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Improving the resource footprint evaluation of products recovered from wastewater: A discussion on appropriate allocation in the context of circular economy

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ABSTRACT

Shifting from a linear to a circular economy has consequences on how the sustainability of products is assessed. This is the case for products recovered from resources such as sewage sludge. The “zero-burden” assumption is commonly used in Life Cycle Assessment and considers that waste streams are burden-free, which becomes debatable when comparing waste-based with virgin material-based products in the context of the growing circular economy. If waste streams are considered as resources rather than waste, upstream burdens should be partly allocated to all products to allow a fair comparison with their virgin material-based equivalents. In this paper, five allocation approaches are applied to allocate the resource use of upstream processes (consumer goods production) to products recovered from the processing of sewage sludge in the Netherlands, which produces biogas, (phosphorus-based) chemicals and building materials.

Except for the approach which allocates 100% of the impact from resource recovery processes to the preceding consumer goods, the allocation approaches show a resource use 27 to 80% higher than with the “zero-burden” assumption. In this particular case, using these allocation approaches is likely to find little support from recyclers. The producers of household products, recyclers and policy makers should find a consensus to consider the shift from a linear to a circular economy in sustainability assessment studies while avoiding discouraging the implementation of recovery technologies. This paper suggests starting the discussion with the approach which allocates the impacts from upstream processes degressively to the downstream products as it best translates the industrial ecology principles.

1. Introduction

Until recently, household wastewater treatment was mainly considered as a step to reduce the emission of harmful substances to the environment and recover water for human activities. However, households’ wastewater contains large amounts of substances that could have a secondary use in the economy. This is the case for nutrients and organic matter which could be valorized as fertilizers and biogas (energy), amongst others (Verstraete and Vlaeminck, 2011). Resource recovery from wastewater streams is increasingly seen as one option to help tackling challenges such as the resource efficiency of regions and countries and the low revenues from wastewater treatment (IWA, 2016; Spinosa et al., 2011). Using sewage sludge as a fertilizer has been considered for many years but is often limited by a heavy metals content that exceeds the maximum allowed in regulation (Linderholm et al., 2012). To overcome this challenge, technologies to extract the useful compounds of sewage sludge and produce “heavy metal free” fertilizers such as struvite are being developed. The wastewater sector is also developing several other innovative technologies, e.g., to recover metals and ammonia or to produce biogas, bio-plastics, biodiesel, esters, fish or microbial protein from sewage sludge (Alloul et al., 2018; Puyol et al., 2017; Verstraete et al., 2016). Therefore, the wastewater

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treatment sector is increasingly positioning itself as a key player in the shift towards a circular economy (IWA, 2016). However, this requires a paradigm shift related to the main goal assigned to wastewater treatment today, i.e., to avoid pollution of receiving water bodies. Renaming wastewater treatment plants (WWTP) into water resource recovery facilities (WRRF) boosts the shift from the "water cleaning" to the "resource recovery" approach by considering giving a second life to resources in wastewater as a major goal of the wastewater treatment chain (Vanrolleghem and Vaneeckhaute, 2014). This paradigm shift has consequences on how the sustainability of products obtained from wastewater is to be assessed. Life Cycle Assessment (LCA) is a tool commonly used to assess the sustainability of products and services. It is a recognized methodology to assess the environmental burdens of a system and follows the framework of International Standards Organization (ISO) 14040 and 14044 (ISO, 2006a, 2006b). It allows comparing the environmental impact of different steps of a process, identifying the steps which could be improved and avoiding environmental impact shifting from one step to another. However, some methodological approaches commonly used in LCA become debatable when it comes to compare products from sewage sludge valorisation in circular systems with virgin material-based products. The "zero-burden" assumption was described by Finnveden (1999) as an approach followed in comparative waste-LCA and which considers that "*those parts of the systems which are identical in all systems which are compared, can be disregarded*". Finnveden (1999) further specifies that if different amounts of waste are produced in the compared scenarios, the upstream processes should be included in the system boundaries. If this definition is strictly followed, the processes upstream waste production have to be included in the system boundaries when comparing products recovered from waste with their virgin material-based equivalents. In practice today, this approach is not implemented because the concept of "zero-burden" assumption has become broader, considering that waste streams do not bear any burden, even in a broader context than waste-LCA. However, since the definition of the "zero-burden" assumption twenty years ago by Finnveden (1999), a new paradigm has emerged, the one of circular economy. The Ellen MacArthur Foundation defines "designing out waste" as one of the three principles of circular economy (Ellen MacArthur Foundation, 2017), which means that no waste should be produced by circular systems, only by-products (Djuric Ilic et al., 2018) and resources used in further processes. As the "zero-burden" assumption applies to waste streams (Ekvall et al., 2007), it might become obsolete and inconsistent in the assessment of circular systems. In the field of wastewater treatment it means that in practice, if wastewater streams are considered as a resource and not as a waste, the upstream environmental burdens should be partly allocated to the downstream products to allow a fair comparison with the equivalent virgin materials-based products. A similar paradigm shift can be observed in the solid waste management sector in which there is a growing discussion on the necessity to allocate part of the impact from the upstream processes (i.e., the production of the products which will turn into waste) to the recycled products (Chen et al., 2010; Oldfield and Holden, 2014). The recent ecoinvent model "allocation at the point of substitution" also follows this approach and allocates the environmental burden of primary production to solid waste streams by considering them as co-products (Weidema et al., 2013). However, this approach is not yet applied to wastewater streams. It has been recently discussed by Pradel et al. (2016), who reviewed the modelling approach followed by 44 LCA studies assessing the environmental sustainability of sewage sludge management. This study shows that the sludge is always considered as a "burden free" flow. The authors stress that such an approach can be followed when comparing different sewage sludge management options but becomes debatable when comparing the environmental sustainability of products obtained from the valorisation of sewage sludge with virgin materials-based products. In these cases, Pradel et al. (2016) argue that part of the environmental burden of the WWTP should be allocated to the sewage sludge. However, the products from sludge valorisation do not only rely on the treatment of the wastewater to be produced. They also rely on the production of the products ending up in the wastewater streams (i.e., consumer goods). Therefore, the rationale of Pradel et al. (2016) could be extended to the allocation of part of the environmental burden from consumer goods' production to the products from sludge valorisation. The wastewater treatment chain is viewed as a cascade system in which natural resources are first used to produce the consumer goods and then partly used to produce new products from the valorisation of sludge from wastewater. The sector of material recycling is already dealing with such a situation and developed several approaches to allocate the impact of virgin raw material processing to the different products of a cascading chain. These approaches also allocate part of the impact of recycling to the products of the chain. In the context of the Product Environmental Footprint (PEF) initiated by the European Commission (EC, 2013), Allacker et al. (2017) present different "end-of-life formulas" commonly used in literature. An example is the "adapted 50:50″ approach which allocates 50% of the environmental burden of the virgin raw material processing and recycling process to the material being recycled (Allacker et al., 2017). The recovery of resources from consumer goods discarded by households in the sewage system is similar to the recycling of materials. The used products enter a "recycling" process, which starts with the WWTP discharging water and producing sewage sludge and ends with the sludge treatment processes to obtain final products. Therefore, the "end-of-life formulas" applied to recycled materials could also be applied to the products used by households and used to produce products from sewage sludge valorisation.

This study aims to propose alternatives for the zero-burden assumption to consider the shift from a linear to a circular economy in sustainability assessment studies. It starts by rethinking the way wastewater and sludge treatment processes are considered in these studies. Then, allocation approaches inspired by the so-called "end-of-life" formulas are proposed to assess the resource footprint, i.e., the cumulative amount of natural resources consumed, of products from sewage sludge valorisation and consumer goods. This methodological approach is tested on two sewage sludge valorisation scenarios from the WWTP of the city of Eindhoven (the Netherlands). The products recovered from sewage sludge valorisation are compared with equivalent benchmark products.

2. Materials and methods

2.1. A novel approach to assess the environmental sustainability of wastewater-based products

This section aims to present a new approach to assess the environmental sustainability of wastewater-based products in the context of their comparison with the virgin material-based equivalent based on LCA. In Section 2.2, this approach is applied to the case of the Eindhoven wastewater value chain.

2.1.1. Rethinking typical wastewater value chains

The value of any wastewater is the result of upstream processes, i.e., the production of the products consumed and ending in the collection system. This paper proposes to consider these processes as part of a "wastewater value chain" (Fig. 1) to account for their contribution to the value of the sludge-based products.

A "wastewater value chain" starts from the production of food and non-food products that will end in the collection system. It includes the extraction of raw materials and their processing. The products are consumed and part of the food ends up as food and kitchen waste. The consumption of food allows fulfilling the needs of the human body through the uptake of energy and nutrients and results in the production of a mix of water, urine and feces. In parallel, the non-food products (e.g., laundry product) end up in the sewage system.

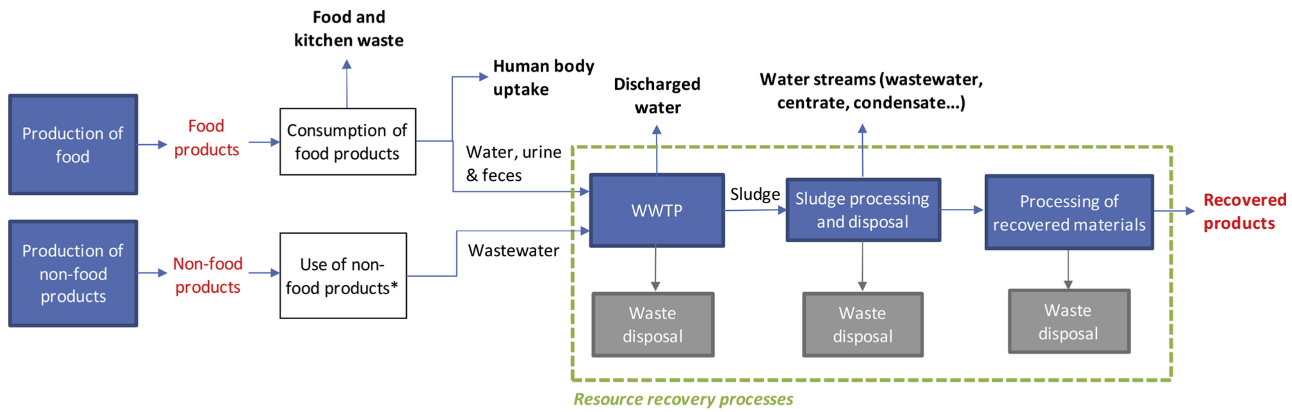

Fig. 1. Wastewater value chain (* non-food products ending in the sewer).

The wastewater enters a recycling process named here "resource recovery processes" which consist of the wastewater treatment at the WWTP, the sludge processing and the processing of the recovered materials. Considering the new paradigm of waste-as-a-resource, several products are obtained along the wastewater value chain: the food and kitchen waste, the human body uptake, discharged and various water streams, and the recovered products (e.g., struvite). This paper focuses on comparing the environmental sustainability of products recovered from wastewater streams with their virgin material-based equivalents. To focus on these products, a division of the value chain into sub-chains is necessary.

2.1.2. Partitioning of the wastewater value chain

In LCA studies, the division of a multi-outputs process chain into sub-chains to focus on the product of interest is common practice. It is generally referred as "allocation". In this paper, in order to avoid any confusion with the next step of the proposed approach, the term "allocation" is replaced by "partitioning". The partitioning of a process burden between its several outputs can be based on different flow properties, e.g., mass, energy, exergy and economic value. While the partitioning of process burdens between physical flows is common, it is not the case for partitioning human consumption between body uptake (which cannot be physically characterized), human excreta and food waste. As human food intake requirements are mostly characterized in terms of energy intake, the partitioning of human consumption can be based on the energy or exergy values of its different outputs. Once the partitioning of the wastewater value chain has been made, several sub-chains are obtained (Fig. 2). Sub-chain 5 can then be analysed to assess the sustainability of the recovered products.

2.1.3. Allocation of the burdens to the different products along the chain

In sub-chain 5, resources are consecutively used to produce consumer goods and recovered products. Then, a similar approach as followed in the sector of material recycling is proposed. It allocates the burdens of the processes along the chain to the different products of the chain (here the consumer goods and the recovered products). Allacker et al. (2017) present 11 end-of-life formulas that can be applied to products used consecutively in a cascade system. Some simply differ by considering avoided virgin production by the recycled product. In this paper, we aim to compare the recovered products with benchmark products so these methods are discarded. Moreover, Allacker et al. (2017) discuss four methods based on the 100:100 principle, meaning that 100% of the impact of recycling is allocated to the recycled products and 100% is allocated to the product producing the recycled material, which results in a double counting of the impact when considering the overall system. To keep a consistent system which results in "physically realistic modelling" (Allacker et al., 2017), these end-of-life formulas were not considered in the analysis either. The five remaining approaches are described in Table 1 and further detailed in Appendix D.

The 0:100, 50:50, "50:50 adapted" and "linearly degressive" approaches imply to know if the recovered products are disposed of, or recycled after use. If recycled, the burden from recycling should be fully or partly allocated to the recovered products. For example, it implies knowing if roadfilling material obtained from sludge incineration ashes is disposed when the lifetime of the road ends, or recycled/reused for another application. However, this study aims to compare recovered and benchmark products for which the disposal or recycling steps are the same so the impact of the downstream steps that should be allocated to the recovered products can be excluded. This has a consequence for the "linearly degressive" approach for which the percentage of impact allocated along the chain depends on the number of times a product is recycled before final disposal. Most of the time, this information cannot be known because of a lack of tracking of materials during their whole lifetime. Therefore, the approach "linearly degressive" was slightly modified compared to the one described in Allacker et al. (2017). Instead of being shared between all the products of the chain until final disposal, the burden of the virgin material is shared between the virgin material-based product (here the consumer goods) and the first product from recycling of this material (the recovered products), but in a degressive manner. This allows applying the principle of degressive allocation without having to know how the recycled products are then used for. Allacker et al. (2017) propose to use the following factor to allocate the impact of virgin material to the different products of the chain:

$$f = \frac{2 \times n - 1}{n^2} \tag{1}$$

Where n is the number of products along the chain. In a typical wastewater value chain, two types of products are obtained (Fig. 1): 75% of the burden of virgin material extraction and processing is allocated to the virgin material-based product, and 25% is allocated to the product obtained from the first recycling process. The responsibility of the recycling processes is equally shared between both products. The approaches proposed are presented in Fig. 3 for the sub-chain 5.

The approach presented in Fig. 3 should also be applied to the sub-chains 1, 2, 3 and 4 in order to quantify the burden from the downstream processes allocated to the consumer goods. The burden of the consumer goods in the sub-chains 1 to 5 are then summed up to obtain the total burden. Therefore, following the proposed allocation approach has an effect on the footprint of both the consumer goods and the recovered products.

2.2. Application to the resource footprint of products recovered from the wastewater treatment chain of the city of Eindhoven

The proposed approach is tested to compare the resource footprint of products obtained from the wastewater treatment chain of the city of Eindhoven with their virgin material-based equivalents (i.e., benchmark products).

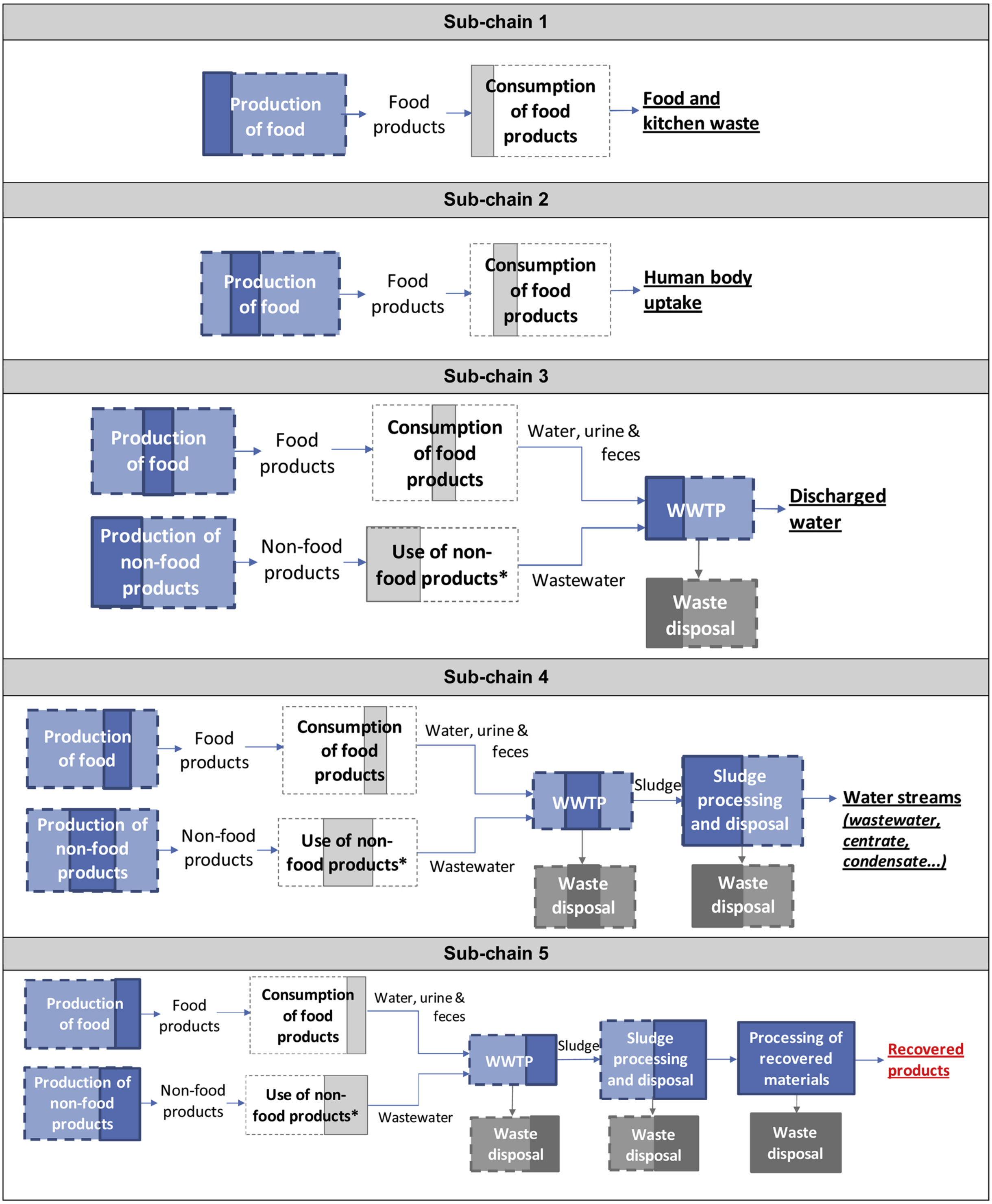

Fig. 2. Partitioning of the wastewater value chain presented in Fig. 1 (the darker portions indicate the partitioning of the processes in each sub-chain; * non-food products ending in the sewer).

2.2.1. Scenarios

The value chain starts with the production of the consumer goods ending up in the sewage system. Because the focus of this study is the testing of a new approach on the wastewater treatment chain, food and kitchen waste are assumed to be incinerated (see Section 4 for discussion). Sewage ends up in the Eindhoven WWTP managed by Waterschap De Dommel, which has a capacity of 680,000 person equivalent (PE; 1 PE defined as 150 g COD day^{-1}). The effluent flows into the river Dommel. Primary and thickened secondary sludge are pumped to a facility in Mierlo, where they are mixed with the sludge of four other WWTPs and dewatered in centrifuges. The centrate is pumped back to Eindhoven WWTP. Two scenarios of sludge treatment were then assessed (Figs. 4 and 5).

2.2.1.1. Baseline scenario. The dewatered sludge is transported by truck to an incineration plant in Moerdijk (N.V. Slibverwerking Noord-Brabant (SNB)) where it is dried and incinerated. Part of the CO_2 produced during incineration is used by a neighboring plant to produce

Table 1
Description of the selected allocation approaches.

Allocation approach	Description
0:100	Full allocation of the recycling impact to the intended product and no burden allocated to downstream products using secondary materials.
100:0	Full allocation of the recycling impact to the product using secondary material, with no burden from recycling operations allocated to the intended product. This approach is usually followed in LCA. In this case study, it is different from the zero-burden assumption as the later does not consider the WWTP as a resource recovery process while the 100:0 applied here does.
50:50	Allocation of the recycling impact to the intended product and 50% to the product using the secondary material.
50:50 adapted	Distributes the impacts due to recycling in a 50:50 manner over the different products in the overall product cascade system but also the virgin material and disposal impact.
Linearly degressive	Uses the 50:50 approach for the allocation of the recycling impact. Allocates the impact of the virgin material in a linearly degressive way to all products in the product cascade system, allocating the highest share of impact to the first product. Same approach with disposal, but allocating the highest share of impact to the last product.

calcium carbonate ($CaCO_3$). All the energy produced during incineration is consumed for drying. In 2013, 36,359 tons of incineration ashes were produced, 78% of which were used as building material (58% as roadfilling material and 21% to produce landfill capping material) and 3% as phosphoric acid for fertilizer production in the EcoPhos plant (Dunkirk). The EcoPhos process produces two other products: calcium chloride ($CaCl_2$) and an iron chloride ($FeCl_3$) solution. The remaining fraction of ashes (18%) was transported to a salt mine in Germany for long-term storage and the waste adsorbents were landfilled. The products of the treatment of sludge are called "recovered products" and the processes from the WWTP to the production of the recovered products are called the "resource recovery processes", including the disposal of waste from the incineration plant. The condensate from sludge drying is treated in the wastewater treatment facility of the incineration plant and discharged.

2.2.1.2. Alternative scenario. The alternative scenario is based on upcoming improvements from Waterschap De Dommel. This scenario consists in subjecting the output sludge of several WWTPs to anaerobic digestion before incineration. The dewatered sludge is transported by truck from Mierlo to Tilburg, pre-treated with a thermal hydrolysis process (THP) and then digested. The biogas is pumped via pipelines to a company that purifies and compresses it to produce biomethane used in city buses. The digestate is dewatered, and the residual sludge transported to the incineration plant. The same valorisation pathways for ashes as in the baseline scenario are considered. The reject water from dewatering is treated in a precipitation process to produce struvite ($MgNH_4PO_4.6H_2O$), a mineral slow-release fertiliser containing nitrogen and phosphorus.

2.2.1.3. Benchmark scenarios. Both scenarios are compared with benchmark scenarios producing equivalent products. In the benchmark scenarios, roadfilling material and landfill capping material are produced from gravel (Birgisdóttir et al., 2007) and bentonite clay (Guyonnet et al., 2009), respectively. CO_2 is produced from the treatment of different industrial gases, H_3PO_4, the $FeCl_3$ solution, $CaCl_2$ and the N and P fertilizers are produced as described in the ecoinvent database (Frischknecht and Rebitzer, 2005). The city buses run on diesel.

2.2.2. Life cycle assessment

2.2.2.1. Goal and scope. The effect of the proposed approach is tested on the comparison of the resource footprint of the recovered products with their virgin material-based equivalent. A first analysis is conducted based on sub-chain 5 only and considers the basket of products recovered from household sewage sludge from Eindhoven during one year (Table 2) as the functional unit. The results of this first analysis are presented in Section 3.1.

The water discharged by the WWTP and the incineration plant are not included in the basket of products because it is released in the nearby rivers and not used in a downstream industrial process. The output wastewater from the sludge processing steps is excluded as not further valorized in an industrial process.

The production of biogas reduces the amount of carbon in the sludge so less CO_2 is produced during the incineration of the sludge in the alternative scenario. However, the amount of CO_2 delivered to produce $CaCO_3$ is assumed to remain the same as in the baseline scenario as the $CaCO_3$ producer requires a continuous supply of CO_2.

In addition to having an impact on the resource footprint of these products, the allocation approaches also have an impact on the resource footprint of the consumer goods. Therefore, a second analysis was conducted considering the basket of consumer goods consumed/used by the city of Eindhoven during one year and ending up in the sewage system as a functional unit (Appendix A). The resource footprint of the consumer goods is the sum of their resource footprint in sub-chains 1 to 5. The results are presented in Section 3.2.

Figs. 4 and 5 present the system boundaries. The packaging of consumer goods is excluded as these do not end up in the sewage. The impact from food preparation is neglected as it represents less than 5% of the resource footprint of food consumption (Notarnicola et al., 2017). For non-food products, only the impacts from the ingredients and their transport to the processing plant are included because of the negligible contribution of their processing step (Golsteijn et al., 2015).

2.2.2.2. Data inventory. *Consumer goods production* - To estimate the resource footprint of the consumer goods, the consumption patterns of food and non-food products released in the wastewater stream had to be estimated. Based on RIVM (2011), 47 products were selected to represent the complete diet of the Dutch population. Their production was modelled using the life cycle databases ecoinvent version 3.3 (Frischknecht and Rebitzer, 2005), the Agri-footprint database (version 3.0; Blonk Consultants (2017)) and the LCA Food database (2.-0 LCA Consultants, 2003). 10% of consumed food is assumed to be wasted (LNV, 2010) and the amount of kitchen waste was estimated based on literature data (e.g., Mahmood et al. (1998) for potato peel) and on the author's estimation.

The non-food consumption patterns were estimated based on RIVM (2006), 2002 and AISE (2014). The composition of the body and house care products was based on the RIVM reports and Golsteijn et al. (2015). Ingredients with renewable origin were assumed to be transported by boat (8000 km) and the ingredients of non-renewable origin by truck (2000 km) (Golsteijn et al., 2015).

Resource recovery processes - Data of the facilities in Eindhoven and Mierlo were retrieved from Blom, 2013Blom(2013). The WWTP treats both household and industry water. The inventory from the plant was allocated to the household stream based on the COD content (74%). Data for digestate dewatering and struvite precipitation were taken from literature (see Appendices). Data on inputs for the incineration and the destination of bottom ashes were extracted from Sijstermans and van der Stee (2013). Chemicals were not included in the assessment. The resource consumption of the incineration plant (which also processes sludge from other WWTPs) was allocated to the sludge from

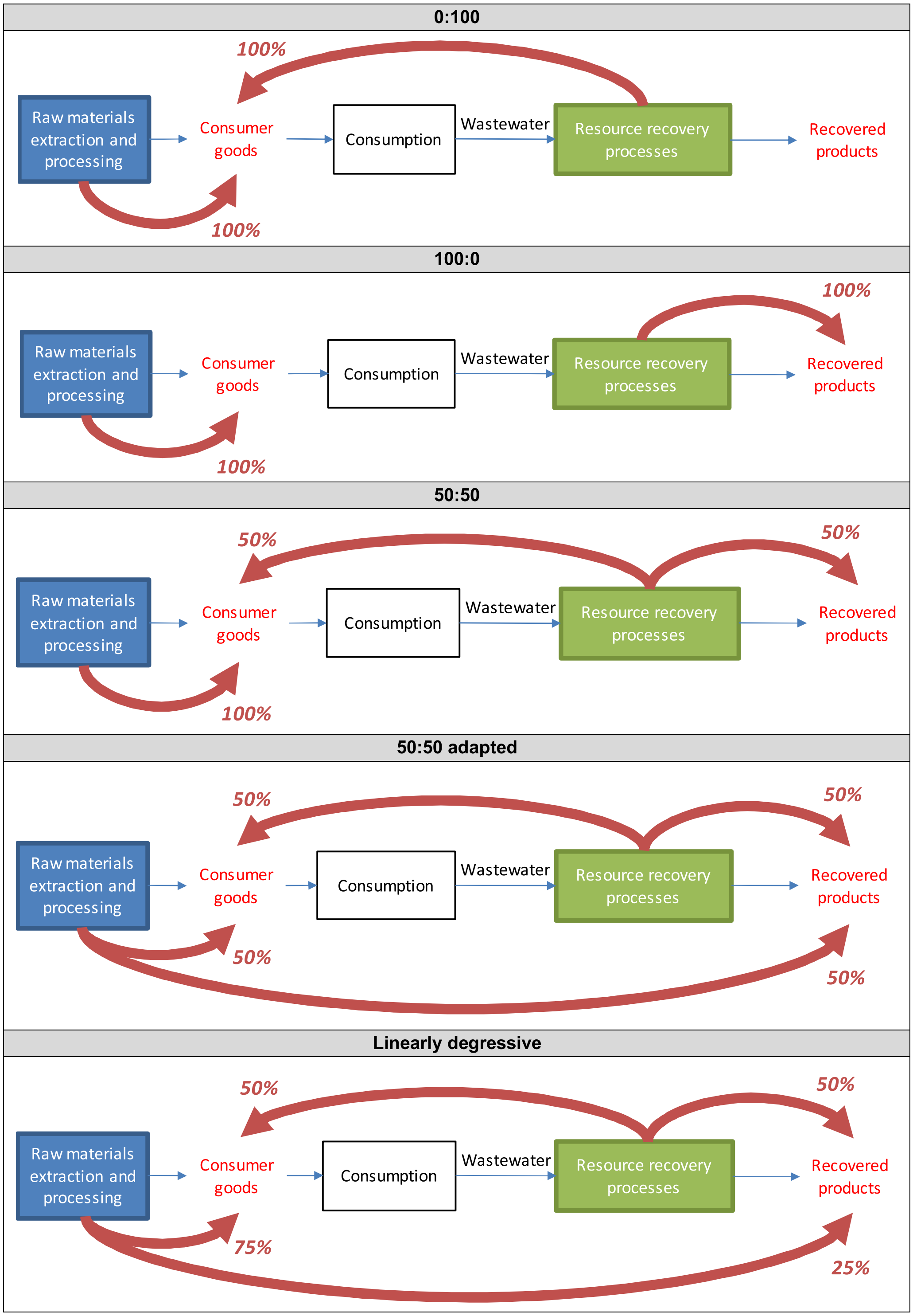

Fig. 3. Visualization of each allocation approach. Red arrows represent the allocation of the environmental burden of processes to specific products (in red: consumer goods or recovered products). Percentages represent the share of the environmental burdens. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

Eindhoven based on its dry solids contribution (13%). The ashes valorized as landfill capping and roadfilling materials are used without any processing step. Data for the EcoPhos process were taken from Jossa et al. (2015).

Based on the inventory, the phosphorus flows within the resource recovery processes were estimated to obtain the final amount of P-containing products in the baseline and alternative scenarios (Fig. 6).

Background processes - The background processes (e.g., production of electricity from the grid and benchmark processes) are modelled based on the ecoinvent database version 3.1 (Frischknecht and Rebitzer, 2005). To be consistent with the co-products partitioning approach of the foreground system, the ecoinvent modelling approach “allocation at the point of substitution” is used.

Ashes used as roadfilling and landfill capping materials are assumed to replace their equivalent products with a 1:1 ratio (Birgisdóttir et al., 2007). A 1:1 ratio is used to estimate the equivalence between the

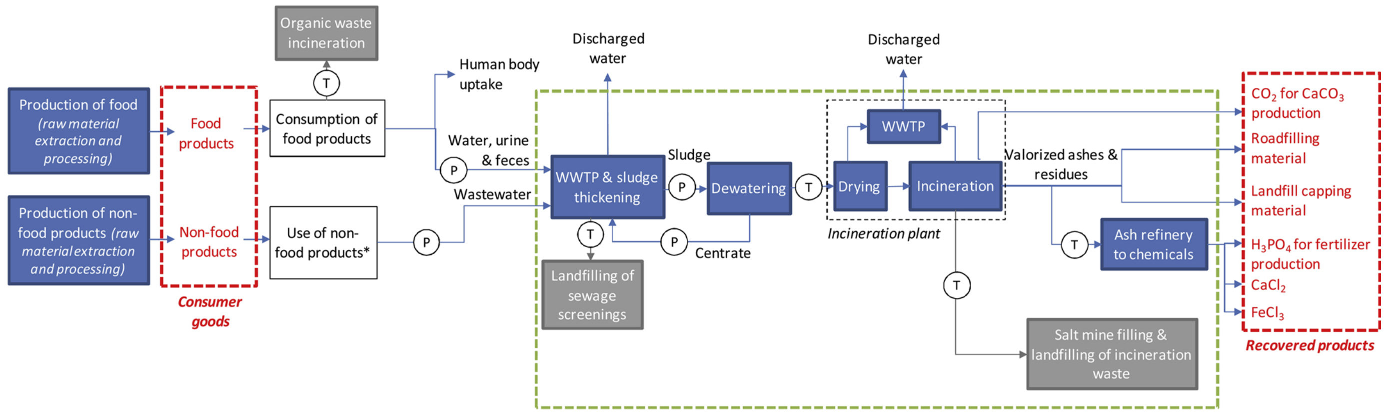

Fig. 4. Baseline scenario (the grey boxes represent the disposal processes; the white process boxes are excluded from the system boundaries; WWTP: Wastewater treatment plant; T: Transport by truck; P: Transport by pipeline; * non-food products ending in the sewer) (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.).

recovered H_3PO_4, $FeCl_3$ solution and $CaCl_2$ and the virgin material-based products, as no impurities which could decrease their value are assumed to be present in the recovered products. 1 Nm^3 of biogas is estimated to replace 0.7 kg of diesel fuel and 1 kg of phosphorus contained in the struvite to replace 1 kg of phosphorus in synthetic fertilizer (Amann et al., 2018; Ishii and Boyer, 2015). The same approach is followed for nitrogen.

2.2.3. Partitioning of the wastewater value chain

Several processes along the chain produce more than one product. As presented previously, the system should be partitioned to allow evaluating the resource footprint of the basket of recovered products only. The processes that produce several products are listed below:

- The consumption of food products produces the proper function of the human body through nutritional uptake of a fraction of ingested food, and the feces and urine;
- The WWTP produces the discharged water and the sewage sludge;
- Sludge processing (alternative scenario, in green in Fig. 5) produces biogas, dewatered digestate sludge, struvite and wastewater;
- The incineration plant produces ashes, CO_2 and discharged water.

For each of these processes, partitioning factors need to be defined. As mentioned in Section 2.1.2, basing the partitioning factors for food consumption on the energy or exergy value of nutritional uptake and feces/urine is the most straightforward approach. Therefore, an exergy-

Table 2
Basket of products chosen to compare the resource footprint of the current and baseline scenarios with their benchmark scenarios (in kg $year^{-1}$ unless specified).

Products	Current scenario	Alternative scenario
Roadfilling material	2.1×10^6	1.1×10^6
Landfill capping material	7.3×10^5	4.1×10^5
Phosphoric acid (H_3PO_4)	2.6×10^4	2.1×10^4
Calcium chloride ($CaCl_2$)	6.6×10^4	5.6×10^4
Iron chloride solution 40% ($FeCl_3$)	3.3×10^3	2.8×10^3
Carbon dioxide for $CaCO_3$ production	2.5×10^6	2.5×10^6
Kilometres driven by city buses	0	2.6×10^6 (*)
Phosphorus fertilizer, as P_2O_5	0	1.1×10^5
Nitrogen fertilizer, as N	0	2.2×10^4

(*) km $year^{-1}$

based partitioning is chosen for each of them to allow for consistency between processes, but also with the exergy-based method chosen to account for resource consumption (see 2.3.5).

Partitioning between nutritional uptake and feces/urine - Based on Mady and Oliveira Junior (2013), the ratio of the energy contained in feces and urine over the energy intake is used as a proxy to estimate the partitioning factor (Appendix C). 19% of the intake energy ends up in the feces and urine and is taken as partitioning factor.

Partitioning between discharged water and sewage sludge – The exergy value of the sewage sludge and the discharged water are calculated,

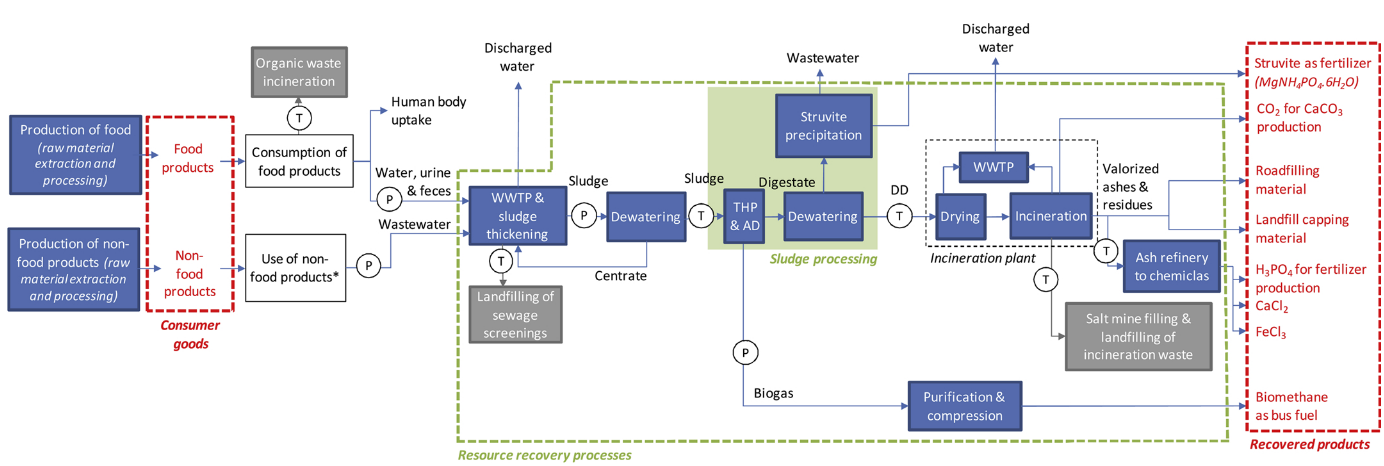

Fig. 5. Alternative scenario (the grey boxes represent the disposal processes; the white process boxes are excluded from the system boundaries; WWTP: Wastewater treatment plant; THP: Thermo Hydrolysis Process; AD: Anaerobic Digestion; DD: Dewatered Digestate; T: Transport by truck; P: Transport by pipeline; * non-food products ending in the sewer) (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

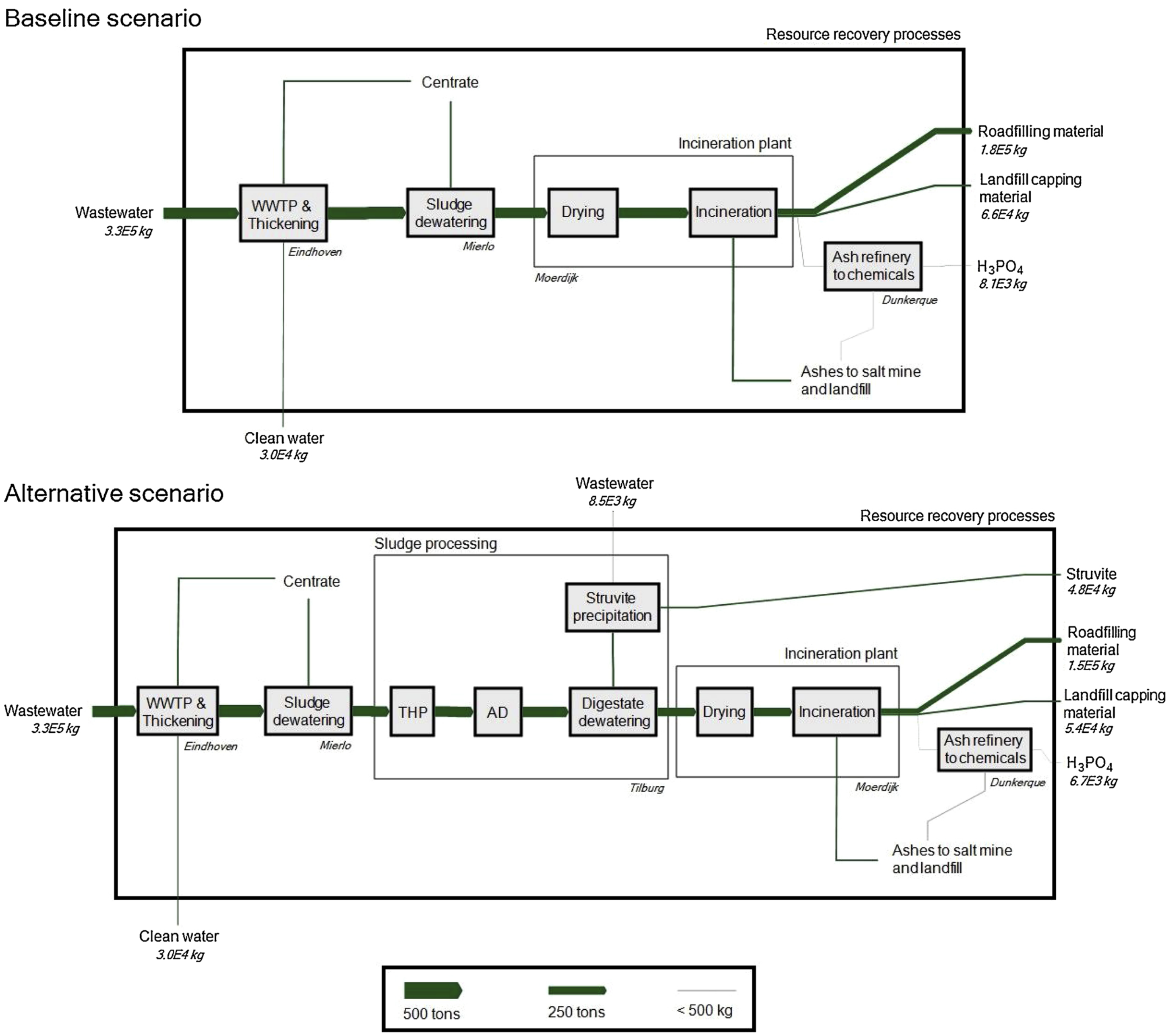

Fig. 6. Phosphorus flows within the resource recovery processes, in kg per basket of recovered products (THP: Thermal Hydrolysis Process; AD: Anaerobic Digestion; WWTP: Wastewater Treatment Plant; italic numbers: amount of phosphorus; italic names: location of facilities).

both based on a mass balance and the COD value and water content of the input and discharged water (Blom, 2013). 34% of the exergy of the wastewater ends up in the sewage sludge and is chosen as a partitioning factor.

Partitioning between the wastewater and the struvite, dewatered digestate sludge and biogas – 55.6%, 42.8% and 0.9% of the exergy of the input sludge ends in the biogas, the dewatered digestate sludge and the struvite, respectively. Therefore, 99% of the input exergy ends up in the struvite, dewatered digestate sludge and biogas.

Partitioning between the ashes, CO_2 and the condensate – 8% of the exergy of the input sludge ends in the condensate so 92% of the exergy ends up in the ashes and CO_2.

The partitioning factors are represented in Appendice E. Applying the partitioning factors results in dividing the process chain in sub-chains that each delivers one single product or basket of products (see Fig. 7 for the baseline scenario).

2.2.4. Allocation between products along the chain

The five allocation approaches proposed in Section 2.1.3 are applied to the wastewater treatment chain and are compared with the zero-burden assumption.

2.2.5. Impact assessment

The resource-based impact assessment method Cumulative Exergy Extraction from the Natural Environment (CEENE) is used. It considers seven resource categories: biotic resources and land occupation, abiotic renewable resources, fossil fuels, nuclear energy, metal ores, minerals and water resources (Dewulf et al., 2007).

3. Results

3.1. Resource footprint of the recovered products

Fig. 8 shows the resource footprints of the recovered products following the different allocation approaches. Two approaches result in a lower footprint of the recovered products than with the zero-burden assumption: the 0:100 approach, which does not allocate any impact from the resource recovery processes to the recovered products, and the

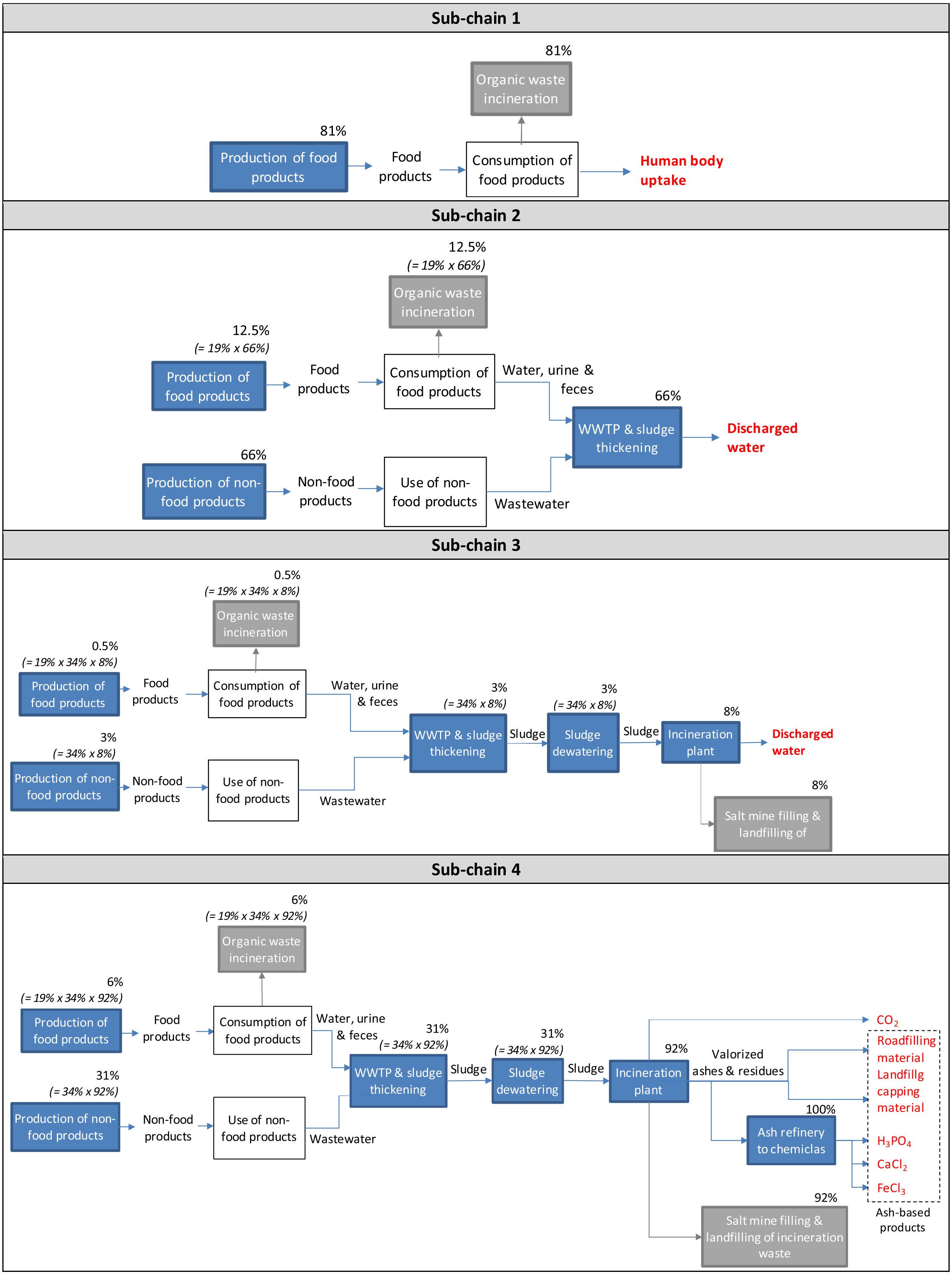

Fig. 7. Partitioning of the studied system (baseline scenario) based on the partitioning factors. The percentages represent the fraction of the resource footprint of the process allocated to the product(s) of the sub-chain. The calculation between brackets refers to the partitioning factors in Appendix E.

50:50 approach, which allocates 50% of the impact from the resource recovery processes to the recovered products. For the baseline scenario, the footprint with the zero-burden assumption is 28, 80 and 64% lower than with the 100:0, “50:50 adapted” and “linearly degressive” approaches, respectively. This difference slightly decreases when implementing the alternative scenario: it becomes 27, 78 and 62% lower

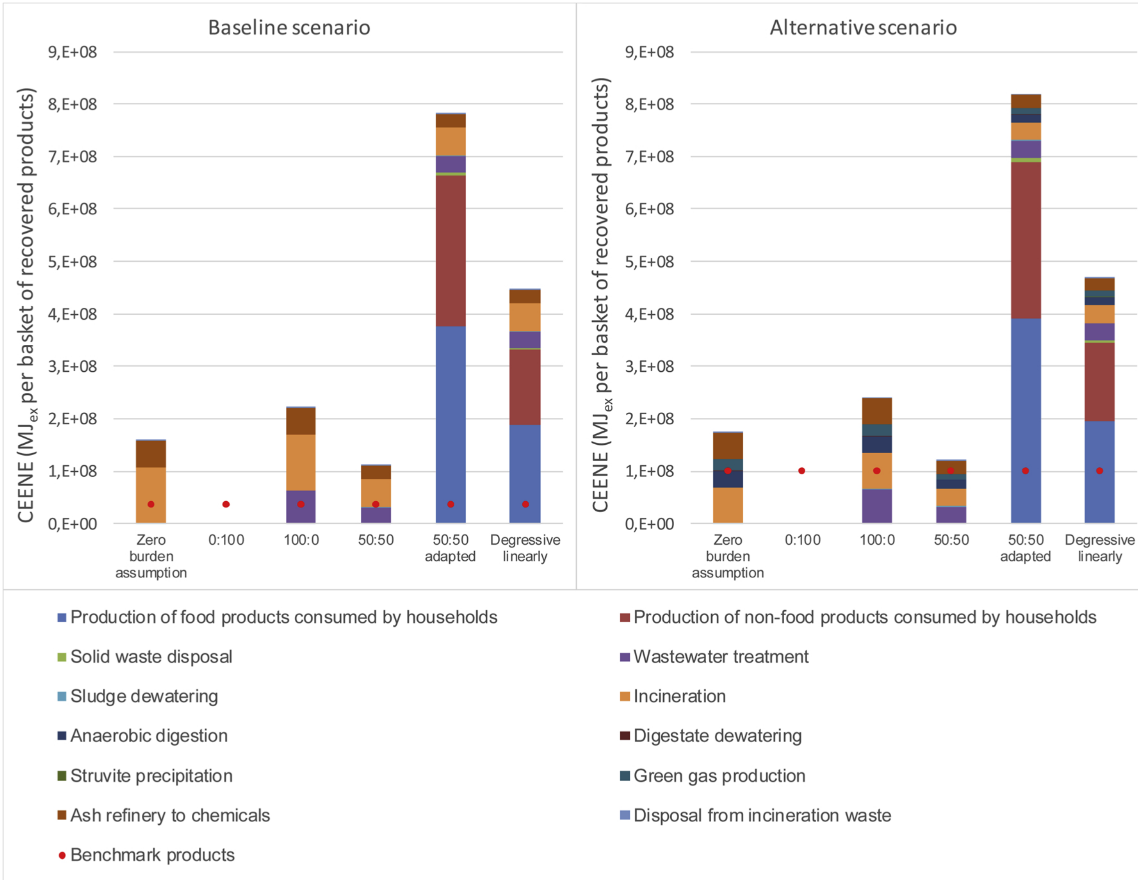

Fig. 8. Comparison of the resource footprint of the recovered products (bars) and the benchmark products (red dots) for the baseline and alternative scenarios, following the zero-burden assumption and the five allocation approaches (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

than with the 100:0, “50:50 adapted” and “linearly degressive” approaches, respectively.

With the 0:100, 100:0 and 50:50 approaches, no impact from consumer goods production is allocated to the recovered products. For the baseline scenario, the process mainly contributing to the resource footprint when following the 100:0 and 50:50 approaches is incineration (48% of the footprint). The second contributor is the WWTP (28%), followed by the EcoPhos process (23%).

In the alternative scenario, the contribution pattern changes for the 100:0 and 50:50 approaches: the contribution of incineration decreases to 28%, followed by wastewater treatment (27%), EcoPhos ash refinery (21%) and anaerobic digestion (12%). Including a digestion step between sludge dewatering and incineration reduces the amount of sludge sent to incineration and the contribution of incineration (e.g., with the 100:0 approach, the impact from incineration decreases from 1.1×10^8 to 6.8×10^7 MJ_{ex} per basket of recovered products).

With the “50:50 adapted” and “linearly degressive” approaches, part of the impact from the production of consumer goods is allocated to the recovered products. The production of consumer goods becomes the first contributor to the footprint, with 85 and 74% of the impact for the baseline scenario for the “50:50 adapted” and “linearly degressive” approaches, respectively. The share of the impact from food products is slightly higher than the share from non-food products (e.g., 48 and 37% of the footprint for the baseline scenario following the “50:50 adapted” approach).

The resource footprint of the benchmark products with the 0:100 approach is higher than the recovered products for both scenarios. This is because no impact is allocated to the recovered products. For all the other approaches, the resource footprint of the recovered products is higher than for the benchmark products. For example, the footprint of the recovered products with the zero-burden assumption in the baseline scenario is 77% higher (1.6×10^8 MJ_{ex} and 3.7×10^7 MJ_{ex} for the recovered and benchmark products, respectively). This is line with Linderholm et al. (2012) who compared the resource footprint of P fertilizer from mineral sources and from the valorisation of the bottom ashes from wastewater sludge incineration. The authors found that the burden of mineral P is around 85% lower than for P fertilizer obtained from bottom ashes. In the case presented here, this difference decreases when implementing the alternative scenario (e.g., the resource footprint of the recovered products with the zero-burden assumption becomes 43% higher than the benchmark products). This is due to the large resource footprint of bus diesel replaced by biogas (53% of the avoided footprint) and synthetic fertilizers replaced by struvite (12% of the avoided footprint). Moreover, the valorisation of the sludge as biogas reduces the amount of sludge to be incinerated, and reduces the amount of resources consumed for incineration. The case that shows the least difference with the benchmark products is the alternative scenario following the 50:50 approach. In this case, the resource footprint is 17% higher than the benchmark scenario.

This case shows that for five out of the six allocation approaches, using products from the valorisation of the ashes of wastewater sludge incineration consumes more resources than using products from raw materials. However, it also shows that including valorisation steps among the resource recovery processes reduces the resource footprint of the recovered products. Other improvement options are still possible. For example, nitrogen is completely lost during incineration, and the inclusion of nitrogen recovery steps such as air stripping of ammonia could reduce the footprint of the recovered products. Moreover, Fig. 6

shows that a large fraction of phosphorus is valorized as roadfilling and landfill capping material while it could be used for the production of higher value products.

As expected, allocating part of the resource use of consumer goods to the recovered products strengthens the conclusions of the comparison and the potential of recovered products to compete with the benchmark products becomes rather limited. However, in the context of a circular economy, considering waste streams as resources is a requirement for a successful implementation of the concept. This also implies that impact assessment approaches account for this change of paradigm and discard the zero-burden assumption. This is not favourable for the recovered products, which resource footprint becomes even larger than the virgin material-based products. This is especially because the resource footprint of consumer goods is more than 30 times higher than the one of the resource recovery processes. It implies that measures to improve the footprint of recovered products should also include measures to reduce the contribution of consumer goods.

3.2. Resource footprint of the consumer goods

The order of magnitude of the resource footprint of the consumer goods is more than ten times higher than the one of the recovered products (Fig. 9). This is due to the large resource footprint of their production, which represents more than 96% of their resource footprint.

The first contributor is the production of the food products (84 to 88% of the footprint), followed by non-food products (12% for all approaches). With the zero-burden assumption and the 100:0 approaches, no impact from the resource recovery processes is allocated to the consumer goods but for the latter, impact from solid waste disposal is allocated. The 0:100 and 50:50 approaches result in a slightly higher footprint as part of the impact from the resource recovery processes is allocated to the consumer goods. However, they only represent less than 3% of the footprint. The 0:100, 100:0 and 50:50 approaches result in a footprint which is only 4, 2 and 3% higher than with the zero-burden assumption for both scenarios. The "50:50 adapted" and "linearly degressive" approaches result in footprints 48 and 23% lower than with the zero-burden assumption for both scenarios. Therefore, while allocating part of the impact of the resource recovery processes to the consumer goods barely changes the resource footprint of these, allocating part of the impact of the consumer goods production to the recovered products highly contributes to decrease the footprint of the consumer goods.

4. Discussion

Choosing one allocation approach of environmental burden over another can appear arbitrary. However, the compliance of the approaches with the concepts of industrial ecology can still be discussed for this case study. Industrial ecology is based on the concept of waste-as-a-resource. It considers products intended to be produced, and secondary resources, which are unintended but can contribute to obtain new products and depend on the intended products to be produced. On the other hand, the unintended secondary resources should be safely managed as a consequence of the production of the intended products. The concept of industrial ecology highlights a "hierarchy of intent" (intended products and secondary resources), and a dependence of all products from the system to one another. First, some allocation approaches do not allocate any impact of virgin raw materials extraction and processing to the recovered products (the zero-burden, 0:100, 100:0 and 50:50 approaches). This does not reflect the dependence of the recovered products to the intended products as they could not be produced without extraction and processing. On the other hand, the 100:0 approach fully allocates the impact of this processing to the recovered products while these processes are a consequence of the production of consumer goods. Therefore, based on the concept of the producer's responsibility often used to promote the implementation of the industrial ecology principles, part of the burden from recovery processes should be allocated to the consumer goods. The "50:50 adapted" approach allocates equally the impact from the raw materials

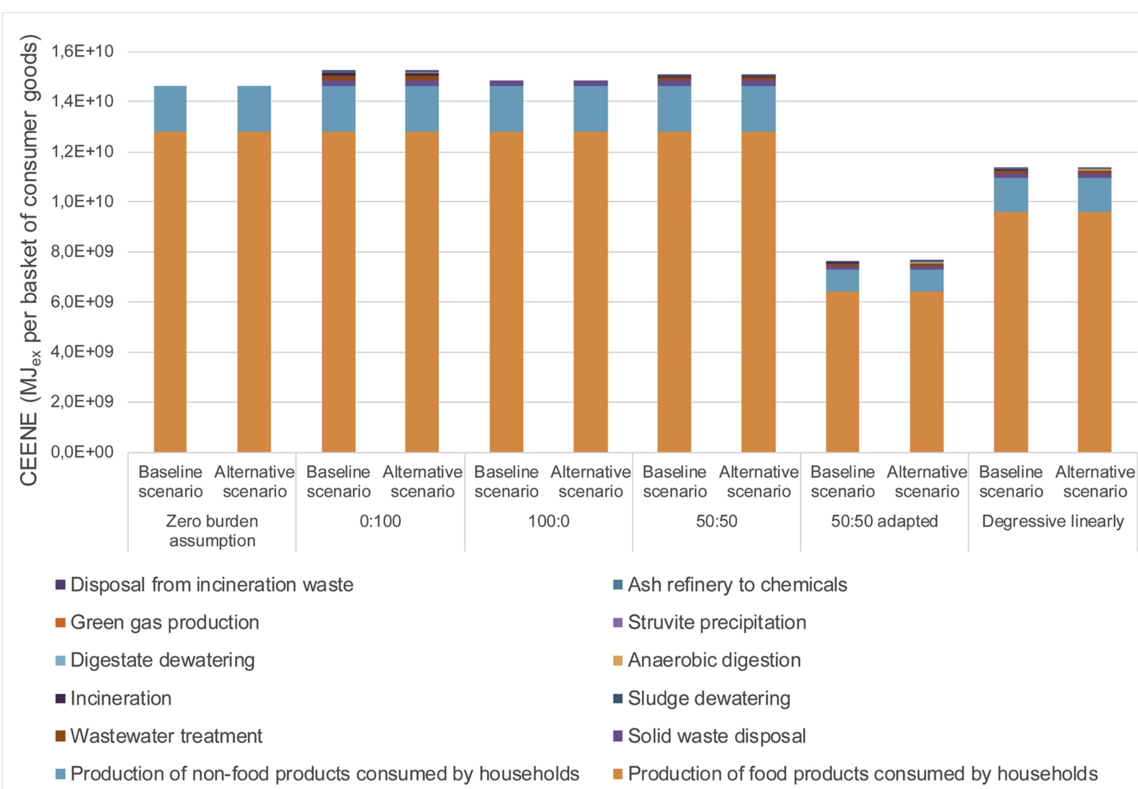

Fig. 9. Resource footprint of the consumer goods with the zero-burden assumption and the five allocation approaches.

extraction and processing to the consumer goods and the recovered products, while the original goal of these processes is to produce consumer goods. This approach considers the dependence of products but not the "hierarchy of intent". The "linearly degressive" approach appears to consider both the dependence of the products to one another and the "hierarchy of intent" and to translate best the concepts of industrial ecology in the modelling.

In this study, the "linearly degressive" approach considers an allocation of the environmental burdens based on a 75:25 ratio based on Allacker et al. (2017). Other approaches could be investigated to define the values for allocating the impact along the chain. One possibility is to consider the ratio of the gate fee at the entrance of the recovery processes over the cost to run these processes. It could represent the share of the impact from these processes that can be allocated to the waste treatment function, and allocated to the consumer goods. The remaining fraction can be fully allocated to the recovered products. A similar approach can be applied to allocate the impact of consumer goods production.

The results presented in this study are obtained using the resource-based method CEENE. Sensitivity and uncertainty analyses could be conducted to identify the most important parameters and the significance of the results. Moreover, other conclusions might be drawn when using other resource-based methods that consider issues related to resource availability or scarcity such as the ADP (van Oers et al., 2002) and the Ecological scarcity (Frischknecht and Büsser Knöpfel, 2013) methods. Using such methods could potentially change the difference of resource footprint between the recovered and benchmark products. Similarly, other results might be obtained when conducting an emission-based impact assessment in which the emissions of the different processes along the chain (e.g., release of heavy metals in the Dommel river after the WWTP) would be allocated to the different products.

Another point of attention when applying the proposed approach is the consistency of the modelling approaches followed in the foreground and background systems. Several allocation approaches were tested in the foreground system but the allocation approach used to model the background system is "fixed" ("allocation at the point of substitution" from the ecoinvent database). The approach "allocation at the point of substitution" should in principle consider all waste streams as co-products of the process they are produced from. However, some discrepancies and unclarity can be found with this approach. While the approach is applied to municipal solid waste, it is not clear in what extend it is also applied to other waste streams such as sewage sludge. Similarly, the end-of-life formulas applied in the foreground system are not applied in the background system modelled with the ecoinvent database. Applying them in the background system would make the study more consistent and probably change the results of the analysis. However, the implementation of such an approach in LCI databases would require a deep rethinking of how products and processes are linked to each other.

In the two studied scenarios, solid waste from food consumption is assumed to be incinerated without valorisation. This assumption was made to simplify the scenarios (in the Netherlands, only 2.5% of municipal waste is disposed of without further valorisation; OECD (2018)), as the focus was on the wastewater treatment chain and not solid waste management. If solid waste valorisation is considered, the end-of-life formulas should also be applied to the solid waste treatment processes. It highlights the complexity of the practical implementation of the approach, especially for the calculation of the footprint of the consumer goods.

Another point is that the approach presented in this study can only be applied when comparing sewage sludge valorisation and benchmark products, or to account for the credits of avoided production. A study that would not compare the recovered and benchmark products and would not account for the credits from avoided production would require knowing the fate of these products, i.e., if they are further recycled after use or disposed of. Accounting for these steps might slightly change the difference of resource footprint between the recovered and virgin material-based products. It is therefore important to keep in mind that the analysis is conducted up to the gate of the recycled products, as indicated in the system boundary section, which provides insights in the context of a comparison. This means that the presented resource footprint of the products only represents the partial resource footprint of these products, as it does not include downstream processes such as further recycling or disposal. However, as highlighted in Allacker et al. (2017), the feasibility to access downstream information is very low as producers most of the time lose track of their products after use.

5. Conclusion

The paradigm shift from a linear to a circular economy is changing the practice of product design, production and consumption. Similarly, the practice of sustainability assessment should adapt to this new paradigm. The goal of this study was to propose a novel approach to assess the environmental sustainability of products obtained from the valorisation of household wastewater sludge. This approach was applied to the wastewater and associated sludge treatment chain of Eindhoven. First, the process chain had to be partitioned based on partitioning factors. Exergy-based factors were chosen. Secondly, five approaches presented in Allacker et al. (2017) were tested. The results show that discarding the zero-burden assumption and applying the different allocation approaches only has a large impact on the resource footprint of the consumer goods when following the "50:50 adapted" and "linearly degressive" approaches. However, it has large consequences on the footprint of the recovered products. Except with the 0:100 and the 50:50 approaches, discarding the zero-burden assumption results in a resource footprint 27 to 80% higher than with the zero-burden assumption. While environmental impact assessment methods should apply the paradigm shift from a linear to a circular economy by considering wastewater as a resource, the interest of discarding the zero-burden assumption in this case becomes debatable for stakeholders producing these recovered products. A discussion on the "fairness" of each of these approaches resulted in selecting the "linearly degressive" approach as it shares the impacts over the process chain the most consistently according to the principles of industrial ecology. However, it is a data-intensive approach as data on consumer goods consumption need to be gathered. The selection of an approach could depend on the incentives that policy makers want to give to each of the actors along the chain. A similar idea is followed in the BPX30-323-0, the French repository for good practices on communication of the environmental impact of products. It proposes to choose different allocation factors to pull the market of recycled products depending if the market for secondary materials is in equilibrium or not. The 0:100 and 50:50 approaches are the most favourable for the producers of recovered products compared to the zero-burden assumption followed today in LCA studies. The "50:50 adapted" and "linearly degressive" approaches are the least favourable but might be interesting approaches for policy makers as they provide an overview of the contribution of consumption to the footprint of recovered products. The results of this analysis encourage policy makers to take action towards less resource-intensive consumption patterns. An interesting future analysis could be to evaluate the impact of those consumption patterns on the resource footprint of the recovered products.

The study also shows that policy makers could more extensively use LCA results to encourage resource recovery steps from sludge (e.g., anaerobic digestion, struvite precipitation) and define a hierarchy for the management of sludge ashes (e.g., fertilizer production prior to roadfilling material, prior to landfilling). More studies should be reviewed and conducted to support policy making in this way. Moreover, aiming for recovered products with a lower footprint than virgin material-based equivalents with the "linearly degressive" approach would strongly position the wastewater sector as a key player of a sustainable circular economy.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.resconrec.2019.03.029.

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