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## Review

## Phosphorus recovered from human excreta: A socio-ecological-technical approach to phosphorus recycling

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## ABSTRACT

This article provides a comprehensive and cross-disciplinary overview of the phosphorus cycle through the wastewater and agri-food system. While mineral phosphorus stocks are finite, the use of mined phosphorus is accompanied with many losses, leading to pollution of water bodies. Recovering phosphorus from human excreta can contribute to more efficient use of phosphorus to ensure its availability for food production in the future. Phosphorus can be recovered through different recovery technologies and consequently used in agriculture via different recycling routes. Each recycling route has its own particularities in terms of interactions with technologies, actors and the environment to bring the recovered phosphorus back into agriculture. In this literature review, we adopt a socio-ecological-technical approach to map three phosphorus-recycling routes, via municipal sewage sludge, struvite recovered from municipal wastewater and source-separated urine. We firstly show that improvements are still needed in all three routes for achieving high P recovery efficiency, and a combination of these recycling routes are needed to achieve maximum recovery of phosphorus. Second, we identify key issues for each recycling route that currently limit the use of recovered phosphorus in agriculture. We indicate where interaction between disciplines is needed to improve recycling routes and identify gaps in research on how recovered phosphorus accesses agriculture.

## 1. Introduction

Phosphorus (P) is an essential nutrient in agriculture to produce food. The major source of P is apatite (rock phosphate), a finite source estimated to be depleted in 50–100 years (Cordell et al., 2009) or, when taking into account technical advancements and the exploration of new stocks, 100–400 years (Dawson and Hilton, 2011). Meanwhile, large amounts of P are lost at several points in the agri-food system and end up in water bodies where it contributes to eutrophication. To ensure food production in the future and to avoid pollution of the environment, the use of P in global agricultural food production must become more efficient. This means that losses should be avoided (Withers et al., 2014; Cordell et al., 2011) and P should be recovered from waste streams and recycled back into the agricultural production cycle (Childers et al., 2011; Cordell et al., 2011; Dawson and Hilton, 2011).

A particularly underutilised waste flow containing P are human excreta (Cordell et al., 2011). Of the 14 MT of mined mineral P globally

used in agriculture, 3 MT is consumed through food. Most of this P is excreted by humans and enters the wastewater system as little P remains in the body of an adult human (Withers et al., 2014; Cordell et al., 2011; Mihelcic et al., 2011; Childers et al., 2011) (Fig. 1). Also at a regional scale, P flow analyses indicate the potential to recycle P from human waste (Metson et al., 2018; Keil et al., 2018; Kamal et al., 2019). The potential and need for P recycling from human excreta to support more sustainable P-use in the future is evident, but less is known about *how* the human excreta could be recycled back into agriculture.

To support recycling of P from human waste, technologies are designed for its recovery. Urine diversion toilets (UDTs), for example, collect urine which, when separated from the excreta, can be used as a fertiliser (Vinnerös and Jönsson, 2002). Another example is the development of struvite installations to precipitate P during the wastewater treatment process (Doyle and Parsons, 2002). Yet, rather than introducing new technologies in the wastewater domain; closing the P-cycle implies an entire re-configuration of both the wastewater system

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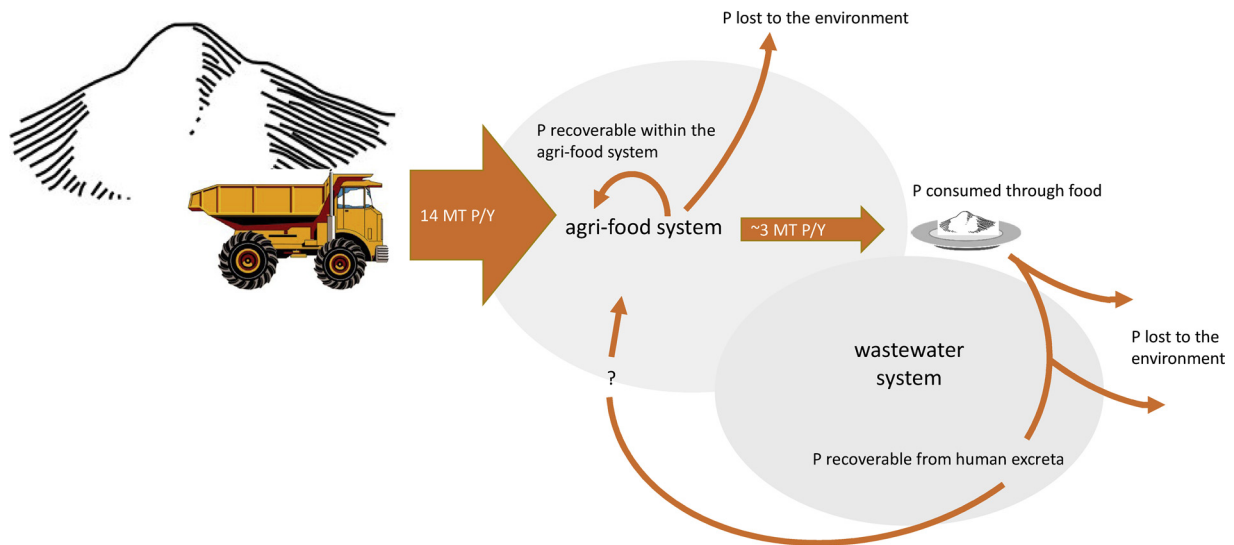


Fig. 1. P inefficiencies in the human food chain and the recycling of P from human excreta.

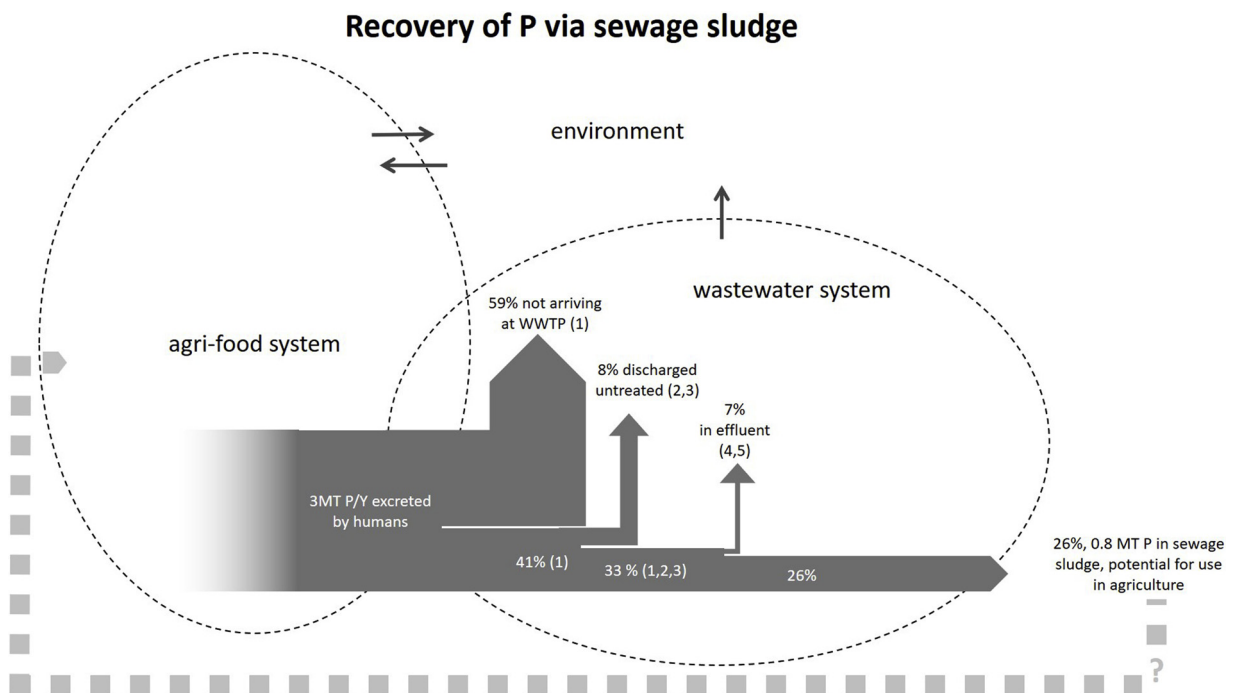


Fig. 2. P recovery via sewage sludge (in percentages of P excreted). References: (1) WHO, 2019; (2) Baum et al., 2013; (3) Langergraber and Muellegger, 2005; (4) Le Corre et al., 2009; (5) Doyle and Parsons, 2004.

and the agri-food system and requires them to become more connected (Fig. 2). Closing the P-cycle requires changes to the collection of human excreta (Mihelcic et al., 2011); to its treatment (Dawson and Hilton, 2011); to the connection to agriculture (Sharpley et al., 2016); and back to food. This in turn raises social questions as to how consumers relate to food cultivated on human excreta, or whether people would use UDTs properly. This has led to a plea for a socio-technical approach for the recycling of P from human waste (Cordell et al., 2011).

In addition to the need for a socio-technical approach to P-recycling, Childers et al., 2011 mention the importance of including ecological sciences to find suitable P-recycling options. Indeed, when making the connection between wastewater systems and the agri-food systems, ecological aspects of P-recycling become more prominent. For instance, the plant-soil interactions determine the efficiency with which P transfers from soil to plant and thus need to be understood. We

therefore propose to extend the socio-technical approach to P-recycling from human waste to a 'socio-ecological-technical' (or SET) approach.

By identifying SET-interactions through three different recycling routes of excreted P, we argue that a clearer understanding can be developed of how P from human excreta can be recovered and re-used for human food production and consumption. We describe in the methodology Section (1.1) how we interpret and apply a SET-approach to the circularity of P from human excreta. We map the state of the art knowledge for P recovery in different disciplines as a basis to situate further (disciplinary and cross-disciplinary) research and to identify knowledge gaps across the cycle.

After outlining the SET-approach in the following section, we describe the losses of mineral P through the agri-food system in a background section. We explain our methods and then discuss the results of our literature review, focussing on three recycling routes: via municipal

sewage sludge, struvite recovered from municipal wastewater and source-separated urine.

### 1.1. Social-ecological-technical approach

While we aim to broaden a socio-technical approach through the inclusion of ecology, a similar attempt has been made within socio-ecological systems (SES) research to include technology (Leach et al., 2010; McGinnis and Ostrom, 2014). For example, in the domain of urban studies (Wolfram and Frantzeskaki, 2016; McPhearson et al., 2016); water management (Butler et al., 2017) and extreme event studies (McPhillips et al., 2018), socio-ecological system approaches have been broadened to socio-ecological-technological systems research to avoid separate solutions for social, ecological and technical challenges.

In this article, we approach both the agri-food and the wastewater system as a heterogeneous SET-system, with many intertwined interactions between the social, ecological and technical. To methodologically account for the relationship between these aspects, we first map social-ecological-technical interactions in current P use and future circular routes. Second, the social, ecological and technical aspects will be treated in symmetry (Law, 1992), meaning that they are all potentially involved in making P-recycling work. By doing so we give a less central place to technology than usually seen in literature in P-recovery. How a technology contributes to making P from human waste circular, could be seen as an outcome of all heterogeneous interactions rather than the intrinsic characteristics of technology alone (Law, 2002; Zwartveen, 2017). We thus include social, ecological and technical studies that contribute knowledge on the recovery of P, its connection to agriculture, and the interactions within the agri-food system that together shape the recycling potential of P from human waste.

Here, we use the term 'recycling routes' to structure the different steps that P can follow when moving through the wastewater system into the agri-food system and consequently back to the wastewater system. Along the routes, P interacts with its environment, technologies, actors and institutions. Through these SET interactions along the recycling routes, recycled P differs from mineral P, sometimes in terms of its chemical form, but also the associations of farmers or consumers, for example waste or fertiliser. By taking the toilet as a starting point for describing the routes, we follow most literature on P-recycling, which is dominated by discussing recovery processes in the wastewater treatment domain.

### 1.2. Global P balances

While 14 MT mineral P is globally used in agriculture, only 3 MT is contained in human food and is consequently excreted by humans (Withers et al., 2014; Cordell et al., 2011; Mihelcic et al., 2011; Childers et al., 2011). This means that globally, ~80 % of mineral P that is not consumed is thus lost or retarded on its route from 'farm to fork'. The exact amount of P lost throughout the different steps in agricultural production depends on the definitions and scope taken by different researchers, and for example differs when livestock is taken into account. Also, some 'losses' will actually be recycled within the agri-food system depending on how system boundaries are set for time-scales and spatial scales.

Mineral P is generally applied in excess, providing legacy P in the following years, so oversupply of P cannot be considered a net loss on the long term (Sattari et al., 2012). Part of the mineral P input is lost from the agricultural land via erosion processes, yet to which extent is unclear. Lui et al. (2008) estimated that 0.5 MT P from mineral fertiliser is lost through erosion. If both applied mineral P and existing P in soils are included, Dawson and Hilton (2011) estimated that 8–30 MT of P are yearly eroded worldwide. Not all eroded P should be considered a net loss from the agri-food system, as part of the eroded P accumulates on arable soils and can be used in the following years (Sattari et al.,

2012). During crop production, 3 MT of P is removed from, or retarded in, the agri-food system via crop losses, due to pests, crop diseases and other crop failures. Another 2–2.2 MT P from crop residues are recycled on the field (Lui et al., 2008; Cordell et al., 2009). As the P coming available from crop residues can be used in a following cropping season, this flow should not be considered a net loss from the agri-food system on the long term. Another 6 % of mineral P used in agriculture is diverted from the food chain after the products are harvested and 7 % of mineral P used in global agriculture is diverted from the food chain during distribution, retail and cooking (Cordell et al., 2009).

While the boundaries of the agri-food system and consequently, the definition of losses, is ambiguous in scientific literature, humans undoubtedly excrete a considerable amount of P: 2.7–3.5 MT P. As about 20 % of the mineral P that now enters agriculture is excreted and ends up in the waste stream of human excreta (Childers et al., 2011; Cordell et al., 2011; Mihelcic et al., 2011; Dawson and Hilton, 2011), several papers (for example Mihelcic et al., 2011; Cordell et al., 2011) plea for recovery and re-use of this excreted P in agriculture. Recovered P could replace 20 % of the agricultural demand for mineral P, they argue, thus reducing the pressure on mineral P stocks (Cordell et al., 2011). Re-using human excreta for agricultural food production is attractive as it is relatively easy to control this P flow (compared to erosion losses for example) and because recovering this flow could prevent P release into the environment (le Corre et al., 2009; Cordell et al., 2011).

While global P balances indicate the need for P recycling, in this article we question the conclusion that P recovered from human excreta could replace 20 % of mineral P input in agriculture as it implicitly assumes that P can be recovered from human excreta and be applied on agricultural land with an efficiency of 100 %, without being hindered by social, ecological or technical processes.

## 2. Methods

Our analysis is based on an explorative literature review and P balances for three different recycling routes. These routes are selected as they are used in practice and have been relatively well-studied, and are thus well-described in the literature<sup>1</sup>. We analyse the following recycling routes via which P can purposively be moved from the wastewater system into the agri-food system:

- Via the *sewage sludge route* the sludge, a by-product of domestic wastewater treatment, is applied on agricultural land. This route is practiced in several countries worldwide. The treatment of sewage sludge before application on agricultural land can vary (for example digestion, thermal treatment, composting).
- The *struvite route* currently gains attention as the most promising P-recycling route. Through struvite precipitation, P is recovered at the wastewater treatment plant in granules, which results in a slow-release fertiliser.
- The *source-separated urine route* has gained academic attention since the end of last century. It proposes to collect the urine separate from the faeces with UDTs (Urine Diversion Toilets) and subsequently use the urine as a fertiliser.

Based on 55 articles, we mapped the different SET-interactions that shape the circularity of P from human waste. Articles were selected per recycling route that were either highly cited compared to other articles within the specific route or that were recently published, and were thus expected to discuss advancements in their field. As articles on P recycling have been published within different domains, a quantitative

<sup>1</sup> Several recycling routes have been omitted from the review because they were either 1) not well-covered in the literature (for example, P-recycling via the cultivation of algae on wastewater) or 2) not distinct enough from the chosen routes (for example, struvite precipitation from source separated urine).

selection, for example based on citation rates, could not be made. Also, review articles were prioritized in this phase of the research.

To structure the issues raised in the explorative literature review P balances were made, including the social, ecological and technical diversions of P along the different routes. These P balances had, as a starting point, the technical recovery potential of each route, but were expanded with the social, ecological aspects encountered in the literature review. We further describe the P balances and the assumptions made to draw these in the results section. Where the P balances showed blind spots in the initial literature review (for example, an unexplained loss of P along the route), the review was extended with new literature derived through additional searches for the specific issue to fill that gap. As the P balances are based on a review of literature from different disciplinary backgrounds and using different scientific methods, these P balances are not thorough but rather serve as a framework for describing the SET interactions along the recycling routes.

### 3. Results

#### 3.1. Sewage sludge route

In Europe, approximately 25 % of P from human excreta is brought back to agriculture, all via sewage sludge application (Cordell et al., 2011). All wastewater treatment plants (WWTPs) produce sewage sludge as a by-product of wastewater treatment, but not all recycle the sewage sludge in agriculture due to socio-ecological constraints in the agri-food system. Other options used to dispose of sewage sludge are storage in landfills or incineration.

##### 3.1.1. Recovery

In domestic wastewater P originating from human excreta is mainly present as orthophosphates ( $\text{PO}_4^{3-}$ ,  $\text{HPO}_4^{2-}$ ,  $\text{H}_2\text{PO}_4^-$ ), which are soluble and bioavailable to any organism (Doyle and Parsons, 2002). At the WWTP, most P will be encapsulated in the sewage sludge while currently, depending on discharge legislations, a certain percentage is allowed to be discharged into surface waters with the effluent. In the European Union, not more than 1 or 2 mg P/L (depending on the number of inhabitants connected to the WWTP) is allowed to remain in the effluent, and in addition, the P concentration of the influent should at least be reduced with 80 % (Doyle and Parsons, 2002; Le Corre et al., 2009). For the purpose of drawing P balances for the sewage sludge route, we here assume that 20 % of P leaves the WWTP via the effluent, and 80 % enters the sewage sludge, although this percentage of P in the sewage sludge is likely to be less as higher P concentrations in the effluent – either due to less strict legislation or non-compliance with legislation – results in lower P concentrations in the sewage sludge (Doyle and Parsons, 2002). The wastewater treatment processes co-determine the form in which P will be present in the sewage sludge, and consequently, how and whether P becomes available to crops. P is removed from the wastewater biologically by specific bacteria, possibly aided physio-chemically with the addition of Fe or Al. When the P in wastewater is taken up by bacteria, the bacteria sink during sewage treatment and end up in the sewage sludge in organic forms. Precipitation with Fe or Al can result in metal-bound P which will not easily become bioavailable to crops once applied on agricultural land (Torri et al., 2017). Raw sewage sludge can contain pollutants from both household wastewater and industries, such as heavy metals and organic contaminants.

When we zoom out from the WWTP and include how – and how much – excreted P actually enters a functioning WWTP, we notice that not all human excreta is collected by the sewage system and actually reaches functional WWTPs due to socio-technical challenges. According to the World Health Organization (2019), 45 % of the global population has access to safely managed sanitation, meaning a connection to the sewage system, pit latrines or compost toilets. In 2017, 41 % of the global population had access to a toilet with a sewer connection. In the

P balance in Fig. 2 we assume that 41 % of the global population with access to the sewer system means 41 % of globally excreted P entering the sewer system, though in reality food consumption patterns and associated P in excreta could differ between these groups.

Baum et al. (2013) estimated that 39 % of all wastewater arriving at WWTPs is treated. According to the WHO, 80 % of all household wastewater collected by sewage systems received at least secondary treatment. Yet, this number does not account for losses of wastewater to open drains or losses within the sewage infrastructure (WHO, 2019). While a total of 41 % of P from human excreta enters a sewer system, based on the WHO estimates, only 33 % worldwide enters a functioning WWTP. As at the WWTP 20 % of entering P is allowed to remain in the effluent, all sewage sludge produced worldwide at functioning WWTPs approximately contains 26 % (see Fig. 2) of humanly excreted P, which equals 0.8 MT P/year.

Before the sewage sludge can be used on agricultural land, it should be treated, both for safety reasons and to dewater the sludge for ease of transport and application (Fytili and Zabaniotou, 2008; Winker et al., 2009; Torri et al., 2017). Treatment options for sewage sludge are dewatering with presses, centrifuges or drying beds, and the sewage sludge can consequently be composted, pasteurized, digested, stored or disinfected with lime (Fytili and Zabaniotou, 2008), which change the form in which P is present in the treated sewage sludge. Yet, not all sewage sludge produced at WWTPs is applied in agriculture, as its access into the agri-food system is hindered by socio-ecological constraints, which we will describe below.

##### 3.1.2. Access into the agri-food system

In global assessments, P recovered from human excreta might seem urgently needed, but these recommendations do not necessarily resonate with local farming realities (Sharpley et al., 2016). The recovered P products have specific characteristics that make them distinct from mineral P, which means that they cannot ‘just’ replace mineral P fertilisers – both in terms of their ecological functioning in the soil and in terms of social meanings attached to the product. These differences from mineral P fertilisers makes it challenging for sewage sludge to find its application in agriculture, and thus access the agri-food system.

##### 3.1.3. Accumulation of P in the soil due to its bio-availability and transportation

As sewage sludge application on agricultural land is mainly a waste disposal method and the resulting product is relatively bulky and thus costly to transport, Bloem et al. (2017) warn for P hotspots in the close vicinity of urban areas. Such P-hotspots lead to P losses to the environment via leaching and run-off. Indeed, with sewage sludge P is often applied to agricultural soils in excess, i.e. more than crops require. This, together with the slow release of the mainly organically bound P from sewage sludge, leads on the long-term to an accumulation of P in the soil, resulting in P levels higher than required for crop growth (Elliott and O'Connor, 2007). Because of this oversupply and accumulation of P, soils can become saturated after a certain period, with the risk of P leaching, which is both a loss of P from the agri-food system and pollution of the environment (Siddique and Robinson, 2003).

As described above, the form in which P is present in the sewage sludge and hence how P will behave in the soil are both determined by the wastewater treatment and sewage sludge treatment. In general, sewage sludge contains relatively low concentrations of water-soluble P, and the bio-availability of P from sewage sludge also depends on its interactions with the soil (Torri et al., 2017). Sewage sludge treated with Al or Fe makes P less available (Torri et al., 2017), as does sewage sludge with added calcium (Siddique and Robinson, 2003; Kahiluoto et al., 2015). The wastewater and sewage treatment thus co-determine with the receiving soil the rate at which P becomes bio-available. Because not all P in sewage sludge is immediately available to the crop, farmers may apply more P from sewage sludge than the crop needs, while the effect of oversupply (the leaching of P to surface waters) will



be noticed later.

### 3.1.4. Legislation for banning or facilitating sewage sludge application in agriculture

To avoid oversupply of nutrients and to ensure safe food production, legislation on sewage sludge application in agriculture is set up by governments. These legislations balance between safety issues and the need to recycle sewage sludge as a resource for agriculture (Christodoulou and Stamatelidou, 2016). The European Union, for example, both encourages the application of sewage sludge in agriculture and aims to prevent risks (Fytali and Zabaniotou, 2008). Legislation for sewage sludge application in agriculture differs considerably: from stimulation (Japan, Australia, EU, UK) to banning (in certain German states). In several countries, legislation on the use of sewage sludge is stricter and more limiting than the legislation for manure (Christodoulou and Stamatelidou, 2016; Bloem et al., 2017).

For sewage-derived products, different countries have included different metals in their regulation for land application to avoid leaching of heavy metals to water bodies and to avoid high concentrations of heavy metals in the food chain via food crops. For the European Union for example, cadmium, copper, nickel, lead, zinc, mercury and chromium concentrations in sewage sludge and soils are regulated in the sewage sludge directive of the European Union (CEC, 1986). Some EU countries have set their limits at lower concentrations of heavy metals in the sewage sludge than the European Union, and some countries also added limits to the ratio of arsenic in the sewage sludge. Taking into account processes in the soil, some countries set regulations for maintaining soil pH within set limits to avoid leaching of heavy metals (Singh and Agrawal, 2008). While these regulations make sewage sludge application in agriculture more bothersome, the other solution to avoid contaminants in the food chain leads to a disruption of the P-cycle: the application of sewage sludge is on non-agricultural land (Torri et al., 2017).

### 3.1.5. Public perception and communication about accompanying contaminants

Although public perception is a seemingly 'social' issue, it is closely related to 'technical' questions on oversupply of P and accompanying pollutants. Opponents of sewage sludge application on agricultural land have concerns about concentrations of heavy metals, potentially toxic organic and chemical pollution, pathogens, odours, health effects and excessive application of nitrogen (Elliott and O'Connor, 2007) and the leaching of P when sewage sludge is applied in excess.

Literature on public perception of sewage sludge mainly focusses on the United States where sewage sludge has been applied on agricultural land on a large scale since the 1970s (Beecher et al., 2005). Initially, the public was not involved in decisions on sewage sludge applications, which resulted in resistance. Since the mid-1980s, experts attempted to better explain the facts about sewage sludge application and the risks for the public. For example, by stating that the application of treated sewage sludge on agricultural land was safe, when existing regulations are followed. Also, the term 'biosolids' to refer to treated sewage sludge was introduced to avoid the negative connotation of sewage sludge (Beecher et al., 2005). Yet, opposition against the application of sewage sludge on agricultural land remained (Elliott and O'Connor, 2007). According to Beecher et al. (2005) stakeholders got actively involved in the US from 2005 onwards, which resulted in more understanding by the public about the issue of sewage sludge application on agricultural land. Yet, a critical issue remains whether the experts informing the public about risks are considered neutral and objective because of contradicting outcomes of research and doubts about the influence of research funders on the research outcomes (Beecher et al., 2005). Closing the P cycle from toilet to food thus requires active participation of all stakeholders, and could improve P efficiencies through the cycle.

## 3.2. Struvite route

A recycling route where oversupply of P, public acceptance and legislation seems less of a concern is struvite precipitation. Would struvite precipitation be a future development that could optimise P-recycling from human excreta? We will here analyse the potential contribution of using struvite installations at all functioning WWTPs to P-recycling from human excreta.

Struvite ( $\text{MgNH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$ ) precipitation is a method to recover P from wastewater. Struvite can form naturally in the wastewater treatment infrastructure, most notably in anaerobic digesters where uncontrolled struvite precipitation causes clogging in pipelines, resulting in increased pumping and maintenance costs (Doyle and Parsons, 2002; Cieřlik and Konieczka, 2017). This process triggered the innovation of controlled struvite precipitation, to avoid blocking infrastructure and to recover P from wastewater (Vaneekhaute et al., 2017).

Desmidt et al. (2015) mention five full-scale operating struvite precipitation installations globally in 2015, using municipal wastewater as input, of which some installations are installed at multiple sites. Yet, the practice of struvite precipitation has been growing since 2015. Struvite precipitation is attractive from an operational point of view and can potentially result in a relatively clean, slow release P product (Cieřlik and Konieczka, 2017) although concerns are raised about the possible accompanying pollutants like weed seeds or pathogens in actual practice (Vaneekhaute et al., 2017; Cieřlik and Konieczka, 2017). With an N:P ratio of 1:1, struvite precipitation at the WWTP has the additional advantage that it concomitantly removes – and thus recovers – part of the nitrogen contained in the wastewater (De-Bashan and Bashan, 2004).

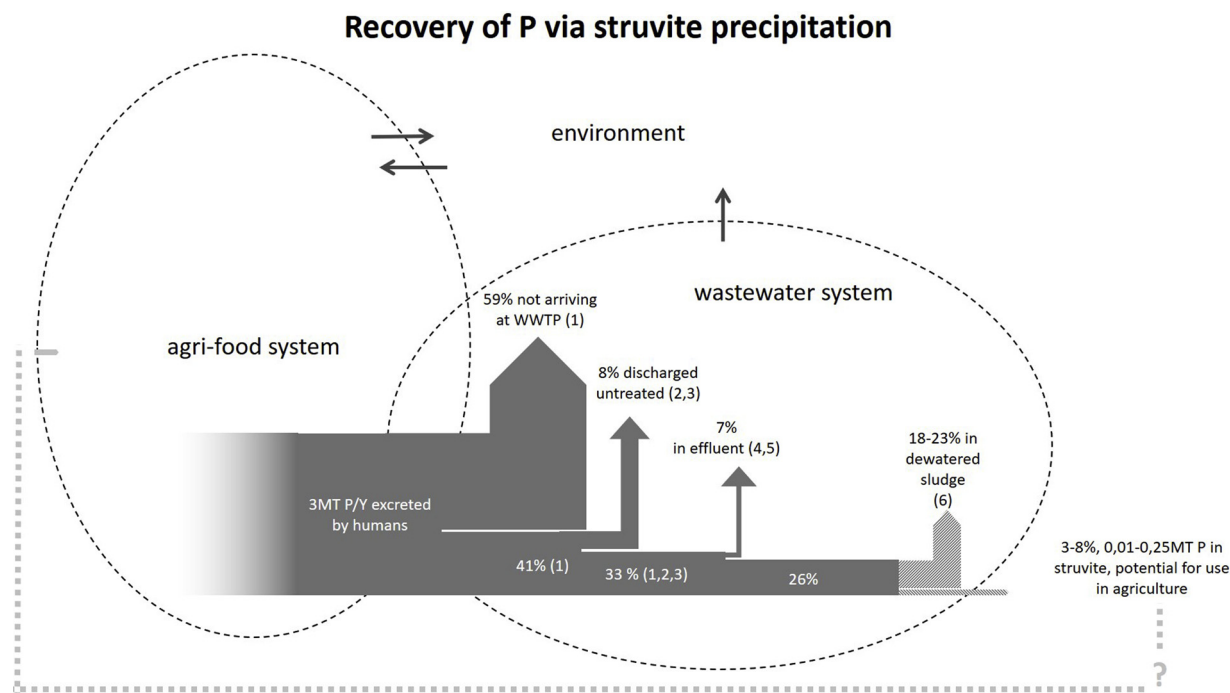
### 3.2.1. Recovery (struvite)

During the sewage sludge digestion phase, part of the P in the sludge will be released, and becomes available in the form of orthophosphates (Cieřlik and Konieczka, 2017). The anaerobic digestion thickens and dewateres the sludge. The resulting wet by-product, the digester supernatant, can be used to recover P in the form of struvite. The digester supernatant would normally be redirected within the WWTP. This flow is usually chosen for struvite precipitation as, compared to the other flows within the WWTP, it contains high concentrations of P which have been released from the sewage sludge during the digestion process and a lower concentration of suspended solids as these have settled in the digested sludge (Gaterell et al., 2000; De-Bashan and Bashan, 2004). Via struvite precipitation from the digester supernatant, 10–25 % of the influent P can be recovered (Egle et al., 2015). The rest of the P entering the WWTP, 75–90 % is partly contained in the treated effluent (ca. 20 %) and partly in the digested, de-watered sludge (ca. 55–70 %). This digested, de-watered sludge diverts from the struvite route and could possibly be recycled as a soil conditioner with lower P concentrations than untreated sewage sludge.

The recovery efficiency of P via struvite precipitation is often defined as the percentage of P in the sewage sludge digester supernatant (thus one specific P containing flow at the WWTP) that can be recovered as struvite. This results in recovery efficiencies of 60–94 %, depending on the specific technology used (Doyle and Parsons, 2002). Here, we are rather interested in the potential contribution of struvite to closing the P cycle from toilet to food, and thus analyse the recovery efficiency as the percentage of P that can be recovered from human excreta.

As explained above for the sewage sludge route, not all P from human excreta arrives in the sewage system. About 33 % of all human excreta ends up in the WWTP (WHO, 2019), where struvite could potentially be precipitated from the digester supernatant, a sub-flow from sewage sludge. Sewage sludge contains 26 % of P from all P globally excreted by humans.

When all functioning WWTPs would install an anaerobic digester to process their sewage sludge, or could bring their sewage sludge to a



**Fig. 3.** P recovery from human excreta via struvite precipitation (in percentages of P excreted). In grey: hypothetical situation, assuming all WWTPs to anaerobically digest sewage sludge and precipitate struvite from the digester supernatant. References: (1) WHO, 2019; (2) Baum et al., 2013; (3) Langergraber and Muellegger, 2005; (4) Le Corre et al., 2009; (5) Doyle and Parsons, 2002; (6) Egle et al., 2015.

WWTP with an anaerobic digester and struvite reactor, this would result in recovery efficiencies of 3–8 % (10–25 % of 33 % of all P excreted) (Fig. 3). This is currently not yet the case, so actual recovery of P via the struvite route is much lower. While some WWTPs have an anaerobic digester to produce biogas and to facilitate the dewatering of sludge, the sludge can also be processed differently, for example by composting or drying, possibly as a preparation for disposal or re-use via the sewage sludge route (Christodoulou and Stamatelatu, 2016). How struvite can be applied in agriculture, and which social, ecological and technical challenges it faces will be discussed below.

### 3.2.2. Access into the agri-food system (struvite)

Academic accounts of the actual agricultural use of struvite precipitated from wastewater are rare. The scientific knowledge is mainly based on pot-experiments, of which only few are multiple-year studies. Experiments have shown similar yields for plants fertilised with struvite compared to mineral P (Johnston and Richards, 2003; Cabeza et al., 2011; Huygens and Saveyn, 2018), yet as we will further explain below, prices for struvite are high compared to mineral fertilisers, and the slow release of struvite has been of concern. Agricultural issues which were important for the sewage sludge route – legislation and public perception – are less-often mentioned for struvite. Legislation in the European Union for example provides flexibility: while struvite is listed as a waste-product, not a fertiliser, member states can assign fertiliser-status to struvite (Desmidt et al., 2015). Also, struvite use creates less concerns about heavy metals, as several studies showed concentrations below legal limits for fertilisers (Münch and Barr, 2001; Ueno and Fujii, 2001; Uysal et al., 2010).

### 3.2.3. Including advantages in the wastewater treatment and fertiliser value in struvite prices

When compared to mineral P, prices for struvite are high (Cornel and Schaum, 2009). Gaterell et al. (2000), estimated the cost of production and distribution of struvite in the UK, which could range between 146 and 1195 £/ton (approximately 243–1992 €/ton), largely depending on the recovery efficiency of struvite at the WWTP.

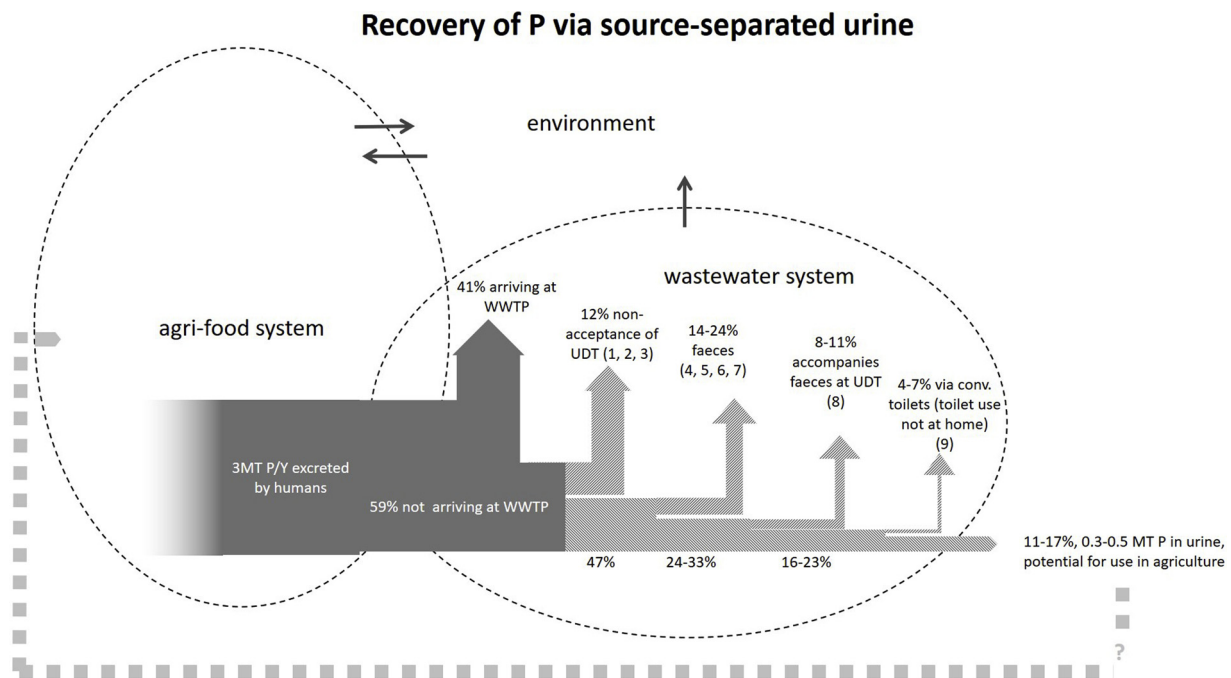
Vaneekhaute et al. (2017) mentioned commercial struvite prices of 45 €/ton in Belgium, 109–314 €/ton in Australia, and 250 €/ton in Japan. As the percentage of P in the struvite is variable, an indication of struvite prices per ton P would be better to compare struvite prices with mineral P prices. For a WWTP, the profits of producing struvite (compared to usual operation costs, without struvite precipitation) might range from a loss of 7800 euro per year to a gain of 89.400 per year (Vaneekhaute et al., 2017). Cieřlik and Konieczka (2017) stated that the recycling of P via struvite precipitation is “economically unjustifiable” (p.1731) as the price of struvite is three times higher than the price of mineral P (Weigand et al., 2013, cited in Cieřlik and Konieczka, 2017).

The possible profit of struvite precipitation is not only the selling of struvite as a fertiliser. The profit is also the advantage of avoiding uncontrolled struvite formation at the WWTP and the environmental benefit of reaching low P concentrations in the effluent. One possibility to make buying struvite attractive for farmers is subsidies (Cornel and Schaum, 2009). From an ecological perspective, one could also focus on the advantages of struvite compared to mineral P, which is its capacity to slowly release P compared to mineral P.

### 3.2.4. Slow release of P versus crop demand for P

As struvite is sparingly soluble, it is considered a slow-release fertiliser compared to mineral fertilisers, containing P that will become plant-available slowly over time without damaging the plant with an oversupply or the risk of leaching (Gaterell et al., 2000; De-Bashan and Bashan, 2004; Le Corre et al., 2009).

A meta-analysis by Huygens and Saveyn (2018) seems to indicate that precipitated P-sources like struvite have comparable agronomic efficiency in terms of biomass production and P-recovery efficiency, although most experiments are carried out in pots, and thus do not necessarily mimic field circumstances (Talboys et al., 2016). The current focus on pot experiments in the literature should shift more to field experimentation (Degryse et al., 2017; Huygens and Saveyn, 2018), to generate a more integrated insight in fate and efficiency of P from struvite in the soil. To become available to plants, P should be dissolved



**Fig. 4.** diversions of P via source-separated urine for agriculture (in percentages of P excreted). In grey: hypothetical situation, assuming the introduction of UDTs in households without access to safely managed sanitation. References : (1) Lamichhane and Babcock, 2013; (2) Ishii and Boyer, 2016; (3) Pahl-Wostl et al., 2003; (4) Mihelcic et al., 2011; (5) Roy, 2017; (6) Karak and Bhattacharyya, 2011; (7) Langergraber and Muellegger, 2005; (8) Vinnerås and Jönsson, 2002; (9) Rossi et al., 2009.

in the soil solution. Struvite is insoluble but the roots of crops produce organic acids, which increase the solubility of struvite. Notably, the capacity to release P by acid production differs between crop species, with some species such as faba bean (*Vicia faba* L.) having been documented to be particularly effective (Li et al., 2007). Thus choice of crop types may play a role in successful use of struvite. In addition, several studies tested whether soil acidity favours the release of P from struvite, by applying grounded struvite to plants in pots, but they found no difference in P-uptake on slightly acid (pH 4.7) or neutral (pH 6.6) soils (Cabeza et al., 2011). When struvite granules are directly applied to the soil, which resembles actual farming practices, the acidity of soils does matter. Degryse et al. (2017) found for the application of struvite granules in a soil with pH 8.1 a solubility of struvite granules of 0.1 mg struvite per day, while for a soil with pH 6.1 the solubility of struvite granules was 0.4 mg struvite per day (Degryse et al., 2017).

Several solutions are proposed to better match the release pattern of struvite with crop demands. The low solubility of struvite could create a reduction of crop growth due to a shortage of P at early growth stages, which could be compensated by mixing with more conventional, more soluble P sources where struvite is more effective during latter growth stages (Gaterell et al., 2000; De-Bashan and Bashan, 2004; Talboys et al., 2016). Considered from an crop-perspective, applications could focus on situations where the low solubility is an advantage, such as fertilising grasslands or forests (De-Bashan and Bashan, 2004; Huygens and Saveyn, 2018). If a specified market for struvite – with its slow-release of P – can be identified, this would also help to match struvite prices with production costs (Desmidt et al., 2015).

### 3.3. Source-separated urine route

Source-separated urine contains high concentrations of nutrients while concentrations of heavy metals are low compared to the faeces fraction. Also, urine is a relatively safe P-containing product, as it does not contain pathogens if effectively separated from the faeces. Source-separating urine is proposed as a solution in distinct settings: in remote and rural areas, where mineral fertilisers are inaccessible and safe

sanitation infrastructure is often lacking (Winker et al., 2009; Karak and Bhattacharyya, 2011) and for the development of new, sustainable buildings (Vinnerås and Jönsson, 2002). When sewage infrastructure is already in place, this could create a lock-in for a transition to source-separated urine (Bell, 2012). Taking this lock-in into account – and realising that via the improvement of WWTPs other P-recycling routes are possible – source-separated urine seems most viable in areas currently without sewage infrastructure. We thus focus on the 59 % of the global population whose excreta currently does not arrive at a WWTP.

At the end of the 20<sup>th</sup> century several experiments have been carried out with urine collection and use as a fertiliser, which are the main basis for this section to describe the social, ecological and technological interactions that facilitate or hinder P recycling via source-separated urine.

#### 3.3.1. Recovery

Currently flush-toilets connected to a sewage system are the most common ‘modern’ toilet (Bracken et al., 2007; Langergraber and Muellegger, 2005). To collect urine separately from faeces, special toilets should be used, for example urinals (Langergraber and Muellegger, 2005) or urine diversion toilets (UDTs) which separate the urine fraction from the faeces through a front bowl for the urine and a rear bowl for the faeces (Simha and Ganesapillai, 2017). As urine contains 50–70% of excreted P, source-separating toilets seem a promising recovery technology (Mihelcic et al., 2011; Roy, 2017; Karak and Bhattacharyya, 2011 and Langergraber and Muellegger, 2005).

An important premise for the separate collection of urine and faeces at the toilet is that people are using these toilets. Several studies have assessed the willingness of toilet users in different case studies worldwide to buy or use UDTs (Lienert and Larsen, 2010; Lamichhane and Babcock, 2013; Pahl-Wostl et al., 2003; Ishii and Boyer, 2016). About 80 % of the respondents in these studies indicate that they are willing to use a UDT, yet willingness to pay for UDTs (Lamichhane and Babcock, 2013; Ishii and Boyer, 2016) or move into apartments with UDTs (Pahl-Wostl et al., 2003), appeared to be 60–67 %. UDT costs and its proper installation are indeed more expensive for UDTs than for conventional

toilets (Lamichhane and Babcock, 2013). From a psychosocial analysis, Rosenquist (2005) concludes that UDTs should satisfy human needs of status, social comfort, safety and physical needs. Uddin et al. (2014), indicated that in a Bangladesh community, the financing of UDTs and the organisation of subsidies had hindered UDT installation. They encountered initial resistance from the population, which was countered by raising awareness about the environmental benefits and involvement of local institutions in the project.

For mapping the potential P recovery for the source separated urine recycling route, we assume that all households whose wastewater currently does not arrive at a WWTP would be offered a UDT. Assuming that 80 % of these households accepts UDTs, 47 % of globally excreted P continues within the recycling route and 12 % of the excreted P is lost due to non-acceptance (Fig. 4). Yet, this is based on small case studies in very specific circumstances, and research shows that acceptance can be increased.

As UDTs separately collect urine and faeces, part of the P excreted is diverted via the faeces (30–50 %), which can be recycled via composting, a P recycling route which we will not discuss in detail here. In a source-separating toilet, some urine will accompany the faeces. Vinnerås and Jönsson (2002) have studied the actual use of UDT's in an apartment block in Sweden. They concluded that 32 % of the urine excreted at the UDT accompanied the faecal part, and thus did not end up in the urine tank. They explained this loss of P as “incorrect use of the toilet” (Vinnerås and Jönsson, 2002, p. 278). This is probably related to the need to prevent faeces from entering the urine fraction (and thus preferring a loss of urine to the faeces fraction over faeces entering the urine fraction) and to the specific use-requirement of a UDT. UDTs require men to urinate in a seated position to ensure that the urine is collected by the front-bowl (Jönsson, 2001). Preventing men from urinating in a standing position at a UDT and hence losing urinated P via the back-bowl reserved for faeces is often tackled using design elements, such as additionally including urinals in public buildings or designing valves in the urine pipeline which only open when the user sits (Werner et al., 2008). Yet, the topic of urinating position could also hint at the need to better understand the practice of urination as a social issue (van Vliet and Spaargaren, 2010). As the faeces (containing 30–50 % of P in excreta) divert from the source-separated urine route, 14–24 % of P, together with 8–11 % of P from urine that accompanies the faeces due to the use of UDTs, diverts from the source-separated urine route.

Another diversion of P that occurs via the source-separated urine route is the use of conventional toilets. When someone installs a source-separating toilet at home, this person will also go to other toilets, for example at work, which could be conventional toilets. Rossi et al. (2009) estimated that 70–75 % of the daily produced urine can be collected with a source separating toilet at home, the other 25–30 % will be excreted at non-source-separating toilets. So, as people are mobile and also excrete at toilets not at home, another 4–7 % is lost to conventional toilets (Fig. 4). Yet, the study of Rossi et al. is based on a situation in a Swiss city. How this relates to remote areas without access to a sewage system - regions where source-separating toilets could recover the P that would otherwise not be collected by the sewage system - should be better understood.

Taking into account these diversions of P, we estimate that 11–17 % of all P in human excreta could be recovered via source-separating toilets at households currently not connected to a WWTP, thus recovering 0.3–0.5 MT P per year. If also urine would be collected in areas already covered by the sewage system, this percentage would be higher – although competing with other recycling routes such as sewage sludge or struvite. When moving on to the next section on source-separated urine use in agriculture, socio-ecological interactions in P-recycling via urine become more apparent.

### 3.3.2. Access into the agri-food system

Although promoted in scientific literature as a low-cost fertiliser

which is locally available, acceptance of farmers to apply source-separated urine on their fields and matching of the nutrient ratio of urine with crop requirements hinder urine use in agriculture. The timing of P release is not an issue for urine, as it acts similarly to mineral P. Legislation on urine use in agriculture is rarely mentioned as a relevant issue in scientific literature (with one exemption: O'Neill, 2011, confirming the lack of legislation on urine use as fertiliser). We will describe below the issue of acceptance and the matching of nutrient ratio's, and discuss the transportation issue which would become important when source-separated urine would also be collected in cities.

### 3.3.3. Acceptance of urine as a safe fertiliser

An under-researched, but clearly essential issue is the acceptance of farmers to use urine as a fertiliser. Lienert et al. (2003) researched whether Swiss farmers accept to use urine. In a survey of organic and integrated production farmers (both focussed on sustainable agricultural production), 57 % of the respondents found the use of urine a good idea. When asked for the willingness to buy urine as a fertiliser product, 42 % were willing to do so, but only at the same or a lower price than the fertilisers already used at the farm. The main concern of the responding farmers were micro-pollutants and hormones – a seemingly ‘technical’ question that requires a clear answer to counter the acceptance issue of urine's use in agriculture. Also, the respondents preferred the recycled fertiliser product to be granular (Lienert et al., 2003). Andersson (2015) approached the question of acceptance differently, by undertaking action research through joint experiments with farmers. She argues that such an approach might motivate farmers to use source-separated urine as the benefits of increased crop growth with this cheap fertiliser are experienced and shown to community members (Andersson, 2015).

### 3.3.4. Adjusting the N:P ratio of urine to match crop demands

Urine is often referred to as mainly a nitrogen (N) fertiliser due to its relatively high N:P ratio (Germer et al., 2011; Sangare et al., 2015). In fact, the N:P ratio of urine is often unknown and varies. It depends on several factors, such as a person's diet, the amount of drinking water consumed, physical activities, body size and the environment. Heinonen-Tanski et al. (2007) for example took samples of urine from two different households, both located in the same region and both sampled in the same time of the year. Measured N:P ratios were 8:1 and 42:1, indicating the possible variety of N:P ratios in urine. Karak and Bhattacharyya (2011) provide a review of urine analysis, showing that N:P ratios often vary around 10:1. Possible N:P ratios in plants vary considerably, but values below 10:1 indicate N shortage and those above 20:1 P shortage. Yet, as P is not taken up as efficiently as N by plants, urine is typically seen as mainly a N-fertiliser.

When determining the amount of urine to apply on agricultural land, application should be limited based on the N requirements of the crop rather than the P requirements to avoid an oversupply and consequently losses of N (Germer et al., 2011). Additional P from another source such as wood ash could be added to the urine to match crop requirements with the relatively high N-content of urine (Heinonen-Tanski and van Wijk-Sijbesma, 2005; Pradhan et al., 2009). Another solution is to apply urine on crops with a high demand for N, such as cabbage (Pradhan et al., 2007).

### 3.3.5. Transportation of urine when collected in cities

When UDTs would replace existing sewage infrastructures in urban areas, the transportation of the bulky urine with relatively low P concentrations per mass product would be a challenge. Several authors mention this challenge to transport urine from its collection point to agricultural sites (Kirchmann and Pettersson, 1994; Germer et al., 2011; Simha and Ganesapillai, 2017). Estimates on the maximum distances that can be reached when transporting urine vary, and are both a sustainability challenge (not consuming more energy than saved at the



WWTP by source-separating the urine (Jönsson, 2002; Tidåker et al., 2007) and a financial challenge (Jönsson, 2002). A possible answer to the transportation challenge could be to precipitate struvite from urine. Precipitation of struvite from urine is less expensive than the precipitation of struvite from sewage sludge and even possible with low-cost precipitation technology. Yet, with the current fertiliser prices the precipitation of struvite from urine will not be cost-effective (Etter et al., 2011).

#### 4. Discussion

Triggered by the potential contribution that P recycled from human excreta could make to slow down P depletion, we researched what phosphorus recycling from human excreta in agriculture actually entails in terms of social, ecological and technical (SET) interactions that together make phosphorus recycling possible.

The literature review indicates a clear imbalance in how the social, ecological and technical sciences are involved in understanding P recycling from human excreta. Within the wastewater system we have mainly encountered technical research focused on improving recovery efficiencies while the recycling steps through the agri-food system are dominated by socio-ecological research. While disciplinary research is indeed needed to improve specific steps in P recycling, interaction between disciplines is crucial in understanding P recycling and making it actually work. Here, we explain with two examples why interaction between the ecological, social and technical sciences is needed in optimising P recycling from human waste, and thus optimally making use of P contained in human excreta to partially replace mined fertilizer input.

First, the poor linkage between wastewater treatment and agriculture creates the problem that P is first considered waste in the wastewater sector and should later be treated as a valuable resource in agriculture. This essential transformation – from waste to resource – is highly challenging in all three routes. Sewage sludge, the major wastewater at the WWTP, is still treated as waste when applied on agricultural land, which supports the oversupply of P and hence unrecoverable P losses to the environment. Crucial in making the use of sewage sludge more efficient is starting to see it as resource, so taking into account the nutrient demand of crops, while knowing the P release pattern of the specific sludge and using this to determine application rates, rather than basing sewage sludge application rates on waste-disposal objectives. The wastewater and sewage sludge treatment could be adjusted to the agricultural demand to reach an optimal P release pattern for crops that are cultivated in the vicinity of the WWTP. In addition, the use of Fe or Mg in the treatment of wastewater should be avoided as it hinders P recycling further in the cycle. To effectively match the P release pattern of sewage sludge with crop demands, interactions between WWTP engineers, local agricultural producers and agronomists are essential. Also for the struvite route, a better connection between the domain of wastewater treatment and agriculture would be beneficial. The price of struvite cannot compete with the price of mineral fertilizers, partly because its high production costs. Yet, struvite recovery also produces environmental benefits: less P is discharged via the effluent (entering surface waters) or the sewage sludge (which has to be disposed as waste). If the alternative waste disposal and ensuing pollution are discounted, struvite prices might become more competitive.

Secondly, while zooming out from the recovery technologies to the wider network that makes P recycling possible, a key loss of P is made in the absence of sufficient sewage infrastructure, resulting in 59 % of P never entering a working WWTP. Part of this flow could be recycled via composting toilets, but greater parts are likely lost in open defecation and disposal of sewage to the environment via open drains. To incorporate this P flow into recycling strategies, decisions have to be made on how to collect human excreta, be it via new or improved connections to the sewage system, the introduction of source-separated toilets or other novel options. These decisions should not only be made

by those operating in the wastewater system and thus likely treating human excreta as waste to be disposed. Also potential users of the recovered P-product in the agri-food domain need to co-decide to ensure the recovered P-product can indeed become a valued resource in agriculture, and be used for food production to further close the P-cycle.

While literature on P recycling via sewage sludge, struvite and source-separated urine enabled us to map the routes till their application on agricultural land, little is known about how different (compared to mineral fertilizer) the recovered P from sewage sludge, struvite or source-separated urine would exactly move through the agri-food system, once applied on agricultural land. Would the use of recycled P from human excreta lead to similar inefficiencies of P in the agri-food system (including losses like erosion and food waste) as P from mineral fertilisers? For example, sewage sludge application with its additional organic matter applied to the soil improves soil structure, retains water better, thus limits the risk of erosion (Bloem et al., 2017). Agricultural yields could also reduce when high ratios of salts accompany the recovered P from human excreta, when sewage sludge is applied in excess (Bloem et al., 2017); when excessive amounts of N in the recycled P-product lead to lodging (Mantovi et al., 2005); or when the application of urine leads to soil acidification (Sangare et al., 2015). Also, it should be further explored whether the association of food cultivated on human excreta would hinder – or imply adaptations in – its retail and consumption.

The P balances are based on little quantitative data, and we had to extrapolate results from context-specific case studies to global recovery. Hence, the P balances serve as a framework to discuss – in more qualitative terms – what P recycling entails. Via the sewage sludge route, we estimated a SET-recovery efficiency of 26 %, meaning the possibility to replace 5 % of mineral P input in agriculture. Via the struvite route, 3–8 % of P from human excreta (hence replacing ca. 1 % of mineral P input in agriculture) can be recovered if all currently functioning WWTPs could make use of anaerobic digestion, hence replacing ca. 1 % of mineral P input in agriculture. When UDTs would be introduced in households currently without safely managed sanitation, 11–17 % of all P in human excreta could be recovered, hence replacing 2–3 % of mineral P input in agriculture. These percentages do not yet take into account the challenges that will be encountered in getting recovered P into the agricultural system, as these could not be quantified.

To recover as much P from human excreta as possible for agricultural use, all P recycling routes need to operate in parallel, including those not mentioned in this review (Cordell et al., 2009). Ideally, the sewage sludge, struvite and urine routes are combined with additional P recovery routes (such as composting of faeces) to make optimal use of P and to avoid losses to the environment. Also, the treated effluent that is allowed to be discharged to the environment contains 10–20 % P from human excreta and could be used to fertilise agricultural land. As different recycling routes face different agricultural challenges, they can be used as fertilisers for different agricultural settings.

The three analysed recycling routes have little overlap. The recovery of P via source-separated urine, makes use of P not entering WWTPs, while sewage sludge and struvite do. Also in terms of its application, urine use as fertiliser is mainly promoted in areas where fertilisers are not available – meaning that P recovered via the source-separated urine route does not replace mineral P input in agriculture, but it has the potential to increase agricultural production. The sewage sludge route and struvite route overlap within the wastewater sector, both making use of P contained in sewage sludge. Yet, as via struvite precipitation only a part of the P is recovered from sewage sludge, and in general more P is applied via sewage sludge than crops require, the two routes do not necessarily compete.

#### 5. Conclusions

By employing a social-ecological-technical (SET) approach we have shown the many interactions within the wastewater and agri-food

system that enable P recycling. Our analysis suggests that a SET approach helps in defining the different steps that recovered P needs to take along the recycling routes; where P possibly diverts from the recycling routes and how P-recycling could be improved. As the potential to replace mineral fertilisers with recycled P from human excreta appears lower from a SET-point of view as compared to literature on recovery technologies and global P balances, more sustainable re-use of P in the agri-food system is urgently needed.

### Declaration of Competing Interest

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

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