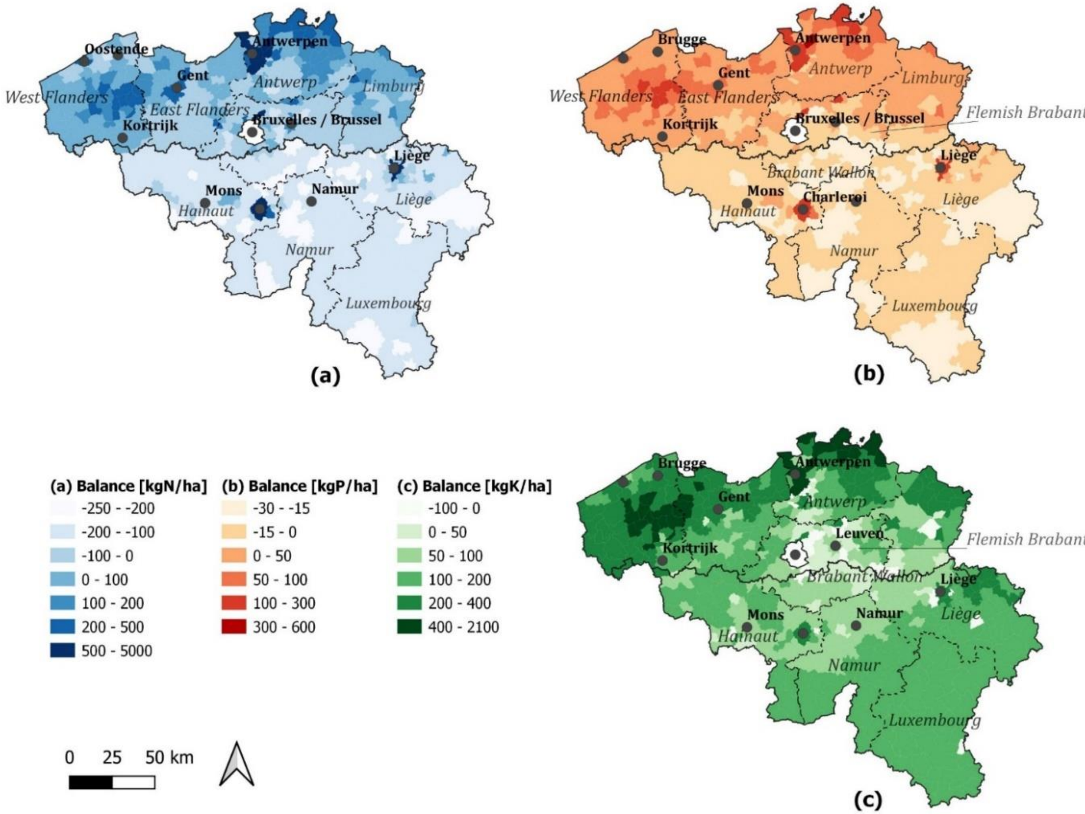


Figure 4.4 Nutrient balances (supply through manure and human excreta minus crops' needs) for N (a), P (b), and K (c) in Belgian municipalities

4.4.2 Interregional transfers: Flemish manure to Walloon agriculture?

Virtually no manure is currently exchanged between Flanders and Wallonia. The manure that is not used in Flanders is currently exported, usually after processing. In 2014 69% and 77% of the exported N and P were directed towards France, and 26% and 18% of N and P towards the Netherlands (VLM, 2015a). Only a 0.16% of the exported N and 0.12% of P reached Wallonia, corresponding to 45 tN and 13 tP (Figure 4.5). These amounts represent roughly 0.4% and 0.2% of the N and P that we estimated to be the imported nutrients in Wallonia, as the difference between applied and supplied manure. It is however not clear where the manure is imported from.

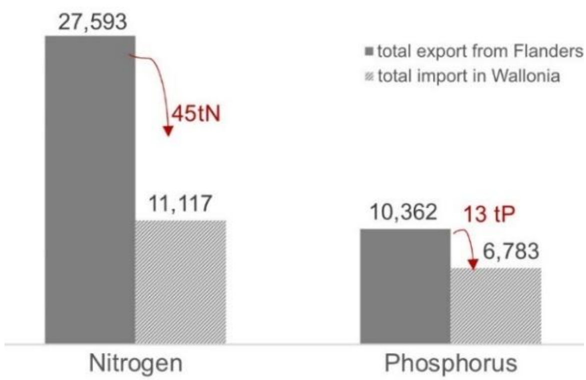


Figure 4.5 Comparison of Flemish exports with Walloon imports of nutrients in manure for 2014 in tons.

The main reason of the mismatch between supply and demand in the two regions is the legal barrier in transporting manure from the nutrient-rich Flanders to nutrient-deficit Wallonia. Imports of what are considered to be waste are strictly regulated in Wallonia (SPW, 2018), and only limited transfers in farms that cross the regional border are allowed, despite the apparent nutrient deficits in the region. This may change after the new European Fertilizer Regulation comes into effect in 2022 (EC, 2019), aiming to facilitate the trade of bio-based fertilizers in the European Union.

Next to regulatory barriers, further agronomic and environmental concerns could hinder manure transport. For example, surface and groundwater bodies in Wallonia are vulnerable to nitrate, ammoniac and orthophosphate pollution (REEW, 2019c, 2019g) and 60% of the agricultural lands in the region lie within zones vulnerable to nutrient pollution (REEW, 2019d). In addition, elevated nutrient concentrations in soils will result in a lower demand for additional nutrients to be supplied through manure application and so a larger gap between supply and demand. For example, the annual fertilization guidelines that the Flemish government publishes every year (VLM, 2019b, 2015b) are determined from a legal point of view, taking into account crop needs but not the fertility status of the soil. Based on these guidelines, the Flemish Manure Bank estimated maximum allowable amounts of 116.2 ktN and 23.3 ktP (VLM, 2015a) in 2014 (Figure 4.6). The amount for nitrogen is lower than both the actual application of N through manure and synthetic fertilizers in the same year, and the crop needs in N as we calculated them in this study, based on (Tits et al., 2016). To the contrary, our estimation of the P demand is almost half of the maximum allowable 23.3 ktP (Figure 4.6), indicating that the guidelines will need to become stricter if further accumulation of P in the Flemish soil is to be avoided.

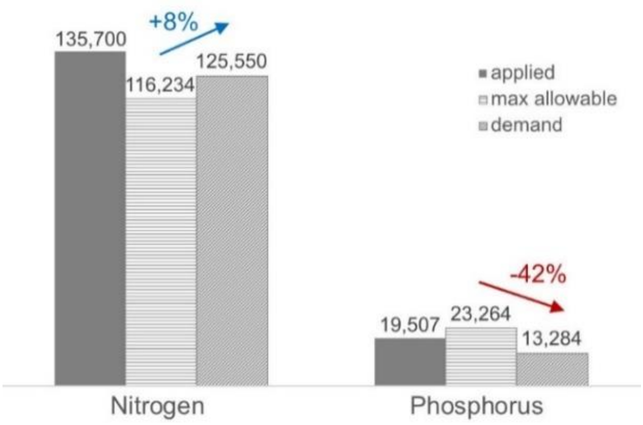


Figure 4.6 Total applied and maximum allowable N and P in Flanders for 2014 in tonnes (VLM, 2015a) and comparison with the demand as estimated in this study with data from the Belgian Soil Service (Tits et al., 2016)

4.4.3 Human excreta as a nutrient source

In an intensive, export-oriented, and livestock-dominated agro-food system like the one in Belgium the "manure issue" is debated and heavily regulated, especially in Flanders. Nonetheless, we showed how human excreta also represents an important source of nutrients, containing 19% of all N and 9.7% of all P available in local secondary sources at the country level. Yet it is a resource largely underutilized. Around 50% of sewage sludge generated in municipal wastewater treatment plans in Wallonia is stabilized with lime and applied to agricultural fields (REEW, 2019f). In Flanders, though, sewage sludge reuse is strictly regulated to the degree that it is virtually illegal. These strict limits for heavy metal emissions reflect the historically high levels of heavy metal concentrations in the region, mostly due to industrial effluents, landfill leaching and transboundary pollution (De Temmerman et al., 2003; Fierens et al., 2016; Vandecasteele et al., 2002). As a result, sludge is mostly incinerated and so hardly any of the nutrients in human excreta are actually reused (Coppens et al., 2016). However, as industrial and municipal waters are increasingly treated separately, treatment technologies advance, and new topics such as the Circular Economy emerge, legislation may need a make-over, a trend already observed in Europe with the upcoming revision of the Sewage Sludge Directive. Opportunities and ways to avoid and reuse food waste and other solid organic wastes are addressed in several official publications in Flanders (Braekevelt and Vanaken, 2017; OVAM, 2020; VMM, 2019), Brussels (IBGE, 2018b), and Wallonia (SPW, 2018). Yet municipal solid bio-waste, such as food and garden waste, contain just a fraction of the nutrients in human excreta. In Flanders 15.5 ktN and 1.5 ktP were contained in all solid bio-waste generated in 2009 (Coppens et al., 2016); the respective values in human excreta were 21.1 ktN and 3.5 ktP. Similarly in Brussels 0.16 ktP and 0.63 ktP were produced in 2014 through solid waste and human excreta respectively (Papangelou et al., 2020).

Re-introducing human excreta into the food system avoids the wastage of a constant, local, secondary source of nutrients (Harder et al., 2020; van der Kooij et al., 2020), and the additional need for synthetic fertilizer that are environmentally costly to produce. To achieve the re-introduction in an efficient and safe way, new technologies are being developed, which can recover nutrients from waste streams and deliver them in a form that is practical to transport and to use in agriculture, like nutrient solutions, precipitates or treated ashes and slags (Harder et al., 2019). Digested or composted sewage sludge can be valuable agricultural inputs, since the plant availability of their nutrient content is mostly independent of soil properties, unlike most of the recovered products mentioned above (Trimmer et al., 2019). In addition, digestates and compost are also rich in organic matter that can improve soil health and productivity (Harder et al., 2019; Toffey and Brown, 2020). High heavy metal concentrations in urban effluents are often hindering their reuse; however, in cities where municipal and industrial wastewaters are not mixed, heavy metal concentrations are likely to be low. This is the case in Brussels (Papangelou et al., 2020).

Developing and strengthening urban and peri-urban agriculture, while taking into account the availability of secondary nutrients when deciding what crops to grow, can therefore be a win-win solution. Nutrient demanding and high value crops, such as vegetables, are ideal. In the case of Belgium we found that the two provinces surrounding Brussels, Flemish Brabant and Walloon Brabant, are nutrient-deficient. Reusing effluents from Brussels in these two provinces would be an example of a tight loop of nutrient flows at the peri-urban scale. Analyses at such regional and peri-urban scales that combine material flows and environmental concerns with considerations of justice and the preferences of stakeholders' will greatly facilitate the way towards more sustainable, circular and just urban and regional food systems.

4.5 Annex

4.5.1 Additional details on methodology and input data

Table S34 Fertilizing recommendations per crop and type of soil for nitrogen (kg N/ha·yr), phosphorus (kg P_2O_5/ha·yr) and potassium (kg K_2O/ha·yr). Average data from (Tits et al., 2016). When there is no recommendation in an agricultural region, but the crop is nonetheless cultivated in this region, we used the recommendation for the next agricultural region.

	Dunes-Polders			Sandy soil			Kempen			Sand-clay soil			Clay soil		
	N	P	K	N	P	K	N	P	K	N	P	K	N	P	K
winter wheat	188	32	31	154	35	47	136	44	52	153	51	40	150	60	23
winter barley (fodder)	147	42	49	127	46	66	123	49	73	129	73	61	125	81	63
winter triticale				127	25	56	110	35	62						
sugar beets	172	49	228	160	62	256	167	51	269	156	73	254	154	86	249
fodder beets				181	43	263	160	52	277	173	75	254	58	81	215
maize	173	46	106	173	45	107	173	56	116	159	70	105	152	88	100
fodder maize	179	47	162	183	43	200	177	57	214	164	71	185	156	89	180
potatoes	215	65	250	207	57	260	189	67	290	204	72	248	204	105	245
early potatoes				155	42	260				152	49	247			
fried potatoes				180	49	214	175	72	243	185	76	221			
chicory				20	41	129				13	57	124			
fiber flax				24	84	107				25	53	106			

cauliflower				206	33	225				215	41	233			
courgettes										192	41	85			
peas							37	58	146	27	65	104	30	79	100
celeriac root										205	31	226			
onion										98	78	229			
leek				214	26	206				215	30	205			
red cabbage				168	41	163									
spinach				171	30	250				172	44	246			
Brussels sprouts				197	64	170				199	63	167			
green beans				73	49	107				78	58	104	78	76	96
white cabbage				215	33	162									
carrots				97	44	253				99	60	248	102	104	259
p. grassland, graz.	354	36	28	365	33	44	360	40	59	363	49	21	371	55	24
p. grassland, mow.				216	56	234	215	62	238	209	79	219			
grass-clover, mowing				138	48	243	136	63	227	139	74	210			
p. grassland, standw.	318	34	29	332	26	39	334	34	59	338	49	27	329	50	15
horse grassland	143	29	47				142	40	64	138	47	29			
rye-grass				128	20	119	130	35	119	117	41	102			
apples, low-stem				79	25	43	84	20	63	61	34	34	59	43	21
grapes, open-air				137	82	137				87	98	117			
cherries										52	47	43	50	57	34
pears				124	13	72				118	29	44	116	34	36
tree nurseries				159	46	119	161	58	135						

Table S35 Aggregated data for the nutrient needs of the food group "fruit and vegetables" for the agricultural regions in Flanders. The weighted averages are based on the surface area each specific crop occupies to the total area for fruit & vegetables (Statbel,

		Polders	Sandy soil	Kempen	Sandy-clay	Clay soil
Fruit & vegetables grown outdoors	N	61	160	130	151	63
	P	41	35	49	49	62
	K	100	83	57	174	52
Fruit & vegetables grown indoors	N	169	154	121	170	169
	P	44	38	26	44	47
	K	116	101	54	117	115

Table S36 Maximum applicable amounts of mineral, organic and total nitrogen according to crop, in kgN/ha·yr and kgP2O5/ha·yr [Source: (Gouvernement Wallon, 2005), values in italics from protect'eau, P-values from (VLM, 2015b)]

	mineral N[(1)]	organic N	mineral N	total N	P_2O_5
Beet	180	210	120	330	55
Maize	180	210	120	300	70
Colza	225	185	145	330	55
Potatoes	225	185	145	330	70
Winter wheat	170	70	150	280	70
Spring cereals	100	70	80	200	70
Winter barley	170	70	160	280	70
Chicory	30	70	0	120	55
Flax	60	70	50	80	55
Beans	50	70	40	80	55
Peas	30	70	0	30	55
Spinach				200	55
Fava beans				60	55
Carrots				120	55
Brussels sprouts				180	55
Natural meadow	0	70	0		90
Pasture		200	150	350	90
on grassland		*230*		*350*	90
on cropland		*230*		*250*	55
in vulnerable zones				*170*	

(1) when only mineral N-fertilizers are used (no manure)

Table S37 Application rates for synthetic fertilizer nutrients in Flanders, per type of crop. Values in italics represent missing values filled in with the average for Flanders, except for tomatoes, for which the same values as for leek was used [Source: (Departement Landbouw en Visserij, 2020) for N & P, (Lenders and Deuninck, 2016) for K].

crop	Nitrogen [kgN/ha.year]			Phosphorus [kgP/ha.year]			Potassium [kgK/ha.year]	
	2013	2014	2015	2013	2014	2015	2013	2014
permanent grassland	128.0	138.5	126.4	1.3	1.2	1.0	12.0	13.0
temporary grassland	73.0	75.6	84.3	0.4	0.2	0.3	6.0	8.0
clover grass	126.7	116.7	105.0	1.2	0.7	1.1	9.0	6.0
wheat	176.4	177.6	180.5	3.5	5.9	5.0	6.0	8.0
barley	142.5	134.0	147.6	6.3	5.9	8.6	10.0	9.0
triticale	68.5	101.2	102.9	1.0	2.0	3.9	60.0	25.0
grain maize	56.1	58.5	52.2	3.2	3.4	3.2	33.0	36.0
fodder maize	55.0	57.6	53.5	3.6	3.6	4.1	26.0	28.0
potatoes	135.5	137.1	132.5	5.0	4.2	5.9	139.0	138.0
new potatoes	135.7	144.4	126.5	4.9	5.1	6.8	113.0	135.0
fodder beet	89.7	78.7	84.2	3.8	3.6	2.8	51.0	44.0
sugar beet	95.7	94.3	91.2	2.9	3.3	3.5	81.0	82.0
pears	54.6	67.7	88.4	3.9	6.0	6.3	35.0	45.0
apples	56.5	64.3	72.9	4.2	6.4	4.5	23.0	19.0
strawberries open air	129.1	118.1	146.3	29.4	23.9	27.8	*34.0*	*34.0*
strawberries greenhouse	189.8	146.9	174.7	58.2	55.1	50.3	*34.0*	*34.0*
tomatoes	1560.2	1547.0	1066.8	311.5	307.7	235.1	*102.0*	*95.0*
leek	116.8	160.2	140.4	3.6	3.0	3.3	102.0	95.0
other crops	44.2	41.1	33.4	3.1	3.6	3.0	*34.0*	*34.0*
Flemish average	106.9	109.6	109.8	3.4	3.7	3.6	34.0	34.0

Table S38 Manure application rates per province for 2014. Rates for N and P are calculated from the total amounts of nutrients applied in Flanders (VLM, 2015a)**, disaggregated into provinces assuming the same distribution in provinces as for 2013 (VLM, 2014). To estimate K application we assumed an average K:N ratio of 1.19 based on manure composition data from (Braekman et al., 2014)**

province	N [kgN/ha]	P [kgP/ha]	K [kgK/ha]
Antwerp	188.4	35.0	223.5
Limburg	130.5	25.2	154.8
East Flanders	115.6	22.0	137.19
Flemish Brabant	69.9	31.2	82.9
West Flanders	370.2	26.4	439.2
FLANDERS	**152.7**	**30.2**	**181.1**

Table S39 Nutrient application rates in Wallonia through manure and synthetic fertilizers. [1]7-year averages (2011-2017) from (REEW, 2018a)**; [2]using K:N and P:N ratios of 1.19 and 0.26 in manure, based on average nutrient concentrations for bovine manure from (Thibaut, 2016)**

	N [kgN/ha]	P [kgP/ha]	K [kgK/ha]
manure	97.7[1]	25.4[2]	116.4[2]
synthetic fertilizer	100.6[1]	5.56[1]	24.0[1]

4.5.2 Additional Results

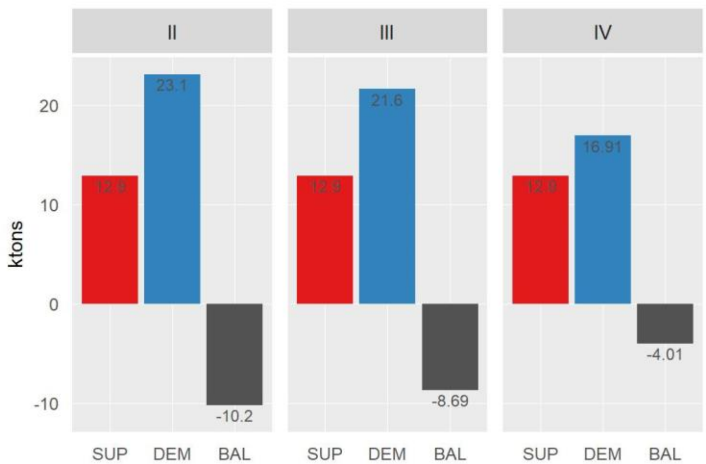


Figure S 6 Effect of the assumption of soil type III for estimating the crops' P demand in Wallonia on the total demand and balance of the region.

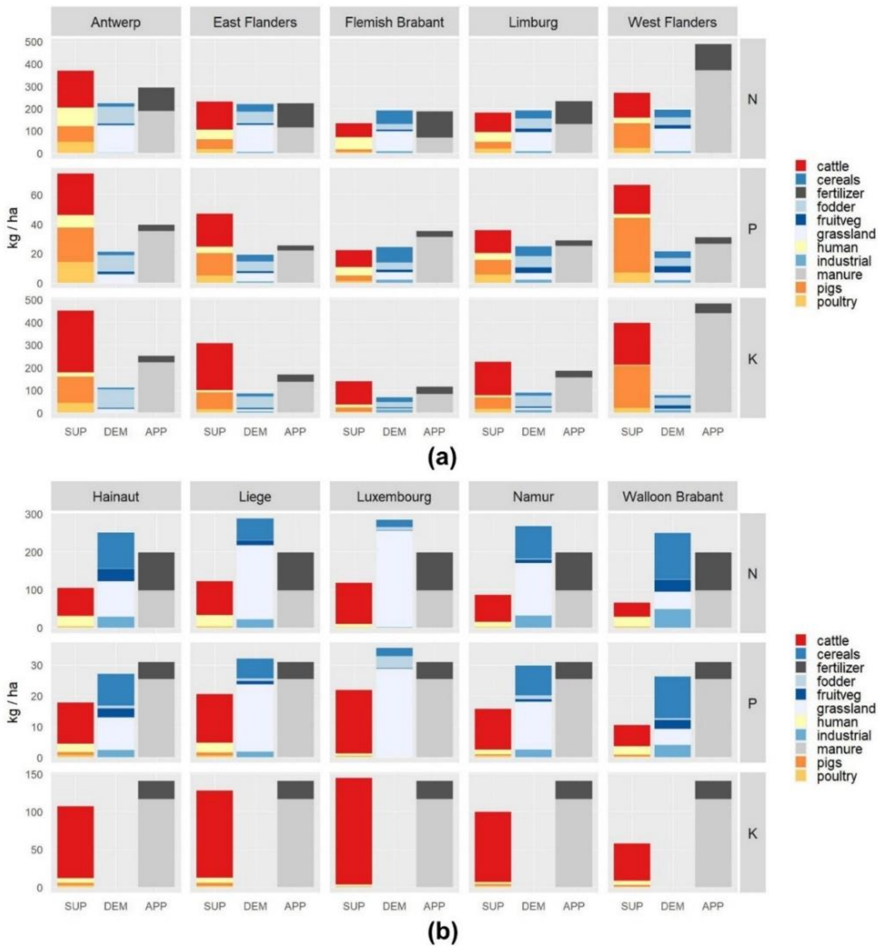


Figure S 7 Supply (red-orange columns), demand (blue columns) and actual application (grey columns) of N, P, and K per province in Flanders (a) and Wallonia (b), in kg / ha of nutrient per year.

Chapter 5

A resource-based phosphorus footprint for urban diets[5]

5.1 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 (Helena 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., 2018, 2015) 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

[5] This chapter is based on: **Papangelou A**, Towa E, Achten W M J and Mathijs E (2021) A resource-based phosphorus footprint for urban diets *Environ. Res. Lett.* accepted

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 (Lenzen et al., 2007; Munksgaard and Pedersen, 2001). Studies on phosphorus footprints are relatively new in scientific literature (Hu et al., 2020; Geneviève S Metson et al., 2016) and we can classify them in three broad categories: P emission footprints, P use footprints, and 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 (MRIO) models and their environmental extensions to assess nutrient emissions (Hu et al., 2020; Li et al., 2019). 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 take a cradle-to-grave approach, account for all inputs, losses and emissions at each stage of the supply chain, and distinguish between primary and secondary inputs, e.g. mineral fertilizer and manure respectively. Yet they have only been applied to two cases so far, both of single products produced in a single country, i.e. oats (Grönman et al., 2016) and beef (Joensuu et al., 2019) produced and consumed in Finland.

There are, therefore, only two P-use footprint studies published to this day that address a region's consumption, rather than a single product. Nonetheless, reducing resource inputs, especially in the case of P that is considered a scarce element, is an important part of circularity strategies and assessments (Navare et al., 2020). Further, neither of the published studies accounts for the use of secondary phosphorus sources, such as manure or treated urban waste. In this study we developed a P footprint approach that combines P flows up- and downstream the city, focuses on resource use, rather than emissions, and accounts for both primary (mineral) and secondary (manure) P inputs into the food system. We then applied it to the food consumption of Brussels Capital Region (BCR).Our objectives have been to:

(i) quantify the primary and secondary P embodied in the diet of the inhabitants' of Brussels,

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

(iii) compare how different interventions affect the P footprint and thus the different parts of Brussels' hinterland.

5.2 Methodology

5.2.1 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 (Grönman 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 5.1b), in the form of mineral fertilizer (P_f, primary) and manure (P_m, 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) :

$$P_{f_{i,l}} = Q_{i,l} \times f_{conv,i} \times \frac{1}{Yield_{i,l}} \times Fert_{i,l} \qquad \text{(Eq.1)}$$

$$P_{m_{i,l}} = Q_{i,l} \times f_{conv,i} \times \frac{1}{Yield_{i,l}} \times Manure_{l} \qquad \text{(Eq.2)}$$

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; $Fert_{i,l}$ (kgP/ha) is the P fertilizer application rate in region l for crop i, and $Manure_l$ (kgP/ha) 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^P), sewage sludge (Q_{ss}^P) and food waste (Q_{fw}^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 (Grönman 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:

$$FP_p = P_f + P_m - P_m^{out} - Q_{ss,re}^P - Q_{fw,re}^P \qquad \text{(Eq.3)}$$

where P_f and P_m are the sums of the $P_{f,i,l}$ and $P_{mi,l}$ over all products i and producing regions l, P_m^{out} the manure excreted by the animals producing animal products for Brussels, and $Q_{ss,re}^P$ and $Q_{fw,re}^P$ 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 Multi-Regional 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 1a offers an overview of the system and methodology, further details of which are presented in the following sections.

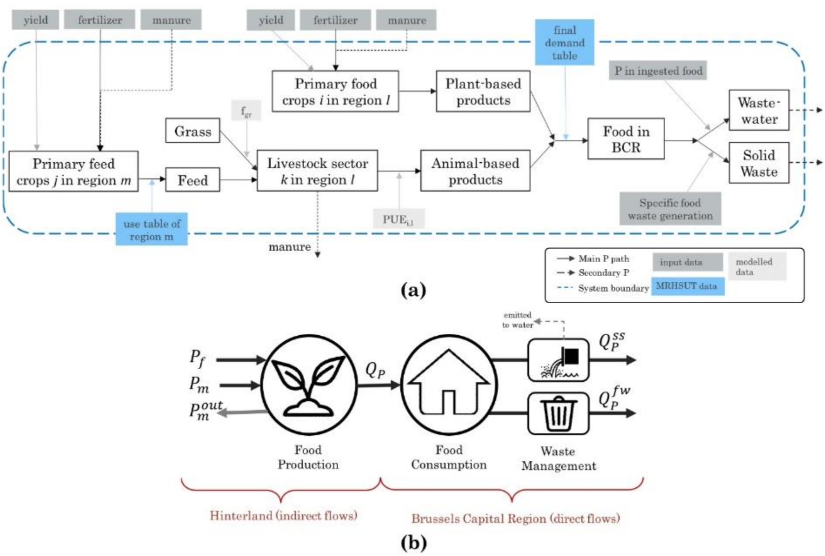


Figure 5.1 Detailed (a) and simplified (b) system diagram. The detailed diagram (a) gives an overview of the methodological steps followed and the different types of input used. Note that only the arrows with left to right direction represent P flows, while the rest simply indicate where different input data were introduced in the model. The simplified diagram (b) includes the aggregated P flows used for the calculation of the P footprint. P_f: fertilizer P input, P_m, manure P input, P_m^{out}: manure P output, Q_P: P in food, Q_P^{SS}: P in sewage sludge, Q_P^{fw} : P in food waste, $PUE_{k,l}$: livestock phosphorus use efficiency, f_{gr}: share of grass to total P ingested, MRHIO: Multi-Region Hybrid Input Output

5.2.2 Estimation of P inputs and outputs

- *Food consumption and primary crops*

The starting point of the analysis is the final demand for food products ($\sum_{i,l} Q_{i,l}$) 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 5.1). 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 & 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, 2017b), 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 primary crops using the conversion factor f_{conv}. Conversion factors account for the mass fraction of the primary crop to the derived product (f_{prim}, values from (FAO, 2000)), as well as its monetary value share (f_{value}), to avoid double-counting (Scherer and Pfister, 2016). A full list of the conversion and their constituents can be found inAnnex2. 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 inAnnex1. 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.

Table 5.1 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_{conv}). (nec: not elsewhere classified).

Food product	Represen-tative product	Final consumption households [tDM][1]	DM [%][2]	Energy [kcal/ 100gr][2]	P [mgP/ 100gr][2]	f_{conv} [-][3]
Wheat	flour for white bread	69’934	85	327.0	90.0	1.01
Cereal grains nec	corn	1’048	89	361.0	99.0	1.00
Vegetables; fruit; nuts	*weighted average*	56’445	15	50.4	33.4	1.00
Crops nec	*weighted average*	20’650	55	161	57	1.00
Poultry	*weighted average*	37’760	30	161	204	*na*
Meat animals nec	pork steak raw	52	*28*	*126*	*210*	*na*
Animal products nec	honey	41	*28*	*126*	*210*	*na*
Raw milk	milk	228	13	65	89	*na*
Fish and other fishing products	fish, lean raw	1’467	19	76.0	199	*na*
Products of meat cattle	beef entrecote	5,187	33	177.0	166	*na*
Products of meat pigs	pork steak raw	9,969	28	126	210	*na*
Products of meat poultry	poultry	1,946	30	161	200	*na*
Meat products nec	sausage pork-beef raw	2,211	36	226.0	414	*na*
Vegetable oils and fats	oil, salad	8’620	99	883.0	0	1.87
Dairy products	*weighted average*	14’546	19	105	175	*na*
Processed rice	rice, hulled	42	87	347.0	145	1.00
Sugar	sugar, granulated	40’907	100	400.0	0	6.57
Beverages	Beer, wine	7’977	12	57	18	1.37 1.43
Fish products	fish, lean raw	10	19	*76*	*199*	*na*

[1](Towa et al., 2020), [2](Nubel, 2018), [3](FAO, 2000) and (Scherer and Pfister, 2016), in [kg primary crop/kg derived product]

5.2.3 Feed requirements of livestock

For animal products, we substitute $Q_{i,l}$ in (Eq.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 (Eq.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 ($PUE_{k,l}$, Eq.4). For the cattle and milk sectors, we further used a f_{gr} factor to differentiate between P ingested through feed and through grazing (Eq.5). The values for $PUE_{k,l}$ and f_{gr} are based on P inputs and outputs into the livestock sector extracted from country- and region-specific studies (seeAnnex1 for details and sources).

$$PUE_{k,l} = \frac{P\ in\ animal\ product\ produced\ by\ sector\ k}{P\ digested\ from\ animals\ in\ sector\ k} \qquad \text{(Eq.4)}$$

$$f_{gr} = \frac{P\ ingested\ through\ grazing}{P\ ingested\ as\ feed} \qquad \text{(Eq.5)}$$

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.

5.2.4 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 (Departement Landbouw en Visserij, 2020; REEW, 2018a); rates for manure use are based on actual livestock numbers and manure management practices (Papangelou and Mathijs, 2021b). 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; Le Noe et al., 2020; Le Noë et al., 2017; Rothwell et al., 2020; Smit et al., 2015; van Dijk et al., 2016), supplemented and cross-checked with official statistics (Agreste, 2017; DEFRA, 2012; DESTATIS, 2013; Eurostat, 2020).

For all other countries and world regions we combined data on fertilizer use from the International Fertilizer Association (IFA, Heffer, 2013) with data on cultivated areas from (FAOSTAT, 2020a) to derive P-fertilization rates for (Eq.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^{out}). Details are given in the Annex1.

5.2.5 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 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 Table S47). 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.

5.2.6 Scenarios and indicators

In order to compare the effect that different interventions could have on Brussels' P footprint, we developed five scenarios (Table 5.2). 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 seeAnnex1 Table S49)

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, 2021b). 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.

Table 5.2 Description of the five scenarios developed and their impact on the calculation of the P footprint (Eq.3)

Scenario	Description	Factors affected in the FP_P estimation (Eq.3)
Baseline	-	-
Reuse	All food waste and sewage sludge are treated and reused into food production	$QTY^P_{ss,re} = QTY^P_{ss}$ $QTY^P_{fw,re} = QTY^P_{fw}$
FWA	Food Waste Avoidance: all food waste is eliminated	$QTY^P_{fw,re} = QTY^P_{fw} = 0$ P_f, P_m, P^{out}_m ↓ due to the avoided production
Vegetarian	Isocaloric substitution of meat products with dairy and plant-based (Table 1)	P_f, P_m ↓ due to the avoided feed production P^{out}_m ↓ due to fewer animals in production
Vegan	Isocaloric substitution of all animal products with plant-based ones (Table 1)	P_f ↓ due to the avoided feed production P_m, $P^{out}_m = 0$ due to no animals in production
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.	P_f, P_m, P^{out}_m change, following the yields, fertilization rates, and PUEs in Flanders and Wallonia

5.3 Results

5.3.1 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 in 2011. The total indirect inputs were 11.4 kgP/cap/yr , 4.6 kgP/cap/yr of which were supplied to crops as fertilizer (P_f), and 6.7 kgP/cap/yr, as manure (P_m, Figure 2b).. Each inhabitant in Brussels is consuming 0.7 kgP/cap/yr directly through the food they buy (Figure 2a), more than 10 times less than the amount used to produce this food. 0.15 kgP/cap/yr are thrown away, while the rest 0.55 kgP/cap/yr are digested and eventually end up in the city's wastewater management system.. Two thirds of the 0.7 kgP/cap/yr are imported in Brussels through animal products, especially domestically produced meat (Figure 5.2a).

95% of all the P inputs into the system are used by the livestock sector (Figure 5.2b). 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 alsoAnnex1, section 2.3).

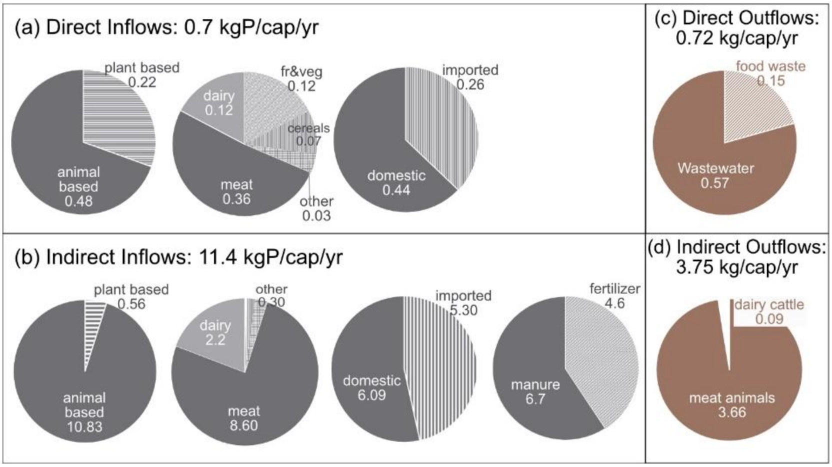
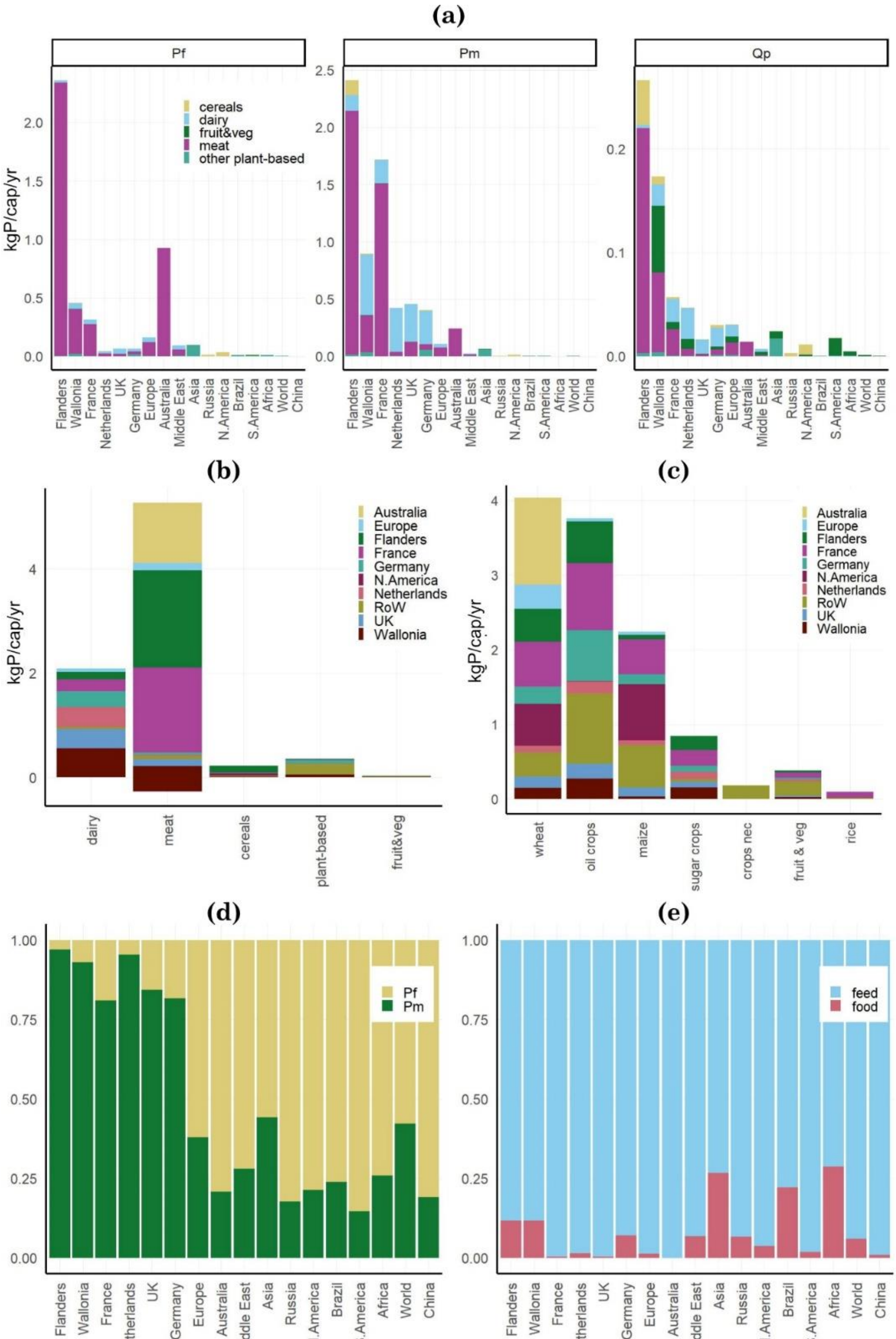


Figure 5.2 Direct and indirect in- and outflows in kgP/cap/yr and breakdown into their different components. Inflows (from left to right): animal- and plant-based food products, five main food groups, domestic and imported products, manure and fertilizer (for indirect flows only). Outflows: wastewater and food waste (direct flows), and manure produced by dairy cattle and meat animals (indirect flows).

5.3.2 Domestic and global P flows

Around 60% (0.44 kgP/cap/yr) 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) is from meat, mostly poultry, imported from Flanders (Figure 5.3a). 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 or 50% of total P_f) and for manure (2.13 kgP/cap/yr 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 5.3c), 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 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 for Flanders, and 0.04 and 0.6 kgP/cap/yr 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 mobilised to provide the city's diet.

Figure 5.3 (next page) (a) Indirect P flows through fertilizer (Pf) and manure (Pm), and direct flows through food items (Qp) per producing country or region in kgP/cap/yr; (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 ; (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)
Pf
Pm
Qp
kgP/cap/yr
cereals
dairy
fruit&veg
meat
other plant-based
Flanders
Wallonia
France
Netherlands
UK
Germany
Europe
Australia
Middle East
Asia
Russia
N.America
Brazil
S.America
Africa
World
China
(b)
(c)
Australia
Europe
Flanders
France
Germany
N.America
Netherlands
RoW
UK
Wallonia
dairy
meat
cereals
plant-based
fruit&veg
wheat
oil crops
maize
sugar crops
crops nec
fruit & veg
rice
(d)
(e)
Pf
Pm
feed
food

5.3.3 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 . If all food waste is avoided, the FP_P is further reduced to 6.8 kgP/cap/yr , 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 and 0.9 kgP/cap/yr , even if little manure or none at all are produced in these cases. Finally, a local diet made up from food produced in Belgium only, has a footprint of 11.9 kgP/cap/yr , 55% higher than the baseline.

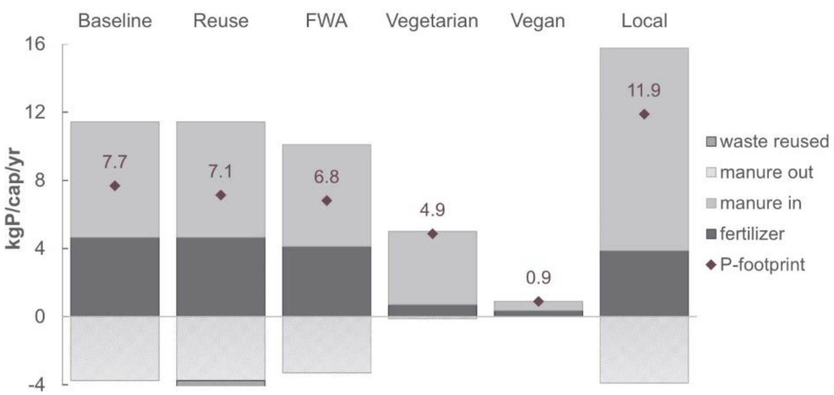


Figure 5.4 Comparison of the annual per capita P footprint of food consumption in Brussels, and its components under alternative scenarios. For an overview of the scenarios see Table 2.

5.4 Discussion

5.4.1 A wider perspective on P footprints

Several recent studies have proposed approaches to estimate and evaluate P footprints and have applied them at global and national scales (Hu et al., 2020; Leach et al., 2012; Li et al., 2019; Metson et al., 2020; Oita et al., 2020). What these studies have in common is the conceptualization of the P footprint from an emission point of view, as they quantify the P emissions from different economic activities into the environment. Such an exclusive focus on emissions could lead to a bias towards end-of-pipe and efficiency-oriented solutions while ignoring approaches such as sufficiency and absolute reduction of resource use that are increasingly recognized to be key for sustainability transformations (Haberl et al., 2020; O'Neill et al., 2018). Besides, phosphorus is not only a pollutant, but also a critical resource for agriculture and food security (Cordell et al., 2009; Elser and Bennett, 2011).

A resource-based P footprint can address these concerns, but so far only two studies have proposed 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 for Rwanda to 6.09 kgP/cap/yr for the USA and 7.02 kgP/cap/yr for Argentina. For Belgium, they report 5.21 kgP/cap for 2007, a value close to the 4.6 kgP/cap 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 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, 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 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 local, renewable nutrient source that should not be disregarded when analyzing the circularity and resource use of food systems.

5.4.2 Towards a more circular urban food system

Strategies considered to promote circularity in the food system 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 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 (Smit et al., 2015), whereas the rates in all other regions are below 10 kgP/ha. Even though the high availability of manure in Flanders also means lower P-fertilizer usage than 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 and disregard 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 Annex).

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 (Vanessa 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 (Geneviève S 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 (see also Figure S11 in Annex). 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.

5.4.3 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: (i) data are often the result of modelling, thus carrying the inherent assumptions of the models that generated them; (ii) 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 (iii) 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 the Annex). 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 prioritise 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 5.3d). 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 (Mekonnen and Hoekstra, 2018; Potter et al., 2010). 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.

5.5 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.

5.6 Annex

5.6.1 Details on methodology

- *Assumptions used on the decision of representative products, and their allocation to primary crops*

Three datasets are needed to estimate the fertilizer inputs P_f:

a. MRHSUT for the food products imported into Brussels and their exporting countries and regions (Towa et al., 2020)
b. Crop yields from (FAOSTAT, 2019)
c. P fertilizer use from IFA (Heffer, 2013), combined with cultivated areas from (FAOSTAT, 2019) to derive fertilization rate in kgP/ha.

MRHSUT and data from IFA are given for 2011; for crop yields and cultivated areas we used 5-year averages around 2011 (2009-2013), to eliminate possible anomalies in single-year datapoints. Since the datasets have different aggregation levels for food products and world regions, we used the following assumptions to combine them:

- Beverages were assumed to be beer (primary crop: barley) when origin was Flanders, Wallonia, or the Netherlands, and wine for all other regions (primary crop: grapes).
- Crops nec (almost all coming from Asia) → coffee (80%, 5% tea, 15% spices)
- Sugar → sugar beet when region of origin is in Europe, sugar cane when not
- Vegetable oils → rapeseed oil when origin in Europe, palm oil when not (and soybeans when it's for feed)
- Fruit & vegetables imported from abroad (not Wallonia) assumed to be fruit when imported from South America, Asia and the rest of the world (mainly Africa), and vegetables from all other regions. In practice this would roughly mean that we assign bananas, exotic fruits such as pineapples and avocados, and citrus fruit, which comprise roughly 22% of all fruit & veg consumption (Statbel, 2017b), to S.America, Asia and the rest of the world, which export to Brussels roughly 23% of all fruit & veg imported according to the MRHSUT.

- *Yields, fertilization rates, manure*

Data from the Belgian agricultural census were used for crops grown in Flanders and Wallonia (Statbel, 2014). As these datasets do not include information on yields for fruit and vegetables. we used FAOSTAT data instead (FAOSTAT, 2019) and a weighted average was estimated for the group, assuming a distribution of different items as shown in Table S40. The distribution is based on the actual amounts of the items purchased in Belgium (Statbel, 2017b) and crops that are widely grown in Wallonia (Statbel, 2015a). Fertilization rates used for Flanders and Wallonia, as well as the other four main exporting countries into Brussels (France, the Netherlands, Germany, the UK, plus Europe) are given in Table S41.

Table S 40 Calculation of a weighted average yield for fruit&veg from Wallonia, based on the quantities of each product consumed in Brussels (Statbel, 2017b)

	tDM/yr	%	yield [100kg/ha]
potatoes	6,894	50%	464
apples	2,539	18%	355
pears	874	6%	336
cherries	1,091	8%	53
onions	985	7%	478
carrots	815	6%	621
lettuce	560	4%	396
total Wal	13,758	100%	411.66
TOTAL FVEG	26,071	53%	

For all other crops imported from outside Belgium, 5-year averages (2009-2013) for the yields from FAOSTAT were used (FAOSTAT, 2019). The food group C_OTCR, including mainly coffee and tea, cocoa and spices deserves some special attention. This is because the yields and fertilization rates for these three sub-groups vary wildly, so that an average makes little sense. Taking into account (i) information on consumption in Brussels (distribution in the group is: 80% coffee and tea, 15% spices, 5% cocoa), (ii) the different regions of origin of the group C_OTCR in the MRHIO model and (iii) production and exporting information from the regions of origin, we take the following assumptions to operationalise the group C_OTCR:

- The imported amounts from Asia are all coffee coming from Vietnam
- The amounts coming from Latin America are all cocoa.
- C_OTCR imported from other countries are spices, which we exclude from the calculation (crops coming from Africa, North America and Middle East, together making up ~5% of all C_OTCR or 0.6% of all food into Brussels)

We approximated manure application onto crops and grassland, using the approach from the study by Sheldrick and colleagues (2003). We used the specific P excretion rates and ratios of P in manure to P excreted from Sheldrick's study (Table S42), with data from FAOSTAT on livestock production in the world regions (FAOSTAT, 2020b) to calculate total amount of P in manure produced in each country or region. We then converted the absolute amounts to application rates in kgP_{manure}/ha, by dividing with the total area of agricultural land (FAOSTAT, 2020a).

For the estimation of the outflowing manure from the livestock sectors supplying Brussels we multiplied f_PperSl with the corresponding weight of animal product.

Table S 41 Synthetic fertilizer and manure application to agricultural land in the six main supply regions of Brussels [all values in kgP/ha]

	wheat	other cereals	sugar beets	rapeseed	potatoes	viniculture	fertilizer to '	manure to '	Source
Flanders	0.7	2.7	4.2	3.3	7		5.22	29	(1), (2)
Wallonia							5.36	8.6	(2), (3)
France						13	6.61	7.8	(4)-(6)
Netherlands							7.47	21.2	(7)
Germany							7.22		(8)
UK	12.7	14.2	11.4	11.4	51.5		8.29		(9)
Europe							7.16	3.6	(10), (11)

(1) (Departement Landbouw en Visserij, 2020), avg2010-2, (2) Chapter 4, (3) (REEW, 2018a), avg2010-2, (4) (Agreste, 2017), (5) (Le Noe et al., 2020), (6) (J. Le Noë et al., 2018), (7) (Smit et al., 2015), (8) (DESTATIS, 2013), avg2010-2, (9) (DEFRA, 2012), p.14, (10) (van Dijk et al., 2016), (11) (Eurostat, 2020)

Table S 42 Information on P excretion (f_PperSl in kgP excreted/ kg slaughtered animal) and ratio of P in manure to total P excreted (f_MtoExc), used to estimate global manure production and application. [Source: f_PperSl from table 2, and F_MtoExcr combining tables 3 and 4 all from (Sheldrick et al., 2003)].

factor	World region	cattle	pigs	sheep	goat	horse	poultry
f_PperSl	*all regions*	0.04	0.05	0.13	0.17	0.03	0.10
f_MtoExc	Africa	0.10	0.71	0.10	0.10	0.10	0.70
	Northern America	0.30	0.80	0.09	0.00	0.31	0.80
	Central America	0.10	0.70	0.13	0.10	0.09	0.70
	South America	0.10	0.70	0.10	0.09	0.10	0.70
	Western Asia	0.10	1.00	0.10	0.10	0.11	0.70
	Southern Asia	0.46	0.71	0.10	0.32	0.35	0.70
	Eastern Asia	0.49	0.80	0.10	0.10	0.46	0.80
	Eastern Europe	0.50	0.80	0.11	0.14	0.46	0.79
	Western Europe	0.65	0.90	0.10	0.11	0.67	0.90
	FSU	0.40	0.70	0.10	0.12	0.40	0.70
	Oceania	0.30	0.79	0.10	0.00	0.67	0.79
	World	0.31	0.80	0.10	0.16	0.22	0.77

- *Conversion of animal products to feed crops*

$PUE_{k,l}$ and f_{gr}

We derived the *PUE* and f_{gr} factors from previous studies on P flows in the food system of the main animal rearing regions supplying Brussels with meat and dairy, combined with national statistics (Table S43). We used the P SFA at the European level (van Dijk et al., 2016) to derive the factors for the rest of Europe, as well as to double check the values from the individual studies. We used the average of these values for the sectors in the rest of the world. The amount of P digested as feed by the animals in livestock sector *k* and animal-rearing region *l*, supplying Brussels with animal products *i* is:

$$Q_{P,k,l}^{feed}\ \left[\frac{tP_{feed}}{a}\right] = \sum_{i}(Q_{i,l}^{an.prod.} \times Pc_i)\ \times PUE_{k,l}\ \times (1 - f_{gr,l})$$

where

$Q_{P,k,l}^{an.prod.}$ is the annual consumption of animal product *i* imported from animal-rearing region or country *l* [tDM/a],

Pc_i is the P content of animal product i [-]
$PUE_{k,l}$ is the phosphorus use efficiency of livestock sector k in region l (Table S43)
$f_{gr,l}$ is the share of grass in the diet of cattle, assumed the same for milk and meat cattle, and 0 for poultry for pigs.
The amount of grass digested by cattle is:

$$Q_{P,k,l}^{grass}\left[\frac{tP_{grass}}{a}\right] = \sum (Q_{i,l}^{an.prod.} \times Pc_i) \times PUE_{k,l} \times f_{gr,l}$$

Table S 43 PUE of all livestock sectors and share of grass to total P intake for cattle for Brussels' main supply regions with animal-based food products

region	Phosphorus Use Efficiency (PUE)				f_{gr}
	cattle	milk	pigs	poultry	
Flanders[(1)]	0.014	0.041	0.079	0.041	0.376
Wallonia[(1)]	0.014	0.045	0.099	0.051	0.750
France[(2)]	0.009	0.086	0.062	0.036	0.328
Netherlands[(3)]	0.011	0.191	0.069	0.043	0.650
Germany[(4)]	0.022	0.094	0.077	0.043	0.317
UK[(5)]	0.011	0.060	0.077	0.043	0.550
Europe[(6)]	0.023	0.074	0.077	0.043	0.474
RoW[(7)]	0.013	0.157	0.133	0.076	0.474

[(1)] (Papangelou and Mathijs, 2021b), [(2)] (Le Noë et al., 2017), [(3)] (CBS, 2012; Smit et al., 2015), [(4)] (Theobald et al., 2016), [(5)] (Cooper and Carliell-Marquet, 2013), [(6)] (van Dijk et al., 2016), [(7)] average of all the above values.

Disaggregating feed intake per livestock sector using MRH Use Tables

The following procedure was used to disaggregate the total feed intake of animals in sector k and animal-rearing region l [6]into feed crops j, cultivated in region m:

1) We extracted from the use tables the columns referring to the livestock sectors and aggregated them into 4 sectors: cattle (A_CATL + A_OMEA), milk (A_MILK), pigs(A_PIGS), poultry (A_PLTR, to obtain the amounts of each crop j from country m used in each livestock sector k: $Q_{k,m,j}^{feed\ use}\left[\frac{tDM}{a}\right]$.

[6] For the six regions supplying 92 of all animal-based food product: Flanders, Wallonia, France, Netherlands, Germany, UK (**Error! Reference source not found.**).

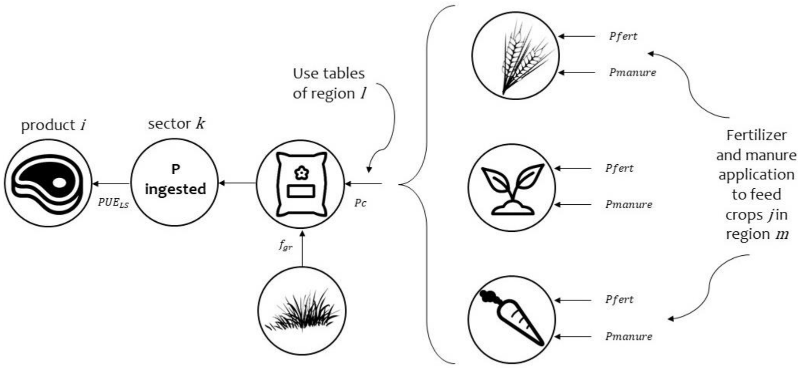


Figure S 8 Simplified schema for the estimation of the virtual P flows related to animal products

2) We assigned a representative product to each feed group, and further disaggregated the “food products nec” group (C_OFOD) into primary products using the same use tables. We then converted DM to P using P content of the different feed crops from feedipedia (Table S44). When assigning representative products, we took the cultivating (exporting) country/ region into account: oilseeds and oilseed products are assigned to rapeseed when coming from Europe, and soybean when coming from America and Asia; sugar crops and “Food products nec” are beets and wheat bran when coming from Europe and sugar cane and copra oilmeal from other regions. We then summed the P amounts to obtain the total amount of P ingested by all animals of sector k in each animal-rearing region l:

$$Q_{P,k,l}^{feed\ use}\left[\frac{tP}{a}\right] = \sum_{m,j} Q_{k,m,j}^{feed} \cdot Pc_j$$

3) We finally allocated each feed crop j used in sector k to the part of the production that is exported to Brussels, by applying the factor f_{alloc} to each element of the use table $Q_{k,m,j}^{feed\ use}$.

$$Q_{k,m,j}^{feed\_BCR} = f_{alloc_k} \times Q_{k,m,j}^{feed\ use} = \frac{Q_{P,k,l}^{feed}}{Q_{P,k,l}^{feed\ use}} \times Q_{k,m,j}^{feed\ use}$$

The values $Q_{k,m,j}^{feed\_BCR}$ where subsequently used in equations 1 and 2 in the main text to estimate the amounts of fertilizer and manure that go into their cultivation, and thus into the production of animal products.

Table S 44 Feed products into livestock sectors, assigned representative products, nutrient content and technical conversion factors to primary crops. [Source: DM and Pc from (INRA et al., n.d.), f_{prim} from (FAO, 2000)]

Product in MRHSUT	Representative product	DM [%as fed]	Pc_j [gP/ kgDM]	f_{prim}
Food products nec	wheat bran	87.0	11.1	0.18
Food products nec	copra oilmeal	91.5	5.8	0.36
products of Vegetable oils and fats	rapeseed cake	92.3	7.3	0.60
products of Vegetable oils and fats	soybean cake	88.7	6.1	0.79
Oil seeds	rape forage	12.1	5.8	1.00
Oil seeds	soybean forage	24.0	2.7	1.00
Cereal grains nec	maize grain	86.3	3.0	1.00
Wheat	wheat grain	87.0	3.6	1.00
Vegetables; fruit; nuts	fodder beet root	16.1	2.4	1.00
Processed rice	rice bran	90.0	13.9	0.08
Sugar	sugarcane forage	23.2	1.3	1.00
Sugar	sugarcane molasses	73.0	0.7	0.04
Sugar	beet molasses	75.4	0.3	0.04
grass	elephant grass	17.9	2.9	1.00

Table S 45 Quantities of total animal products consumed in BCR, according to their region of origin and respective shares to total mass of animal products consumed.

region	QTY [tDM]	share
Flanders	35,017	48%
Wallonia	14,388	20%
FR	7,631	10%
NL	4,565	6%
EU	3,202	4%
DE	3,086	4%
UK	2,048	3%
ROW	1,623	2%
AU	1,338	2%

- *Food waste and wastewater*

Table S 46 Information on food waste and wastewater generation in BCR

Variable	Value	Unit
per capita food waste generation	89.48[1]	kg/cap.yr
P content of household food waste	0.086%[4]	%
DM content of household food waste	31.8%[4]	%
Share of animal products in food waste (in FM)	30%[2]	%
Share of plant products in food waste (in FM)	70%[2]	%
P inflow in N.WWTP (2011)	568[3]	tP
P removal efficiency N.WWTP (avg 2011-2015)	85%[3]	%
P inflow in S.WWTP (2011)	164[3]	tP
P removal efficiency S.WWTP (avg 2011-2015)	60%[3]	%

[1] (Papangelou et al., 2020), [2] (Zeller et al., 2020), [3] (Papangelou et al., 2020 SM1) [4] Average of corresponding food groups – see Table S8

Table S 47 Breakdown of food waste quantities (QTY_{fw}), as well as dry matter (DMc) and phosphorus content (Pc) per food group, and estimation of food waste generation rates for each of the five main food groups (r_{fw})

food group	QTY_{food}[1]	r_{group}[2]	QTY_{fw}	r_{fw}[3]	QTY_{fw}[3]	DMc[4]	Pc[4]
	kg/cap.yr	%FM	kg/cap.yr	%	kg/cap.yr	%	%FM
cereals	76.64	0.70	62.6	*0.10918*	*8.4*	85%	0.09%
fruitveg	356.19				*38.9*	15%	0.03%
other plant-based food	140.88				*15.4*	55%	0.02%
meat	175.23	0.30	26.8	*0.10902*	*19.1*	30%	0.21%
dairy	71.00				*7.7*	19%	0.17%
	819.9		**89.48**[2]		**89.48**	***31.8%***	***0.0862%***

[1] Final household consumption [2] (Zeller et al., 2020) [3] *Calculated here* [4] Average value for the food group (see Table 1 in main article)

- *Comparison of baseline, vegetarian and vegan diets*

Table S 48 Amounts of food supplied to Brussels (final household consumption from MRIO, QTY_{supply} in tDM) and food consumed (supply minus food waste QTY_{cons} in gr/cap·d) for three of the scenarios: baseline (base), vegetarian (veg) and vegan.

food group	QTY_{supply} [tDM]			QTY_{cons} [gr/cap.d]		
	base	veg	vegan	base	veg	vegan
cereals	71 024	95 557	105 123	147	198	218
fruit &veg	56 445	75 943	83 545	683	919	1 011
other plant-based	78 155	105 152	115 678	270	364	400
dairy	14 774	19 877	0	58	79	0
meat	57 125	0	0	144	0	0
TOTAL	**277’523**	**296’530**	**304’346**	**1’303**	**1’559**	**1’629**

Table S 49 Daily per capita intakes of energy, protein and phosphorus through food consumed in Brussels for three of the scenarios: baseline (base), vegetarian (veg) and vegan.

food group	$Energy_{cons}$ [kcal/cap.d]			Protein [gr/cap.d]		
	base	veg	vegan	base	veg	vegan
cereals	481	648	712	14.6	19.7	21.7
fruit &veg	344	463	510	8.8	11.8	13.0
other plant-based	595	801	881	6.1	8.2	9.0
dairy	61	82	0	3.5	4.7	0.0
meat	227	0	0	28.4	0.0	0.0
TOTAL	**1’709**	**1’994**	**2’103**	**61.4**	**44.4**	**43.7**

food group	Phosphorus [gr/cap.d]		
	base	veg	vegan
cereals	0.19	0.25	0.28
fruit &veg	0.33	0.44	0.48
other plant-based	0.08	0.11	0.12
dairy	0.34	0.45	0.00
meat	1.00	0.00	0.00
TOTAL	**1.94**	**1.26**	**0.89**

5.6.2 Robustness check for some model choices & parameter values

- *P additives into animal diets*

In order to test the robustness of our final results to the exclusion of P additives from the animals' diets, we performed a check calculation under the assumption that 50% of the P ingested by all animals in Flanders and France through feed directly comes from mineral P sources and not from feed crops. We performed the test for Flanders, because it provides almost half of all the animal products consumed in Brussels, and France because it provides another 10%, and data used to model the livestock sector in France comes from a different study that the one for Flanders. The choice of 50% is meant as an exaggerated upper limit, since shares of P additives are typically lower and only for cattle.

We found that under such an assumption the manure use for the cultivation of feed would strongly decrease by -48% and -45% for Flanders and France respectively. The decrease would be partly offset by an increase in the use of mineral fertilizer by + 17% and +67%. The increase in the mineral fertilizer is because the ratio (P in fertilizer applied / P in feed produced) is less than one, because high manure use makes up for the difference in P needs of the crops. When the fertilizer is directly fed to the animals, though, in the form of P additives into compound feed, the ratio is 1:1 for 50% of the P in diet. As a result, the total P footprint related to animal production in Flanders would be reduced by 1.4 kgP/cap·a (-25%) and for France by 0.5 kgP/cap·a (-18%). These changes correspond to reductions of -10% and -4% to the total P footprint of Brussels' food consumption.

- *Phosphorus Use Efficiencies for livestock sectors*

Using Phosphorus Use Efficiencies for each livestock sector has been a pivotal step in our calculation. To derive these values, we used data from country- or region specific studies from the peer-reviewed literature, cross-checked with national statistics when possible. We generally found efficiencies ranging from 1-2% for cattle meat to around 8-9% for milk (up to 19% for the Netherlands). Nonetheless, because the final result is potentially sensitive to these values, we performed a check calculation for an assumed 5-fold increase in the

PUE of all animal sectors in Flanders and France. This change would cause an almost linear equivalent reduction to the required inputs in both Flanders and France and would ultimately decrease the total footprint 8.8 kgP/cap·a, a change of -66%. This is an important quantity, however, a 5-fold increase in the PUEs means efficiencies of up to 30-40% in some cases, which are unrealistic.

- *Fertilization rate for fruits and vegetables*

As one can notice in Table 3 of the main text, the estimated P inputs into the production of fruits and vegetables is lower than the actual P content of these fruit and vegetables. This is possible because we used uniform country-level fertilization rates, since country- *and* crop-specific data are scarce. For Wallonia, for example, we use a uniform P-fertilization rate of 5.36 kgP/ha (REEW, 2019e). However, more than 70% of the crops that Wallonia exports to Brussels are fruits and vegetables, that typically require more P-fertilizer than cereals. Another explanation would be the planned under-fertilization of some soils in Wallonia, so that soil excess and legacy P can be used.

To test the possible effect of using a different fertilization rate to the end result, we applied a rate of 80 kgP/ha, which is double the highest value in our existing data (42 kgP/ha for vegetables from N.America), and by far higher than literature values, e.g. (Nesme et al., 2016 Table S2). We found that in that case the total P inputs into fruit and vegetables would rise to 0.16 kgP/cap·a, from the current value of 0.03 kgP/cap·a. This is an important increase for the specific food group, however, it represents only a 20% increase of the inputs into all plant-based food products and a 1% increase of the P footprint, from 13.3 to 13.4 kgP/cap·a.

- *Different allocation methods*

Finally, we ran the model with conversion factors f_{conv} derived using two alternative allocation methods for crops that give more than one product: once a mass allocation, where P inputs are allocated to each product according to their relative mass share, and once an “equal” scenario where P inputs are allocated equally to derived products. This change affected those crops that give both a primary food product and a secondary one used for animal feed, notably oil and sugar crops.

Because the mass and equal allocation approaches give more weight to the by-products used for feed, they increase the P footprint by 22-23%.

An overview of the alternative scenarios used in the robustness checks and the respective results are presented in Table S50 and Figure S9.

Table S 50 Overview of alternative parameter values and assumptions used for the robustness check and detailed results of the analysis.

Alternative scenario	Parameters/ assumption	P_f	P_m	P_m^{out}	FP_P
Baseline		4.66	6.77	-3.75	7.69
Precision AG BE	Fertilization rates in Belgium: -10%	4.65	6.59	-3.75	7.49
Precision AG RoW	Fertilization rates in Belgium: -10%	4.21	6.28	-3.75	6.74
Veg fertilization	Fertilzation rate for fruit & veg = 80 kgP/ha	4.79	6.77	-3.75	7.81
P additives	50% of animal P intake in Flanders & France through P additives	6.76	4.99	-3.75	8.00
PUE	PUE of all livestock sectors in Flanders & France increased 5 times	2.53	3.59	-3.75	2.36
Other meat	Food group "Meat products nec" is pork meat instead of beef	4.06	6.86	-3.68	7.25
Mass allocation	$f_{conv} = f_{alloc} / f_{prim}$, where $f_{alloc} = f_{mass}$	5.29	7.93	-3.75	9.47
Equal allocation	$f_{conv} = f_{alloc} / f_{prim}$, where $f_{alloc} = 0.5$	5.48	7.68	-3.75	9.40

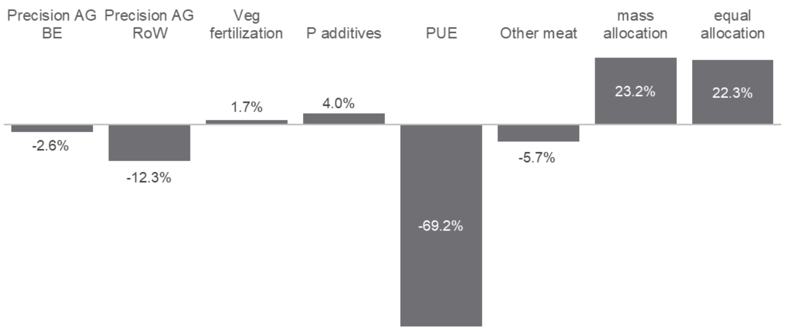


Figure S 9 Graphic representation of the robustness check results. For details on the alternative scenarios see Table S11.

5.6.3 Results for additional exploratory scenarios

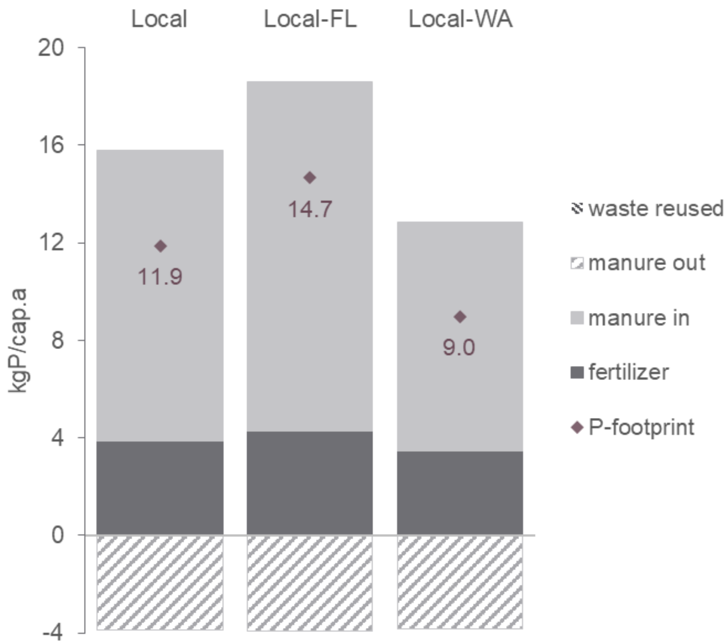


Figure S 10 Comparison of the P footprint and its constituents for the local scenario and 2 sub-scenarios: Local-FL, where all food previously imported from abroad is now coming exclusively from Flanders, and Local-WA, where all food previously imported from abroad is now coming exclusively from Wallonia. As a reminder, for the Local scenario the imports were assumed to be distributed equally between Flanders and Wallonia.

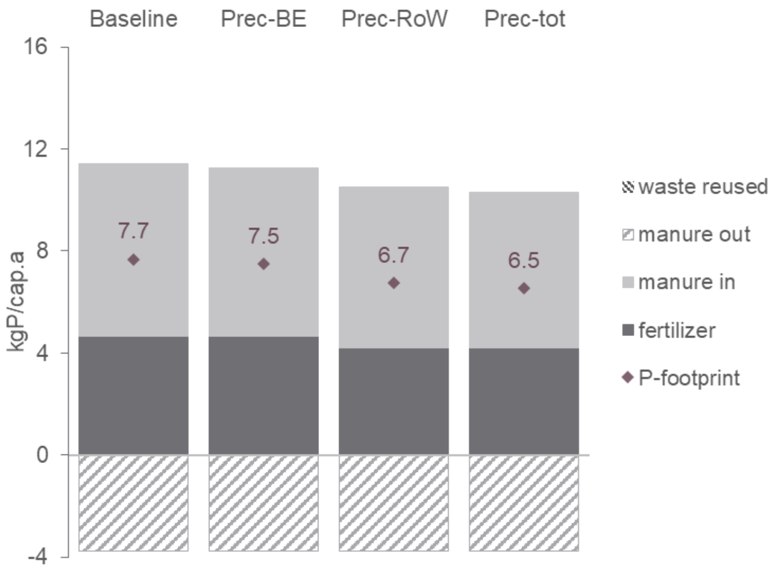


Figure S 11 Comparison of the P footprint and its constituents for the baseline scenario and 3 explorative scenarios of precision agriculture adoption: Prec-BE, where the adoption of precision agriculture practices in Flanders and Wallonia was assumed to reduce the fertilization needs of all crops in these regions by 10%; Prec-RoW, where the adoption of precision agriculture practices in all producing regions outside Belgium was assumed to reduce the fertilization needs of all crops in these regions by 10%; and Prec-tot, where the 10% reduction in fertilization rates was applied uniformly in and outside Belgium.

Chapter 6

General Conclusions

A vision should be judged by its clarity of values, not by the clarity of its implementation path.
Donella Meadows

6.1 Synthesis

6.1.1 Summary of key findings

In this thesis I studied the food system of Brussels from a circularity perspective using nutrient cycles as the main analytical tool. In this concluding chapter I summarize and synthesize the key findings, and discuss their implications for research and action.

Phosphorus flows in Brussels Capital Region have a linear pattern. Approximately 0.8 kgP/cap·a (700 tons) enter the city each year. 80% of those 700 tons end up in wastewater and the remaining 20% in solid organic waste, mostly household food waste. Sewage sludge from the wastewater treatment plants is either exported for incineration or landfilled; organic wastes are primarily incinerated and the ashes are used in road construction materials. All of the P currently flowing out of the city is either lost in the environment or locked-in in the technosphere. Reusing this P could provide an alternative to synthetic fertilizers without requiring additional net energy inputs for treating the effluents.

Nonetheless, urban effluents cannot be reused within the city's boundary, due to the negligible agricultural activity therein. Urban P will have to be exported to peri-urban fields or further into the agricultural lands of Flanders and Wallonia. Flanders is already over-supplied with nutrients, due to the abundance of secondary nutrients in livestock manure, the low nutrient use efficiency of the livestock sector, and the system's high dependency on imported feed. A total of 5.4 kg/cap·a (34'600 tons) of primary P enter Flanders, 15% of which are turned into food products, and 73% are lost as emissions into the environment or as non-recovered waste. In Wallonia, on the other hand, nutrients flow in tighter loops. Fewer animals and an extensive arable zone in the region's North lead to negative nutrient soil budgets and higher nutrient recycling rates: around half of the sewage sludge produced in Wallonia, for example, is valorized in agriculture. A total of 1.3 kgP/cap·a are entering the region (4'600 tons), and only 1% of this amount is lost, mostly thanks to the negative soil budgets that provide crops with P accumulated in the soil and reduce the dependency on external inputs.

Beyond these regional trends, there are important differences in the way nutrients circulate *within* each region, and in the potential of local secondary nutrient sources to cover crop needs. A mismatch in supply and demand for nutrients closely follows the pattern of segregation between areas dominated by livestock and crop production: supply is concentrated where animals produce manure, demand where crops need fertilization. Cities are also important hotspots of nutrient supply: human excreta contain 17% and almost 10% of all secondary nitrogen and phosphorus in Belgium, a nutrient source that to this day is largely ignored.

There is, therefore, some potential demand for additional secondary P into agricultural soils. Yet simply transporting P from the city to the rest of the country will not automatically close the loop for Brussels. It is important to know what part of the nutrient demand generated by Brussels these effluents will actually cover. 7.7 kgP/cap·a are going into producing the food for Brussels, more than 10 times the actual P content of food. 80% of these inputs occur outside Belgium, as fertilizer and manure inputs into feed crops. Because of the high P footprint of livestock rearing, switching to a vegetarian or vegan diet could reduce the footprint down to 4.9 kgP/cap·a and 0.9 kgP/cap·a. A switch to a local-only diet, though, would raise the footprint up to 12 kgP/cap·a. Since the ultimate goal of circularity is the absolute decrease of resource use, this means that an urban food system based on local-only food would be less circular than the current one. At the same time, the secondary-to-total input ratio of the local diet (62%) is higher than the current value (47%), because manure, rather than synthetic fertilizer, is the main source of P in Belgian agriculture.

In order to operationalize the work carried out throughout this thesis, the focus was placed gradually on one of three scales: city, country, and global hinterland. In the rest of this section I am attempting a synthesis of the work while coming back to the overall research questions.

6.1.2 What metrics to use to assess circularity in urban food systems?

Different sets of metrics to assess circularity were tested in the main chapters of this thesis:

(i) Three different recycling rates (RR) at the city scale in chapter 2: a closed-loop RR ("Food Circularity"), an open-loop RR ("Weak Circularity") and a rate of reuse within the city boundary ("City Circularity").

(ii) A set of five metrics to assess agro-food system circularity at different levels[7] in chapter 3 that included: total input, phosphorus use efficiency, losses, recycling rate and the secondary-to-total input ratio.

(iii) The P-footprint in chapter 5, i.e. the total indirect P input into the city, differentiated into primary (fertilizer) and secondary (manure) input.

Each application clarified different aspects related to the circularity assessment of the system. There are two main components in circularity as conceptualized in this study: the end goal, i.e. the absolute reduction in total resource use, and the means through which the goal

[7] As a recap: "scale" refers to spatial scale, the geographical aspect of the system boundary, e.g., city scale, national scale; "level" refers to hierarchical system level, the functional aspect of the system boundary, e.g. whole system or sub-systems such as agriculture, consumption or waste management.

is achieved; closing, narrowing and slowing down nutrient loops, i.e. using secondary instead of primary resources. Keeping these in mind, as well as the usefulness of working at different system levels discussed in chapter 3, circularity assessments of urban and regional food systems should contain at least:

- The **total absolute input**, direct and indirect, and its distribution into primary and secondary inputs;
- The **secondary-to-total ratio** of inputs into agriculture, as an indication of the extent of secondary resource use;
- The **closed-loop recycling rate**, indicating the extent to which waste generated in the system is valorized as a secondary resource, and
- The **rate of system losses**, including emissions into the environment, e.g. nutrient surpluses into soil, and the nutrients in waste streams that are either disposed of in landfills, or reused in ways that do not profit from their nutrient content, e.g. incineration or production of bio-plastics.

This set covers the most important aspects of circularity in food systems as defined in the introduction (see also section 3.2.2) and is fairly simple to implement and communicate, an important characteristic when it comes to negotiating and collaborating with stakeholders. Shortcomings and potential improvements are discussed in section 6.2.2.

6.1.3 At what scale to analyze and measure circularity of urban food systems?

Each scale and level of analysis offers different insights into the system's function and involves different implications for circularity.

The **city scale** is the most useful for zooming into consumption and waste generation. Data at the city level can allow for the detailed estimation of consumption by inhabitants, visitors, and commuters, and of waste and wastewater generated. In this way, a city scale analysis is a good starting point to assess the indirect inputs and estimate recycling rates.

The **national or regional scale**, where the whole country or a large surrounding region is considered to be the city's hinterland, is perhaps the most relevant from a circularity assessment perspective, because it includes agricultural lands. It is these lands that produce a variety of food consumed in the city and generate the demand for secondary nutrients. In addition, many of the most important datasets needed for a nutrient flow analysis or a circularity assessment are available at the national and regional scales, e.g. agricultural census and trade data. Finally, it is at this scale that environmental policies are drawn and environmental indicators, such as orthophosphate and nitrate concentrations in water bodies, are monitored. To be able to account for sub-national differences in geographies and production systems, though, an appropriate disaggregation into lower spatial scales is a necessary component of a country-scale study.

The **global scale**, corresponding to a footprint approach, offers the additional insight into indirect flows and an overview of the city's actual hinterland. Nonetheless, it is generally unrealistic to analyze all production system contributing to the nutrient footprint of a city in equal detail.

As it is often the answer in research, there is not one definitive and universal answer to the question "what scale to look at?". It will certainly have to extend outside of the city's boundaries, to include the places where the food is produced. A global perspective gives valuable insights into the actual places affected by urban consumption, but requires lots of data and a great effort to produce insights that are often not possible to control at the city or regional level. A national or regional scale seems to be the most appropriate one, at least for dense and intensively cultivated regions such as Belgium.

The analysis of nutrient flows at gradually expanded scales offered insights on how different territorial aspects play out at those scales. In chapter 5 it was shown that agricultural systems and nutrient inputs across the world are mobilized to provide Brussels with food. These indirect flows coming from far away will unavoidably be unidirectional: this loop will never close. At the same time, and as long as no food can be produced within the city boundaries, there is no alternative to unidirectional nutrient flows. In that respect, Brussels exemplifies the notion of the city as a parasite: mobilizing large amounts of resources in the hinterland, having nothing to offer back but waste (Barles, 2010; Buclet et al., 2015). Yet there are two implicit assumptions necessary to reach this conclusion, both of which were questioned in this thesis: first, the idea that nutrient streams flowing out of the city are waste; second, and most importantly, the idea that the city ends at its administrative boundary. In chapter 2 it was argued that the phosphorus contained in Brussels' effluents matches the P applied through mineral fertilizers on agricultural lands in the two neighboring Brabant provinces. Analysis in chapter 4 further indicated that nutrient supply through local re-sources in these two provinces (manure and human excreta) are lower than the crop demand. There is therefore a theoretical potential for a tighter exchange of nutrients between Brussels and its immediate hinterland (the Brabant provinces), and for establishing a more circular food system at this expanded city-region scale.

The potential is, however, only theoretical. Nutrient flows crossing the administrative boundary of BCR into the Flemish and Walloon Brabant provinces would be liable to different regulations, as described in chapter 2 and discussed in chapter 4 : a complete ban of importing "waste" flows into Wallonia, and the absence of sewage sludge land application in Flanders. The administrative status of BCR as one of three Belgian regions has been one of the main reasons that this study has been possible, thanks to the type of data gathered and made available at the regional level. On the other hand, the regional boundary isolates Brussels from the other two regions, and the fact that the city virtually borders both Flanders and Wallonia complicates things further. We see, thus, how the analysis and planning scales that could make sense from a circularity or food system perspective are supra-regional (national, multi-province), while it is almost impossible to implement solutions at these scales, due to regulatory, governance, or cultural barriers. Upcoming changes in European legislation, such as the new Fertilizers Directive (EC, 2018b) and the update of the Sewage Sludge Directive 86/273/EEC may smoothen some of these barriers. Nonetheless, more radical changes will be needed in order to harvest the full potential of local nutrient re-sources: spreading out livestock production to "free up" demand for urban nutrients, for example, or re-configuring City-Regions and granting them the autonomy to plan their agro-ecological food systems (Taylor Buck and While, 2020; Vaarst et al., 2017).

6.2 Implications

6.2.1 Contribution to the literature

Results and methods from this thesis extend and complement mainly two fields of research: the industrial ecology strand on spatially explicit and multi-regional nutrient flow analysis, and the interdisciplinary literature on circularity assessments and metrics for food systems. The most important contributions include:

(i) **A set of metrics to measure (urban) food system circularity** tested at different levels and scales. While research on circularity metrics and assessment frameworks has been flourishing recently, e.g., (Moraga et al., 2019; Parchomenko et al., 2019; Pauliuk, 2018), attempts on addressing food system circularity are still scarce (Fernandez-Mena et al., 2020; Helenius et al., 2020). This thesis contributes a small set of simple metrics based on P flows that cover the most important aspects of circularity in food systems: closing, narrowing and slowing the nutrient loops through the reduction of resource input (Total Input and Nutrient Use Efficiency), minimizing wastes and losses (Losses), and increasing the inputs of recycled materials (Secondary-to-total Input and Recycling Rate). The thesis further offers a case to conceptualize circularity as a means towards achieving absolute reduction in resource use, and thus proposes total direct (chapter 3) and indirect (chapter 5) inputs as the main indicators in food system circularity assessments.

(ii) One of few **systematic food and nutrient flow analysis of an urban food system at multiple scales**. Although there is a wealth of research on food and nutrient flows in Paris and its hinterland at different scales e.g. (Chatzimpiros and Barles, 2013; Esculier et al., 2018; Tedesco et al., 2017; Verger et al., 2018), it has been carried out in several separate studies, each with its own rationale and focus. This thesis contributes a study that deals with a system at successively expanded scales, with a constant focus on circularity, and can thus offer insights into tensions and opportunities across and within scales.

(iii) A **detailed case study on nutrient flows in the whole of Belgium**, with a focus on the different possible interactions between Flanders and Wallonia. The two regions are typically studied separately (Coppens et al., 2016; Renneson et al., 2016; Tsachidou et al., 2019; van der Straeten et al., 2010), however, addressing nutrient cycles at the federal or intra-regional level could potentially foster circularity and improve nutrient management in both regions. The analysis pointed to the need to challenge imposed system boundaries (e.g. geographical, administrative) and revealed some implications of adopting a territorial approach in food system planning in Belgium.

(iv) A **methodology to assess a resource-based P footprint**, that takes into account inputs from secondary resources. MRIO models, upon which footprint studies are usually based, now include nutrient emissions in their environmental extensions (Zoran J.N. Steinmann et al., 2017; Tukker et al., 2014), enabling the development of several emission-based P footprints, e.g. (Li et al., 2019; Metson et al., 2020). Yet only two studies so far have quantified the embodied P of whole diets, adopting a resource-use, rather than an emission-based perspective: one for the USA (Metson et al., 2012), and one for the EU (Nesme et al., 2016). Both of these studies only account for synthetic fertilizer, though, while chapter 5 also includes the net use of manure. The inclusion of manure is pivotal from a circularity perspective because most P inputs into food production come from manure (almost 60% of indirect P inputs in the case of Brussels). Additionally, the exclusion of manure fails to account for the fact that food systems are not only nutrient consumers, but also nutrient producers. Although manure is often treated as a waste stream, and sometimes it is, it is also local, renewable nutrient source that should not be disregarded when analyzing the circularity and resource use of food systems.

6.2.2 Limitations of the study

The down side of making a study that spans different scales and levels, is **losing some detail and depth** in other aspects. The multi-scale approach required a great variety of data coming from different sources, and a great effort to gather and harmonize them. Given the time and resource constraints, other important aspects of the food system circularity had to be sacrificed, most notably:

- the **dynamic aspect**. A multi-year approach would offer a better insight into the drivers of the different flows and their development in time, as well as a better integration of environmental aspects that unfold over long periods of time, such as soil fertility and nutrient concentrations in water bodies. An important omission in this regard is excluding legacy P, an important yet under-studied resource towards better phosphorus management in Europe (Bouwman et al., 2017; Le Noe et al., 2020; Withers et al., 2019, 2014).
- the **socio-economic aspect**. Despite my initial ambition of adopting a territorial ecology approach, it has not been possible to delve deeper into the territory aspect in this thesis. Information flows, power relations, and social outcomes are some of these territorial aspects that have not been addressed (Barles, 2017), and would greatly enrich the analysis of Brussels' food system, its circularity, and relationship to the hinterland.

In addition, the patchwork of input data means that there are **discrepancies in their quality** and assumptions replaced hard data in several instances while developing the model. Consequently, results come with an inherent **uncertainty** that readers should carefully consider when interpreting the results. This kind of uncertainty is shared among such studies at high system levels that rely mainly on national statistics, i.e. secondary data. Coupling high level analyses with plot and farm level ones, where primary data can be collected through soil testing or monitoring fertilizer inputs, for example, can increase confidence in high level studies and complement them with the farmers' point of view.

A further point of concern could be the simplicity of the set of metrics and the reasoning behind the choice of each indicator. The **set includes 5 P-based metrics, which may appear too simple** to represent a reality that is too complex. Yet reality is far more complex than any metric will ever be able to capture. Metrics that pretend to reduce this complexity into a list of numbers are not the right tools, no matter how long the indicators list, how fancy the equations, or how high the input data requirements. A reasonable approach is to acknowledge the complexity and address it in a transdisciplinary approach including stakeholder involvement, visioning exercises, and knowledge co-creation (Lang et al., 2012). A simple set of simple metrics can provide the platform for transdisciplinary collaboration. If stakeholders and scientists from other fields can easily grasp the metric, they will be able to work with it, use it in their own analysis, and enrich it with their own insights. An overcomplicated metric, on the other hand, that can only be understood and applied by its developer, will remain idle and ineffective.

In this study I chose to work **only with theoretical best case scenarios**, to test the application of the different metrics, making the implicit assumption that the full amount of P in waste streams could be reused. 100% recycling is, however, impossible, because losses will occur in several parts of the treatment process and transport. Further, P recycling does not simply mean diverging the amounts currently landfilled onto agricultural fields. It will further require a coordinated effort including the choice of treatment technologies, the installation of treatment plants, the location of the reuse sites and the organization of the logistics, while navigating between different authorities and existing practices and infrastructures (e.g. the manure digestion plants in Flanders). In addition, nutrients in synthetic fertilizers are considered to be more easily available to plants and therefore not substitutable 1:1 by organic fertilizers. This can be true for some types of organic fertilizers, however, recent research reports substitutability in the range of 1:1, especially for treated sewage sludge and ruminant and pig manures (Hamilton et al., 2017; H. Kahiluoto et al., 2015).

6.2.3 Research outlook

The limitations of this study listed above present opportunities for future research, the most important of which are:

(i) Integrating the set of circularity metrics into a broader framework that will assess circularity at a higher level, including for example socio-economic aspects or **a holistic vision of the Circular Economy for cities and food systems.**

(ii) Expanding the current steady state study into a **multi-year analysis** of the food system; accounting for the historical development of the food and nutrient flows and **incorporating stakeholders' views**, in an attempt to gain a deep understanding of the system, its evolution and territorial aspect. Such an approach could include the **co-creation of realistic scenarios** together with stakeholders, and back-casting exercises to promote concrete actions.

(iii) Testing the approach adopted here in **different case studies**, and setting a standard or **common framework for measuring circularity** that will enable cross-city comparison and promote the harmonization and comparability of data and models.

(iv) Developing and making widely available a **dataset on global manure use**, that will be based on actual manure application rates and practices in different parts of the world. Such a database would be ideally easy to combine with data on synthetic fertilizers (IFA, FAOSTAT) and will therefore make footprint-like approaches more accessible to more researchers.

6.2.4 Implications for policy and action

- *A circular food system for Brussels and Belgium*

If I was asked to write my thesis as a single sentence, this sentence would be: **reduce manure, eliminate food waste, reuse human**

excreta. Great effort and thought goes into food waste in all three Belgian regions: authorities have been quantifying food waste flows in detail (e.g. differentiation on avoidable and unavoidable food waste in Flanders (Roels and Van Gijseghem, 2017)), setting standards for their reduction, or debating the most appropriate treatment technology for them (Bortolotti et al., 2018b). All these efforts are of course useful, necessary and motivated by European legislation. What is remarkable though is the absence of discussion on the valorization of human excreta, despite their higher nutrient content compared to solid organic waste, and the comparable advantage of using treated sewage sludge as organic fertilizer, rather than composted organic waste (Hamilton et al., 2017; Trimmer et al., 2019). Researchers have been calling for the **reframing of human excreta as an integral part of the food system** (Harder et al., 2020), and as human-derived resources (Trimmer et al., 2020), an approach that would benefit nutrient management and food system circularity in Brussels and Belgium.

A further obstacle towards increased nutrient and food system circularity is the isolation of the Belgian regions that hinders the exchange of secondary nutrient flows between the regions. Shifting legal and logistical barriers, however, and **addressing food system circularity in an intra-regional approach** has a greater potential to properly address challenges related to food system circularity than each region trying to achieve it within its own boundaries. Ultimately, though, a re-distribution of the manure produced in Belgium can only make sense if the livestock production system becomes better embedded in the local context. As long as the system is heavily depended on imported feed, excessive amounts of manure will be produced, nutrient budgets will remain high and nutrients from urban effluents will always be in disadvantage (Clercq et al., 2015), ending up in ashes, landfills, and water bodies.

- *Circularity towards a more sustainable food system*

Throughout this thesis I worked with nutrients, and even more so with phosphorus, to analyze and understand food systems. The motivations for this choice are explained in the introduction: nutrients make good trace elements for material flows in the food system, and constitute, together with biomass, the backbone of reuse in food systems. Nonetheless, such a choice remains a reductionist one made to operationalize the analysis. On the one hand, nutrient content alone is often not enough to warrant that a material flow is adequate for reuse, and specific properties of a nutrient re-source need to be taken into account when deciding if and how to reuse it. Manure, for example, is not a uniform type of material. Especially manure from intensive pig or poultry farming is a liquid slurry that cannot be reused in agriculture without proper treatment (García-Albacete et al., 2012; Weidner et al., 2019). Urban sewage sludge can contain heavy metals and other contaminants that accumulate in the soil (Alloway and Jackson, 1991; Bisschops et al., 2019), and organic fertilizers are often considered to have a lower relative agronomic efficiency than synthetic ones. Nonetheless, new studies increasingly show that organic fertilizers have comparable agronomic efficiencies with synthetic and mineral ones (Huygens and Saveyn, 2018; Lemming et al., 2017; Vaneeckhaute et al., 2016), on top of a series of other benefits, such as providing the soil with multiple nutrients and organic matter to improve its structure and overall quality (Harder et al., 2019; Toffey and Brown, 2020; Withers et al., 2014). In addition, advances in treatment processes and alternative management practices for human excreta, e.g. source separation, can help address contaminants such as pathogens and heavy metals (Amann et al., 2018; Bisschops et al., 2019). Newly revised legislation on the use and trade of organic fertilizers (EC, 2018b), as well as the upcoming revision of the Sewage Sludge Directive 86/278/EEC, reflect these new developments and will further promote and facilitate the reuse of secondary nutrients, and hopefully the reduction in the amounts of synthetic and mineral fertilizer produced and used.

Some of the concerns about secondary nutrient sources are inherent to them, e.g. the pathogen content. Most, however, are the result of the system that produces them: heavy metals in sewage sludge, for example, comes from industrial wastewaters, or from rainwater and runoff in combined sewerage systems; high amounts of difficult to valorize manure concentrated in specific locations originate from the existence of intensive livestock farms that rely on imported feed. As a result, we cannot address food system circularity by focusing on resource reuse alone. Ultimately, we cannot address food system sustainability by focusing on circularity alone. A more circular food system may be preferable than a linear one, but looking at circularity alone entails the danger of achieving a false sense of accomplishment and neglecting other major issues of the food system: energy use, for example, and the resulting GHGe; land use and its effect on biodiversity; the great power imbalance in food supply chains, where few agri-food businesses make the rules of the game; the problem of hunger still prevalent in an otherwise affluent world and the increasing rates of obesity. Envisioning a future sustainable food system will have to take into account all these different aspects and the systemic interaction and feedbacks between them.

- *Circularity for a more sustainable and just society*

While the concept of decoupling economic growth from resource input and environmental degradation with the help of the CE proved to be sticky and popular, taking up a big part of the sustainability and ecological transition debate in the last years, academics from different fields have been critical. Criticism mostly focuses on three aspects that are missing or are problematic: (i) the bio-physical constraints of recycling, especially the fact that no perfect and perpetual closing of the loop is possible due to the second law of thermodynamics that dictates that recycling needs energy inputs and results in materials and products of lower quality (Giampietro, 2019; Korhonen et al., 2018); (ii) the lack of attention to possible rebound effects that CE strategies may have, i.e. when reused resources do not replace virgin ones but rather lead to additional production (Korhonen et al., 2018; Zink and Geyer, 2017); and (iii) the sole focus on *closing* the loop (recycling) while neglecting other important aspects like narrowing (locality) and slowing (sufficiency) the loop (Korhonen et al., 2018; Moreau et al., 2017). Such critiques conclude that just because a system or a product is more circular (mostly meaning its content its recycled) it is not necessarily more sustainable too (Blum et al., 2020) and that environmental impacts should always be assessed to avoid trade-offs.

Additionally, much of the academic and activist world has joined their voices to critique the eco-modernist and technocratic vision of the mainstream CE narratives (Calisto Friant et al., 2020; Giampietro, 2019; Hobson and Lynch, 2016). These narratives present the CE as the silver bullet that will finally achieve the decoupling of GDP growth from resource consumption and pollution, and thus allow our society's quest for perpetual economic growth and unlimited consumerism. In this regard, even though the CE is being served as a disruptive and transformational world vision, it is in its roots nothing more than another business-as-usual scenario (Clube and Tennant, 2020), a weak sustainability vision where a combination of better recycling technologies, digitalization and "new" business models, such as the sharing economy and the service economy, are enough to bring about transformational change. Nonetheless, nothing transformational can happen without a thorough rethinking of the current economic and political system that produces so much wealth at the expense of the majority of the people living on this planet today, those still to come and the planet itself. What we need is a Transformational Circular Society more than anything else (Calisto Friant et al., 2020; Jaeger-Erben et al., 2021).

When I started working on the Circular Economy five years ago, the concept was just starting to gain its now extraordinary popularity. The EMF report "The Growth within" had just been published (EMF, 2015a), as had one of the first academic papers conceptualizing the CE (Ghisellini et al., 2016); only months before, the European Commission had launched its very first action plan (EC, 2015). Papers, reports, conferences and events on every possible aspect of the CE started appearing everywhere. Critical voices saw the Circular Economy as just another attempt to greenwash and a distraction to the fact that the carefree green growth that many CE reports were implying is a delusion, because perpetual growth is a delusion. Yet I often have the impression that we just put too much expectation on the Circular Economy, and so we doomed it to fail us. Although recycling will not save the world indeed, I find the idea of circularity to be rooted on a strong sustainability vision, carrying undertones of sufficiency and symbiosis, and having an earthly connection to the circle of life. For these reasons I think circularity can be very much a part of the bundle of solutions to our Anthropocenic problems, along with degrowth, doughnut or well-being economics, commoning, renewable energy sources, or making extreme wealth illegal. We just need to keep hoping and envisioning and working towards a future that will be more just, more caring, and more circular too.

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List of publications

Peer-reviewed publications

Papangelou A, Achten W M J and Mathijs E (2020) Phosphorus and energy flows through the food system of Brussels Capital Region *Resour. Conserv. Recycl.* **156** 104687 Online: https://doi.org/10.1016/j.resconrec.2020.104687

Bortolotti A, Kampelmann S, De Muynck S, **Papangelou A** and Zeller V (2019) Conditions and concepts for interdisciplinary urban metabolism research–the case of an inter-project collaboration on biowaste *Flux* **116–117** 112–27 Online: https://www.cairn.info/revue-flux-2019-2-page-112.htm

Papangelou A and Mathijs E. (2021) Assessing agro-food system circularity using nutrient flows and budgets. *J. Environ. Manage.* **288**, 112383. https://doi.org/10.1016/j.jenvman.2021.112383

Papangelou A, Towa E, Achten W M J and Mathijs E (2021) A resource-based phosphorus footprint for urban diets *Environ. Res. Lett.* **accepted**

Papangelou A and Mathijs E (2021) *The potential of reused nutrients to cover crop needs in dense livestock-dominated regions*. **Submitted**. https://ees.kuleuven.be/bioecon/working-paper-series/

Contributions to international conferences

Papangelou A., Achten W.M.J., Mathijs E. (2019): Towards circular urban food systems – phosphorus and energy flows in Brussels Capital Region. IPW 9, ETH Zürich, CH, 8.-12.07.2019 (*poster presentation and invited poster pitch*).

Papangelou A., Achten W.M.J., Mathijs E. (2019): A multi-resource assessment of circular solutions for the food system in Brussels Capital Region. 13th ISIE-SEM Conference, Berlin, DE, 13.-15.05.2019 (*oral presentation*)

Papangelou A., Mathijs E. (2018): Circular Economy for Urban Food Systems. Comparing solutions for Brussels Capital Region. 1st Life Cycle Innovation Conference - LCIC 2018, Berlin, De, 29.-31.08.2018 (*oral presentation*)

Papangelou A., Mathijs E. (2018): Closing the loops in urban food systems. Phosphorus flows between Brussels and its hinterland. GRC Industrial Ecology, Les Diablerets, CH, 20-25.05.2018 (*poster presentation*)

Outreach activities (selection)

Axinte L, Moriggi A, Nieto Romero M, **Papangelou A**, Pearson KL, Vasta A (2020) *Once Upon the Future – Everyday adventures that change the world.* A children's book on sustainability with six stories inspired by our research; forthcoming in 2021 by Babidi-bú libros.

"Once Upon the Future", 22-24.09.2021, online. Participation to **Science is Wonderful 2021**, the yearly science fair of the Marie-Sklodowska-Curie Actions during the European Research & Innovation Days.

"A Belgian Odyssey of phosphorus", 08.09.2020, online. Presentation for a general audience during **the Pint of Science 2021** festival.

"How poo connects humans to the universe", 19.02.2020, Brussels, Belgium. A talk during the **TEDxYouth @EEB3** (European Brussels School 3) event.

"The Circle of food". Three-minute video pitch of my research in the **Science Figured-Out** platform.

"Orange bags and purple carrots – the story of organic trash in Brussels". 14.06.2019, Brussels, Belgium. An informal talk on organic waste management in Brussels.

"Fall in love with Science", 14.02.2019, Brussels, Belgium. Participation in a "speed-dating" evening where the public can meet and chat with scientists.

Curriculum Vitae

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PROFESSIONAL EXPERIENCE

04.2016 – 03.2019	**Marie-Sklodowska-Curie ITN Research Fellow** Department of Earth & Environmental Sciences, KU Leuven, Belgium SUSPLACE project (Sustainable Place-shaping)
02.2014 – 12.2015	**Project Coordinator & Research Associate** Institute for Water, Wastewater &Resources (IWAR), TU Darmstadt, DE CuveWaters project – Integrated Resources Management in Central-Northern Namibia
10.2012 – 05.2013	**Research Assistant** Chair of Ecological Systems Design, ETH Zürich
09.2011 – 12.2011	**Intern** EAWAG – Swiss Federal Institute for Water Research, Cairo, Egypt ESRISS project – Egyptian-Swiss Research on Innovations in Sustainable Sanitation

EDUCATION

2016 – 2021	**PhD in Bioscience Engineering** KU Leuven, Belgium
2019 – 2021	**Post-grad Certificate**, Science Educ. & Comm. KU Leuven, Belgium
2010 – 2013	**MSc in Environmental Engineering** Swiss Federal Institute of Technology (ETH), Zürich, Switzerland
2004 – 2009	**Diploma (MEng) in Civil Engineering** Aristotle University of Thessaloniki, Greece

LANGUAGES

Greek, English, German, French