Current and potential recycling of exogenous organic matter as fertilizers and amendments in a French peri-urban territory

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Title

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Declaration of interest

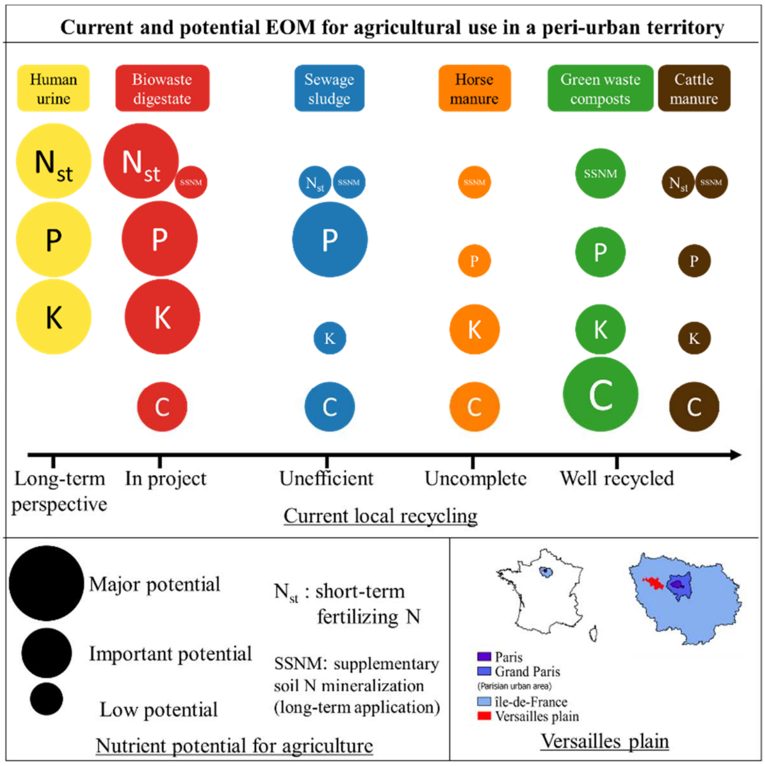
None.

Abstract

Animal breeding, industries and cities produce byproducts and wastes that can be recycled in agriculture as exogenous organic matters (EOMs), promoting soil carbon storage and nutrient recycling. Peri-urban areas show a strong potential for recycling urban EOMs in agriculture. This potential needs to be better quantified in regard to farmer fertilizer demand. We investigated this question in a peri-urban territory, namely, the Versailles plain, which is located near Paris, France. We coupled interviews with farmers and waste managers with regional datasets to identify accurate flows of organic matter and their associated nutrients. The EOM potential for C storage was assessed with the AMG soil carbon model. Potentials for P, K and (short-term) N fertilizer substitutions were based on fertilizer equivalent coefficients. Additionally, we assessed the potential additional mineral N release related to increased soil organic matter after repeated EOM applications. For social and technical reasons, local EOM recycling was unequal; it was high for manures and compost and low for sewage sludge. Inputs of efficient C into soils due to EOM recycling corresponded to 20% of the inputs from crop residues. Current EOM use met 44% and 50% of the crop P and K demands, respectively. The potential for N fertilizer substitution was very low but increased when considering the additional mineral N available after long-term EOM applications. EOM could reach higher

potential, especially when considering the anaerobic digestion of biowaste and potential source-separation of human urine (up to the whole P and K demand and half of N demand).

Graphical abstract



Keywords

Organic waste management

Peri-urban agriculture

Nutrient recycling

Soil organic carbon storage

1 Introduction

The FAO has stated that agriculture is facing three main challenges worldwide (FAO, 2020): increasing productivity and incomes, adapting to climate change and decreasing its greenhouse gas emissions. Among other practices, the recycling of exogenous organic matters (EOMs) can contribute to a more sustainable agriculture. EOMs are organic byproducts and wastes from animal breeding, industries, and cities that are spread on agricultural soils for their amendment or fertilizing value. When used as soil amendments, EOMs promote soil carbon storage, increasing soil fertility and crop production (Diacono and Montemurro, 2010). Through their use as fertilizers, they decrease agricultural dependencies to nonrenewable resources such as P or K mineral fertilizers (Akram et al., 2018; Caniani et al., 2019). The substitution for N fertilizer (Gutser et al., 2005) avoids the production of mineral N fertilizer through processes with high energy costs. As nonchemical fertilizers, EOMs are necessary for organic agriculture (Nowak et al., 2013). Research remains necessary to maximize those desired effects while avoiding potential environmental issues and considering all the diversity of EOMs.

Several authors have estimated EOM resources in different contexts to assess either their potential for agricultural use (Jia et al., 2018) or their environmental impacts (Le Noë et al., 2017; Lwin et al., 2017). Looking at the agricultural sector, animal effluents are the main EOMs. In China, Jia et al. (2018) showed that manure production was of the same order of magnitude as crop residue production. The fluxes of nutrients in animal effluents can be of the same order of magnitude as inorganic fertilizers (Coppens et al., 2016), as shown for P (Klinglmair et al., 2017; Senthilkumar et al., 2012). Manures are also potential resources for biogas production at national scales (Burg et al., 2018; Chávez-Fuentes et al., 2017).

In contrast, organic wastes originating from urban areas are abundant in densely populated areas and yet little recycled. Currently in France, one-third of agro-industrial wastes, 15% of household wastes, and 70% of sewage sludge are used in agriculture (Houot et al., 2014). Biowastes are another resource for biogas production (Guo et al., 2019), and digestates are recycled in agriculture with positive environmental impacts (Ascher et al., 2019; Bacenetti et al., 2016; Tsydenova et al., 2019). Sewage sludges show a strong potential for P recycling but face environmental and social barriers, including the question of heavy metals becoming (Lwin et al., 2017). The separate collection and reuse in agriculture of human urine has been studied as a substitute for mineral fertilizer (Karak and Bhattacharyya, 2011) and to close the nutrient loop in agriculture (Esculier et al., 2018; Trimmer and Guest, 2018). The potential of these urban EOMs is particularly important in peri-urban areas, close to dense populations with large waste emissions. Peri-urban areas import nutrients (Bittman et al., 2019, 2017) and produce high quantities of urban EOMs; green wastes, sewage sludge and the organic fraction of municipal solid waste, which can be composted or anaerobically digested. Moreover, the presence of farmland is an interesting outlet for those EOMs. Waste recycling is important to optimize urban metabolism (Agudelo-Vera et al., 2012) and to close nutrients loops, which can decrease the impacts of agriculture (Wielemaker et al., 2018). Waste recycling is also a way to recouple agricultural systems and cities, a key component in the future of peri-urban sustainable development (Petit et al., 2017; Tedesco et al., 2017). Assessing the potential of urban EOMs in peri-urban territories is thus a research question of interest.

However, estimating the relevant potential use of EOMs in agriculture does not require only the quantification of their nutrient contents. First, the potential for nutrient recycling of EOMs has to be compared to crop nutrient demands (Akram et al., 2018; Caniani et al., 2019), which can vary among territories (Trimmer and Guest, 2018). This comparison helps to estimate realistic spreading amounts and to avoid nutrient excess with large negative environmental impacts. The specific spatial organization of each territory has also to be considered. It conditions the distance between treatment plants and agricultural areas, given that several urban EOMs (e.g., composts or digestates) cannot be economically transported on long distances. Some local policies may also impact organic waste treatment types (Bahers and Giacchè, 2019), and thus the type of available EOMs. Finally, local social and technical barriers and opportunities influence EOM related practices, e.g., for sewage sludge or urine derived products (Case et al., 2017; Joncoux, 2013; Simha et al., 2017). Additionally, the use of urban EOM will often depends on livestock density as animal effluents are usually preferred to urban EOMs. Thus, it is important to study EOM potential at a relevant territorial scale considering all these aspects and to understand the processes leading to potential local EOM recycling. Few studies focus specifically on EOM potential at this territorial scale while integrating the question of EOM demand by farmers.

The aim of this study was to quantify the proportion of the current and potential recycling of organic wastes in agriculture that satisfies the crop requirements and contributes to soil carbon storage at the scale of a peri-urban territory. The case study was the Versailles plain, located west of Paris, France.

2 Materials and Methods

In this study, we built and analyzed a material flow analysis of organic products and wastes emitted or used in a peri-urban territory, in terms of mass, nutrients, and organic matter. For that purpose, we used interview of farmers and waste managers, regional datasets from other studies, and modelling.

2.1 Study area

The Versailles plain (VP) is a territory of 27 cities located 30 km west of Paris (Figure 1-A). It is delimited by the cities involved in the Versailles Plain Association, which is composed of municipalities, farmer representatives, and diverse local actors, whose goal is to promote a coherent agro-urban territorial project. It included 142,826 inhabitants on 23,666 ha in 2017 (DGCL, 2017; OpenStreetMap, 2018). The VP is characterized by urban cities with concentrated populations, with few agricultural activities in the southeast of the VP, and more rural cities with less dense populations, where agriculture has a greater importance, in the western part (Figure 1-B and 1-C). The VP is bordered by the "Grand Paris" metropole on the east, which represents more than 7 million inhabitants. In this strong peri-urban structure, it is interesting to study the coupling between cities, which are organic waste producers, and agriculture, which is a potential outlet for them.

The agricultural lands represented 13,162 ha (56% of the total area) in 2014 and were mainly composed of cereal croplands: winter wheat (48% of the used agricultural area – UAA), rapeseed (19% of the UAA), barley (10%), and grain maize (8%). A minor area was dedicated to grasslands (4%), market gardening (2%) and arboriculture (0.5%) (Agence de Services et de Paiement, 2015). Animal breeding was scarce, excepted equine breeding. The livestock density (Eurostat, 2013) on the VP was 0.1 livestock unit per hectare UAA. Agriculture was mainly conventional, with recent development of organic agriculture (from 5% of the UAA in 2014 to 10% in 2019).

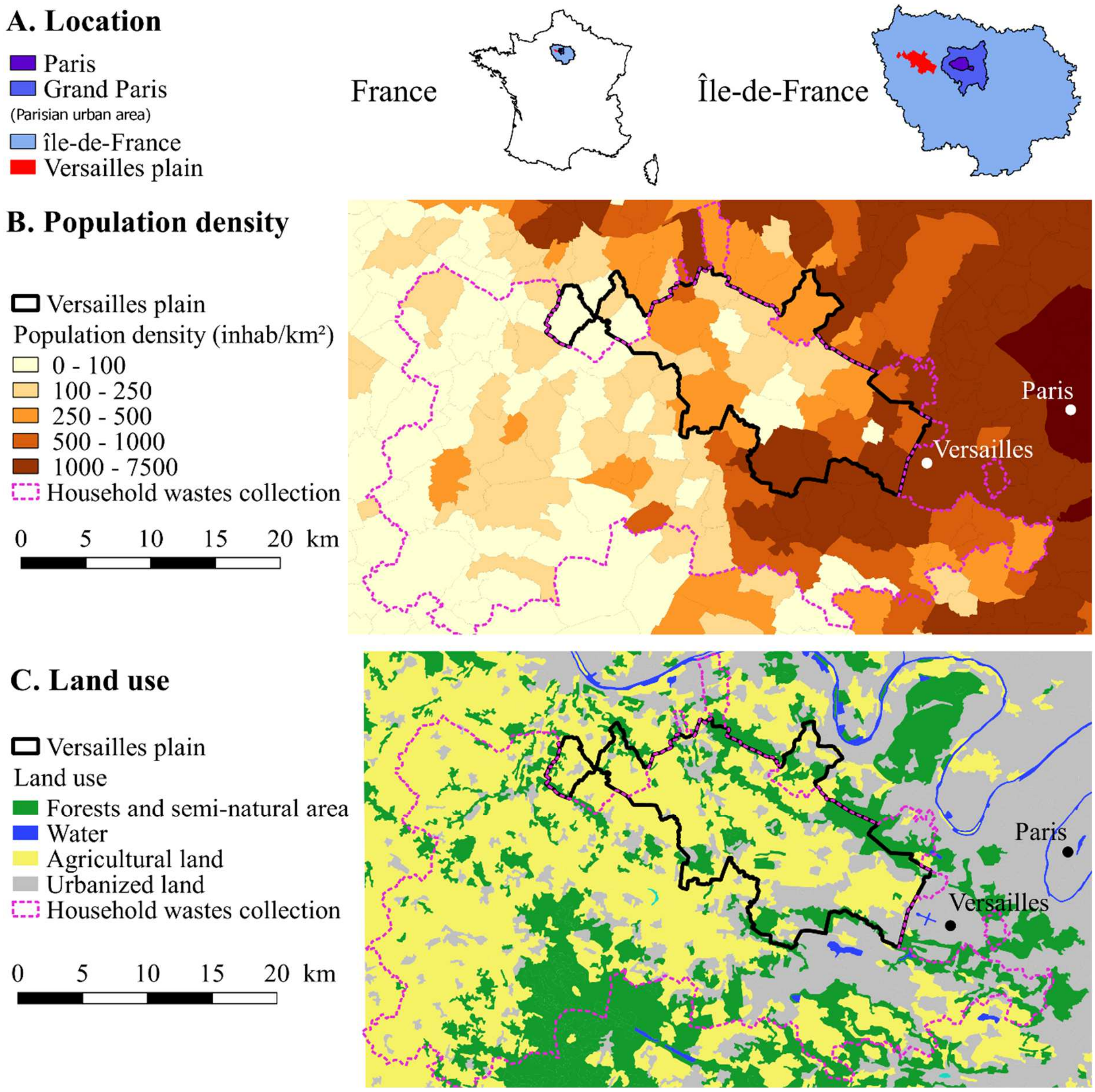


Figure 1. Presentation of the study area, the Versailles plain: (A) Location in France, (B) population density, and (C) land use. The incinerator on the VP serves the territory delimited in pink.

2.2 Data acquisition

2.2.1 EOM use on the Versailles plain

We investigated agricultural practices associated with EOM use on the VP through semi-directed interviews with farmers. The interview guide was constructed as 4 parts to understand local strategies for fertilization and soil amendment: (1) use of EOMs, expected benefits, and issues, (2) potential interest in EOMs that farmers were not currently using, (3) soil fertility perception and fertilization practices, and (4) other farming practices. We interviewed 25 cereal farmers and 4 arborists and market gardeners, representing together 40% of the UAA. A second set of preexisting 20 interviews (additional 20% of the UAA) gave us partial information about farmer practices and the use of EOMs (Cormier, 2018).

We extrapolated the EOM use by the interviewed farmers to the whole VP. We assumed that the average uses of each EOM (in t ha^{-1} $year^{-1}$) were the same for interviewed and non-interviewed farmers for each of the following types: conventional cereal farmers, organic cereal farmers, organic arborists, conventional arborists, organic market gardeners and conventional market gardeners. As a last type, breeding farmers had specific farming practices and were all interviewed, they were thus not concerned by data extrapolation.

We estimated a potential digestate demand (in ha), extrapolating to the whole VP the proportion of the area cultivated by the interviewed farmers interested in digestate.

2.2.2 Organic waste production, management, and treatment

We quantified EOM fluxes within the VP and across the VP limits to understand local waste management strategies. All EOMs that could be used in agriculture were investigated. We conducted interviews with local waste treatment stakeholders to collect locally accurate data and to understand EOM-related strategy. We used official regional databases when needed (Supplementary Table 1) and different hypotheses for flux computations (Supplementary Table 2, Supplementary Table 3).

2.2.2.1 Manure from horse and cattle breeding

We quantified approximately 750 cows, 620 sheep, and 300 goats in three different farms on the VP, together with 7000 chickens in five poultry farms. The farmers were all interviewed about their agricultural practices and manure treatment. Moreover, we used the horse manure quantification from Dhaouadi (2014), which found 23 horse establishments with 912 horses.

2.2.2.2 Sewage sludge production and management

Thirteen wastewater treatment plants served the VP, nine being inside its borders. Their size and sludge production were investigated through an official database (Ministère de la Transition Écologique et Solidaire, 2017). We led nine interviews with managers from urban communities and wastewater treatment plants to obtain information about sludge chemical characteristics, treatment processes, and the geographical destination of sludge.

According to the French legislation (*décret 97-1133 du 8 décembre 1997*, *arrêté du 8 janvier 1998*), sewage sludge spreading is controlled by official documents, called "spreading plans", based on sludge analysis, quantity and location of spreading. We used those data to quantify spread sludges issued from the VP wastewater treatment plants or spread on the VP but produced outside the VP.

2.2.2.3 Green waste and biowaste collection and treatment

Five waste collection operators (e.g., urban communities) were present on the VP and were interviewed. Green wastes were treated in 3 composting platforms, while all municipal solid wastes were incinerated in the same plant. There were also two anaerobic digester projects. We conducted interviews at all of these waste treatment plants to get information on the following:

- the waste management policy;

- the fluxes of green wastes, incinerated household wastes, and home individually composted biowastes [we deduced household biowaste fluxes by assuming that 28% of household wastes are biowastes (Observatoire Régional des Déchets d'Île-de-France ORDIF, 2017) (Supplementary Table 2)];

- the production and destination of composts.

Green waste and household waste emissions and treatments were cross-verified through 3 datasets of a regional agency (ORDIF). The total mass of industrial biowastes (agroindustrial wastes and wastes from catering and retail) at the city scale and their fraction considered to be potentially available for anaerobic digestion were taken from a study by ADEME (2013).

2.2.2.4 Potential for anaerobic digestion of biowastes

The potential production of digestate from two anaerobic digestion projects was assessed from manager interviews, as a first digestate scenario (S0). Additionally, we estimated the potential for digestate production if all biowastes were treated by anaerobic digestion. Digestate fluxes were computed using mass and nutrient balance between input biowastes and produced digestate found by

Schievano et al. (2011): 0.91 $tFW_{digestate}$ $tFW_{biowaste}^{-1}$, 0.28 $tDM_{digestate}$ $tDM_{biowaste}^{-1}$, 0.91 $kgN_{digestate}$ $kgN_{biowaste}^{-1}$, 0.94 $kgP_2O_{5digestate}$ $kgP_2O_{5biowaste}^{-1}$, 0.94 $kgK_2O_{digestate}$ $kgK_2O_{biowaste}^{-1}$, and 0.21 $kgC_{digestate}$ $kgC_{biowaste}^{-1}$. We designed three biowaste digestion scenarios, considering different hypothesis of biowaste availability and different catchment territories. The scenario (S1) considered the biowastes emitted on the VP and made a realistic hypothesis about the short-term available fraction of biowastes with regard to the efficiency of home sorting. The available fraction of biowastes (i.e., the realistic collected fraction) is approximately 40% of total household and industrial biowastes (ORDIF, 2017) (Supplementary Table 2). The scenario (S2) considered the same biowaste availability hypothesis as (S1), but the biowastes were emitted in a larger catchment area, corresponding to the actual area from which household wastes are collected to be incinerated on the VP. Indeed, the incinerator located on the VP collects biowastes from a territory 3 times bigger than the VP (468,268 inhabitants and 89,630 ha in 2016) (Figure 1). Additionally, we considered industrial biowastes emitted from the same area (i.e., the catchment area of the incinerator), considering it a coherent territory. The scenario (S3) considered 100% availability of biowaste emitted within the same large territory as (S2). (S3) was unrealistic but provided a maximal potential of digestate production.

2.2.2.5 Potential for human urine use

Source-separation and spreading of human urine has been proposed to close N loop, even if it is a long term perspective. To estimate the potential order of magnitude of this process, we estimated the nutrient fluxes produced in human urine incoming to the wastewater treatment plant on the VP. Two territories were considered: the cities located on the VP served by wastewater treatment plants on the VP, and all the cities served by plants on the VP (including cities outside VP). For each territory, the nutrient fluxes in urine were computed by multiplying the number of inhabitants served by the corresponding wastewater treatment plants, by the nutrient emission in urine per inhabitant per year; issued from Larsen et al. (2013) (Supplementary Material 2).

2.3 Data analysis

2.3.1 Material flows aggregation of wastes and EOMs

The previously described datasets allowed the quantification of waste fluxes, in mass. When a flux was described by different sources, we kept the highest confidence data, in the following order (Supplementary Table 1, Supplementary Table 3): data from the official spreading plan, accurate figures given by concerned actors, regional databases, order of magnitude obtained during interviews, and global estimation found in French literature. For each product, we used the mean flux of the years 2014 to 2018 as a tradeoff between year representability and recent figures. The hypotheses made to evaluate all the fluxes are summarized in Supplementary Table 2.

2.3.2 Wastes and EOM nutrient flows and EOM fertilization and amendment values

The chemical compositions of each EOM were assessed from analyses provided by interviewed managers or from analyses of these EOM that we performed in previous studies. When not available, e.g. for biowastes, green wastes, or poorly used EOM, we used average composition from the literature (Supplementary Table 4). EOM and waste flows were converted using several units: fresh weight (FW), dry matter (DM), total carbon (C), total nitrogen (N), phosphorus equivalent (P_2O_5), and potash equivalent (K_2O). The phosphorus and potash fertilizing potentials of EOMs were considered equivalent to their respective mineral fluxes (Linères and Morel, 2004; Wen et al., 1996). We evaluated the short-term N fertilization potential for each EOM using the mineral-fertilizer equivalent unit (N_{eq}, tN_{eq}):

$$N_{eq} = N_{tot} \times K_{eq} \quad (1)$$

where N_{tot} is the flux of total N of each EOM (tN_{tot}), and K_{eq} is the mineral fertilizer equivalent coefficient of this EOM, as given by local farming advisors (*Chambre d'Agriculture d'ïle-de-France*) (Supplementary Table 5).

The potential of each EOM to increase SOC (amendment value) was determined using an "efficient carbon" unit. Efficient carbon is defined as the flux of C coming from EOM that will be integrated into SOC after one year; it has previously been used as a proxy of the potential increase of SOC (Le

Noë et al., 2017). Efficient carbon is a relevant simplified measure to compare practices independent of soil types, as C inputs to soils have a major influence on C storage (Kong et al., 2005). Efficient C fluxes (C_{eff}, tC_{eff}) were computed using the AMG model humification parameter (Clivot et al., 2019; Levavasseur et al., 2020) (Supplementary Table 5):

$C_{eff} = h \times C_{tot}$ (2)

where C_{tot} is the total C flux due to the EOM (tC_{tot}), and h is the fraction of C assumed to be integrated into the SOC, with the remaining fraction being released as CO_2.

With the AMG model (Clivot et al., 2019), we simulated a regular, homogeneous application of each EOM on the whole UAA on the VP for 20 years; the entire stock of yearly produced EOM was applied on the whole UAA each year. We obtained the supplementary carbon storage induced by each EOM. The potential increase in yearly mineralized N from increased soil organic matter (SOM) after 20 years of application was designed as a supplementary N release occurring after long-term regular EOM applications (Gutser et al., 2005) and called Supplementary Soil N Mineralization (SSNM) in this study. The increase in SOC was converted into an increase in soil organic N assuming a mean C:N ratio of SOM of 11.0 on the VP (Noirot-Cosson et al., 2017; Zaouche et al., 2017). We calculated the corresponding increase in yearly SSNM (N_{SSNM}, kgN ha^{-1} $year^{-1}$) using the same equation used for organic C yearly mineralization (Clivot et al., 2019) (Supplementary Material 6). The dataset of EOM fluxes and nutrient equivalents is available in Supplementary Material 7.

2.3.4 Farmer nutrient demand

We compared the flux of nutrients from EOMs to two scenarios of crop nutrient demands. In the first scenario, the territorial average demands for chemical fertilizer for N, P_2O_5, and K_2O were deduced from sale datasets obtained from local farming advisors (*Chambre d'Agriculture d'Île-de-France*) for conventional cereal farms and applied for the entire UAA: 150 kg N ha^{-1} $year^{-1}$, 40 kg P_2O_5 ha^{-1} $year^{-1}$, and 40 kg K_2O ha^{-1} $year^{-1}$. These figures were consistent with national statistics (AGRESTE, 2014; UNIFA, 2011a, 2011b) and interviews with farmers (Supplementary Material 8). Fruit production and market gardening were omitted.

The second fertilizer demand scenario for the VP was a hypothetical of 100% legume-cereal organic agriculture on the entire UAA. We estimated the average nutrient demand for organic cereal farms based on data from local experts and local practices. The crop succession corresponded to a typical rotation: alfafa-alfafa-wheat-2nd cereal-maize-fava bean-wheat-2nd cereal, with a 4 t ha^{-1} poultry feces supply on the 2nd cereal and maize crops and a 6 t ha^{-1} manure supply on the maize. This corresponded to an average application of 60 kg total N ha^{-1} year^{-1} (i.e., 9 kg N_{eq} ha^{-1} yr^{-1} and 22 kg N_{SSNM} ha^{-1} year^{-1} after 20 years of EOM application), 48 kg P_2O_5 ha^{-1} year^{-1}, and 41 kg K_2O ha^{-1} year^{-1}. We estimated the available N related to the EOM application as the sum of N_{eq} and SSNM, i.e., 31 kg N ha^{-1} yr^{-1}. Our goal was not to propose a realistic scenario but to compare the potential of urban EOMs in organic agriculture in an ideal low input system in the context of the local development of organic agriculture. The high P and K demands were explained by the actual organic fertilization and may not be representative of minimal crop demand.

2.3.5 Efficient C from crop residues and roots

We compared the flux of efficient C from EOMs to that of crop residues and roots. This was estimated for each main crop (and cover crop) from the yield, area of cultivation, and frequency of residue incorporation with the AMG model (Clivot et al., 2019) (Supplementary Table 8). Yields and frequency of residue incorporation were based on our interviews with farmers, while the area of cultivation was collected from a spatial dataset based on the farmer declarations for the European Common Agricultural Policy (Agence de Services et de Paiement, 2015).

3 Results

3.1 Use and perception of EOMs by farmers

The term "soil fertility" had numerous understandings among farmers (Figure 2 A). Some definitions focused on crop production; a fertile soil prevents water stress, does not seal, has few weeds, and has high nutrient concentrations and retention. A fertile soil was also seen as easy to work with and having fewer issues than less fertile soils (it is "easier for everything”). A soil can also be considered fertile if it is easy to till or offers multiple crop choices. The addition of EOMs was the most frequent practice to maintain soil fertility, often combined with other practices (organic agriculture or no tillage) (Figure 2 B). On the VP, the addition of EOMs was already considered a way to improve soil fertility or to fertilize crops: 68% of farmers are EOM users.

Most reasons for using EOMs (fertilization or SOM increase, as in Figure 2 C) were linked to common "fertility" notions (high yields, P and K content, or no sealing, as in Figure 2 A). Other issues cannot be solved using EOMs (larger choice of crops). Some fertility components could be improved through SOM increase, but EOMs were rarely seen as a solution for those problems (e.g., soil structure or water retention). The low interest in soil structure that we found differs from the results of Case et al. (2017) and may depend on soil type or cropping system. The low interest in water retention may be because water availability is already high in our study area and because it can be improved only with large increases in SOM (Houot et al., 2014; Rawls et al., 2003).

Barriers against EOM use were mainly economical (cost or specific machinery), logistical (workload, transport, storage, low availability, or administrative constraints), technical (difficult fertilization control) and social (odors, pressure from neighbors, lack of confidence, or "dirty products") (Figure 2 D). Availability was an important reason for whether or not EOMs were used. Some farmers saw EOMs as expensive products, while others saw them as cheap amendments or fertilizers. Those barriers have been discussed in previous studies (Case et al., 2017; Houot et al., 2014; Joncoux, 2013). Regarding logistical constraints, the main EOMs produced on the VP (compost and manures) are spread very locally, i.e., within 10 km of the production sites. Therefore, the use of EOMs is spatially heterogeneous, with high local EOM use in the eastern densely populated part of the VP, where the composting plants are located.

We identified typologies of EOMs. In the first group, animal effluents and composts benefit farmers close to the production source (availability and price criteria). Slurry, cattle manure, and compost were seen as "good quality" products and thus are used close to production sites. Horse manure is more difficult to spread and provides less nutrients. It is less efficiently recycled in agriculture. A second group includes sewage sludges, which could be considered "all or none" products. Among farmers, 58% would not use sludge because of its "waste" status and the environmental risks. In contrast, it can also be seen as a free source of organic matter, N, P and Ca (limed sludge). There could still be a demand for it; 22% of the VP farmers use sludge, and 19% more would spread it if it were offered to them and if the contamination risks were low. A third type of EOM is the processed and often imported high quality organic fertilizers, such as amended compost (P, K), composted pig slurry (P, N), or poultry feces (N). Those EOMs are often imported large distances, e.g., from Brittany (400 km) or Belgium (350 km). Among farmers, 73% were interested in digestate (76% of the agricultural area), although it is not produced on the VP.

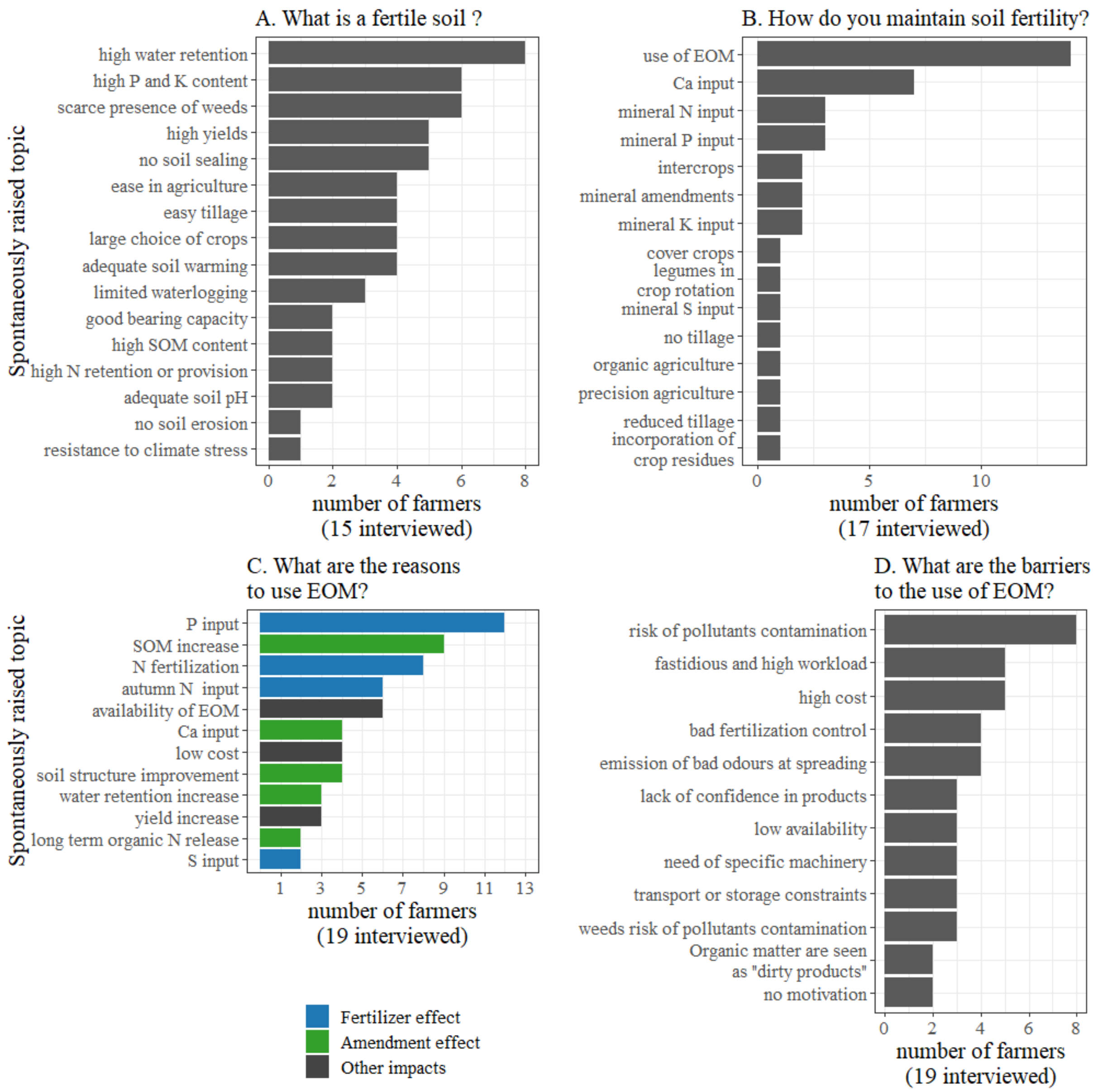


Figure 2. Answers of farmers during their interviews about (2.A) the definition of soil fertility, (2.B) fertility maintaining practices, (2.C) reasons to use EOMs, and (2.D) barriers to EOM use. Bars show number of farmers who raised the topic on the vertical axis to answer the question. A total of 25 farmers were interviewed.

3.2 EOM fluxes at the territorial scale

We identified all the EOM fluxes used and produced on the VP (Figure 3). Each EOM was subjected to different management practices, which explained how much they were locally recycled (Figure 4). The database of EOM fluxes and nutrient equivalents is described in Supplementary Materials 7, 9 and 10.

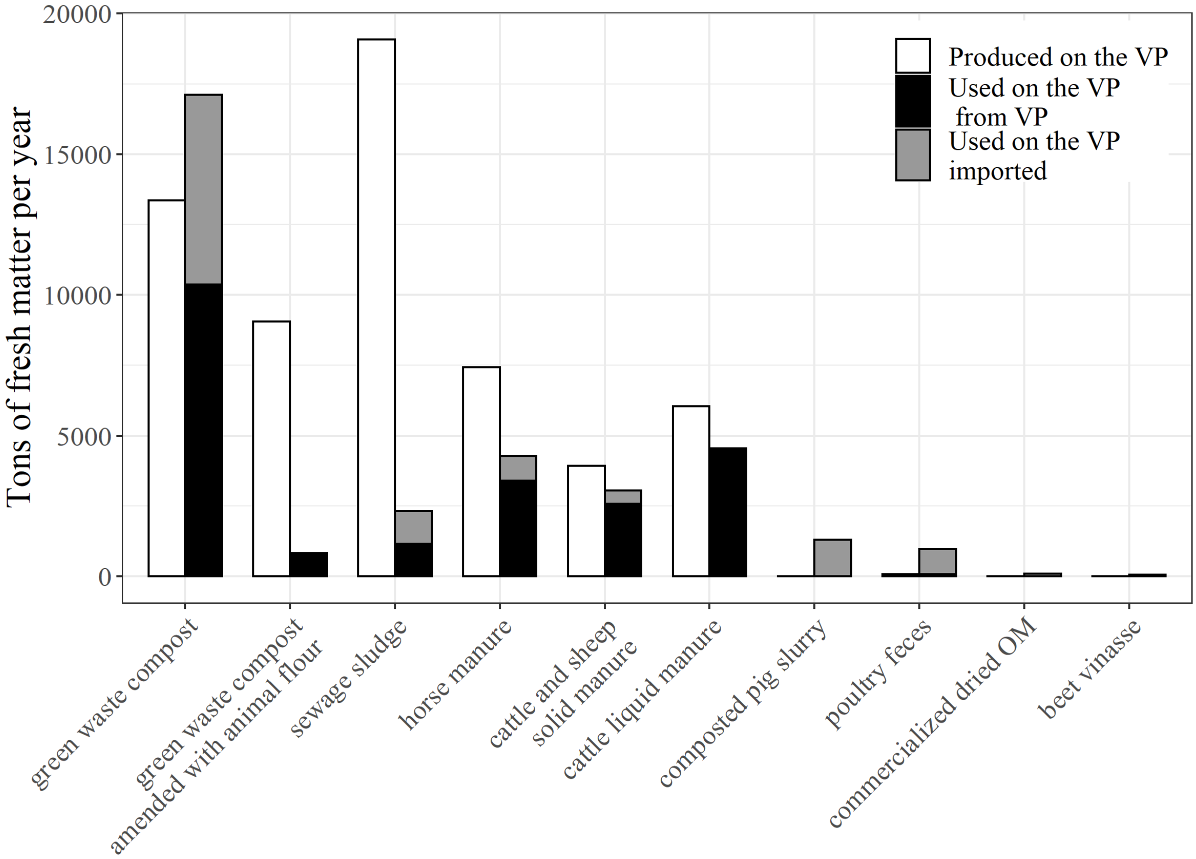


Figure 3. Production and use of each EOM on the VP. White bar shows the production on the VP, and gray and black bars show the use by VP farmers as a function of the origin of the products.

3.2.1 Sewage sludge

The rural western part of the VP was served by small plants (200 to 10,000 served inhabitants). Major plants (45,000 to 200,000 served inhabitants) served the eastern part of the VP and other populated cities (e.g., Versailles). Therefore, 58% of the wastewater treated on the VP came from outside the territory, implying high sewage sludge production (Figure 3).

The sewage sludges produced in VP was composted (60%), directly spread in fields (32%), spread after storage in reeds basins (2%), industrially valued (<1%), incinerated (<1%), or had unknown destinations (4%) (Figure 4). Sewage sludges were composted in platforms located at a minimum distance of 50 km, and composted sludges were used near them, outside of the VP. Considering the direct spreading of the sludge, the largest plants spread their sludge at distances greater than 20 km, outside of the VP. This high transportation distance of sludge is common in the Paris region (Esculier et al., 2018). Thus, only 8% of the sludge treated on the VP were locally recycled (DM basis), whereas 54% of the sludge spread on the VP were imported from 4 wastewater treatment plants, located at 10 km outside VP (Figure 3). Sludge local recycling was thus nonoptimal.

Wastewater treatment plant managers currently want to increase sludge agricultural valuation, for transport and economic reasons. In rural areas, small reed-basin plants are being developed, with local spreading as the first option for sludge treatment.

3.2.2 Green waste composts

Compost production was located in the eastern part of the VP due to the emission of urban green wastes and the presence of the three composting plants. Of the green waste treated on the VP, 77% was imported (Figure 4). 33% of the total treated green waste was turned into “standard” compost (mainly spread within 15 km), 21% into enriched compost (mainly exported on longer distances), 18% was exported for sludge composting, and 28% has no agricultural uses (landscaping or biomass heating). Among the total green wastes, 28% was treated and spread on the VP, mostly as “standard” compost, which is the main EOM used on the VP (Figure 3). This was more than the VP production of green wastes, and VP soils thus benefited from green waste imported from nearby urban areas. Standard green waste composts were spread very locally; however the amended compost, with a higher financial value, was sold to farmers in a larger area and was thus mostly exported from VP (Figure 4). Local recycling of green waste seemed already optimized due to the transport costs and local demand from farmers.

3.2.4 Animal manure

Effluents from cattle, sheep, and poultry were mainly recycled on the VP (90% of poultry feces and cattle slurry and 75% of cattle and sheep manure), due to their high agricultural values. Even if they crossed the VP border, they travelled small distances (c.a. 10 km). Although the VP was an area with few breeding, manure and slurry were among the most used EOMs by farmers after composts (Figure 3).

There was high production of horse manure (Figure 3). Only 50% of the horse manure was recycled on the VP, through straw-manure exchange. The remaining horse manure could be exported to nearby regions as amendments or resources for anaerobic digesters (Figure 4). This lower local recycling efficiency was explained by its lower agricultural value. Composting or anaerobic digestion of horse manure could improve its value and local use.

3.2.3 Biowastes

In the territory of biowaste collection, the fermentable fraction of household waste represents half of the total biowaste production, with the second half being wastes from agro-industries, catering, retail, and public establishments. Biowastes valued through home composting represented only 0.8% of the biowastes emitted on the VP. Biowastes were largely imported to the VP and not yet recycled (Figure 4), due to the barriers of the need for separate collection and the lack of treatment options. The recycling of biowastes should increase due to the selective sorting obligation planned in 2023 (Directive 2018/851 of the European Parliament and of the Council).

3.2.4 The Versailles plain, a hub of urban waste

The VP contains many waste treatment plants, which contribute to import organic wastes (wastewater, green wastes, and biowastes). Of the transformed products, 77% of DM is exported (sorted green wastes, composts, and sludge) (Figure 4). In contrast, 40% of the DM of EOMs used in the territory is imported in diverse forms. Some come from less than 50 km (composts, sludge, and beet vinasse), while others come from different regions (transformed pig slurry, poultry feces, and other products). Thus, there is high import and export of wastes and EOMs (Figure 4). This is accentuated by the important nutrient loss occurring during wastewater and biowaste treatments; 67% of N inputs are lost, as well as 69% of P_2O_5 and 86% of K_2O (Supplementary Material 10).

The spatial use and production of EOM is heterogeneous. The western part of the VP is closer to waste treatment plants, and farmers use more EOMs in this area. The correlation between the use of EOMs and their availability close to the fields is a strong conclusion of our interviews (Figure 2).

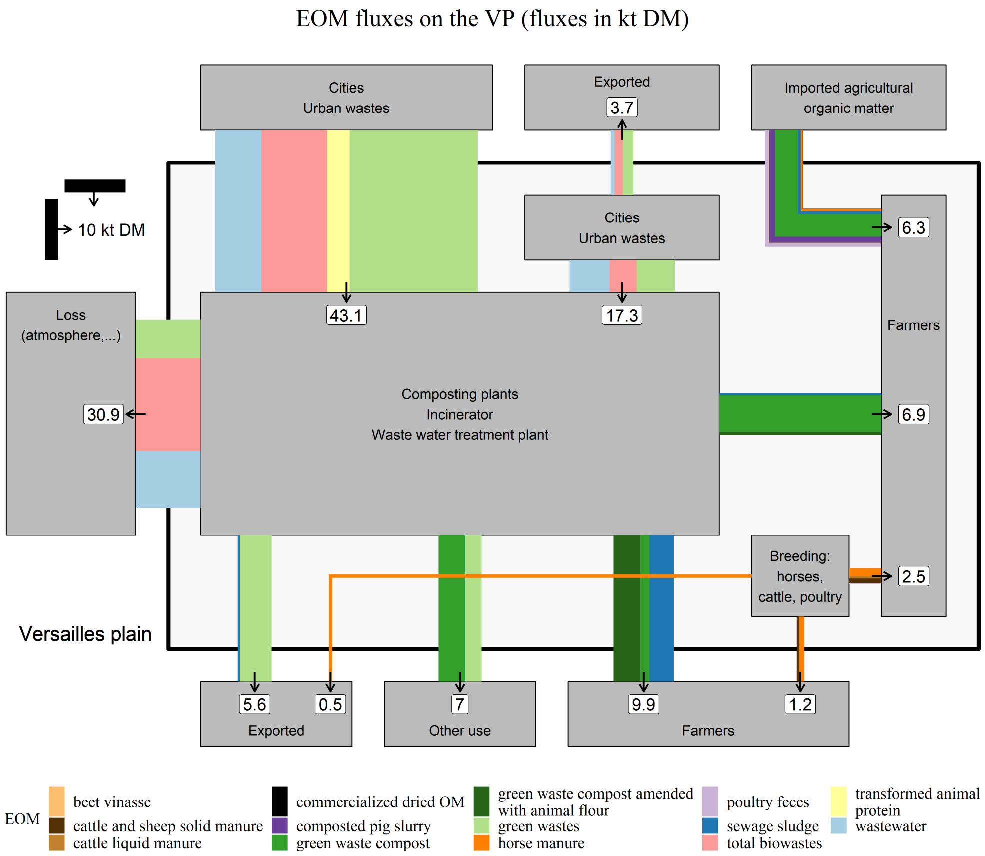


Figure 4. Sankey diagram representing fluxes of organic matters at the scale of the VP. Widths of lines are proportional to fluxes, and numbers indicate the value of the flux, in kt DM.

3.3 Quantification of the fertilizer and amendment potential of EOM application

The current use of EOMs on the VP already represents approximately half of the current mineral fertilizer use for P_2O_5 and K_2O (Figure 5). EOM production on the VP (including EOM export not used on the VP) could even closely meet P agricultural demand. Imported poultry feces and composted pig manure contribute 34% and 12%, respectively, of the P and K used from EOMs on the VP. This imported K flux could be replaced by the greater use of amended compost or horse manure. Only sewage sludge could replace the organic P imports. The production of EOM P on the VP is larger than that of K, However it is two times less locally recycled than K. The potential available EOM P and K is lower for the demands of 100% organic agriculture, as sewage sludge use is forbidden in organic agriculture (Commission Regulation (EC) No 889/2008, Appendix 1).

The short-term N fertilizer equivalent of EOM, either used or produced on the VP, was very low in comparison to the N fertilizer demand. Imported poultry feces and composted pig manure contribute

42% of the short-term N fertilizer equivalent from used EOM. This could only be replaced with sludge recycling. The use of urban EOMs for N fertilization will be efficient only with long-term use; SSNM related to long-term EOM use currently represented 9% of the nitrogen demand and could represent up to 13% at its maximal potential. EOMs cannot currently easily replace mineral N fertilizers. Organic farms currently use compost or manure to increase long-term N release, as well as EOMs with high N fertilizer potential. Considering only N fertilizer equivalent inputs, the current EOMs used or produced on the VP could not cover the 100% organic agriculture scenario, due to the lack of local EOMs with high fertilization potential. However, SSNM could cover the N demand.

The use and production of EOMs represented an efficient carbon input of 0.22 t C ha^{-1} $year^{-1}$ and 0.30 t C.ha^{-1}.$year^{-1}$ into agricultural soils, respectively. Composts had a major role in that input. In comparison, we estimated the efficient C input in soils from crop residues at 1.07 t C ha^{-1} $year^{-1}$, consistent with the value found by Le Noë et al. (2017) at the national scale. Compared to a scenario without EOMs, the current use of EOMs could induce a mean C storage of approximatively 0.12 t C ha^{-1} $year^{-1}$ (AMG, 20 years), which is 0.27% of the agricultural topsoil C stock. This storage was important compared to the "4p1000project" target. (4p1000.org). The impact of EOM use on climate change should consider the emissions during treatment and spreading, and only additional treated EOMs promote atmospheric C sequestration.

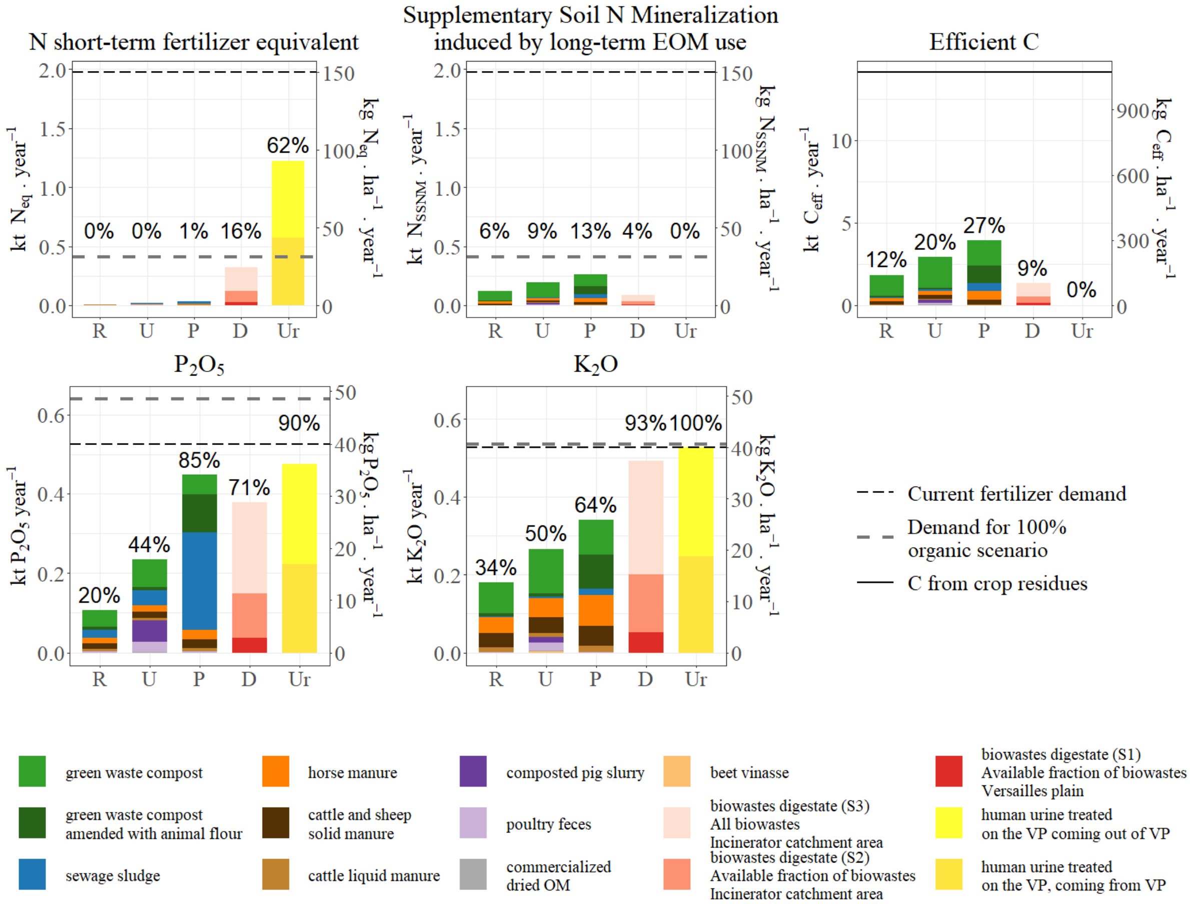


Figure 5. Nutrients available through EOMs on the VP. R : EOMs currently used on the VP coming from the VP. U: EOMs currently used on the VP (imports included). P: EOMs currently produced on the VP (not necessarily used there - imports excluded). D: three scenarios of biowaste digestion. Ur: two scenarios of human urine selective sorting. Figures show the ratio compared to the current demand for each nutrient (N, P, and K) or compared to the current humified C due to crop residues (C).

3.4 Agricultural potential of biowaste digestate and human urine recycling

Considering the two anaerobic digester projects (S0), the total production of digestate may represent 10,500 t FW year^{-1} and 80 t N_{tot} year^{-1}, which could be easily spread on the VP (Table 1).

Taking a reasonable biowaste treatment scenario (S2), digestate production could fertilize 680 ha, which is much less that the potential digestate demand (Table 1). The digestate would, however, become the second most spread EOM on the VP, providing high potential N for short-term mineral N fertilizer substitution, a service not yet provided by EOMs. The digestate would also provide high K and reasonable P inputs (Figure 5). Without biowaste import (S1), digestate production would represent 20% of the digestate production of scenario S2. In scenario S2, the VP would thus largely benefit from the high import of biowastes. The unrealistic maximal potential biowaste digestion scenario (S3) indicates that biowaste digestate could feed the total mineral P and K demand and would be a large source of short-term fertilizer equivalent N (16%). The digestate could not, however, meet the current demand, except with 100% organic agriculture.

We confirmed the high importance of the nutrients associated with human urine (Figure 5). The high nitrogen efficiency makes urine the EOM with the best potential for N mineral fertilizer substitution. It could meet the current P and K demand, as well as a substantial part of the N demand (62%). In 100% organic agriculture, N, P, and K demands would be met. Here again, the imports of wastewaters increased the EOM potential. This recycling would cause the spreading of urine on a large proportion of the UAA (Table 1).

Table 1. Biowaste digestate and human urine potential on the VP. Four scenarios of digestate availability are shown. (S0): short-term digestate production potential. (S1): low biowaste availability, emitted from the VP. (S2): low biowaste availability, emitted from a larger collection area. (S3): all biowastes available, emitted from the same large area as in S2. Farmer demand was estimated from interviews. Application dose corresponds to mean N demand on the VP.

EOM	Scenario	Production			Spread area (150 kg N_{eq} ha^{-1})			Farmer demand	
		kt FW	t N_{tot}	t N_{eq}	Application dose (tFW ha^{-1})	ha	(%UAA)	ha	(%UAA)
Digestate	(S0) Digestate soon produced on the VP	10.5	80	48	30-35	320	(2.4%)	10,003	(76%)
Biowaste digestate	(S1) biowaste digestate	5.8	43	26		172	(1.3%)		
	(S2) biowaste digestate	21.5	170	102		680	(5.2%)		
	(S3) biowaste digestate	68.1	535	321		2,140	(16.3%)		
Human urine	Urine emitted and treated on the VP	66.0	572	514	19	3,427	(26.0%)	N.A.	N.A.
	Urine treated on the VP	141.6	1,226	1,103		7,353	(55.9%)		

4 Discussion

4.1 Looking at regional to very local scales: methodology and uncertainty

The use of EOMs was assessed with multiple interviews. This approach is time-consuming; however, those fluxes could not have been precisely defined with regional databases as is usually done in existing studies (Cooper and Carliell-Marquet, 2013; Le Noë et al., 2017; Senthilkumar et al., 2012). It was also useful to better understand the challenges and opportunities in EOM management..Some fluxes could not be determined at the actor scale because these actors were not able or willing to communicate the quantity of wastes and their becoming (some composting plants and industries). The regional datasets were therefore necessary to assess fluxes such as biowastes. At an intermediate scale, urban communities, composting plants, and the incinerator were interesting contributors as they manage all the wastes from the inhabitants. Their number was relatively small, which allowed the obtaining of high quality information in reasonable time.

The multiple scale approach enabled a cross-validation of datasets, as done by Cooper and Carliell-Marquet (2013) (Supplementary Table 3). Official data on sewage sludge were considered to be close to exact. Fluxes deduced from several sources varied from 10% (green wastes from cities, slurry, and composts) to 30%-40% (green wastes from professionals, household wastes, and manure). Digestate and urine fluxes were only used as orders of magnitude. It is interesting to rely on local EOM analyses to reduce uncertainty on EOM characteristics.

Morée et al. (2013) estimated such uncertainties as up to 45% with a model-based approach. Using the Monte-Carlo method, Le Noë et al. (2017) estimated nutrient flux uncertainties of 10% to 30%. Cooper and Carliell-Marquet (2013) found confidence intervals from 30% to 100% for P fluxes in manures, sewage sludge, and some biowastes in the UK. Thanks to our methodology, our uncertainty was lower (sludge, manure, and compost) or within the same order of magnitude as those studies for regional fluxes (green wastes and biowastes). Considering such uncertainties, the results concerning the importance of EOMs in terms of P and K fertilization, SOM maintenance, and their low participation in N fertilization, remained conclusive (Figure 5). Although the figures were more uncertain, the high potential of biowastes and human urine resources was convincing (Figure 5).

It is difficult to define a relevant spatial limit, especially when different matters have different reference travel distances (e.g., 10 km for compost and 50 km for sludges). It is possible to consider territories of different sizes for different products (Bahers and Giacchè 2019). Herein, we distinguished the local or external use with a single border, considering similarly manure spread just outside the border and equine manure exported to further regions. This approach was objective but required attention for export interpretation.

4.2 The Versailles plain as an example of "urban EOM – enriched" territory

Currently, EOMs are major contributors to P and K fertilization and to SOM maintenance on the VP, but had only a minor role in N fertilization. We found that P used from EOM was close to 40% of P fertilizer demand (Figure 5). Similar results were found in different territories, e.g. in France (Le Noë et al, 2017, Senthilkumar et al., 2012, Verger et al., 2018) and in the Netherlands (Coppens et al, 2016). Importance of EOM could be higher in regions with a higher livestock density (Bittman et al., 2017). We found that the potential resources of P in agricultural and urban wastes (not only currently used, but also potentially available) were higher and comparable to P fertilizer demand on the VP. Other studies found P resources comparable or higher than P fertilizer demand, in Europe (Caniani et al., 2019, Coppens et al., 2016, Klinglmair et al., 2017), in Pakistan (Akram et al., 2018), or in cities for urban agriculture (Wielemaker et al., 2018). We found similar results for K element but this element remains poorly investigated in the literature. In Pakistan, Akram et al. (2018) found high unused resources of K, comparable to two times the crop demand. The total N input from EOMs was often reported as lower than N inputs from synthetic fertilizers in previous studies, often ranging from 5% to 50% of N fertilizer demand (Aramaki and Thuy, 2010; Caniani et al., 2019; Coppens et al., 2016; Le Noë et al., 2017; Verger et al., 2018). The low breeding density on the VP and the fact that the main used EOM had a low mineral-fertilizer equivalence explained why we estimated a very low N potential from EOM on the VP.

Compared to other territories, the VP has a low availability of EOMs from animal breeding and a high availability of urban EOMs. Le Noë et al. (2017) found that in the intensive crop-specialized region englobing Paris — and the VP — the fluxes of N and P from human excreta were 60% and 35% of the

N and P fluxes from animal effluents, respectively. The authors concluded that these fluxes were much more important than in regions with intensive livestock specialized systems (10% for N and 5% for P). On the VP, the contrast is strongly accentuated, with N and P from wastewater emitted by VP inhabitants being 723% and 965% of the N and P from animal excreta. Similarly, the P cycles were studied in several regions in France by Senthilkumar et al. (2012). In Centre, a region with low animal breeding, 4.2 $kgP.ha^{-1}.yr^{-1}$ from animal excretion are used as fertilizers, and 1 $kgP.ha^{-1}.yr^{-1}$ are exported. On the VP, with lower breeding activity, animal excretion used on the VP represented 4.0 $kgP.ha^{-1}.yr^{-1}$, including 2.7 $kgP.ha^{-1}.yr^{-1}$ imported from other regions as poultry feces or dried pig slurry. However, composts and sludges used on the VP are twice as important (3.5 $kgP.ha^{-1}.yr^{-1}$) compared to all regions described by Senthilkumar et al. (2012).

Verger et al. (2018) studied the nutrient fluxes on the Saclay Plateau, located 10 km southeast of the VP. Both territories have comparable agricultural systems, areas, and inhabitants (15,594 ha and 175,006 inhabitants on the Saclay Plateau; Verger et al., 2018). The Saclay Plateau has a lower UAA (4039 ha), a higher manure availability, and a similar compost availability. Conversely, the potential for urban EOM production is lower due to the absence of household waste treatment and wastewater treatment plants. The two comparable territories have different resources, highlighting the diversity of agro-urban territories.

The situation of the VP (low breeding EOMs and high urban EOMs) may not be unique, and the relative importance of urban EOMs in local farming systems could thus appear in other regions. Parchomenko and Borsky (2018) also identified "cold spots" of animal P around major agglomerations in Denmark, where the importance of urban EOM use would be interesting to investigate. Wielemaker et al. (2018) highlighted the potential for near self-sufficiency in Rotterdam by coupling new sanitation systems and urban agriculture. In contrast, in Rennes, an important metropole in Britany, the peri-urban areas are already saturated with animal effluents (Bahers and Giacchè, 2019) without the potential use of urban EOMs, highlighting the need to diversify study cases.

4.3 Increasing local and adapted EOM recycling

We obtained the quantity of some EOMs used on the VP in 2010 (Dhaouadi, 2014; SystèmesDurables, 2010); in eight years, sewage sludge recycling decreased (380 t FW recycled versus 1,200 t FW in 2010), green waste imports increased (from 26 kt FW to 46 kt FW). In 2018, anaerobic digestion was about to appear in the territory. Use of EOMs in agriculture increased, and 23% of the interviewed farmers using EOMs started to use EOMs or increased their use of EOMs less than 3 years ago, possibly due to better availability.

Even if the number of farmers using sludge could double under sanitary conditions (section 3.1.), urban sludge production would be still far greater than potential recycling (Figure 3). Recycling all sludge would face strong social barriers about the quality of the products and the will against receiving "trash from the cities". Changing treatment of sewage sludge could increase their agricultural usability (e.g. drying) or the efficiency of nutrient removal (e.g. P removal). This could slightly impact territorial nutrient fluxes. Composting practices could also lightly influence nutrient fluxes (amendment, choice of inputs).

The nutrient fluxes of EOMs are low compared to nutrient fluxes within biowastes, losses during waste treatment, chemical fertilizers, or food fluxes (Coppens et al., 2016; Esculier et al., 2018; Verger et al., 2018). Similarly, on the VP, wastewaters and biowastes are major fluxes of nutrients that are currently lost. We highlighted the huge potential of selective sorting of urine and anaerobic digestion of biowastes to close the nutrient loop in peri-urban areas.

Improving EOM recycling requires overcoming the well-known technical and social barriers for EOM use for farmers (Joncoux, 2013) and development of adaptive policies with citizen agreement (Bahers and Giacchè, 2019). If it is already a challenge for existing EOMs, such as equine manure and sewage sludges, it is even more complex for biowaste anaerobic digestion or human urine recycling. Such an important use of digestate could cause environmental risks under inadequate practices, especially of ammonia volatilization (Möller, 2015; Nkoa, 2014), and may face social rejection. Instead of centralizing biowastes from broad territories, small and integrated territorial digestion projects could be socially acceptable and environmentally friendly (Bacenetti et al., 2016). Urine selective sorting is one-step behind in term of practical application and deserves more research interest, e.g., regarding its environmental safety. Improving the EOM recycling would need to produce EOM with adequate characteristics with regards to farmer needs, to control and adequate use of EOMs, and to develop local waste management policies oriented towards the promotion of organic waste collection, local treatment, and local use.

Finally, considering biowaste recycling and more local sewage sludge valuation, VP could closely reach the self-sufficiency in terms of P and K. EOM contributions to N fertilization would however remain low in a conventional agriculture scenario. In a much more ideal scenario, the use of human-urine would allow the self-sufficiency in terms of P and K, and could be a major contributor of N fertilizer.

4.4 Urban EOMs cannot easily replace animal EOMs

On the VP, even in a very low breeding system, local manure is still important for crop fertilization; it is the second most used EOM. The local system requires imported animal EOMs as good quality organic fertilizers, in particular for their N short-term fertilization value. It is thus difficult to replace missing animal EOMs with urban EOMs without anaerobic digestion or selective urine sorting. Moreover, other benefits of crop-livestock integration, such as a decrease of income fluctuation or the possibility to diversify crops to feed livestock, (Ryschawy et al., 2017) cannot be replaced by urban EOMs.

Nowak et al (2013) showed that organic farms strongly relied on EOMs from conventional and organic breeding farms; a very low amount of nutrients came from urban EOMs. On the VP, organic farmers still use high quantities of often-imported animal effluents. However, they also use composts as an available resource to bring organic matter and long-term N release. Extrapolating and increasing organic scenarios in a territory have to consider nutrient supply (Muller et al., 2017), which has no clear answer in low breeding farm systems. P and N short-term resources from urban EOMs are scarce, because of the interdiction against spreading sewage sludge in organic farming. Nowak et al. (2013) proposed to fill the nutrient demand of organic farms by allowing sewage sludge in organic farming. Digestate (Clements et al., 2012) and human urine could also be solutions to bring nutrients to organic cereal farms, even if urine is currently forbidden in organic agriculture.

5 Conclusion

We have herein investigated the importance of urban EOMs in agriculture fertilization in a waste-rich agro-urban territory, the Versailles plain. We used interviews and regional datasets to build an accurate flow analysis while interpreting the main constrains of local EOM recycling. The main EOMs produced and used are composts (1.4 t FW year^{-1} ha^{-1} UAA used) and equine and bovine effluents (0.9 t F W year^{-1} ha^{-1} UAA used), which are locally used, and sludges, which are mostly exported (1.4 and 0.2 t F W year^{-1} ha^{-1} UAA produced and exported, respectively). Social and technical constraints, as well as waste treatment plant locations, were major factors determining EOM use. Urban EOMs show strong potential for P and K fertilization. Organic N fertilization has to consider short-term N potential and longer-term increases in mineral N release related to increased SOM after repeated EOM applications. However, it cannot meet agricultural N demand. The current EOM production in VP could represent 85%, 64%, and 14% of P, K, and N fertilizer demand, respectively. Contribution of EOM to efficient C inputs in soil was currently 20% of the one from plants residues and was thus non-negligible. Anaerobic digestion of biowastes and source-separation of human urine could improve nutrient recycling in agriculture. Further research could investigate peri-urban diversity, to generalize such a study and help improve local EOM recycling by considering the environmental, logistical and social aspects of the territory.

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