Environmental assessment of urine, black and grey water separation for resource recovery in a new district compared to centralized wastewater resources recovery plant

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Highlights

- Source separation scenarios were compared to centralized resource recovery facility
- Urine and black water separation reduce by 45 % and 34 % greenhouse gas emissions
- Nitrogen recovery can lead to neutral GHG emissions plant
- Greywater reuse increases climate change impact due to high energy consumption for MBR

Abstract

Current trends show that wastewater treatment plants (WWTPs) will intend to shift towards water resources recovery facility (WRRF), however nutrients recovery in WRRF is limited by the need of highly concentrated stream for the process. Source separation can help to increase this potential, but assessment is necessary to evaluate the whole system at district scale.

In this study three scenarios based respectively on urine diversion, blackwater and greywater separation were compared to the conventional end-of-pipe strategy with adjunction of treatment at the centralized WRRF for producing renewable fertilizers and energy. Life cycle assessment was used to compare environmental impacts. The study was performed to represent the implementation of a new district in an urban context. Treatments have been chosen among the best technologies available for resource recovery.

Results show that for maximizing nutrients recovery and limiting the greenhouse gas emissions, urine and blackwater separation are better scenarios than conventional mixing option and centralized WRRF. Indeed it allows to mitigate by at least 60 % the nitrous oxide (N₂O) emissions and to avoid nitrogen fertilizer production which emits large amount of greenhouse gases (8.6 kg CO₂-Eq/kgN). Urine source separation is particularly beneficial by recovering nitrogen at a low environmental footprint: impact on climate change decreases by 45 % compared to the Reference. The separation of blackwater treated at decentralized scale shows a decrease of 34 % of impact on climate change compared to *Reference*, thus a little worse than *Urine* due to a higher external energy demand at decentralized scale for nitrogen recovery. Phosphorus can be recovered in all the source separation system without additional climate change impact.

If the priority is given at water reclamation the treatment of blackwater and greywater separately is a valuable option. However energy balance and greenhouse gases emissions analysis do not support this scenario if highly energy consuming technology is used for greywater treatment (more than 0.56 kWh/m³ treated). It would become acceptable in case of decarbonized energy or if tap water production is a high greenhouse gases emitter.

Keywords

Source separation, wastewater treatment, resource recovery, life cycle assessment, urine, blackwater

Graphical abstract

Abbreviations

BOD₅: Biological oxygen demand at 5 days

BW: Scenario with blackwater separation and treatment in the district, greywater is sent to the WRRF

BW-GW: Scenario with blackwater and greywater separation and with a decentralized treatment

COD: Chemical oxygen demand

GHG: Greenhouse gases

LCA: Life Cycle Assessment

TN: Total nitrogen

TP: Total phosphorus

U: Urine

WRRF: Water resources recovery facility

WWTP: Wastewater treatment plant

1. Introduction

Current trends show that wastewater treatment plants (WWTPs) will intend to shift towards Water Resources Recovery Facility (WRRF), with the aim, not only to preserve water bodies but also to produce resources (Guest et al., 2009; Diaz-Elsayed, et al., 2019) such as water, energy and carbon, phosphorus and nitrogen compounds. However the current processes can only be applied on highly concentrated stream, conventionally nutrients are recovered from liquid portion of the anaerobically digested sludge or directly from solid sludge or even sludge ash. The amount of nutrient recovery is thus limited by the capture efficiency in primary and secondary sludge from waterline treatment.

In addition to improvements developed for WWTPs, several studies investigated the possibility of a shift in sanitation systems, where wastewater can be segregated at the source (within households). Indeed, a large proportion of pollutants (nutrients and organic matter) in wastewater comes from human excreta (Larsen and Gujer, 1996). Although urine alone represents only 1% of the total effluent volume, between 70 % and 80 % of total nitrogen and around 50 % of the phosphorus in wastewater come from urine. Therefore, source separation would be efficient to concentrate the nutrients and hence avoid downstream dilution and mixing with other molecules.

Two strategies have been tested in northern Europe: 1) projects with a urine source separation approach (Larsen et al., 2013) and 2) projects with separation of blackwater at the source, i.e. effluent from domestic toilets (Skambraks et al., 2017). As low volume is involved, urine can be collected by truck and urine source separation aims to produce a fertilizer rich in nutrients. Blackwater separation is able to recover the organic matter fraction from blackwater, which represents around 50 % of organic matter coming from the wastewater. It allows energy production (via biogas produced from anaerobic digestion) but also nutrients extraction in the following treatment steps. Moreover the greywater can be reused for non-potable uses. This option of blackwater/greywater separation could be therefore a more promising approach because the entire wastewater can be turned into a resource. In order to optimize blackwater treatment through anaerobic digestion and avoid dilution with too large flush water, vacuum collection would need to be implemented with vacuum toilet. High energy consumption of this technology forces the treatment to be decentralized. Therefore source separation transforms the whole wastewater management system with changes in the collection, treatment and valorization.

Several studies have already attempted to evaluate the efficiency and the environmental impact of different resource recovery solutions by modelling and comparing them to conventional sanitation systems thanks to life cycle assessment. It can be highlighted that the boundaries of these studies can

integrate different subsystems, such as potable water production, biowaste management, fertilizer production and obviously waste water management, making the comparison difficult.

However the majority of the studies conclude that there is an environmental benefits of urine source separation. Several keys point can be highlighted: 1) the nitrogen of urine must be recovered and valorized as a fertilizer (Bisinella de Faria et al., 2015); 2) the spreading of urine or by-product is a sensitive parameter to avoid emissions into air (Tidåker et al., 2007). Acidification is most of the time increased by urine source separation due to higher emissions of NH_3 in the air (losses during storage and spreading) and NO_x (transport of urine and urine collection equipment). Moreover urine source separation scenarios considers most of the time a low water flush (0.05 L/flush) (Igos et al., 2017; Ishii and Boyer, 2015; Landry and Boyer, 2016). This value is very unlikely for a large deployment, the toilet paper is not trained enough, as it has been reported in real pilot project (Winker et al., 2013).

Regarding blackwater separation scenario, conclusion is less obvious and depends on 1) energy consumption for vacuum sewer and 2) the choice of greywater treatment. Vacuum sewer energy consumption varies from 8 $\text{kWh}_{\text{el}}/\text{m}^3$ (Thibodeau et al., 2014b) to 15 $\text{kWh}_{\text{el}}/\text{m}^3$ (Meinzinger et al., 2010) which is similar to real case pilot projects. With the lowest value, energy balance can be positive but it is not the case with the highest. Finally greywater treatment is also a key parameter which can switch the conclusion. Indeed rustic treatment of greywater (septic tank and reed bed filter) allows to have a positive energy balance (Thibodeau et al., 2014b) but not with high-tech processes as membrane bioreactor (Remy, 2010). In this latter study, large amount of greywater need to be recycled to have an environmental benefit.

Moreover, several parameters are less integrated, as for example the direct emissions from wastewater treatment in the atmosphere and the emissions from spreading of by-product (Ishii and Boyer, 2015; Landry and Boyer, 2016). These emissions have been proven to be a major contributor to environmental impact (Bisinella de Faria et al., 2015) and cannot therefore be omitted.

The majority of these studies compare the alternative scenarios with a conventional wastewater treatment plant without any resource recovery. The functions of those systems are thus not equivalent which be in favor of source separation system where best technology available are applied. When plant retrofitting is considered (Bisinella de Faria et al., 2015; Igos et al., 2017; Wilsenach and van Loosdrecht, 2006), only the redesign of the treatment process for nitrogen removal and not nutrient recovery is included.

Finally all of these studies implement source separation to a large extent in the city. Only one investigates the implementation at the district scale (Kavvada et al., 2017) as it is the case in real pilot projects. Only few studies analyze source separation at district scale for on-site sanitation (Mbaya et al.,

2017; Xue et al., 2016). Greywater treatment and reclamation has yet shown the importance to take into account the scale and the different type of urbanism (Bonoli et al., 2019; Jeong et al., 2018) and the same effect can be supposed in source separation scenarios. Therefore a tool capable of district scale analysis is necessary.

To fulfil these gaps, this study aims to develop a methodology and a tool allowing the modelling and comparison of different wastewater management options with resource recovery, in a new district. The developed methodology and tool should enable evaluating to what extent source separation (blackwater, urine, and greywater management) could help increasing the resource recovery and decreasing environmental impacts of wastewater management for a new district compared to the system without separation. It could identify the potential interest for shifting from centralized WRRF to a scenario of source separation with better environmental performance. The tool is based on mass and energy balance modelling of the involved processes, from collection to effluent discharge in the environment. For illustrating the methodology and the tool capabilities, three scenarios representative of several pilot projects already undertaken around Europe have been modelled and compared. The simulation results are then used for Life Cycle Assessment (LCA) of the proposed scenarios and of the centralized WRRF in order to compare their environmental performances.

2. Materials and methods

2.1 General methodology

The general methodology is composed of three steps:

1) The district typology and topography is generated according to the topologies identified by (Bonhomme et al., 2012). A geographic information system is used to associate the buildings and their characteristics (surface of floor, number of inhabitants and employees, height). Influent characteristics and transport requirements are then calculated and the decentralized technologies are defined.

2) Modelling of the different wastewater recovery facilities treating effluents from the new district and from the existing city. For this purpose, the SUMO[®] (Dynamita v19 2019) software is used with an integrated model based on an activated sludge and anaerobic model coupled to gas exchange.

3) LCA of the defined scenarios, based on data obtained from the first two steps. To this end, an automated link between the SUMO results (output) and LCA Umberto software (input) was used, previously implemented by (Bisinella de Faria et al., 2015).

The different steps of the modelling strategy provide an analysis with different boundaries. Results on mass and energy balance are analyzed at the level of the processes involved, from the effluent collection to the effluent treatment, while the LCA results are analyzed with extended boundaries to take into account the benefits provided by fertilizer production, water reused or energy production.

This study focuses on the new district, however, the new district is part of a city, where the number of inhabitants is larger than the new district. The impact of the new district needs to be highlighted. For this purpose the centralised WRRF treating the city effluent before the implementation of the new district has been modelled (*City* scenario). Then, the results obtained from the *City* are subtracted to the results of simulation of the whole system including the new district (Equation 1). In this sense, only the extra environmental costs of the district are taking into account.

$$Results_{District} = Results_{District+City} - Results_{City} \quad \text{Equation 1}$$

The methodology description follows the three steps cited above.

2.2 Generation of district data

2.2.1 District generation

A new district is considered, built in a city which is already equipped with a WRRF treating 56 003 PE with a flow rate of 12 900 m³/d. The urban planning of the district is described as discontinuous blocks according to the work of (Bonhomme et al., 2012). With a surface of 6.25 ha this type of planning is composed in majority of collective buildings for residence purposes. Besides the 921 inhabitants, 469 external people are employed on site which together represents 697 PE to treat. The planning of the district is presented in Figure S 1 in SI. A flat topography has been considered in this study.

2.2.2 Studied scenarios

Three different source separation and one reference centralized scenarios are compared (Figure 1) with an increasing decentralized scale of treatment meaning that the treatments of wastewater are more and more located in the district and less in the centralized WRRF:

- i) the centralized WRRF scenario (*Reference*) where nitrogen and phosphorus recovery take place,
- ii) the urine source separation scenario (*Urine*) with treatment at the centralized WRRF,
- iii) the scenario of decentralized treatment of blackwater and centralized treatment of greywater in WRRF (*BW*),
- iv) the scenario of black and greywater source separation and their treatment in a decentralized system (*BW-GW*).

The three source separation scenarios are representative of several pilot projects already undertaken around Europe. Phosphorus recovery is performed through struvite precipitation in each scenario. Nitrogen recovery within separated waste effluents is the only treatment which has been added since it is not implemented in these projects. The choice of technology for nitrogen recovery has been based on two criteria: 1) the lowest energy consumption and 2) a demonstration of the process efficiency with at least pilot plant scale (see Table S 1 in supplementary information). Transmembrane chemisorption (TMCS) is currently one of the best emerging technologies for nitrogen recovery with its low energy consumption (2.55 kWh_{elec}/kgNH₄-N_{recovered} (Böhler et al., 2018) compared to 76 kWh_{elec}/kgN for nitrification/distillation (Udert and Jenni, 2013; Udert and Wächter, 2012) and 7.2 kWh_{elec}/kgN for ammonia stripping (Maurer et al., 2003)). It consists on hydrophobic membranes which allow ammonia gas to transfer to an acidic solution and form a nitrogen salt. This technology has been tested at pilot scale for urine treatment (Damtie et al., 2020; Zhang et al., 2013) but it is already installed at full scale in the WWTP of Altenrhein and Yverdon-les-Bains (Böhler et al., 2018) treating approximately 37 m³/day of rejected water from anaerobic digestion. Therefore relevant and realistic data are available for this study. More information on treatment is given in the following section 2.3.2.

Figure 1: Scheme of the four scenarios and of the WRRF. WRRF: Wastewater recovery facility, AD: Anaerobic digester, PPTSTRU: Struvite precipitation, TMCS: transmembrane chemisorption, UASB: upflow active sludge blanket, MBR: membrane bioreactor, ANAER: anaerobic tank, ANOX: anoxic tank, AER: aerated tank, PANOX: post-anoxic tank, PAER: post aerobic tank, THK: thickening, PTHK: post-thickening, DW: dewatering.

In the *Reference* scenario, wastewater is collected with a gravity sewer line. The volume of flushed water is 3 L after urination and 6 L after defecation (Arocha and McCann, 2013).

In the *Urine* scenario, urine is collected in tanks and then transported once a week by truck (10m³ trucks, two rounds per week) to the WRRF. The remaining wastewater is transported as usual through the gravity sewer line. Considering 3 L or 6 L toilet flushes, it is considered that only 0.2 L of every flush

goes into the urine compartment and storage, while the remaining volume is flushed in the feces compartment. Only 80 % of urine is separated which corresponds to the best performances observed in pilot projects (data from (Jönsson et al., 1998) and (Peter-Fröhlich et al., 2007) respectively for Understenshöjden et Palsternackan projects in Sweden and SCST project in Germany). The remaining 20 % is transported with feces. This urine recovery is influenced by the design of the toilet which allows more or less urine retention but also the willingness of the user to facilitate the separation. Indeed, urine separation toilet still requires gents to sit down when urinating.

In both *BW* and *BW-GW* scenarios, blackwater is collected through a vacuum toilet (1.2 L per flush, Evac Toilet (EVAC, n.d.)) followed by a vacuum sewer until reaching a vacuum station in the middle of the district where the treatment takes place. The greywater obtained in the *BW* scenario is sent to the WRRF through a gravity sewer line. In the *BW-GW* scenario, blackwater and greywater are treated in decentralized systems in the middle of the district.

The influent characteristics used for each scenario are presented in the Table 1 and are extracted from the literature review described in SI (Section 1.3).

Table 1: Influent characteristics for each scenario (see Figure 1 for the definition of the flow)

Scenario	All	Reference	Urine		BW and BW-GW			
Name	City	District	Yellow water	Grey brown water	Black water	Grey water	Remaining BW	Unit
Flow rate	12900	111.4	1.7	109.7	6.1	95.5	5.5	m ³ /d
COD	601	864	4154	814	7897	507	565	g COD/m ³
BOD₅	261	375	3143	333	2700	283	331	g O ₂ /m ³
TN	68	88	3546	36	1429	12	450	g N/m ³
NH_x	50	67	3331	18	1191	2	213	g N/m ³
TP	11.	14	304	10	184	5	40	g P/m ³
SPO₄	6	12	290	8	168	3	19	g P/m ³

As explained in the general methodology in section 2.2, each scenario has been modelled at the city scale with both decentralized treatment and the modeling of the centralized WRRF. The results analysis allows to take into account only the impact of the new district by the subtraction with the result of the modeling of *City* scenario (Equation 1).

2.3 Modelling of wastewater management

2.3.1. Wastewater collection

Gravity sewer is designed to achieve the transport of the peak flow by choosing the appropriate couple diameter / slope of the pipe according to (BERLAND, 2014; Bourrier, 2008; Monfront, 2009; Okun et al., 2010). This couple needs to fulfill three self-cleansing conditions to avoid sedimentation in the pipes. Moreover, when depth is too high (assumed to be 4 meters in our model), a pumping station is placed and designed.

Vacuum sewer works with air transport instead of water transport as in gravity sewer. A lift in series allows to maintain the transport of the wastewater. The design rules of (Bowne et al., 1991; Islam, 2017; Roediger Vacuum, 2012; SQAEE, 1994) have been followed with a minimum slope of 0.002 m/m and a maximum depth is 1.5m below surface. The pressure drop is calculated based on the recommendations of (Bowne et al., 1991; Islam, 2017; SQAEE, 1994) for static pressure drop and of (Jinming and Jingxuan, 2006) for frictional pressure drop.

Urine transport by truck is optimized to minimize the total distance thank to an ant algorithm (Bullnheimer et al., 1999; Dorigo and Gambardella, 1997), to solve the vehicle routing problem. It has been assumed that urine is stored for one week before collection by a 10 m³ truck and centralized at the WWRF, 10 km away.

More information can be found in supplementary information 1.1.

2.3.2. Decentralized treatments

In this section the decentralized treatments implemented in *BW* and *BW-GW* scenario are presented, more information about their design can be found in SI in section 1.2. For both, blackwater are digested in an upflow anaerobic sludge blanket (UASB) reactor which allows reaction and decantation to take place in the same unit. Bacteria are aggregated in granules and flocs and are maintained within the reactor by balancing the settling velocity and the upflow velocity of the liquid.

This reactor has been successfully applied by (de Graaff et al., 2010) when treating blackwater at ambient temperature and with a SRT of 75 days. A production of methane between 54 % and 60 % of the blackwater COD have been respectively reported by (de Graaff et al., 2010) and (Tervahauta et al., 2014) studies. The higher value has been chosen in this study.

After digestion, the effluent is processed for P and N recovery through struvite precipitation and transmembrane chemo-sorption (TMCS). (de Graaff and van Hell, 2014) chose a molar ratio of 1.5 Mg/P to precipitate the P with struvite, leading to the removal of 86 % phosphate and 4 % for other phosphorus compounds. In the TMCS process only the gas form of ammonia can transfer through the

pores. Thus both pH and temperature values have to be controlled. However, optimal values of pH and temperature are still not defined and such topics need to be more intensively investigated. In this study, applying a pH of 9.6 and a temperature of 45 °C allows to obtain 88.5 % of ammonia. To achieve such alkaline pH values, caustic soda is injected before the membrane separation. 90 % of ammonia is assumed to be removed in the TMCS module. A heating system is required to increase the temperature from the operational temperature of UASB (25°C) to 45°C.

Greywater produced in the *BW-GW* scenario, is also treated in a decentralized system, by using a membrane bioreactor (MBR). Aeration is used to prevent membrane clogging which involves a higher energy consumption compared to conventional activated sludge treatment systems. However, membrane separation, although being energy intensive, offers a better quality and stability of output effluents as stated in literature (Winward et al., 2008). Water recovery hence seems very promising with this technology whether applied on small or large scales. In this present study, the MBR is designed to perform biological removal of phosphorus and denitrification. The effluent of blackwater treatment is sent to the MBR with a mass flow of nitrogen. The sludge obtained through the MBR is also sent to the digester for mineralization and the out coming phosphorus can then be captured with struvite precipitation.

2.3.3. Modelling the centralized WRRF

As previously described, the advanced WRRF (Figure 1) is designed to recover as much as possible of nutrients and organic matter. WRRF is based on biological removal of phosphorus coupled to sludge digestion and struvite precipitation of digested sludge. Magnesium is added with a constant ratio between magnesium added and to total phosphorus in entrance.

Nitrogen will also be recovered after precipitation by using the TMCS with the same operating conditions than at decentralized scale: pH of 9.6 and a temperature of 45 °C. Moreover, to prevent fouling of the membrane, solid separation is needed. Pre-treatment is modelled by considering a retention of 100 % of the solids.

In case of urine separation, the collected urine is added to the output effluent of the digester to undergo struvite precipitation and TMCS filtration. In the *BW* scenario, both greywater produced and the effluent obtained after blackwater treatment, are both mixed with the city sewage in the main sewer line.

The same discharge regulations have been applied to each scenario fixed at 10 mgN/L and 1 mgP/L. Even if regulation standards could allow higher concentrations in *BW-GW* scenario, due to the small load to treat, the purpose of this study is not to increase the release of nutrients. The risk of eutrophication is therefore maintained at the same level for all scenarios.

To achieve these standards, controls in the process are performed by using the SUMO software. The two recirculation flowrates, r_{ANOX} and r_{ANAER} , are proportional to the influent flowrate entering the primary decanter. Methanol flow is controlled to achieve the 10 mg TN/L (total nitrogen) by improving the post-denitrification efficiency. Iron chloride is added to respect 1 mg TP/L (total phosphorus) for the chemical removal of phosphorus. Finally, magnesium is added for struvite precipitation according to a fixed ratio of 1.49 mol Mg / mol TP_{in} .

2.4 Life cycle assessment

LCA is the third step of the methodology and was applied following its four steps (ISO 14040-14044): 1) goal and scope definition, 2) life cycle inventory (LCI) building, 3) life cycle impact assessment (LCIA), and 4) results interpretation.

2.4.1 Goal and scope

The goal of LCA study is to assess and compare several sanitation systems for a new district which aims firstly to protect the receiving environment from eutrophication and secondly to maximize the resource recovery.

The function of the system can be defined as “*collect and treat wastewater produced in the district while respecting the discharge limit of 10 mg-N/L and 1 mg-P/L, and this for one year*”. The functional unit is thus 1PE considering the EU normalized definition: 1 PE = 60 gDBO₅/d.

The system boundaries (Figure 2) includes the collection and treatment of wastewater (foreground processes) but also production and transport of chemicals, electricity and heat and infrastructure (background processes). As the WRRF systems produce fertilizers and treated water in addition to above defined function, avoided processes are taken into account for fertilizer and tap water production.

2.4.2 Life Cycle Inventory

The inventory has been built by using the mass and energy data obtained from the two previous steps: simulation results of collection and transport at district scale and with centralized facilities (step 1), simulation results of treatment processes (step 2).

The life cycle inventory framework follows the work of (Bisinella de Faria et al., 2015) with the adjunction of effluent collection and transport in the foreground part of the life cycle system. Figure 2 presents the extended system boundaries of the LCA.

Regarding sewer infrastructure, the inventory takes into account the diesel used for excavation, pipe production and transport (in PE for vacuum network and PVC in gravity network) and the material to fill trenches (sand, gravel), the infrastructure for inspection is also integrated (manholes and inspection chambers). The bitumen used for building the road is not taken into account as it is allocated to road creation (Petit-Boix et al., 2014). Regarding urine collection, storage tanks are assumed to be placed in

the district, one per building. The tanks are assumed to be in polyethylene (PE) with 40 years of lifespan, and a mass of HDPE per tank volume of 28.412 kg/m³ is assumed (Ishii and Boyer, 2015).

Evaluation of the energy consumption within the WRRF is one of the most important task and follows the work of (Bisinella de Faria et al., 2015) which takes into account: pumping of wastewater and chemicals in the WRRF, mixing the non-aerated tank, energy for aeration, scraping sides of the decanter and energy for dewatering and thickening the sludge. The energy consumption is also related to the heat demand for digestion but also heat production through cogeneration. In this present study the energy consumption of the TMCS was added in the inventory. Consumption of pumps are taken into account, as well as the heat requirement in order to reach 45°C prior to ammonia stripping, which is supplied by cogeneration or by a boiler if necessary. The energy consumption of TMCS is assumed to be 1.887 kWh/kgNH₄-N_{recovered} from data of the pilot plant of Yverdon-les-Bains in Switzerland (Böhler et al., 2018) (CO₂ stripping and coagulation / flocculation pre-treatments and ventilation are excluded).

N₂O emissions from the biological treatment of wastewater are not incorporated in SUMO model. The emissions factor from (IPCC, 2019) has been used: 1.6 % of the nitrogen entering the water line or MBR. The remaining emissions into air, water or soil are the same as the ones used in (Bisinella de Faria et al., 2015).

By-products (sludge, ammonium sulfate and struvite) are spread on agricultural land and can replace part of nitrogen and phosphorus fertilizer considering the bio-availability of the components and the post-spreading emissions from (Bisinella de Faria et al., 2015).

Environmental impacts were evaluated with endpoint ReCiPe (H, A) and midpoint ReCiPe (H) (Goedkoop et al., 2009). Attributional LCA was performed with UMBERTO software and ecoinvent database v2.2.

A sensitivity analysis has been conducted on several parameters that have been identified as uncertain in the literature and relevant to the results. Four parameters have been studied: (1) Emission factor of nitrous oxide from wastewater treatment, (2) efficiency of urine collection in the urine diverting toilet, (3) energy consumption of MBR for greywater treatment, (4) vacuum sewer energy consumption.

Nitrous oxide emissions from WWTP are difficult to evaluate as they vary during the year and with the operating conditions (Vasilaki et al., 2019). Long-term monitoring is therefore necessary but few studies report these data. Thus, nitrous oxide emissions shows a wide range even within similar groups of wastewater treatment processes. Whereas the IPCC value has been used as a baseline (1.6 % of the nitrogen mass flow entering the treatment plant (IPCC, 2019)), the following range of emissions factors have been studied: the emission factor of an activated sludge with extended aeration (0.19 %) from (Filali et al., 2016); the average value (0.87 %) reported by (Vasilaki et al., 2019), the yearly average of emission factor for two biofilters (nitrifying and denitrifying) treatment plant determined from (Bollon, 2016; Bollon et al., 2016).

Efficiency of urine collection by the toilet is an uncertain parameter due to limited data related to the first generation of urine diverting toilet. Data of (Jönsson et al., 1998, Peter-Fröhlich et al., 2007, Vinnerås, 2002, Vinnerås et al., 2006) range indeed from 50 % to 70 % in average. In our study 80 % was assumed considering recent improvement with the NoMix toilet from Roediger and the more recent toilet Save! from Laufen.

In this study, vacuum sewer consumption is calculated based on the pressure loss according to the sewer design (pathway, elevation, design of the lift). However the topography of the district was considered flat and vacuum sewer consumption can be impacted by this simplification. In this sensitivity analysis, two values from pilot projects have been used: 15 kWh/m³ from Flintenbeite project in Germany (Albold, n.d.; OtterWasser GmbH, 2009) and 5.6 kWh/m³ in the Waterschoon project in the Netherlands (de Graaff and van Hell, 2014).

Finally, the evaluation of MBR energy consumption for decentralised treatment is really challenging as it depends on the type of membrane, the size of the plant and it is in constant evolution. A range from 0.3 to 8 Nm³/h/m² have been applied in MBR treating greywater at pilot scale (Jefferson et al., 2001; Lesjean and Gnirss, 2006; Merz et al., 2007; Peter-Fröhlich et al., 2007). In our study the value of 1.2 Nm³/h/m² of membrane has been used which is equivalent to 1.5 kWh/m³ of water. However the minimum value observed in the literature is around 0.4 kWh/m³ (Jeong et al., 2018, Atanasova et al., 2017). Moreover recent studies show also a decrease of full scale MBR treating mixed wastewater, up to 0.4 kWh/m³ for MBRs commissioned after 2014 (Brepols, 2020). In the sensitivity analysis, the following values have been considered: 1 kWh/m³ and 0.4 kWh/m³.

3. Results

3.1 Material recovery

The recovery efficiency is presented in Figure 3 below and as Sankey diagrams in SI (Figure S 20, Figure S 21 and Figure S 22).

3.1.1 Nutrients

3.1.1.1 *Reference* scenario

In the *Reference* scenario, results show that 14 % of the nitrogen and 82 % of the phosphorus entering the WRRF is captured in the sludge, and only 6 % of nitrogen is converted into a fertilizer (ammonium sulfate and struvite) and 13 % of phosphorus (struvite). Sludge still contains a significant proportion of nutrients which are valorized through sludge spreading but with lower bioavailability than fertilizer.

Not surprisingly nitrogen recovery efficiency into fertilizer is limited because most of nitrogen is lost in atmosphere during denitrification. The nitrogen capture would be slightly higher if activated sludge process was operated at higher rate, or if external carbon source or external sludge was provided. For instance (van der Hoek et al., 2018) found 27 % of the N captured in the digester liquor (against 23 % in our study) but with 44% of external sludge treated in the anaerobic digester. (Shi, 2011) reported 12% in the sludge and 10.9% in the centrate after digested sludge dewatering. (Baker et al., 2001) found a value even lower, i.e. only 10 % in the sludge. (See also Table S 21 for a comparison)

Regarding phosphorus a major part remains in solid sludge in this *reference* scenario due to the combination with metallic cations as chemical phosphorus removal was performed in complement to biological P removal in the WRRF. The limited amount of nutrients captured as struvite in the digester liquors has been already reported by previous studies and our *reference* result is within the range of reported values. The recovery potential as struvite from digester liquor with respect to the WWTP influent phosphorus load is between 10–25% (Egle et al., 2015). Higher values were obtained for systems with pure biological P removal. For instance (Shi, 2011) reported 20.4% of P measured in the dewatering centrate after anaerobic digestion.

3.1.1.2 Source separation scenarios

The present study shows that all the source separation scenarios at the level of new district increase the recovery rate of nitrogen up to 48 % in the *Urine* scenario and 54 % for *BW* scenario. The same trend is observed for phosphorus recovery with 58 % for the *Urine* and *BW-GW* scenarios and 73 % for *BW* scenario, thanks to struvite precipitation. It is assumed that urine contains 76 % of the nitrogen of domestic sewage, and only 80 % of the total amount is here collected and sent to recovery. Results show that 60 % of the nitrogen in *Urine* scenario reach TMCS treatment whilst 88.5 % can be recovered in

the blackwater. Therefore, blackwater separation might be slightly more efficient in terms of nitrogen recovery.

Figure 3: Recovery rates for each scenario and each compound at district scale

Higher phosphorus recovery rate is feasible in source separation scenarios compared to the *Reference* scenario by enhancing biological removal of phosphorus implemented in the WWTP. Indeed, these scenarios showed a slight increase in the ratio of biodegradable COD versus total phosphorus entering the water line and more organic carbon is available for biological P removal as less is used for denitrification (more nitrogen being recovered as ammonia). As showed in Figure S 23 in SI, less iron chloride per entering phosphorus (molar ratio) is thus necessary. Moreover, in both scenarios involving blackwater separation the use of vacuum toilet, reduces the flush water consumption, and in the meantime the effluent concentration is kept to 1 mgP/L. Thus the mass flow in the effluent decreases (7.0 % of the entering phosphorus in the *Reference* scenario and only 6.4 % for *BW* and *BW-GW* scenarios) and the phosphorus recovery in sludge was higher than in the other scenarios.

3.1.2 Organic matter

COD recovery is performed through the production of biogas. In the *Reference* scenario, 35 % of the influent COD is converted into biogas. The digestion of blackwater increased the COD recovery up to 46 % in *BW* scenario and up to 39 % for *BW-GW* scenario. Beside, urine separation was less efficient than the *Reference* scenario (only 32 % of recovery), which can be explained by the fact that 7 % of COD produced by this district is found in the urine and is partly degraded before entering the digester. Moreover by modifying the COD/TN/TP ratio of the wastewater, source separation reduces the need of methanol in post denitrification, necessary to reach the discharge standard for nitrogen. In this perspective, if methanol is considered as the COD inlet, the COD recovery in biogas will be different as

presented in Table S 22 in SI. *Reference* scenario presents the worst efficiency with only 32 % of COD converted into biogas whereas for *Urine* and *BW* scenarios it reaches up to 34% and 49% respectively.

3.2 Energy balance

If we considered the WRRF of the reference scenario treating effluents from the new district and the city, the system consumes 0.33 kWh/m³ of treated water. By implementing cogeneration with biogas production, electricity and heat are produced. The production of electricity represents only 81 % of the consumption. However, the production of heat by cogeneration is enough to cover twice the actual heat demand. The advanced treatment plant of this study presents rather low energy consumption values compared to current wastewater treatment plants which show power consumptions ranging from 0.3 to 0.8 kWh/m³, without any recovery processes. These recovery processes accounted for 3 % of the total electricity demand in WRRFs and 58 % of the demand in heat.

As stated previously, only the additional environmental costs for managing the sewage from the new district are then analyzed. The electricity balance of the four scenarios are compared in Figure 4 for the entire sanitation system with transport and treatment. The black bars represent the balance between extra consumption and extra production compared to the operation of WRRF without the new district. A negative value means higher extra production than extra consumption. In the Figure 4, the electricity needed for the nutrients recovery installed in the WRRF are taken into account in the category “WRRF”, whereas the electricity consumption for MBR in *BW-GW* scenario and nutrients recovery in *BW* and *BW-GW* scenario are detailed in the “Decentralized treatment” category.

For electricity balance, the scenarios are all energy positive except *BW-GW*. Reference scenario and *Urine* scenario are close (-8 to -9 KWh_{elec}/PE/year), *BW* scenario is less beneficial (- 4.4 KWh_{elec}/PE/year) whereas the last scenario *BW-GW* needs 6 times more electricity than it can produce. It has to be highlighted that even if for the first three scenarios the overproduction of biogas compensated the electricity consumption, the WRRF is still not energy neutral. Indeed for the new district, the production of electricity increases faster than the consumption.

Figure 4: Electricity balance for each scenario taking into account transport and treatment needs and production

Urine separation increases the electricity consumption due to a higher demand from the nitrogen recovery process train (see also Figure S 24 in SI with a focus on WRRF balance), energy consumption increasing proportionally with the nitrogen recovered by TMCS. In the meantime, as previously discussed, the production of electricity from biogas decreased in the *Urine* scenario, and leads to a slightly less beneficial balance.

In the *BW* scenario, blackwater is treated in the district and greywater is sent to WRRF. Moreover energy consumption is added for the transport of blackwater by using a vacuum sewer. Results show that the consumption in the WRRF is around two-fold lesser (1.7) than those in the Reference (9.9 kWh/PE/year versus 17.2 kWh/PE/year). This gain is related to a decrease in the need for aeration in the centralized plant since nitrogen is previously removed and treated at the district scale. The production of electricity is shared respectively in the WRRF and in the district, with the same range of production than in the *Reference* scenario. As a conclusion *BW* scenario presents a beneficial energy balance (electricity) but provides less energy compared to the *Reference*.

In the last scenario including decentralized treatment of greywater (*BW-GW*), results show that the greywater treatment needs much more electricity than all the other scenarios, with a total consumption around 100 kWh_{elec}/PE/year compared to around 20 kWh_{elec}/PE/year for all the other scenarios. This is due to the treatment systems for greywater. Indeed, the energy demand of the membrane bioreactor at district scale varies between 0.3 and 8 Nm³/h/m² (1.2 Nm³/h/m² was used in this simulation). Even if the

best performance is used for airflow consumption, such as $0.27 \text{ Nm}^3/\text{h}/\text{m}^2$ (Atanasova et al., 2017), this scenario will still present a higher electricity consumption ($34 \text{ kWh}/\text{PE}/\text{year}$) and not a beneficial balance. Finally, the production of electricity in *BW-GW* scenario is lower than all the other scenarios, even though the organic matter recovery rate is higher (39 % against 35 %). This is due to the fact that at a small scale, cogeneration production unit provides more heat than electricity.

3.3 LCA result

3.3.1 Endpoint impact categories

The first analysis undertaken on endpoint impacts shows (Figure 5) that climate change impact dominates the human health and ecosystem quality categories. For all scenarios, human health category is the most impacted with on average 53 % overall impacts, followed by resources (around 26 %) and ecosystem quality (around 22 %). *Urine* and *BW* scenarios present a reduction of impacts for all three categories of damage (ecosystem quality human health and resources). Benefits are provided by the diminution of fossil depletion and climate change impact on both human health and ecosystems. Moreover the two impacts (climate change and fossil depletion) are correlated. Source separation scenarios did not significantly change particulate matter formation and toxicity towards humans (micropollutants were not considered). The individual impacts will also be analyzed in mid-point impact category hereafter.

Figure 5: Endpoint results for ecosystems quality, human health and resources depletion impact categories

3.3.2 Midpoint methods

The endpoint results revealed that climate change (accompanied by fossil depletion) is the major contributor. The contribution analysis on this impact category is first presented. Then, the midpoint results will be analyzed by comparing the scenarios for each impact category.

Detailed graphics can be found in SI (from Figure S 25 to Figure S 41).

3.3.2.1 Contribution analysis

Figure 6 shows the impact on climate change for each scenario, positive values represent impacts caused by our system. The negative values of the graph represent impacts which are avoided. Avoided impacts are generated outside the boundaries of the studied system, for example avoided conventional fertilizer,

tap water production, but also the emissions linked to spreading of conventional fertilizers. The black bars, which represent the balance between impacts and avoided impacts, help to compare different scenarios: the smallest the bar, the better the scenario. Moreover, when we refer to the “impact of the system”, only the positive values are considered.

It can be concluded that both *Urine* and *BW* scenarios have less impact on climate change compared to the *Reference*. *Urine* and *BW* separation can reduce by 45 % and 34 % climate change impact. On the contrary, *BW-GW* scenario shows an increase by 27 % of climate change impact.

The main contributions for the *Reference* scenario are the WRRF infrastructure, sewer infrastructure and direct emissions: nitrous oxide (N₂O), methane, CO₂ into air and discharge water emissions. Thanks to the substitution of nitrogen and phosphorus fertilizer, separation scenarios can reduce climate change impact. However, the avoided phosphorus fertilizer production contributes less to the avoided fertilizer production and use (from Figure 6) than avoided nitrogen fertilizer production. Indeed avoided P-fertilizer production represents 26 % of the avoided fertilizer production and use for *Reference* scenario and only 8 % for the source separation scenario. Avoided N-fertilizer production contributes to 56 % for the *Reference* and up to 70 % for source separation scenarios. The remaining contributor to this category are the avoided emissions linked to the fertilizer spreading and the post-spreading emissions. This avoided impact of fertilizer production compensates the increase of emissions from the new specific infrastructure for both *Urine* and *BW* scenarios. Indeed, the *Urine* scenario needs collection transport which contributes to 4 % of the total emissions. More recovered fertilizer is spread into fields which increases greenhouse gas emissions (title co-products valorization in Figure 6). While higher recovery is described, a significant decrease of direct emissions is visible (60 %). This is due to less N₂O emissions explained by a reduced amount of nitrogen entering the WRRF. The *BW* scenario is less beneficial than *Urine* scenario for climate change because more energy (heat) is needed for the nitrogen recovery process in the case of blackwater compare to *Urine* (due to nitrogen concentration). The emissions related to this energy demand (boiler operated with natural gas) represent 15 % of the total emissions.

Figure 6: Comparison of the four scenarios on climate change impact (ReCiPe Midpoint H)

For the last scenario with decentralized blackwater and greywater treatment systems (*BW-GW*), the total balance increases by 27 % compared to the *Reference*. The emissions related to energy consumption (heat and electricity) represent 37 % of climate change impact, thereby increasing the emissions compared to the *Reference* scenario. Heat demand is once again a major source of impact with 16 % of the emissions. As described in *Urine* and *BW* scenarios, the direct emissions are reduced significantly (62 %) by reducing N_2O emissions. Nevertheless, the overall emissions are not balanced by avoided impacts, such as production of fertilizer and reuse of greywater for non-potable use.

3.3.2.2 Integrated analysis

All impact results with midpoint methods are summarized in Figure 7. They are normalized with the value of the *Reference* scenario, meaning that if the value is above 1, the impact is higher than the *Reference*. If we consider significant only the variation of 20 % of the impact categories compared to *Reference*, *Urine* scenario have similar results for most the categories (11), and impact better for 5 of them and only 2 worse than the reference: ionizing radiation and water depletion. The degradation of

water depletion is linked to the production of magnesium and is not compensated, in this scenario, by decrease in water consumption.

BW scenario presents 5 improved categories and 4 with a worse impact than *Reference*.

The *BW-GW* scenario represents the worst situation with 6 impact categories with values reaching 1.5 times higher than the *Reference*. These categories are all negatively impacted by to the high energy consumption of greywater treatment.

Agricultural land occupation and natural land transformation are strongly reduced with the source separation scenario thank to the increase of avoided fertilizer production and especially nitrogen.

Figure 7: Normalized comparison of all scenarios to the reference scenario, for each category of impact at midpoint level (ReCiPe H). ALOP: agricultural land occupation, GWP100: climate change, FDP: fossil depletion, FETPinf: freshwater ecotoxicity, FEP: freshwater eutrophication, HTPinf: human toxicity, IRP_HE: ionising radiation, METPinf: marine ecotoxicity, MEP: marine eutrophication, MDP: metal depletion, NLTP: natural land transformation, ODPinf: ozone depletion, PMFP: particulate matter formation, POFP: photochemical oxidant formation, TAP100: terrestrial acidification, TETPinf: terrestrial ecotoxicity, ULOP: urban land occupation, WDP: water depletion.

3.4 Sensitivity analysis

In the following paragraphs, only the impact on climate change has been discussed. The results for the other impact categories are presented in the supplementary Tables S27, S28 and S29.

3.4.1 Nitrous oxide emissions from wastewater treatment

The four emissions factors studied in the sensitivity analysis show that the rank of each scenario is not modified concerning the climate change impact. However with low emission factor (0.19%) *Urine* scenario can only decrease climate change impact by 25 % compared to the *Reference* scenario and only 14 % decrease is observed for *BW* scenario. On the contrary, with high N₂O emissions factor *BW-GW* scenario has a similar impact on climate change than the *Reference* (7 % increase).

3.4.2 Efficiency of urine collection in the toilet

For all the studied efficiencies of urine collection, *Urine* scenario still presents an improvement regarding climate change impact with a GHG emission reduction from 27 % to 45 % compared to *Reference*. However below 65 % of urine collection efficiency, the climate change impact of *Urine* scenario is similar to *BW* scenario.

3.4.3 Energy consumption for the vacuum sewer

Regarding climate change impact, *BW* scenario still presents a better balance (11 % of improvement) than the *Reference* scenario even with 15 kWh/m³ consumed for the vacuum collection of blackwater. With an energy consumption higher than 21 kWh/m³, *BW* scenario presents a worse balance on climate change impact. For *BW-GW* scenario this does not change the conclusion. Even without energy consumption for vacuum sewer, this scenario still increases by 11 % the climate change impact of the system.

3.4.1 Energy consumption of the membrane bioreactor

MBR energy consumption has a strong effect on the climate change impact for *BW-GW* scenario. Indeed the emissions vary from 79 kgCO₂eq/PE/year to 106 kgCO₂eq/PE/year by varying from 0.4 kWh/m³ to 1.5 kWh/m³. Above the value of 0.56 kWh/m³ for MBR electricity consumption, *BW-GW* scenario presents a degraded climate impact compared to the *Reference* scenario. A reduction of more than twice the current value would be necessary to reach a similar impact as reference.

4. Discussion

4.1 Benefits of urine and black water separation scenarios

In this study three different wastewater management scenarios with source separation were compared to a centralized WRRF operating with nutrient recovery. Globally the results shows that *Urine* and *black water* separation can reduce by 45 % and 34 % climate change impact respectively. Climate change is the most important category regarding LCA (contribution to endpoint). High nitrogen recovery is the major explanation for such improvement, because it reduces direct N₂O emissions and decreases the needs for chemical fertilizers. Nevertheless nitrogen recovery process can generate significant energy demand (heat), which negatively impacts the results for the *BW* scenario for instance. The *BW-GW* scenario including greywater recovery is very beneficial regarding the water depletion criterion (WDP), but it is negatively impacted by the high energy needs of decentralized membrane treatment technology (MBR) resulting in more impact on climate change. Moreover to achieve *BW-GW* scenario, all the greywater needs to be reused by replacing tap water, but the flush water volume will only represent 6 % of the available amount of greywater. To reach this water recovery level, new applications need to be found, such as for irrigating green spaces, cleaning roads...

In this study, urine and blackwater separation showed comparable phosphorus recovery thank to the biological removal of phosphorus in the centralized WRRF. In this way, the P from the feces was also recovered in the digestate. (Meininger et al., 2010) showed different P recovery rates when chemical removal of phosphorus was implemented in the WRRF. Indeed, phosphorus recovery was found to be 3 times higher for blackwater separation compared to *Urine*.

Regarding energy, the *Urine* and *BW* scenarios showed close electricity balances compared to the *Reference*. Source separation did not significantly improve the recovery of COD in the form of methane, compared to the WRRF. The extra consumption due to the collection steps using vacuum pumps was compensated by the reduction in aeration in the WRRF. Moreover the production of electricity was not improved with direct digestion of blackwater (*BW* and *BW-GW* scenarios) because of the lower electricity efficiency of CHP with a decentralized scale. (Thibodeau et al., 2014a) found a similar conclusion for blackwater separation: the energy production from blackwater digestion was reduced compared to wastewater digestion, because organic matter from greywater was not degraded. For the *BW-GW* scenario, energy consumption by the MBR was not compensated by the energy production which led to an unfavorable electricity balance. (Remy, 2010) found similar conclusions on MBR consumption and energy balances of the *BW-GW* scenario. Finally, the use of TMCS for N recovery consumed large amounts of heat. If installed at the centralized WRRF, the extra heat requirements can be fulfilled by the extra heat produced by the wastewater coming from the city, which is not the case at

decentralized plants. Moreover blackwater is more diluted than the supernatant from digestate and more energy is required to heat it up to 45°C.

Regarding endpoint damages, the impact of the wastewater management system compared to the *Reference* was reduced for *Urine* and *BW* scenarios but increased for *BW-GW*. The main contributors were climate change and particulate matter formation on human health, fossil depletion and climate change on ecosystem quality.

Regarding midpoint impacts, no clear conclusion can be drawn from the analysis of the 18 impact categories. However, *BW-GW* scenario presented the most categories with worst results, mainly due to the high energy consumption of the MBR. Reduction of climate change impact observed in the *Urine* and *BW* scenarios, was due to the reduction of N₂O emissions and the avoided production of nitrogen fertilizer. (Thibodeau et al., 2014a) reported different conclusions between a *reference* and a *BW* scenarios with an increase of 23 %. However, authors did not take into account any direct emission from WWTP, nor any post spreading emissions from mineral fertilizers.

It has to be emphasized that the WRRF used as a reference already provides high environmental performances. For example, compared to average value for conventional WWTP, a 14 % of decrease in climate change impact is observed for our advanced WRRF (0.9 kgCO₂eq/m³ for our reference compared to 1.05 kgCO₂eq/m³ for conventional WWTP of (Bisinella de Faria et al., 2015) (with the same N₂O emission factors and without sewer). In this perspective, source separation, and especially *Urine* separation is currently the best option to significantly reduce the impact on climate change (in our study, 45 % of decrease for *Urine* scenario at district scale).

Finally, in this study, the European energy mix of 2004 has been used for the LCA analysis. Changing the energy sources will have a major effect on the related impact categories i.e. climate change, fossil depletion, particulate matter formation. Electricity mix varies from a country to another and will change in the near future by increasing the proportion of renewable resources.

Indeed, the European electricity mix of 0.5 kgCO₂-eq/kWh is assumed, however, for example, prospective scenarios in Spain can achieve 0.039 kgCO₂-eq/kWh in 2050 with 80 % GHG emission reduction compare to 2005 (García-Gusano et al., 2017).

Moreover, if the current French mix is used, a low impact on climate change category will be observed since the emission factor in France is only 0.078 kgCO₂-eq/kWh due to the large proportion of nuclear energy. However, the ionizing radiation impact will show an increase since the emission factor is 0.53 against 0.37 kg U235-eq/kWh. Whatever the energy resources, the ranking of the four scenarios (if placed on the same geographic context) will not change since it is determined by the energy consumption. The remaining question is if a green energy could justify the choice of an energy intensive scenario. Even if in the future fossil energy will be more and more replaced by renewable one, the humanity is constrained

at drastically reduce the energy consumption (IPCC, 2018) from now on, since renewable resources are limited and less efficient.

4.2. Possible directions for optimization and future research

From this study several directions for future research can be proposed in relation with energy balance and associated climate change impacts.

First, regarding the nitrogen recovery approaches, technologies still need to be optimized and validated. Attention should be paid to the energy demand and practical applicability regarding different effluent sources. Here the TMCS was assumed to be the optimal process ($2.55 \text{ kWh}_{\text{elec}}/\text{kgN}_{\text{recovered}}$ (Böhler et al., 2018; Zhang et al., 2020)) whereas other commercialized techniques can be much more energy consuming (ex: Nitrification/distillation: $76 \text{ kWh}_{\text{elec}}/\text{kgN}_{\text{recovered}}$ (Udert and Jenni, 2013; Udert and Wächter, 2012), stripping: $7 \text{ kWh}_{\text{elec}}/\text{kgN}_{\text{recovered}}$ (Maurer et al., 2003; Zhang et al., 2020) or even nitrification/denitrification: $4 \text{ kWh}_{\text{elec}}/\text{kgN}_{\text{recovered}}$ (Maurer et al., 2003)). Even if the TMCS process for nitrogen recovery seems relevant to implement, heat consumption should be optimized. Several options should be investigated. Firstly, heat recovery from the effluent could be installed in order to pre-heat the inlet. Secondly, greywater heat can also be recovered. Indeed, this can account for a large source of heat but at a low temperature (below 40°C). Moreover, reducing the heat demand by limiting the flowrate can be an option for blackwater treatment. In a final step, other sources of heat can also be investigated to reduce the impact on climate change. For instance, using a boiler with wood pellets has only an emission factor of $0.016 \text{ kgCO}_{2\text{-eq}}/\text{MJ}$ (Ecoinvent database v3.6), against $0.07 \text{ kgCO}_{2\text{-eq}}/\text{MJ}$ for the natural gas boiler (Ecoinvent Database v2.7).

Future research should also focus on a detailed integration of treatment infrastructure in life cycle assessment in order to refine the results for decentralized infrastructure. Indeed in this study, the infrastructure is only based on the treated PE and Ecoinvent database for WWTP infrastructure.

For future scenarios analysis one should also consider the possible transformation of treatment train regarding emerging technologies on the mainstream. For instance COD recovery can be improved by producing more primary sludge with an enhanced primary sedimentation tank, or high-rate activated sludge. In addition the combination of biological removal of phosphorus and anammox bacteria in the mainstream is a promising alternative but still needs to be demonstrated (for both centralized and decentralized treatment). A current challenge is to guaranty performance of anammox systems in the mainstream under cold climate whereas successful trial was reported by (Cao et al., 2017) under warm climate. However, from an environmental point of view, a mitigation strategy of N_2O emissions should be found for partial nitritation and anammox applications, as a very small increase of such emission can completely reverse the advantages in terms of energy saving (Besson et al., 2017).

To improve the energy balance of *BW* and *BW-GW* scenarios, increasing the heat production from cogeneration can also be an option. By considering the decentralized treatment, co-digestion of blackwater with kitchen waste can be imagined as it is considered in future scenarios deployed in Northern European countries for instance (Skambraks et al., 2017). This strategy shows valuable environmental results (Thibodeau et al., 2014a; Vergara-Araya et al., 2020).

Greywater treatment at small scale is another research topic to investigate as energy consumption of MBR revealed to be highly impacting on climate change. Moreover progress have been made to reduce the energy consumption of full scale MBR up to 0.4 kWh/m³ for MBRs commissioned after 2014 (Brepols, 2020). These progress can be beneficial if they can be extended to small scale MBR treating greywater. Indeed with such level of energy consumption the same climate change impact as the *Reference* scenario could be achieved.

5. Conclusions

A tool was developed to model the collection and decentralized treatment at the district scale. The effect on the centralized treatment plant is taken into account through plant-wide modelling. Results obtained from the simulation are used as inventory for the LCA. This study aimed to compare several options in order to enhance the recovery of resources from wastewater by using a WRRF coupled with, or without, source separation systems. The mains results from this study are:

- Both *Urine* and *BW* scenarios can reduce greenhouse gas emissions by respectively 45 % and 34 %.
- At endpoint level, climate change is the most important category for all the scenarios. Regarding other LCA midpoint categories, *Urine* and *BW* scenarios show similar results in most impact categories, whilst the *BW-GW* scenario only improved water depletion category.
- Nitrogen recovery is the most impacting aspect regarding LCA results. Source separation of urine or blackwater can achieve a much better recovery compared to centralized WRRF. This benefit is related to both chemical fertilizer substitution and reduction of N₂O emissions from biological treatment. However nitrogen recovery technique (here TMCS) is a sensitive consumer of energy, and future effort should be paid to ensure proper design and operation of this process for source separation.
- Greywater water reuse scenario appears globally negative for climate change. This is due to high energy demand of decentralized membrane treatment (MBR) which is not compensated by avoided potable water production. However the benefit of water recycling and reducing the pressure on water bodies should be more considered in the future. Conclusion would be different depending on the context (decarbonized energy or drinking water production with high GHG emissions).

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