Department of Water Management
Faculty of Civil Engineering and Geosciences

# A life cycle perspective of water conservation and resource recovery strategies in the urban water system

**Elisavet Apostolidou**

MSc Thesis, December 2020

Delft University of Technology

Master Programme in Department of Water Management
Faculty of Civil Engineering and Geosciences

# A life cycle perspective of water conservation and resource recovery strategies in the urban water system

by

**Elisavet Apostolidou**

in partial fulfillment of the requirements for the degree of

**Master of Science**
Water Resources Engineering
at the Delft University of Technology

December, 2020

**Thesis committee members:**

| | |
|---|---|
| Prof. Dr. J.P. van der Hoek (chair) | TU Delft<br>Sanitary Engineering |
| Prof. Dr. Ir. J.B. van Lier | TU Delft<br>Sanitary Engineering |
| Dr. Ir. E. Abraham | TU Delft<br>Water Resources Engineering |
| Dr. Ir. L. Zlatanovic (supervisor) | TU Delft<br>Sanitary Engineering |
| Dr. K. L. Lam (supervisor) | TU Delft<br>Sanitary Engineering |

# Acknowledgments

"The greater the storm, the brighter the rainbow" my father always says to me. My academic trip was long but full of challenging and inspiring moments. After seven years of continuous work and progress, this trip comes to an end with the completion of my MSc Thesis. However, this successful work was a team effort and there are many people I would like to thank for their contribution and support.

First of all, I cannot thank Professor Jan Peter enough for giving me the opportunity to work on this interesting project. Even during these trying moments where the communication bonds were weak due to remote work, Prof. Jan Peter was always providing me with important feedback and chance to communicate with other experts on the field whenever needed, which would be impossible without his valuable help.

I would also like to thank from the bottom of my heart my two supervisors; Ljiljana Zlatanovic and Ka Leung Lam. I am very thankful to them, since their guidance and valuable comments in their field of expertise have been very helpful in enhancing this work. They supervised me with unwavering support, while giving me the appropriate directions, total freedom and encouragement. Their confidence and understanding during the difficult times are very well appreciated.

Special thanks to my beloved parents, Stefanos and Kiki, and all my close family for their unlimited support and love. Finally, I would like to whole heartily thank Sherif for always being there for me and believing in me in every step of my academic trip. I would not have achieved anything without them.

Delft, 2020
Elisavet Apostolidou

# Summary

With the rapid urbanization and growing population, some of the main issues in sustainable management of the water systems in cities include the qualitative and quantitative protection of the natural resources. The transition towards water conservation and resource recovery via the waste(water) streams is a major challenge nowadays. To meet the high demands of urban life, resources need to be efficiently used and resource recovery from generated "waste" streams should become a "new normal". Water and wastewater can provide an alternative and environmentally viable source of resources supporting the resilience of natural systems under water stress. There are many resources that can be recovered via the water path, such as water itself, energy and components such as nutrients and metals. However, the urban water system is a complex of interconnected processes which should be explored, one by one, in order to create an overall sustainable value.

This study aims to quantify the environmental impacts of water conservation and resource recovery strategies in an urban water system in terms of life cycle energy consumption, life cycle global warming potential, life cycle water consumption and life cycle freshwater eutrophication potential. In total, eight scenarios (two sets of scenarios) were assessed. The first set of four scenarios involves household-level interventions only: greywater treatment and reuse, rainwater harvesting and use, usage of water efficient appliances, and food waste valorisation via wastewater streams. The second set of four scenarios adds centralised phosphorus and nitrogen recovery to the first set of four scenarios. Life cycle assessment (LCA) methodology was used to assess the environmental impacts of these scenarios, compared to a baseline of residential water end-use without water saving or food waste valorisation via the sewer system. The baseline has also centralised wastewater treatment with biogas and struvite recovery, and co-incineration of food waste and sludge waste for energy recovery.

The LCA deals with complex interactions between a product or process with the environment, taking into account all the implications caused by the production, use, disposal of raw materials, as well as the avoided impacts from resources offset accounting. LCA points out technical factors that require further research and operational conditions with the highest potential for impact reduction, ensuring that only the most promising technologies are pursued before lock-in occurs. The system boundary of this study has been outlined broad enough to reflect all the activities that are involved in the urban water system, including water supply services (purification and distribution), domestic water end-use, wastewater collection and treatment, sewage sludge waste management, or could potentially be involved, such as the food waste stream.

The results revealed that the usage of water efficient appliances (low-flush toilets, water efficient shower heads, waterless washing machines, waterless dishwashers) at a household level, coupled with struvite recovery, biogas recovery and sludge drying at centralised wastewater and sludge treatment facilities can achieve the lowest life cycle environmental impacts among the studied scenarios. This scenario was estimated to offset 79% of life cycle energy consumption, 72% of life cycle greenhouse gas (GHG) emissions, 22% of life cycle freshwater eutrophication potential and 56% of life cycle water consumption. Food waste valorisation via the sewer system provides great opportunities for nutrient and energy recovery from wastewater and sludge waste, but it increases the nutrient discharge to recipient water bodies. Greywater reuse demonstrates an overall better environmental profile than rainwater harvesting, mainly due to lower demands for wastewater treatment and higher efficiency of biogas and struvite recovery. The results, also, revealed that the combination of thermal hydrolysis of sludge and air stripping as methods for nitrogen recovery from digester reject water are quite energy and chemical intensive technologies, leading to high GHG emissions and nutrient releases. Furthermore, sludge mono-incineration and phosphorus recovery from sludge ash can be a more efficient management method compared to sludge co-incineration in regards to life cycle energy consumption, life cycle global warming potential and life cycle freshwater eutrophication potential, only in case of on-site application. However, it returns significant amounts of water back to the environment. The sensitivity analysis revealed that the energy requirements for water heating in showers, dishwashers and washing machines are the most important parameters for defining the life cycle global warming potential of the entire urban water system.

# Contents

# List of Figures

# List of Tables

# 1. Introduction

## 1.1. Principles of water conservation and resource recovery

Resource degradation and availability are global issues of concern. The ever increasing population and the industrial and technological development are critical factors in accelerating the depletion of natural resources. Day by day, essential resources such as water, energy and other materials (e.g., oil, natural gas, coal, phosphorus (Ruz, 2011)) are becoming scarce, making reuse of resources more and more attractive and requisite (van der Hoek et al., 2016). Scarcity of natural resources poses a threat to the continued prosperity of the world's population, especially within the urban fabric which is more vulnerable to experience quantitative and qualitative degradation of the available natural resources. Urban areas consume and transform massive flows of energy and matter, with the urban life been organised in a linear fashion meaning rather than circular. For cities, this means increased waste. The linear methods of production and consumption are unsustainable for the planet. As urbanization continues, the desire for more resources will accelerate if the current consumption trajectory continues. The linear set-up of "take-make-waste" is wasteful by design, while the circular economy of "reuse-recover-recycle" is conceptualised as a continuous cycle of value preservation and resource optimization, presenting sustainable alternatives for eliminating waste (Forum/PwC, 2018). Therefore, cities must evolve and adapt in order to survive and grow economically in an ever changing world with increasing needs for water, energy, raw materials and goods.

Embedding resource recovery as a key consideration in urban water and wastewater management plays a significant role to the efficient tackling of the environmental degradation and climate change. It is evident that resource recovery is an approach to achieve and optimise natural resource and energy efficiency using the waste output of a production process as the resource input creating valuable products as a new output (Iacovidou et al., 2017). Cities are major direct and indirect water users, energy users and nutrient releasers. Water and wastewater utilities are among the largest consumers of energy in municipalities, regions and countries (Xue et al., 2016). Satisfying the demand for water and sanitation services currently requires significant amounts of energy for drinking water abstraction, treatment and distribution, water end-use, wastewater collection, treatment and disposal. Thus, the transition to sustainable urban utilities and water services presents promising opportunities to mitigate water and energy consumption, global warming potential and water pollution. The requirement for safe and reliable water supply is universal and is acknowledged as a basic necessity for human livelihoods (Racoviceanu and Karney, 2010). The need for adequate water infrastructure, as essential to sustain human health and economic growth, encouraged several innovative water conservation and water fit-for-purpose strategies (Racoviceanu and Karney, 2010). Furthermore, water, besides being a resource of its own, is a transport medium of other valuable resources (van der Hoek et al., 2016). Materials and chemicals are added to water by end-users (e.g., households, businesses) during drinking water use and wastewater production. By extension, massive amounts of organic matter, phosphorus, nitrogen, heavy metals, thermal energy, chemical energy pass through the centralised wastewater treatment plants (WWTPs), and are discharged to the environment, rather than reused and converted into renewable energy sources or organic fertilisers.

The urban water cycle, and especially wastewater sector, has many opportunities to meet the high resources demands of the urban life and close water, energy and nutrient loops. Resource recovery from wastewater is more effective at high concentrations. This can be achieved through dewatering processes at treatment plants, but also by limiting wastewater dilution in the collection process. Some of the methods to narrow down the dilution are to reduce the domestic drinking water use (e.g., recycling greywater, usage of water-efficient appliances), adapt separate sewer systems (separate drains for rainwater and wastewater) and reduce the groundwater infiltration arising from the cracks or leaks of the wastewater network by repairing/replacing broken pipes. Another perspective for increased resource recovery efficiency is the enrichment of wastewater with nutrients via the disposal of domestic food waste into the sewers (e.g., usage of kitchen waste grinders) and co-treatment with municipal sewage. All these techniques enhance the motivation for exploring the environmental opportunities and threats arising from the implementation of resource recovery as a design criteria of the urban water system.

The impacts of limiting the wastewater dilution through water conservation strategies and strategies of enrichment of wastewater with nutrients via the application of food waste disposers have been studied using integral hydraulic modelling. However, circular solutions do not necessarily equate to sustainable solutions. Large scale implementation of circular scenarios might overcome one problem but create others, leading to a worse overall environmental performance. Integrated management and exploitation of resources via the urban water cycle should include criteria that respect the natural environment and public health and return water to the environment in a sufficient quantity and quality, without adding pressure on other natural resources. Water conservation and resource-efficient strategies require assessment of the environmental impacts related to the entire life cycle of the urban water system. There are plenty of methods and tools available that evaluate the environmental implications of urban water systems. Some tools aggregate the indicators to an index, such as Ecological Footprint Analysis (EFA), Environmental Sustainability Index (ESI) and some product-related assessment tools focus on production and consumption of goods and services, such as

Life Cycle Assessment (LCA), Life Cycle Costing (LCC), product material flow analysis (Srinivasan et al., 2011) (Appendix A briefly describes three sustainability tools). LCA is a method that quantitatively assess the potential environmental impacts of a product or a process across its life cycle, including the stages of raw material extraction, transportation, manufacturing, usage, end-of-life treatment, recycling and eventually, disposal. This method provides the facts needed to create a sustainable value and plan robust city infrastructure.

### 1.2. Research gaps

Several pieces of research were conducted in the last two decades which focused on the LCA of several water conservation and resource recovery strategies. These studies generated new knowledge, but also revealed significant research gaps. The overall picture emanating from the literature is that companies and municipalities are willing yet lack of knowledge and evidences to transform their activities and meet sustainability challenges in a meaningful way. Research into circular economy implementation is emerging and the empirical knowledge limited, although there is a growing body of evidence suggesting a number of drivers and barriers (Velenturf and Jopson, 2019). The major challenge of transitioning to resource-efficient urban water systems is not the availability of technological options for resource recovery but the lack of planning strategy and design methodology to identify and deploy the most sustainable solution in a given context (van der Hoek et al., 2016). Lam et al. (2020), after an extensive comparison of LCA studies, concluded that there is a need to improve methodological consistency (e.g., multifunctionality, fertilisers offset accounting, contaminants accounting) and ensure transparency of inventory and methods. The current transport infrastructure of urban water is based on the criteria of water quality, water quantity, public health, safety and comfort, excluding "Resource recovery" from the design parameters (AMS). Literature studies on the environment performance of resource recovery-based strategies have very limited scope, focusing on activities that are concern either at dwelling level nor at a plant level. However, the overall sustainability of urban water systems requires integrated modifications and activities that include combined processes, and not individual water, energy and nutrient utilities in order to identify solutions which eliminate resource depletion problems, rather than simply shifting them creating new ones. This scientific research can improve and extent the existing knowledge and provide transparent information to create innovative systems that are above all sustainable and environmentally friendly.

### 1.3. Research questions and hypothesis

The main objective of this research is to conduct a holistic assessment of the environmental performance of several water conservation and resource recovery strategies. It combines location-based (if available) and global data from literature researches, with a theoretical framework based on the LCA methodology to address the following research questions:

I. Which interventions are the most resource efficient and promote an environmentally sustainable urban water system with regards to life cycle energy consumption, life cycle water consumption, life cycle global warming potential and life cycle freshwater eutrophication potential?; Which interventions shift burdens and induce more environmental challenges?

II. What are the key parameters that have the greatest influence on life cycle global warming potential of the urban water system?

III. What is the most efficient point of improvement in the entire urban water system and what measures need to be taken in order to adopt the most promising technologies into the implementation road map?

Previous LCA-based studies have shown that closed loops are not always favourable from an environmental point of view (Haupt and Zschokke, 2017). The main hypothesis, therefore, is that the water conservation and resource recovery strategies, applied either at a dwelling or at a centralised waste(water) treatment level, are not necessarily environmentally sustainable and resource efficient options. Some of the strategies, although recover resources, they can introduce more environmental challenges, increasing the resources consumption and the environmental emissions.

## 1.4. Thesis outline

The report is divided into seven chapters. In the first chapter, the aim of the thesis along with identification of the knowledge gaps are presented. The second chapter deals with the theoretical basis of the urban water systems and the technologies under study. The third chapter presents an overview of the urban water and sanitation services in Amsterdam and dives into the research approach. The forth chapter focuses in the methodology and the steps which were followed in order to obtain the final results. In the fifth chapter, the final outcomes and the results of the sensitivity analysis are presented. Chapter six highlights the major outcomes, cites the limitations of the work and makes recommendations for future research and diffusion of the innovations. The last chapter summarises the conclusions of the research.

# 2. Background and literature review

## 2.1. The water cycle

The water cycle has no starting point nor ending point. Water is a renewable resource, with the evaporation and precipitation replenish the fresh water balance constantly. Most precipitation falls back into the oceans or onto land surface via gravitational flow over the ground as surface runoff. Part of the runoff enters rivers in valleys in the landscape, with stream-flow moving water towards the oceans, while another part infiltrates deep into the ground and replenishes aquifers with freshwater. One of the most important environmental challenges is to provide sufficient and safe water to people. The main human activities concerning the water cycle are the drinking water production and supply, the wastewater and sewage sludge treatment. The water services vary from country to country and their management is a national competence. The drinking water supply, the collection and treatment of wastewater are the main water services in all European countries, while stormwater management do not come everywhere under the jurisdiction of the water services. In a few countries, flood protection and reclaimed water provision is also carried out by water service providers (EurEau, 2018). In the Netherlands, the key players in the water cycle are the drinking water companies, the municipalities and the Water Authorities (Geudens and van Grootveld, 2017). The drinking water companies start the management the water cycle by abstracting water from the water bodies (e.g., groundwater, polders, surface water) as a source for the production of drinking water. The raw water is purified and then supplied to households and businesses. After the water end-use and the addition of several materials (e.g., chemicals, food scraps, human excreta), the water is disposed via the drains as wastewater. Municipalities are responsible for collection of wastewater (usually including rainwater) via the sewerage system, while the Water Authorities are responsible for treating the wastewater in sewage treatment plants (Geudens and van Grootveld, 2017). Many companies pre-treat their wastewater in private treatment plants before the discharge to the sewers. After treatment at the wastewater treatment plant (WWTP), the clean water, also known as effluent, is discharged into the surface water and returns back to the environment. The sewage sludge waste arising from the treatment process is incinerated or transformed into added-value products (e.g., fertilisers), while the disposal of sludge in landfills is still an important disposal route for some European countries (excluded the Netherlands) (Commission et al., 2010). The Water Authorities manage the water system and take measures to ensure sufficient water quantity and optimal water quality according to the water quality standards of each country.

### 2.2. Sewage sludge management

Sewage sludge is a by-product of WWTPs, considered as a valuable source of nutrients and, but also potential threat to humans and the environment due to the presence of organic pollutants and heavy metals. Sustainable solutions and the best available techniques for the treatment and disposal of sewage sludge, including recovery of energy and nutrients, are currently being discussed in the European Union (EU) (Pinasseau et al., 2010). The quantity of sewage sludge generated in WWTPs is increasing with the progressive expansion of wastewater networks due to population growth and industrial development. The situation is crucial especially in large and densely populated cities with high sewage sludge production and limited availability of surface area for its processing and disposal. There are two main pathways of sewage sludge management: organic recycling (e.g., use in agriculture, composting and land reclamation), and thermal treatment for energy and nutrient recovery (e.g., mono-incineration and co-incineration, pyrolysis, gasification, wet oxidation, thermal hydrolysis, hydrothermal carbonization and bio-fuel production by microorganisms) (Đurđević et al., 2019).

In the past 20 years in the EU, sewage sludge treatment has relied on three main processes: stabilisation by anaerobic digestion (AD), dewatering and drying, and thermal treatment (Đurđević et al., 2019). AD is the most popular stabilisation method as it produces biogas which is valuable energy source. Biogas from sewage sludge digestion is usually for combined heat and power (CHP) generation. The electricity generated from biogas engines supplies electricity to the grid, reducing the consumption of raw materials and fossil fuels in power plants, and the possible surplus heat recovered from CHP can replace thermal energy from fossil fuels and reduce the relevant emissions (Li et al., 2017). Biogas can also be upgraded into biomethane (Green Gas) for injection into natural gas networks or as transport fuel (van der Hoek et al., 2017). There are also several nutrient recovery technologies applied to the digested sludge or to the sludge dewatering streams which reduce the formation of struvite scale in the equipment and prevent operational problems, while parallel they can replace the use of synthetic fertilisers in the agricultural sector.

Incineration of sewage sludge waste becomes the fastest growing disposal practice in the EU, as the main alternative to agricultural reuse where the soils are not suitable for application of recycled organics or public disapproval is present. Incineration reduces the mass (up to 75%) and volume (up to 90%) of sewage sludge and safely destroys hazardous substances, such as pathogens and toxic chemicals (Stauffer, B. and Spuhler, D., 2019). Sludge incineration can also be combined with energy recovery. It is usually performed on stabilised and dewatered sludge. However, the high water content of dewatered sludge (approximately 25% dry matter (DM)) has no practical energy value. Thus, in incineration plants and waste-to-energy (WTE) plants, sludge is first being dried and grinding before incineration. Sewage sludge is comparable to wood biomass in terms of energy content but with higher inorganic (ash) content, with a heating value of 17–18 MJ/kg DM for raw sludge, 14–16 MJ/kg DM for active sludge (AS), and 8–12 MJ/kg DM for stabilised sludge (SS) (Đurđević et al., 2019). Combustion-based processes for municipal solid waste (MSW) treatment are a controversial subject around the world. In the absence of effective controls, harmful pollutants may be emitted into the air, land and water which may influence human health and environment (Zafar, 2018). Sewage sludge incineration is a potential source of harmful substances such as dioxins, furans, and heavy metals which are present both in flue gases and in the residual incinerated sewage sludge ash (ISSA). Even though most ISSA is currently landfilled, it has significant potential to be used as a substitute for clay (bricks, tiles, pavers) and raw material for the production of lightweight aggregate and Portland cement (Donatello and Cheeseman, 2013). However, sludge combustion represents a loss of its valuable phosphate content and hinders phosphorus recovery which is in high mass fraction and is typically comparable to that of a low grade phosphate ore (Donatello and Cheeseman, 2013). Thus, ISSA has significant potential to be used as a secondary source of phosphate for the production of fertilisers and phosphoric acid. Although, the phosphorus recovery from diluted ash that is produced in co-incineration plants is not feasible at present and is limited to ashes with high concentrations of phosphorus, like those produced in mono-incineration plants (Fooij, 2015). In recent years, several technologies were developed for phosphorus recovery from ISSA, including pyrolytic processes and various wet processes. In wet processes, acids or bases are added to ISSA, in order to dissolve phosphorus. Afterwards, phosphorous can be recovered through the precipitation of ammonium, calcium, sodium, iron, or aluminium phosphate, which are compounds identical to the ones found in mineral phosphate fertilisers (Đurđević et al., 2019). However, many of the technologies are still in the experimental phase or are not yet feasible. After adopting the Directive concerning Urban Wastewater Treatment 91/271/EEC, EU member states agreed to implement primary, secondary, and tertiary wastewater treatment processes (see Figure 2.1), starting from large urban agglomerations and subsequently moving onto smaller ones (Đurđević et al., 2019).

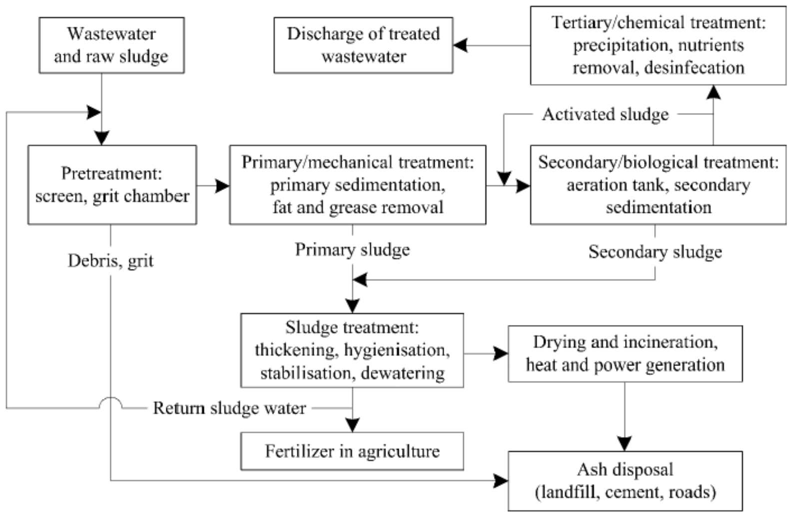



**Fig. 2.1:** Main processes of wastewater and sewage sludge treatment within the EU (Đurđević et al., 2019)

## 2.3. Resource-efficient strategies in the water system

In many parts of the world, the water industry is under intense pressure, as water demand is ever increasing and existing potable water supplies reaching their limit. Population growth, differentiated lifestyle particularly in the urban areas, coupled with climate change consist dominant factors which lead to the growing deficit between the available water resources and the increasing demands. The main impacts of climate change in urban water systems are associated with changes in air temperature, leading to shifts in precipitation patterns which increase the frequency and intensity of flood and drought events (Fidar et al., 2010). Despite the fact that the Netherlands is known for its frequent rainfalls and thus for its abundance in fresh water, the last few years it started a nationwide water shortage across the country with multiple drought and heat waves have been recorded. The quantity and quality of surface water is expected to be under pressure, with more frequent and severe drought events (Duinen et al., 2015), as well as salinization of water resources (van Duinen et al., 2015). Urban water systems are both affected by and contribute to climate change. The excessive water consumption (i.e., households, industry, agriculture), as well as the energy and materials consumption and the environmental emissions (e.g., air, water, soil) during the water and wastewater management (Flower et al., 2007) have resulted in an increasing pressure on the water systems, which adds to the effects of climate change. Water quantity and quality degradation, therefore, constitute one of the main challenges that the Netherlands will face soon.

The requirement for reliable water management system is universal, acknowledged as essential to sustain human health and economic growth. The last few years, governmental authorities and municipalities have shown great interest in increasing the sustainability of products and services regarding the urban infrastructure. Water conservation is one of the most wide-spread water demand management methods to provide adequate and safe water infrastructure by planning and implementing actions that reduce misuse and loss (Kalbusch and Ghisi, 2016). There are many researches related to water conservation strategies at a dwelling level which promote the principle of efficient water use and water fit-for-purpose, minimising the drinking water demand without compromising the quality of services. The technical feasibility of providing water-related services coupled with water fit-for-purpose has been demonstrated in pilot projects in Europe, North America and Oceania (Xue et al., 2016). There is also the Code for Sustainable Homes (CSH) which sets out various water efficiency targets, among them to reduce the domestic consumption of potable water through installing water efficient devices, rainwater harvesting and greywater reuse systems (Fidar et al., 2010). However, the implementation of water conservation methods in an existing water system, as an upgrading strategy, induces changes in the water quantity and quality which can affect the water transport networks.

The requirements for embedding resource-efficient design options into the water management systems is vital due to the current resource overexploitation and its collateral consequences (e.g., water quantity and quality degradation, global warming potential, eutrophication potential). The overuse and misuse of freshwater in households, the carbon intensive energy demands for municipal water supply and wastewater treatment, the excessive use of chemical fertilisers in agriculture and nutrient releases on the water systems are some of the major environmental issues which can be confronted with the resource conservation and recovery. The sustainability, although, is not limited only in the water conservation. Food waste valorisation via their disposal into the sewers is another possible promising technique, which can enrich the WWTPs' organic load and improve the recovery performance in terms of energy and nutrients. The challenge of transitioning to resource-efficient urban water systems is often not the availability of technology for resource recovery, but the lack of planning and design methodology to identify and develop the most sustainable solution in a given context (Lam et al., 2020). The approach of such complex issue as the sustainability yield in the urban water requires multidimensional investigation with well-aimed interventions. European cities lead the way on the overall sustainability of their water management, while the Netherlands wrestling with the issue to shift the parasitic cities into circular ones, focusing on the water-energy nexus (Giezen, 2018). The Dutch government, in line with EU policy and other global trends, had placed a priority in shifting to a circular economy approach (see Appendix B). This section presents the theoretical background of the resource-efficient technologies under study, including fundamental information regarding their performance, benefits and limitations, across relevant LCA studies.

### 2.3.1. Greywater reuse systems

Greywater separation and decentralised treatment and reuse is an attractive alternative to centralised conventional systems with flexibility of capacity expansion, offering the possibility to reduce the drinking water supply, eliminate long-distance water transportation and ease the treatment capacity of the centralised WWTPs. Furthermore, this method decreases the dilution of the wastewater enabling better biogas and nutrient recovery in the WWTP (Kobayashi et al., 2020). While black water is more suitable for recovering biogas and nutrients, greywater is viewed as more feasible and socially acceptable for water reuse because of its relatively low levels of contaminants (Kobayashi et al., 2020). Greywater, which is typically coming from washing machines, kitchen sinks, baths, and hand basins, is a low-polluted domestic wastewater resource which consists of 60-70% of the total domestic in-house water demand (Prajapati, 2018). It can be used as an alternative water supply source to cover all the non-potable in-house water demands, reducing the drinking water supply and the wastewater production. It has been established that greywater reuse, alongside with the offset of municipal water and wastewater collected, treated and distributed, it reduces the energy use for it. This change in the energy consumption pattern, can lead to the mitigation of GHG emissions, and consequently global warming potential (Xue et al., 2016). However, the energy intensity of such systems and thus the overall energy consumption is a high questioned fact.

Human health risks related to greywater reuse raise many questions and consist a primary concern for residential greywater reuse implementation. The low frequency of greywater circulation and the longer greywater residence time in the storage tank undergoes changes in quality which include growth in numbers of microorganisms (e.g., total coliforms, faecal coliforms) increasing potential human health risks from saprozoic (environmental) pathogens (Kobayashi et al., 2020). Alongside, the greywater provides potential for viral infection (Dixon et al., 1999). Results from human health carcinogenic potential showed that larger water storage tanks could add significant impacts, so water reuse applications that occur regularly is preferable (Kobayashi et al., 2020). Greywater reuse also generates aerosols which may transmit inhalable pathogens, like *Legionella*, and adds to the potential health risk. Particularly, Blanky et al. (2017) after a Quantitative Microbial Risk Assessment (QMRA) concluded that the annual risk associated with reuse of treated and chlorinated greywater was not significantly higher than the risk associated with using potable water, whilst the reuse of treated but unchlorinated greywater was associated with significantly higher health risks.

Even though greywater reuse can produce substantial environmental benefits, social impacts cannot be ignored in water management decision-making. The implementation of such recycling systems require some construction interventions at a dwelling level to create a separate greywater collection and distribution system, as well as space availability for the treatment unit. The installations of greywater systems can also result in potential for plumbing cross-connections between drinking water and non-potable water network, caused by back-pressure or back siphonage, fact that adds to the social and economic burdens (Nolde, 2005). An example to consider is the case of the Netherlands, where in 2003, the ministry banned all dual water supply schemes for households in the Netherlands. The reason was some health incidents induced by a wrong connection between the drinking water supply and the B' quality water network in one of the nine pilot projects, which were defined in 1996, to investigate the possibilities of rainwater and greywater usage as an alternative sources of water for non-potable use Agudelo-Vera et al. (2014a). The safeguard of hygienic of greywater reuse requires risk assessments, public education for the proper use of such systems and proper design and colour-labelling of the pipes.

Several greywater treatment methods have been studied in the literature, as a sustainable water management alternative, including biological, chemical and physico-chemical processes (Dominguez et al., 2018). Membrane bioreactors (MBR) are promising and mature technologies in the field of greywater treatment and reuse. MBR is a membrane process (i.e., microfiltration or ultrafiltration) coupled with a biological wastewater treatment process. MBR systems have various advantages, such as process stability, small footprint, low sludge production and high effluent quality for various non-potable water uses (Kobayashi et al., 2020). In contrast, the deployment of such advanced technical solutions as MBRs generates an environmental impact associated with an intensive use of resources (e.g., chemicals and energy) and the construction of the required infrastructure (Dominguez et al., 2018). Thus, the application of this technologies has to be investigated not only by evaluation of the degradation and mineralization yield, but also by thorough environmental assessment.

Research on the literature revealed the existence of several MBR treatment technologies which have been evaluated using LCA method (Kobayashi et al., 2020), (Jeong et al., 2018), (Dominguez et al., 2018). Memon et al. (2007) investigated the MBR system among various greywater treatment technologies applied in 20 development scales. This work quantified the materials and energy required for the construction and operation phases, excluding the impacts of the system on the water transport system. Kobayashi et al. (2020) conducted an extensive comparative LCA examining two greywater treatment solutions; nature-based (constructed wetlands) and engineeredbased (MBRs), at different scales of implementation (household, neighbourhood, community) and for multiple non-potable uses (toilets, laundry, irrigation). On the other hand, Xue et al. (2016) investigated several sanitation services from a life cycle perspective, among them the on-site greywater treatment and reuse with MBR, for the environmental impacts of eutrophication potential, energy consumption and global warming potential. Lastly, Jeong et al. (2018) conducted LCA on small-scale greywater reclamation systems for on-site reuse (irrigation, toilet flushing) by using submerged MBRs, regarding ten impact categories related to the ecosystem, human health and natural resources.

#### 2.3.2. Rainwater harvesting systems

Rainwater harvesting and use is a worldwide climate adaptation strategy which provides the possibility to replace tap water in various uses, enhance the system resilience to drought events and water shortages (Xue et al., 2016), as well as cope with extreme rainfall events (Hofman and Paalman, 2014). Peak flows can be curtailed and flood risks lessened, especially in countries that are prone to flood hazards, such as the Netherlands. Ahilan et al. (2018) studied the influence of domestic rainwater harvesting design practices on water supply and storm water management efficiency, highlighting the reduction of storm water runoff volume and flood peak attenuation. The mitigation of the drinking water demands results in many other benefits arising from the reduction of chemicals and energy usage for the water purification (e.g., eutrophication, carbon footprint). According to Racoviceanu and Karney (2010), rainwater harvesting and use is collectively a great mean to enrich urban living conditions, extend water availability and add to the water efficiency scheme providing various in-house water uses. Unlike with greywater, rainwater as an external source replaces the drinking water but does not recycle the wastewater, so the final wastewater production remains the same. Kirhensteine et al. (2016) stated that the EU policy on certification to promote rainwater harvesting and reuse in buildings could achieve a 5% reduction in potable water use by 2050 but would be applicable only for major renovations or new buildings.

Design configurations and installation of rainwater systems require some technical interventions into the conventional system. The core component of the system is the rainwater tank that allows storage and/or treatment of the collected rainwater. However, limited space availability can often prevent their installation (Campisano et al., 2017). Typically, the collection surface is the building rooftop or terrace which is connected to the tank. The generated runoff after a rain event is delivered to the tank via the collection system (usually a system of gutters and downspouts) and temporarily stored in order to match demand for rainwater for the domestic uses. A separate network is used to supply the collected water to the appliances and/or taps with the use of one or more pumps to assure appropriate head pressure head. In order to ensure a reliable water supply system installation, the household plumbed rainwater tank system can incorporate mains water top-up (e.g., “trickle top-up”, “rainwater switch”) for back-up drinking water supply in case of absence of rainwater source (Umapathi et al., 2013). However, these mechanisms increase the energy requirements, and thus, the operational cost.

Depending on the purpose of rainwater usage (e.g., irrigation, shower, washing machine, flushing toilet) rainwater has to satisfy particular quality standards. The harvesting method, the components of the system configuration, the water purification system and the storage tank are key-factors of this requirement. Rainwater is usually subjected to treatment, including sterilisation and filtration, or/and first flush diversion. First flush is the initial surface runoff of a rainstorm which washes all the pollutants deposited and accumulated on the roof during the dry period, before water is allowed in the store. First flush diversion is increasingly recognised as a useful intervention to reduce both suspended and dissolved contaminate loads in rainwater systems (Martinson and Thomas, 2009). Typical pollutant materials are organic matter, inert solids, faecal deposits from animals and birds, trace amounts of metals, and even complex organic compounds which are enhanced in long dry periods (Hofman and Paalman, 2014). Although, exposure of the roof top to heat and UV radiation inactivates many bacteria, while wind removes some heavy metals accumulated from atmospheric fallout. After first flush has passed, the quality of rainwater improves considerably with almost contamination-free rainwater in terms of E.Coli, acceptable turbidity and neutral pH (Hofman and Paalman, 2014). First flush diversion is considered environmentally and economically preferable method (no energy or chemicals use) when the rainwater serves appliances that do not require such high quality water (e.g toilet, washing machine) compared to other uses (e.g., showers). Gikas and Tsihrintzis (2012) stated that the use of first flush system improves physicochemical quality, but not the sanitary quality. They stated that microbial contamination due to regrowth during storage cannot be avoided and disinfection measures should be undertaken. It was, therefore, advised this method to supply rainwater appropriate for use as greywater.

### 2.3.3. Water saving devices

Installation of water efficient appliances is considered worldwide as an effective way of managing residential water demand, save energy, reduce pollution and keep natural reserves at sustainable levels. The replacement of ordinary plumbing devices with water-efficient ones (e.g., WC, shower, basin tap, kitchen tap, bath, dishwasher and washing machine) is a regular practice in the implementation of water conservation programs in existing buildings and they have been examined by numerous of research projects.

This strategy is undoubtedly a positive step to reduce the drinking water consumption, the wastewater flows towards the WWTPs and therefore to delay the expansion and construction of new facilities. On the other hand, according to Bailey et al. (2019), the decreased water use increases the wastewater concentration of chemical oxygen demand (COD), Total Kjeldahl Nitrogen (TKN) and Total Phosphorus (TPH), limiting the ability of the sewer network to convey the nutrient-rich wastewater. However, the wastewater quality changes as the wastewater travels through the sewer network due to the dilution by rainwater and infiltrating groundwater (Bailey et al., 2019). Rapid changes in the wastewater quality can also change the treatment operations in the WWTP. Appliance-specific water demand management is seen as a way forward to reduce per capita water consumption without necessarily changing user's behaviour. Fidar et al. (2010) presented a methodology that quantifies the resources consumption and the carbon loads associated with the use of several types of water-efficient appliances and evaluated their effectiveness in contributing to compliance with the CSH's water efficiency levels and the water industry's energy efficiency programs. Clarke et al. (2009) and Racoviceanu and Karney (2010) compared the use of water-efficient devices and rainwater harvesting on a life cycle basis, highlighting the magnitude of superiority of the former regarding water saving, energy saving and GHG emissions mitigation, and concluded that the water-saving devices outperforms the rainwater harvesting system environmentally as an overall. However, Kalbusch and Ghisi (2016) pointed out the necessity for further investigation of the technical feasibility of installing such appliances and the occupancy of data on water consumption for both ordinary and water-saving plumbing devices. There is a controversy about the effectiveness of such water saving devices because lowering the fresh water flow rates by too much may lead to significant technological challenges for water utilities and infrastructure (Stavenhagen et al., 2018). The new flow rates and wastewater qualities create a need to revise the design of the water and wastewater transport networks and the pumps to avoid mechanical failures (e.g., flow velocity under the critical velocity, stagnant water, pumping failure due avoid cavitation).

#### 2.3.4. Kitchen waste grinders

Wastewater and food waste from municipal sources are the primary contributors of organic waste within the urban fabric. Food waste (or green waste) generated at consumer level covers approximately 30-60% of total municipal solid waste worldwide (Iqbal et al., 2020), (Tonini et al., 2020). The bulk of solid food waste (including dairy products and thick liquids) which are disposed of via the household waste were accounted 67.7%, in 2019, of which 30.2% were via rubbish bags/bins (Voedingscentrum, 2019). The sub-optimal management of food waste results in lost opportunities regarding environmental and socioeconomic benefits, while their mismanagement can cause severe environmental, health and social impacts (Tonini et al., 2020). Lack of landfill capacity, insufficient thermal treatment capacity and the flourishing environmental awarenes led the Dutch government to change the waste management policies, reduce the landfilling and stimulate recycling (Dijkgraaf and Gradus, 2014). Incineration has been the standard method of disposing and treating the food waste in the Netherlands since the late 1980's (Dijkgraaf and Gradus, 2014). Specifically in Amsterdam, domestic food waste end up to the WTE plant for incineration. Although, a big fraction of food waste are not suitable for incineration due to their high moisture content (Iqbal et al., 2020). Even though incineration has significant environmental benefits over the landfill (e.g., leachate, GHG emissions), there are some resource recovery opportunities which are lost via the food waste combustion. Considering the fact that resource recovery from the wastewater stream is more efficient at high nutrient concentrations, there is an option, instead of sending the food waste directly for incineration, to dispose them into the wastewater system, at a dwelling level, and retrieve valuable nutrients (phosphorus and nitrogen) and biogas, at a centralised wastewater treatment level.

Food waste valorisation for energy and nutrient recovery in the WWTP is an alternative sanitation management of domestic wastewater and food waste (Kjerstadius et al., 2017). In this design practice, food waste is separated from other urban waste flows and is embedded in the urban wastewater flow, where the nutrient-rich wastewater turns into added-value products in the WWTP. The economic dependency of some countries, such as the Netherlands, on the agricultural sector and consequently the conservation requirements for soil quality, productivity, as well as the increased cost of mineral fertilisers generate the necessity of using alternative soil amendments (Marmolejo et al., 2012). As a response to this increasing demand, this strategy of organic solid-based nutrient recycling can produce bio-fertilisers and return the nutrients back to agriculture.

Recovery of resources from wastewater would not only supply nutrients (e.g., nitrogen, phosphorus) in the form of renewable fertiliser, but also reduce the energy and chemicals demands in the production of chemical fertilisers benefiting a number of environmental impact categories such as global warming, eutrophication, and water use (Lam et al., 2020). Furthermore, anaerobic treatment of nutrient-rich wastewater can lead to increased energy recovery (biogas) (Krozer et al., 2010) which will contribute to GHG abatement (Xue et al., 2016). Relevant studies which assessed the environmental impacts of upgrading nutrient recovery-based systems (Guven et al., 2018), (Tidåker et al., 2006), (Kjerstadius et al., 2017) concluded that this is a promising technique to increase nutrient and energy recovery from urban areas, while decreasing the carbon footprint and climate impacts. Recently, the anaerobic co-digestion of sewage and food waste gained great attraction to change the WWTP into a net energy generator (Iqbal et al., 2020) and a mean to recover valuable resources. These advanced technological intervention can potentially lead to reach the targets for phosphorus, nitrogen, and energy recovery, along with reaching the Dutch national environmental objectives for climate change (van der Hoek et al., 2018b).

The global quest to achieve energy self-sufficiency and lower carbon footprint in conventional WWTPs alongside with the current interest for circular economy lead the way for food waste exploitation, and not just food waste disposal. Food waste disposers (or kitchen grinders) are small devices that grind food waste into small particles which can be flushed away via sewers. These domestic units, mounted directly under the kitchen sink and connected to the sewer pipe, are designed to grind biodegradable organics such as meat scraps, vegetables, fruit pits, citrus fruit peelings, coffee grounds and small bones (Marashlian and El-Fadel, 2005). The units are usually made from plastic (small to medium households) or stainless steel (medium to large households). The kitchen grinders, by using mechanical means and with the addition of (cold) tap water, enables the separation of a considerable fraction of food scrubs out of the entire municipal solid waste stream and allows the mixture into the sewage system for nutrient and energy recovery in the WWTP. The diversion of organic matter from the solid waste stream via sewers started as a waste minimisation option, especially in regions with landfilling of food waste.

However, according to Ng et al. (2019), there is lack of complete strategic approaches in dealing with valorisation of food waste for resource recovery. Changes in operational processes and technologies are very much dependent on external factors and actors, with legislation being a dominant one. Before water authorities can implement food waste disposers, changes in legislation and policy are required (van der Hoek et al., 2016). Food waste disposers have been in the market for many years, but authorities are still reluctant to adopt them due to poor knowledge of their impacts on the sewer system and WWTP (Iqbal et al., 2020). They were first introduced back in the 1930s in the US where their usage evolved to reach more than 94% of all cities (Marashlian and El-Fadel, 2005). Although, this use was surrounded by scepticism in certain large cities, such as New York City, which had banned them for a long time due to doubts that the city's old sewer infrastructure could handle the additional load. Today, food waste disposers are sold to households under limited or no restrictions in approximately 50 countries including England, Ireland, Italy, Spain, Japan, Canada, Mexico and Australia. The most realistic penetration rates are 25 and 50%, based on the total of 60 years of marketing garbage disposers in the US (which is considered the oldest market worldwide) which reached a maximum distribution of 50% (Marashlian and El-Fadel, 2005). The values of solid food waste grinded are varying between 75 and 95%, with the latter used in case of a limited number of food wastes could not be ground including highly fibrous wastes and shells of certain seafood (Marashlian and

El-Fadel, 2005). In the Netherlands, disposal of food waste via sewers is illegal (van der Hoek et al., 2016). However, the prospect of installing kitchen grinders in households and/or businesses and co-treating food waste with wastewater has already started being investigated as an option in the city of Amsterdam. The extra organic matter arriving at the WWTPs can be recovered using the existing technology (e.g., mesophilic digestion) or new technology (e.g., fermentation to produce bioplastic) (van der Hoek et al., 2016) after the application of certain modifications in the operation of the WWTP.

Besides the fact that food waste disposers allow diversion of organic waste materials from the residual waste stream, and hence save on their associated collection and management costs, their utilization raises numerous questions. Several studies investigated the additional energy required to run these units, the amount of additional tap water required to flush the particles into the sewage, and the induced changes in the sewer network due to the alteration of sewage quality. The water required in order to flush out the residues differs across the literature, with Marashlian and El-Fadel (2005) reporting 4.3 L/person/day, Evans et al. (2010) 0.29 (for large families) to 6.4 L/person/day (for small families), while Iqbal et al. (2020) mentions the use of 10 litres of freshwater to flush 1 kg food waste. Marashlian and El-Fadel (2005) pointed out that the cost of electricity to run food disposers and its associated pollution is relatively insignificant. Roberts and Davies (2012) investigated the required changes in the sewer network, concluding that a 2% minimum gradient in the pipework can prevent heavy materials, such as bones and egg shells, from clogging of the sewer pipelines. On the other hand, damage in the gravity sewers due to biofilm formation and sulfide production (Zan et al., 2019) and crown corrosion are some of the reported drawbacks of this strategy (Iqbal et al., 2020). Furthermore, the displacement of such significant quantity of organic material and suspended solids from the solid waste to wastewater stream is expected to radically change the WWTPs' operation.

The results from a plant-wide model revealed that the additional food scraps significantly increase the organic and nutrient load on the WWTPs, demanding extra energy for treatment. However, the carbon or COD to nutrients (nitrogen and phosphorus) ratio increases to be more favourable for increased nutrient removal efficiency and with anaerobic digestion reduces the net energy demand (Iqbal et al., 2020). As an overall, the net energy balance was increased (by 80-400%). The study also revealed improved denitrification efficiency with low N/C ratio. Iqbal et al. (2020) highlighted the potential adverse impacts which include damage to sewers (e.g. more crown corrosion), an increase in WWTP sludge production, poorer effluent quality and a less financially economic operation. Most of the potential environmental implications are subject to local conditions and to the particular WWTP's technology. However, so far, there is no reliable information of the quantification of the impacts of this perspective on the WWTPs' actual operating performance parameters, its organic load capacity and the energy and carbon footprint in the context of local conditions (Iqbal et al., 2020).

## 2.4. Nutrient recovery technologies from municipal wastewater

Reuse and recovery of resources is becoming more and more attractive and requisite, while natural resource stocks are shrinking and resource extraction activities are negatively affecting the environment (van der Hoek et al., 2016). Academics, governments and industry discovered that municipal water can be a core for recovery of valuable resources. Water, energy and nutrients constitute intrinsically linked and valuable components of human's life which can be retrieved from the municipal wastewater via innovative methods having multiple benefits. Nutrient recovery (e.g., biofertiliser production) from wastewater helps alleviate eutrophication problems (Hospido et al., 2005) and minimisation of natural reserves depletion. There is, therefore, growing belief that the focus should be to treat wastewater not as a waste but as a resource of water, energy and nutrients, while reducing the environmental impacts of the wastewater treatment process (Ashley et al., 2009). A transition of the WWTPs toward the reuse of wastewater-derived resources is recognised as a promising step to shift the standard wastewater treatment to the current emphasis on sustainability (van der Hoek et al., 2018a).

### 2.4.1. Phosphorus recovery

Phosphorus mining from phosphate rock is associated with economic, as well as environmental concerns (Amann et al., 2018). The importance of developing techniques and technologies to capture and recycle phosphorus is crucial to the future of mankind. While the global threat of climate change must be addressed, the issue of looming phosphorus shortages is exacerbated by a global population of 7.8 billion humans, 70 billion livestock (LUCASSEN, 2019) and the increasing demands for bio-fuels (e.g., biodiesel production). Phosphorus is just as important to agriculture as water. It is an essential and irreplaceable nutrient, worldwide, used in the food production chain as a major limiting factor for plant growth mainly in the form of phosphorus fertilisers. Phosphorus fertilisers are delved from phosphorus-rich deposits which are known to be finite (Smit et al., 2015).

Lack of availability and accessibility of phosphorus is an emerging problem that threatens the global population with food shortages. Research shows that the stocks of mineral phosphorus will be depleted in 50 to 100 years (De Jong, 2017). Current mining practices of phosphorus from raw phosphate rock are accompanied by various environmental impacts, including air pollution, freshwater and marine eutrophication, land degradation through phosphoric acid production and soil contamination through cadmium and uranium application by fertilisers (Amann et al., 2018), (De Ruiter, 2014). Municipal wastewater has a great potential to close the phosphorus cycle and safeguard the world's food chain. Phosphorus recovery from wastewater can substitution a great fraction of the demand for natural phosphate, while simultaneously it can alleviate the environmental burdens from current practices of phosphorus use (Amann et al., 2018). Even though phosphorus recovery from municipal wastewater could decrease the dependency of the countries on the global phosphate rock market, this can conceivably lead to an increase in environmental impacts from fertiliser production.

Despite the fact that phosphorus removal at the WWTPs for cleaner estuaries is one of the few environmental success stories, recycling of phosphorus for agricultural use rises many concerns. In many cases around the world, due to the restrictions for application of biosolids on land, most of the phosphorus is dispersed in the environment and thus lost to future generations (Ashley et al., 2009). Intensive research in recent years has led to the development of a broad spectrum of innovative technologies for phosphorus recovery from wastewater. There are several technologies which can recover phosphorus from the liquid phase (digester supernatant, dissolved P in anaerobic sludge, effluent), sewage sludge and sewage sludge ash (Amann et al., 2018). These technologies have mainly assessed by technical and economic perspective by numerous of comparative studies to identify the most applicable and market-feasible options as an alternative to phosphate mined and chemical fertilisers (Amann et al., 2018). Nevertheless, knowledge regarding the environmental benefit and burdens of this technologies arising from the equilibrium between the resources use, the environmental emissions and the resource recovery is prerequisite.

Environmental assessment of phosphorus recovery is important to eliminate any possibility of replacing one problem (e.g., land degradation and eutrophication potential from phosphate rock mining) with another (e.g., high energy demand and air emissions from the recovery technologies). Amann et al. (2018) thoroughly evaluated many of these technologies in terms of cumulative energy demand, global warming potential and acidification potential with the methodology of LCA. Based on their findings, phosphorus recovery from the liquid phase showed mostly positive or comparably little impacts on emissions and energy demand, but low recovery potential. On the other hand, technologies for recovery from sewage sludge that already or are close to full-scale application were associated with comparatively high emissions and energy demand. Recovery from sewage sludge ash showed varying results, partly revealing trade-offs between heavy metal decontamination, emissions and energy demand. Nevertheless, recovery from ash was correlated with the highest potential for an efficient recycling of phosphorus.

A prerequisite for phosphorus recovery from sludge ashes is that the sludge has to be incinerated separately from other waste, the so-called mono-incineration (Fooij, 2015). EcoPhos is a technology which uses fly ashes from sludge incineration to produce fertilisers. In order to recover phosphorus from sewage sludge ash the phosphorus has to become released, firstly, from the solids. This leaching is carried out by the means of hydrochloric acid (HCl) (Ashley et al., 2009). HCl dissolves the ashes and phosphorus ore but also iron and other metals, enabling the recovery of phosphoric acid, phosphate salts and other salts like iron chloride van der Hoek et al. (2016). Phosphoric acid is a high added-value product which is used as an intermediate in the fertiliser industry, for metal surface treatment in the metallurgical industry and as an additive in the food industry (De Ruiter, 2014). Amann et al. (2018), who thoroughly assessed this technical option, stated that through the utilisation of ion exchangers, EcoPhos can achieve a good heavy metal removal and the phosphoric acid can further be used in the fertiliser industry. They also concluded that EcoPhos can save considerable amounts of energy and improve the overall global warming potential if additional products, such as $CaCl_2$ (de-icing agent) and AlCl3 (aluminium alloy) can be retrieved. However, this method carries some negative environmental impacts related to the use of many chemicals van der Hoek et al. (2016). Figure 2.2 demonstrates the input and output of the EcoPhos technology.

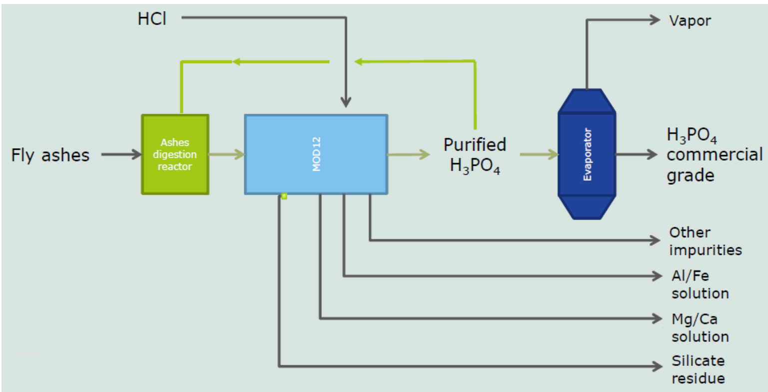



**Fig. 2.2:** EcoPhos technology (De Ruiter, 2014)

### 2.4.2. Nitrogen recovery

Nitrogen is abundantly presented in the atmosphere (almost 80%) in the highly stable and non-reactive form of $N_2$ gas (van der Hoek et al., 2018a). Nitrogen in its reactive forms (ammonium, nitrite, and nitrate) is a critical limiting element for plant growth and production, the content of which is limited in soils. Most naturally occurring reactive nitrogen comes from lightning (2%) and biological fixation (98%) (Sengupta et al., 2015). Natural fixing of nitrogen is insufficient to meet the food and energy demand of the increasing world population, and thus anthropogenic production of reactive nitrogen have started the last few decades (Sengupta et al., 2015). The importance of developing technologies and processes to capture and recycle nitrogen is critical to the future of mankind and the environment. The Haber–Bosch, which was invented in 1909, is a commonly used process for industrial fixation of $N_2$ into ammonia for the production of N-based fertilisers (van der Hoek et al., 2018a). This process managed to more than quadrupled the productivity of agricultural crops (van der Hoek et al., 2018a). However, the intensive food production results in massive discharges of wastewater effluent containing excess nitrogen which are excreted by human metabolism as urea and NH4. The consequent algae growth in receiving water bodies lead to eutrophication problems.

In order to secure the quality of receiving water streams, urea, free ammonia and complex nitrogen compounds are converted to free ammonia compounds in solution by the biological wastewater treatment processes (Ashley et al., 2009). The conventional nitrification-denitrification process converts $NH_4^+$ to $NO_2^-$ and $NO_3^-$ under aerobic conditions (nitrification) and afterwards $NO_3^-$ to $N_2$ under anoxic conditions (denitrification) (Ashley et al., 2009) which is released to the atmosphere. On the other hand, the Anammox ("anaerobic ammonium oxidation") reaction directly converts the $NH_4^+$ and $NO_2^-$ into $N_2$ and water. Although, these techniques are characterised by several environmental and technological limitations. According to van der Hoek et al. (2018a), the first issue is that nitrogen which enters the WWTP is mainly converted to $N_2$ gas and lost to the atmosphere rather than reused. Secondly, both N-dissipation for wastewater treatment and N-fixation for fertilisers production are quite energy intensive techniques. Thirdly, during the biological removal of nitrogen from the wastewater, nitrous oxide ($N_2O$) is released which often exceeds the carbon emissions related to the electricity consumption for the process requirements of WWTPs. The $N_2O$ emissions have a share of 14-26% of the total carbon footprint of a WWTP (Cruz et al., 2019) and 26% of the GHG footprint in the whole water chain (van der Hoek et al., 2018a). According to Cruz et al. (2019), the theoretical energy embedded in ammonium in domestic wastewater represents roughly 38-48% of the embedded chemical energy available in the whole of the discharged bodily waste. The mainstream options for ammonium removal neglect the energy embedded in ammonium. Considering these limitations, it is therefore relevant to examine and develop more sustainable pathways for ammonium management that aim at recovering of nitrogen in the form of added-value products, rather than just removing it from the water stream.

In WWTPs, sludge dewatering processes usually produces reject water with high ammonium content. The reject water is recycled back to the biological process and imposes a high N-load to the plant (Wu and Modin, 2013). A Germany-wide survey revealed average return loads under normal conditions in the range of 10 to 15% of the inflow N-load (Jardin et al., 2006), while in Amsterdam West WWTP the digester reject water contains around 26.6% of the inlet nitrogen (van der Hoek et al., 2018a). Since the reject water stream has a relatively high concentration of ammonium, it might be suitable for nitrogen recovery. Air stripping coupled with absorption is a promising technology to substantially ease the N-loads from the main wastewater treatment process (Jardin et al., 2006), and recover ammonia from the liquid fraction of the anaerobic digestate (Ashley et al., 2009). Air stripping is the process where, by applying air, ammonia is removed from wastewater into the gas phase (van der Hoek et al., 2018a). The ammonium is transferred to the air stream where it is converted to slightly volatile gaseous ammonia that is readily soluble in water (desorption) (Jardin et al., 2006). Although, a balanced ammonium-to-ammonia ($NH_4^+/NH_3^+$) ratio is a function of temperature and pH. According to Jardin et al. (2006) and Ashley et al. (2009), a pH of 10 and a temperature of 70°C-80°C will completely shift the dissociation equilibrium towards ammonia. That means that there will be no ammonium in the water phase anymore. For ecological reasons, the ammonia stripped into the gas phase should not be directly released to the atmosphere, but has to be converted to a recyclable or disposable product. The ammonia is therefore led to an absorber which contains acid, typically sulphuric ($H_2SO_4$) or nitric ($HNO_3$) acid, where the ammonia dissolves and ammonium salts are formed (van der Hoek et al., 2018a). The outlet concentration has reported by Cruz et al. (2019) to be 500-1000 $NH_4$-N mg/L, which is in line with (Wu and Modin, 2013) (1000 $NH_4$-N mg/L). One of the challenges that air stripping can face is fouling of the stripping towers (Cruz et al., 2019).

A thorough research for nitrogen recovery techniques in Amsterdam's wastewater revealed that air stripping from digester reject water has high applicability without interfering to other possible recovery activities, such as biogas production, phosphorus recovery and cellulose recovery (van der Hoek et al., 2018a). The study concluded that among hydrophobic membranes, vacuum membranes and urine treatment processes, air stripping showed the second best performance (24% N-removal) after source separated urine (60% N-removal). Urine collection, even if it has been proved from most of literature findings as the most efficient nitrogen recovery method, requires a completely new infrastructure for wastewater collection and treatment and this is why it is not investigated as nitrogen recovery technique in the current project. Air stripping from digester reject water has the potential of removing 24% of the total nitrogen load with a 90% recovery efficiency, and therefore reduce of $N_2O$ emissions. van der Hoek et al. (2018a) revealed energy requirements of 90 MJ/kg-N removed, amount which is proportionate to the total primary energy requirement for N-fixation (Haber–Bosch process) and N-removal by nitrification/denitrification process (90 MJ/kg N). To be cost-effective and practically feasible, existing recovery methods typically require concentrations above 2-3 g/L NH4-N, which is well above the dilute concentration of ammonium in domestic wastewater (40-60 mg/L NH4-N) (Cruz et al., 2019). In Amterdam West WWTP the inflow nitrogen concentration calculated approximately 80 mg/L. In relation to this, van der Hoek et al. (2018a) concluded that air stripping in the digester reject water is not directly applicable and other alternatives that enhance nitrogen concentration in the digester reject water have to be applied first to ensure maximum efficiency and reduce risks. The research revealed that pre-treatment of sludge by applying thermal hydrolysis process (THP) to increase the nitrogen content, and subsequent application of air stripping in the digester reject water can provide high potential for nitrogen recovery from the return water after dewatering.

## 2.5. Thermal hydrolysis of sludge as a nitrogen content enrichment method

Wastewater sludge management in the Netherlands is controlled by strict regulations related to land application. Sludge digestion is a widespread technology which is used to minimise odour, volume, disposal costs, improve biological activity of the sludge and recover energy as biogas. Primary sludge (PS) is easily biodegradable and results in a volatile solids (VS) removal of 50 to 60% and high biogas generation, in contrast with waste activated sludge (WAS) that due to low loading rates results in low biodegradability in sludge digesters with typical VS removal rates from 25 to 50% and relatively low biogas yield (Oosterhuis et al., 2014). WAS is formed at a consistent rate during the treatment of wastewater, representing around 30% w/w of the COD-load of WWTPs (Nagler et al., 2018). WAS requires pre-treatment to enhance methane-yields and reduce digester volume and sludge waste disposal cost. Pre-treating solid particles in sludge is one of the widest studied processes, including methods of biological, thermal, microwave, mechanical, enzymatic, ultrasonic, pulsed electric fields, freeze/thaw, chemical and wet oxidation (Nagler et al., 2018), (Phothilangka et al., 2008). Among all these strategies, thermal and chemical pre-treatment seem to be the most efficient in terms of full-scale applicability, efficiency and economic profit.

In general sludge disintegration technologies aim to accelerate and enhance degradation of organic matter. Faster anaerobic degradation rates saves additional digester volume and higher bioavailability of organics allows higher biogas generation and less residual solids production. Anaerobic sludge digestion combining with THP of the WAS is a well proven method to remarkably affect the anaerobic biodegradability, viscosity, energy recovery and sludge dewaterability. The first full-scale operation of sludge thermal hydrolysis date from 1995 in Norway, Denmark and the UK (Oosterhuis et al., 2014). The commercial processes of Porteous and Cambi™ are two examples of thermal hydrolysis concepts which have been implemented on many sites worldwide in the past. Across the world there are 23 full scale THP sites either in operation or under construction (Mills et al., 2014). A few water boards in the Netherlands have already decided to implement this technology, and some water boards are considering implementation, having as a reference several pilot-projects in WWTPs across the country (Oosterhuis et al., 2014). The environmental performance and the implementation feasibility of THP depends on local aspects, with the impacts varying between countries due to the differences in wastewater quality, sludge characteristics, design standards, system configurations and effluent requirements (Oosterhuis et al., 2014). Consequently, systematic tools (e.g., mathematical model approaches) need to be developed for a more generic process description (Phothilangka et al., 2008).

During THP, WAS is first hydrolysed at a temperature of 160 to 180°C (Nagler et al., 2018), (Li et al., 2017), (Mills et al., 2014) and then cooled at 35 to 40°C by heat transfer (Li et al., 2017). The heat transferred from hydrolysed sludge is used to warm the digesters. The process needs both grid electricity and natural gas, according to Li et al. (2017), and therefore the emissions of NOx and SO2 add to the environmental burdens of the overall performance. THP uses a high temperature and pressure (7 bar) for 30 min to disrupt and solubilise sludge before feeding it to a conventional digester. The process also homogenises the sludge so that it is more digestible resulting in increased methane production and a smaller volume of digestate (Mills et al., 2014). Phothilangka et al. (2008) introduced an innovative thermo-pressure hydrolysis process which has been tested in full-scale at Zirl WWTP in Austria. Sludge is treated under pressure of 19–21 bar at a temperature of 180°C. In contrast to other established technologies, thermo-pressure hydrolysis represents a continuously operated system involving high pressure pump, controlled pressure release valve and heat exchangers. The process produces a sludge which is partially solubilised and the biological cells are disintegrated. The form of organic material allows more efficient anaerobic digestion.

Implementation of full scale THP provides higher anaerobic degradation efficiency with subsequent increase of specific biogas production by 10% to 60% across different literature (Li et al., 2017). Respectively, Phothilangka et al. (2008) revealed a 75% increase of biogas production after digestion of pre-treated WAS, which is in line with prior studies on the thermal hydrolysis impact under similar operating conditions at 180°C. However, total biogas yield increases only 20-40% when both streams of untreated PS and THP-treated WAS are mixed, in several ratio, in order to simulate the full-scale operation mode of the digesters (Oosterhuis et al., 2014), (Phothilangka et al., 2008). Particularly for the case of the Netherlands, Oosterhuis et al. (2014) reported that despite the fact that the required energy for THP process (steam production) can have a negative impact on the total energy balance of sludge treatment, when a mixture of 40% PS and 60% WAS is digested the amount of heat and electricity generated is sufficient to produce steam for thermal hydrolysis of WAS alone.

A research regarding the THP of WAS in a pilot plant in the Netherlands, in line with many other international studies, revealed that the digestion of mixed untreated PS and THP-treated WAS compared to a conventional system without WAS pre-treatment, shows great advantages. Heating up of WAS results in solubilisation of COD and consequently higher VS removal rates in mesophilic sludge digesters. Results from pilot experiments in the Netherlands showed 62% increase of VS removal at Hengelo WWTP, 55% at Venlo WWTP (at 140°C) and 38% at Amersfoort WWTP (at 140°C) (Oosterhuis et al., 2014). Thermal hydrolysis also decreases the viscosity of sludge allowing the supply of digesters with high VS-content sludge. Enhanced degradation of organic matter and digestion performance, is a result of the increased sludge loading rate, which according to Oosterhuis et al. (2014) is around 2.3 times higher compared to conventional sludge digesters for the Netherlands case. As a result, cake's solids content improves (25% reduction in sludge disposal costs) (Phothilangka et al., 2008), and therefore GHG emissions from transport. Most of the studies revealed increase in the overall biogas generation, which is used as heat input for the THP contributing to smart recycling of thermal energy, saving external resources and approaching $CO_2$emission goals. Following the sludge treatment pathway, the THP boosts the dewaterability as thermal sludge treatment induces a higher breakdown of the cell structure of sludge and the release of intracellular bound water (Phothilangka et al., 2008). An additional advantage of the process is that the sludge is sterilised, which makes land application possible as fertiliser in the form of biosolids (class A) in several countries Oosterhuis et al. (2014). Despite the fact that Dutch legislation does not allow the application of sterilised sludge as a fertiliser in agriculture, all the above-mentioned benefits make the application of thermal hydrolysis of WAS an attractive option.

On the other hand, THP is characterised by interconnected side-effects, which have to be taken into account and managed properly towards the benefit of the WWTP. Thermal disintegration improves biological accessibility of compounds and more nitrogen gets released in form of ammonium nitrogen (NH4-N) by degradation of N-containing organic matter. The consequent substantial load of ammonia contributes to an increase of about 40% of the N return load, according to Phothilangka et al. (2008). Alongside, soluble inert compounds Si are produced when WAS is pre-treated at such high temperatures. However, these soluble organics are highly stabilised and do not cause depletion in rivers. It was also highlighted by the authors that operating temperature should not exceed the optimum range of 170 to 180°C to prevent any formation of unwanted by products. Meanwhile, pre-treatment of sludge with high temperatures above 150°C can also be toxic for the anaerobic digestion culture and it should be carefully considered.

Concluding, the THP process is environmentally feasible only if high-organic-content sludge is used as the treatment object (Li et al., 2017). Both Phothilangka et al. (2008) and Oosterhuis et al. (2014) mention that the high-strength sludge liquor with high ammonia return loads is ready for an efficient side-stream nitrogen removal. Consequently, the combination of thermal hydrolysis of WAS with air-stripping from the digester reject water can convert the drawbacks of THP (increased N-load) into benefits (nitrogen recovery) and shift the WWTP into a valuable source of nitrogen and energy, as suggested by (van der Hoek et al., 2018a).

# 3. Description of the case study

## 3.1. Thesis motivation

Amsterdam is a dynamic environment with an increasing number of tourists and inhabitants (Pinkster and Boterman, 2017). The city is expected to experience an increase of 90.000 inhabitants by 2035, according to the statistics (UN, 2020). The growing and increasingly affluent human population has high requirements for water, food and energy (D'Odorico et al., 2018) with an emergent competition among them. Amsterdam, as a sustainability-oriented community, has to cope with these problems and determine innovative and long-lasting ways to satisfy the growing demands adopting practices that maintain values of materials and products within a sustainable circular economy (Velenturf and Jopson, 2019).

The "New Urban Water Transport Systems" (NUWTS) project has been established to investigate new urban water transport systems and technologies as enablers for resource recovery. Regional water authorities, drinking water companies, municipalities and engineering consultants contribute in this initiative to promoting reliable and sustainable urban water systems which could function as a template design in every developed urban environment with similar characteristics. However, this design challenge needs to be explored in terms of environment, cost and human health. This MSc thesis, as part of this research initiative, aims to provide the data needed to ensure development of only promising technologies are pursued before lock-in occurs.

## 3.2. Water and sanitation services in Amsterdam

### 3.2.1. Overview of the urban water system

#### 3.2.1.1 Drinking water supply

Waternet is responsible for the drinking water supply in and around Amsterdam, managing two water treatment plants (WTPs). The Leiduin WTP is located in Amsterdam's Southwest and the Weesperkarspel WTP lies to the Southeast (Barrios et al., 2004). Both plants produce 57.2 million $m^3$ drinking water for distribution in the city (van der Hoek et al., 2016). This project focuses on the latter, where raw water is firstly subjected to pre-treatment at Loenderveen pre-treatment plant and then distributed to Weesperkarspel for the main treatment process. As stated by Barrios et al. (2004) and Roest et al. (2016), the drinking water from Weesperkarspel used to be approximately 28 million $m^3$, while the up-to-date Waternet's database recorded 24.4 million $m^3$ for 2019. The 68% of this water is distributed to households, 28.5% to businesses (e.g., offices, hotels, restaurants, industry), while the 3.6% is lost as leakage from the distribution network (van der Hoek et al., 2016). Such low leakage rates are typical in the Netherlands and are the lowest among European capitals (Vanham et al., 2016).

Having a closer look at the water treatment process, the abstracted raw water, which is a mixture of seepage water from Bethune polder and water from Amsterdam-Rhine canal (Barrios et al., 2004),(Bosklopper et al., 2004), is transported to Loenderveen plant. A series of treatment activities including coagulation, sedimentation, self-purification in a lake water reservoir and rapid sand filtration take place (Sobhan, 2019). Until 2016, HCl was added prior to transportation to Weesperkarspel plant in order to avoid scaling in the pipelines (Chiou, 2018). However, this practice has recently eliminated and no conditioning is applied anymore. The pre-treated water is then pumped and transported over a 14-kilometer pipeline to Weesperkarspel (Sobhan, 2019). Figure 3.1 shows the treatment process at Loenderveen WTP. At Weesperkarspel WTP, the purification of water is achieved by a series of further treatment processes including ozonation, softening, activated carbon filtration and slow sand filtration, which are depicted in Figure 3.2.

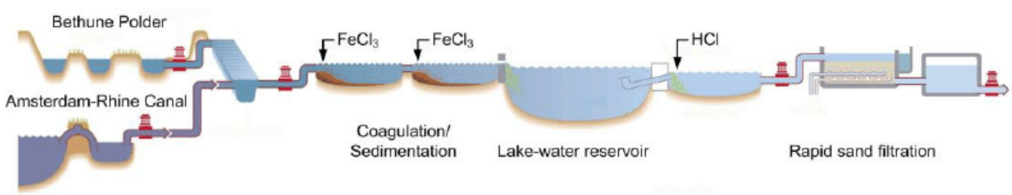



**Fig. 3.1:** Pre-treatment scheme at Loenderveen (Sobhan, 2019)

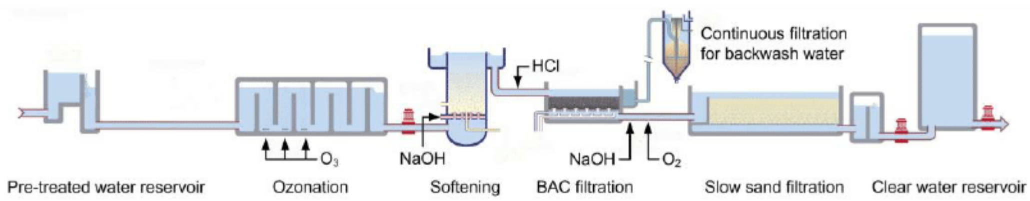



**Fig. 3.2:** Treatment scheme at Weesperkarspel (Sobhan, 2019)

However, in the last few years, Waternet made several process modifications in the WTP as a step towards sustainability. Based on data retrieved by the company's database, since 2018, HCl dosing for conditioning in softening process has fully replaced with $CO_2$dosing. Part of the $CO_2$comes from the conversion process of biogas produced in the WWTPs of Amsterdam to Green gas. Biogas consists of 55-65% methane and 35-45% $CO_2$. Before biogas can be injected into the natural gas network, it is upgraded in such a way that it consists of at least 88% methane. According to van der Hoek et al. (2017), during the production of Green gas from biogas, the $CH_4$ content is increased to 88% with an efficiency of 95% (0.7 $m^3$ of Green gas produced from 1 $m^3$ of biogas). The remainder is $CO_2$. Part of this biogas produced in Amsterdam (10500 $Nm^3$/day) is converted to Green gas as fuel for cars (70%) and $CO_2$(30%), while the rest 25000 $Nm^3$/day is burned at Amsterdam West WWTP for its operations. Particularly, 7350 $Nm^3$/day Green Gas and 3150 $Nm^3$/day $CO_2$ are produced, which with a density of 1.986 kg/$m^3$ results in a $CO_2$ production of 2283 tons of $CO_2$/year (van der Hoek et al., 2017).

Another significant modification in the WTP was the recycling of the pellets produced in softening process. In order to avoid the use of imported garnet sand for softening and enhance the reuse of the full grown pellets produced and managed as waste materials since then, a closed loop in the process was introduced (van der Hoek et al., 2017). Garnet sand, which used to come by boat from Australia (high transportation cost), got replaced partly with pellets produced from the pellet softening process itself (12% of the output pellets (van der Hoek et al., 2017)) and calcite pellets coming from North Holland. The remaining 88% of pellets produced has already been studied to be used in road construction or in the WTE plant for flue gas treatment (van der Hoek et al., 2017). Last but not least modification in the water purification process was the shift of the water quality requirements to a more conservative side by setting the goal of total hardness from 1.5 to 1.4 mmol/L.

#### 3.2.1.2 Domestic water end-use

Final destinations of the drinking water are residential, commercial and industrial areas of Amsterdam. From the total drinking water distributed to households, 2.6% evaporates and is lost to the atmosphere (van der Hoek et al., 2016). It is fact that the Netherlands has decreased continuously the domestic water use over the last two decades (Vanham et al., 2016). More specifically, during the period of 2003 to 2017, household water usage per inhabitant fell by over 9% (CBS Netherlands, 2020). This decline is mainly due to the shift into more efficient appliances (e.g., water-efficient washing machines, dishwashers, toilets with low flush system) and raising public awareness. However, low gas prices and national campaigns to promote hygienic practices, led to changes in routines by increasing showering frequency (Agudelo-Vera et al., 2014b) and duration (Vanham et al., 2016). Dwellings in Amsterdam use 133.8 L of drinking water per person daily, whilst the total water use in Prinseneiland neighbourhood is differentiated at 112.7 L based on measurements in the area (Bailey et al., 2020) (see Table 3.1).

**Table 3.1:** Appliance-specific water use in Amsterdam and in Prinseneiland (Bailey et al., 2020)

| **Average water use (L/cap/d)** | | |
|---|---|---|
| **appliances** | **Amsterdam** | **Prinseneiland** |
| bath | 1.0 | 0.8 |
| bathroom tap | 5.8 | 4.9 |
| dishwasher | 1.4 | 1.2 |
| kitchen tap | 9.6 | 8.1 |
| shower | 62.7 | 52.9 |
| toilet | 35.3 | 29.7 |
| washing machine | 12.3 | 10.3 |
| outside tap | 5.7 | 4.8 |
| **total** | **133.8** | **112.7** |

#### 3.2.1.3 Wastewater collection and treatment

After water use and addition of several materials and chemicals (i.e., human waste, food scraps, oils, soaps and chemicals) wastewater returns to the water chain and coupled with infiltrated ground water and storm water is transported via sewers towards Amsterdam West WWTP. The infiltration rate in Amsterdam is approximately 9.9% (van der Hoek et al., 2016). Amsterdam West is the biggest WWTP out of the 12 operated by Waternet (van der Hoek et al., 2016), (van der Hoek et al., 2017). The WWTP, which started operating at 2005, has a treatment capacity of almost 1 million (1.014.000) people equivalent (PE) (van der Hoek et al., 2018a) and 168.000 $m^3$/d influent water at dry-weather flow (Zhou et al., 2019). The current wastewater flow that reaches the WWTP is estimated around 74.9 million $m^3$/year (van der Hoek et al., 2016). The sewer system is a looped and partly combined network (i.e., storm water and wastewater) [1].

Focusing on the WWTP's operation, after primary treatment, the wastewater is subjected to a series of biological treatment processes. The design is based on the modified University Cape Town (mUCT) process with biological phosphorus and nitrogen removal (van der Hoek et al., 2018a). After the secondary settling, the WAS and the PS are introduced into a mixed storage tank. The plant does not only process

[1] The simulation of the hydraulic performance in Prinseneiland was conducted for dry weather flow data (Bailey et al., 2020).

the sludge from its own treatment, but also sludge coming from the other WWTPs of Amsterdam[2]. The external sludge is introduced in the storage tank where is co-treated with the PS and WAS from its own wastewater.

During the sludge management, the sludge is firstly thickened and after digested in a mesophilic digester where biogas is produced. Taking into account all 12 WWTPs operated by Waternet, the (maximum) biogas production is 35500 $Nm^3$/day, which contains 65% $CH_4$ and 35% $CO_2$(van der Hoek et al., 2017). The Amsterdam West WWTP produces the 25000 $Nm^3$/day of this total amount which are used to generate energy (CHP) to satisfy its own needs. In order to eliminate the extensive scaling problems used to occur after digestion process, a struvite recovery installation (AirPrex technology) was introduced in 2014, making Amsterdam West one of the Europe's largest struvite production facilities (Zhou et al., 2019). The remaining sludge is dewatered, the digester reject water returns back to the biological treatment and sludge is transported to the WTE plant (AEB) for incineration. Figure 3.3 illustrates the process configuration of Amsterdam West WWTP.

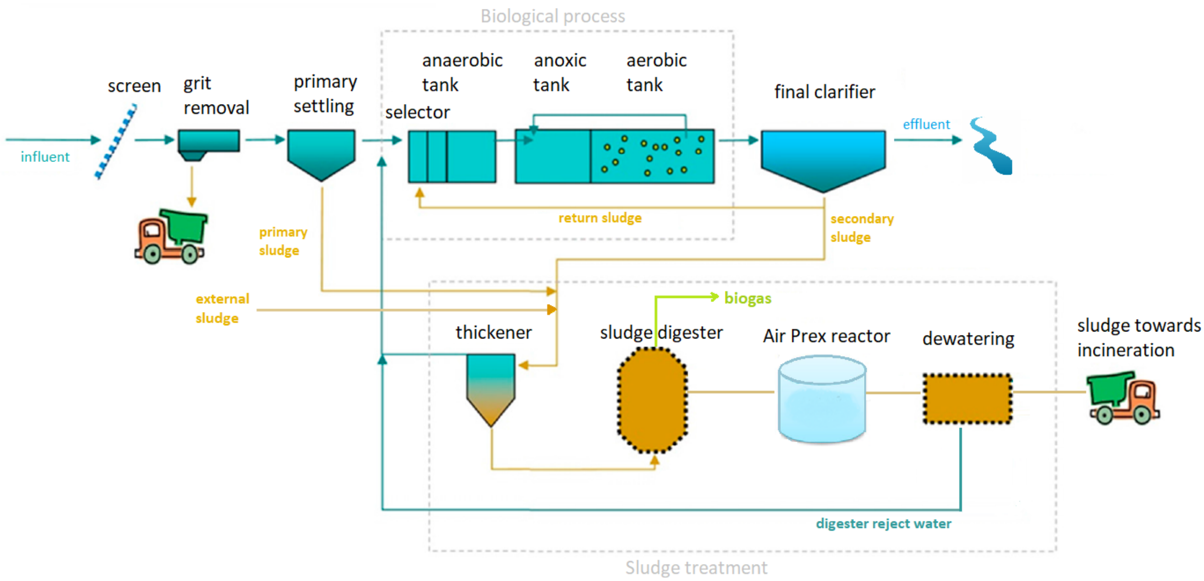



**Fig. 3.3:** Amsterdam West plant treatment process (personal modification, van der Hoek et al. (2018a))

[2]External sludge is coming from Amstelveen, Blaricum, Hilversum, Horstermeer, Huizen, Loenen, Maarseen, De Ronde Venen, Uithoorn WWTPs (Fooij, 2015).

Diving into the phosphorus management technologies, Amsterdam West plant follows a struvite precipitation process in order to eliminate the massive scaling problems and blockades of buffers and pipes after digestion of the sludge. The AirPrex process is based on the addition of magnesium chloride ($MgCl_2$) to digested sludge, during aeration with compressed air, leading to struvite (magnesium ammonium phosphate (MAP) ($NH_4MgPO_4$:6($H_20$))) formation which precipitates and is removed from sludge (van der Hoek et al., 2017). Struvite has a high commercial value as fertiliser. In Amsterdam West WWTP struvite contains approximately 14.36% of the influent phosphorus (van der Hoek et al., 2016). The WWTP produced approximately 1000 tons struvite annually, having a struvite harvesting efficiency of approximately 21% with regard to the dissolved ortho-phosphorus in the digester sludge (Zhou et al., 2019). However, the biggest fraction of phosphorus escapes from the system via the sewage sludge waste. Digested sludge waste has an excessive P-concentration of about 75.69% of the total inflow phosphorus (van der Hoek et al., 2016).

Nitrogen is also recovered through the struvite precipitation process but in very small percentage because the main focus of this process is phosphorus recovery. Particularly, around 0.2% nitrogen of the total influent in Amsterdam West plant is recovered through struvite based on the nitrogen cycle (van der Hoek et al., 2018a).

#### 3.2.2. Water-related energy use

Natural gas contributes to about one-third of the country's energy mix (Gerbens-Leenes, 2016). In the country, 38% of energy consumption goes to heating with half of this is used by residential buildings. Almost 90% of Dutch houses have a gas-fired boiler. The Netherlands wants to remove gas as source of heating for the residential units, provide renewable energy to cover the energy demands and improve the insulation of the houses to reduce part of it. Government wants to reduce $CO_2$emissions from the built environment by 80%-95% by 2050 (Ministry of Economic Affairs and Climate, 2016). The current national electricity mix uses 78.7% fossil fuels (51% natural gas, 26.5% coal, 1.1% oil), 3.1% nuclear and 18.3% renewable (2,1% bio-fuels, 3.7% waste, 0.1% hydropower, 9.2% wind, 3.2% solar) (IEA, 2018).

Significant quantities of energy are used to withdraw the raw water, treat it to potable quality, distribute it to the end-users, as well as collect and treat the wastewater produced. A number of recent studies confirmed that the biggest share of the water-related energy use is attributed to the domestic water end-use (Fidar et al., 2010). The municipal water system in the Netherlands requires approximately 10.2 GJ per capita per year, dominated by direct domestic energy consumption for water heating with a share of 92% (Gerbens-Leenes, 2016). This finding is in line with other studies which reveal fractions of 84-97% for water heating requirements in different case study areas around the world (Kenway, 2013). Households are major stakeholders of water-related energy use and their respective GHG emissions. In the Dutch households, the top three energy consumers are showers (58%), washing machines (9%) and dishwashers (8%) (Gerbens-Leenes, 2016). Figure 3.4 indicates thoroughly the shares of the water-related energy use within a Dutch household.The rest energy requirement, apart from water heating, is attributed to water supply (6%) and wastewater treatment (2%) (Gerbens-Leenes, 2016). Particularly, the energy consumption of water purification is approximately 1.1 $MJ/m^3$ (data for Loenderveen and Weesperkarspel WTPs (Roest et al., 2016), (Barrios et al., 2008)), 0.4 $MJ/m^3$ for water distribution and 4.3 $MJ/m^3$ for other services (UV discinfection, membrane filtration, flotation, aeration) (Gerbens-Leenes, 2016). The energy dedicated to wastewater transport and treatment services is 1.05 $MJ/m^3$ and 3.35 $MJ/m^3$ respectively (Gerbens-Leenes, 2016).


**Fig. 3.4:** Shares of water-related energy requirements of a Dutch household (Gerbens-Leenes, 2016)

The drinking water company of Amsterdam uses approximately a total of 105 GWh per year to satisfy the energy requirements to manage the entire water cycle in Amsterdam. The 10 GWh is generated by its own solar installations (18 GWh from the end of 2020), the 20 GWh is coming from its own biogas generation from wastewater, while the rest is purchased from the grid and turned by the company into green energy (50% renewable energy and 50% natural gas) with Guarantee of Origin (GO) (personal communication, Ruijs Freya (Waternet, 2020)). The origin of energy strongly determines the total environmental impacts. For a sustainable solution, it is of great importance to select green energy sources if possible, otherwise technologies with minimal energy usage (see Appendix B for information regarding the transition towards green energy).

#### 3.2.3. Overview of the sewage sludge waste management

To operationalise the transition from linear to circular flows of energy, water and resources in practice, public and private parties have to seek possibilities to implement technological and system innovations from a shared vision and in partnership (Government of the Netherlands, 2016). Within the Urban Harvesting Concept in Amsterdam, the water company (Waternet) and the WTE company of Amsterdam (AEB) collaborate the last few years on innovation projects in the field of water, energy, waste and material flows. Particularly, the agreement sets AEB responsible to manage the biogas and the sewage sludge waste that are produced in Amsterdam West WWTP for energy recovery. The WWTP uses the energy recovered from biogas for its own needs, while the WTE company uses the energy recovered from sludge incineration to provide district heating. The neighboring location of the WWTP and the WTE plant add value to this synergy by increasing energy efficiency from biogas (from 50% to over 90% (Simoës and Veldman, 2007); only 3% being lost as a gas flare (van der Hoek et al., 2016)) and eliminating the transportation cost of the bulky sludge waste.

Amstedam West, WWTP with a design capacity of 1 million PE, generates approximately 100000 ton stabilised and dewatered sludge per year, with DM content of approximately 26% (Fooij, 2015). The stabilised sludge requires additional drying to reach a certain amount of DM content (>=90% DM) for energy recovery with incineration. The annual quantity of sewage sludge waste and biogas produced in the WWTP is 26000 ton DM and 9.125 million $m^3$, respectively. Biogas, apart from methane and carbon dioxide, it contains water vapour, ammonia, hydrogen sulphide and other corrosive trace elements (siloxanes) which are present in smaller concentrations and are removed from biogas before its use in CHP units (Đurđević et al., 2019). The specific annual quantities of sewage sludge and biogas yield in the WWTP were calculated 26 kg DM/PE and 9.125 $m^3$/PE, on average, while in the EU the specific quantities vary between 20–35 kg DM/PE and 6.6–9.5 $m^3$/PE, respectively (Đurđević et al., 2019). The heating value of the dried sludge is approximately 8 MJ/kg DM (personal communication, David van Diepen, AEB) and for biogas 23 MJ/$m^3$ for 65% methane and 35% $CO_2$content (Center, 2012).

Biogas engines are marketed with capacities in the range between 10 kWel and 5 MWel (Đurđević et al., 2019). AEB achieves overall efficiency of 94%, with electrical efficiency being 40%, thermal efficiency 38% and heat 16%. Taking into account that the average biogas heating value is 23 MJ/m$^3$ (6.4 kWh/m$^3$), CHP unit generates 2.56 kWhel/m$^3$ of electricity and 3.46 kWhth/m$^3$ of heat per unit of biogas. Consequently, the amount of electricity generated from biogas is approximately 2.56 kWhel/m$^3$ biogas and 3.46 kWhth/m$^3$ biogas, with a total energy generation of 54896 MWh/year. According to van der Hoek et al. (2017), part of the energy retrieved from biogas (total 33889 MWh; 20000 MWh electricity and 50000 GJ heat) is provided to Amsterdam West WWTP to cover its operational needs resulting in 1.8 million m$^3$/year avoided natural gas and consequently 3200 ton $CO_2$-eq/year avoided GHG emissions.

The average energy consumption of Amsterdam West WWTP is 0.93 kWh/m$^3$ (3.35 MJ/m$^3$), with major electricity consumers in WWTPs are aeration, pumping, sludge thickening, and dewatering (Gerbens-Leenes, 2016). Furthermore, around 61% of this energy is electricity (0.57 kWhe/m$^3$) and 39% thermal energy (0.36 kWhth/m$^3$) Đurđević et al. (2019). Thus, taking into account the average amount of 63.6 million m$^3$ inflow wastewater in the WWTP (van der Hoek et al., 2018a), the average energy requirement of the plant are 59 kWh/PE. The annual required energy of the WWTP is approximately 59000 MWh. Consequently, biogas provides 57.4% energy autonomy in the WWTP, which is in line with the findings of a similar WWTP in case study in Croatia (Đurđević et al., 2019).

Energy recovery from sewage sludge waste incineration can be performed in mono-incineration or co-incineration plants. Based on data regarding the case study WTE plant characteristics, the boiler efficiency of the energy company AEB is 85%, the conversion efficiency from steam to electricity is about 35% in the Waste Fired Power Plant (WFPP) (25% in the conventional WTE plant), and 100% conversion efficiency to heat. The sewage sludge waste with a heating value of 2.2 kWh/kg DM (8 MJ/kg DM) could be converted into 0.65 kWhel/kg DM of electricity and 1.87 kWhth/kg DM of heat. The total amount of energy (65637 MWh/year) generated from sewage sludge incineration is used by the energy company to provide district heating.

Nevertheless, the contract with AEB expires on January 1, 2023 and the garbage plant HVC (Huisvuilcentrale) in Alkmaar takes over the management of the sewage sludge (Water Board AGV, 2020). The HVC Alkmaar main goal is to process the sludge, no longer as a waste but as a source of valuable materials, in the most sustainable way possible and at the lowest possible social costs in the long term. Sustainability mainly concerns the recovery of energy and raw materials from the sewage sludge and processing of sewage sludge with the lowest possible $CO_2$emissions, following a treatment method where sludge waste will be being dried. The SS after drying process where reaches a dry matter content >90% can be used as an alternative fuel in coal-fire power plants, waste incineration plants, in cement industry (cement kilns), or as a source of phosphate fertiliser (agricultural application) (Mills et al., 2014), (Teoh and Li, 2020).

#### 3.2.4. Overview of the food waste management

The city of Amsterdam does not currently have a comprehensive food waste management and valorisation program. A sorting analysis of the food waste from the Dutch households showed that solid food waste, including sauces, fats and dairy products, amounts to 30.4 kg/person/year via garbage bins, 5.9 kg thick liquids via sink/toilet and 4.9 kg of solid food wasted via other pathways (Van Dooren et al., 2019). By and large, the municipality of Amsterdam has no recovery strategy for any organic waste at the household level (Coudard, 2019). Currently mixed garbage, included the food waste, are transported to the energy plant where they are incinerated, without any separation, to recover heat and electricity. The inherent variable composition of moisture content and thus the calorific value very much depend on the region of origin and the dietary habits of the population. Typical values for European food waste are 24% DM content and heating values from 22 MJ/kg DM (Banks et al., 2018) to 37.7 MJ/kg DM (Pham et al., 2015).

### 3.3. Water conservation and resource recovery strategies within the urban water system

#### 3.3.1. Research approach

Motivated by the desire to develop sustainable urban water systems, oriented towards water conservation and resource recover, and identify the key processes that are likely to pose the greatest environmental impacts within the system-wide, an evidence-based research was conducted via the LCA method. This work introduces a full system analysis of the life cycle energy consumption, water consumption, global warming and freshwater eutrophication potentials of alternative urban water systems relied on the principles of water fit-for-purpose and resource recovery. The set of scenarios included three water conservation scenarios and one food waste valorisation scenario, all investigated individually, for future implementation (2040-2050) and 100% penetration rate. The scenarios were evaluated analysing data from a sample of 418 residential units for a neighbourhood in Amsterdam. The scenarios were modelled for maximum occupancy of 1.8 persons per household according to projections obtained from the municipality of Amsterdam (Bailey et al., 2020).

The environmental performance of the water system was examined under four main planning scenarios, divided into two main categories (eight scenarios in total), having as a reference axis the baseline (comparative LCA). The scenarios of the first category included i) the greywater treatment and reuse to supply the toilet and washing machines "GTR1", ii) the rainwater harvesting and use "RHU1" to supply the toilet and washing machine, iii) the usage of water-efficient plumbing devices (toilet flush, shower head, washing machine, dishwasher) "ECO1", iv) the food waste valorisation "FWV1" where food waste is disposed into the sewer system via the usage of kitchen waste grinders. On the second category, it was assumed application of the same interventions at a dwelling level, but also additional interventions at a plant-wide level for all the options (GTR2/RHU2/ECO2/FWV2). The additional interventions included application of THP and air-stripping in the WWTP to recover nitrogen from the digester reject water, as well as technology of phosphorus recovery from the ISSA applied applied at a sludge mono-incineration plant.

Furthermore, they were assumed different sludge waste management methods between the two categories of scenarios and the baseline. The baseline was modelled for sludge waste co-incineration with the rest of MSW (WWTP adjacent to WTE plant; 0 km distance), as it is the case in the existing situation. In the first category of alternatives, sludge waste was sent to a drying plant (WWTP in Amsterdam to garbage plant in Alkmaar; 40 km distance), as scheduled after 2023. n the second category, sludge waste was sent to a mono-incineration plant in order to perform phosphorus recovery from ISSA (WWTP in Amsterdam to mono-incineration plant in Dordrecht; 100 km distance).

The first category of scenarios was chosen for their ease of implementation and data availability, while the second one was selected as a mean to investigate a more integrated approach which aim to the maximisation of resource recovery. The selection of the technologies for nitrogen and phosphorus recovery was based on the researches of van der Hoek et al. (2016) and van der Hoek et al. (2018a), which suggested environmental attractive solutions for nutrient recovery from the wastewater of the WWTP under study.

### 3.3.2. The water cycle company of Amsterdam

Waternet is the public water service of Amsterdam and the Regional Water Authority Amstel, Gooi, and Vecht (van der Hoek et al., 2018a). Waternet is the first company in the Netherlands that combines services regarding the entire water cycle in one organisation. Its activities concern drinking water supply, sewerage, wastewater treatment, surface water management, control of the canals in Amsterdam and flood protection (van der Hoek et al., 2016). It controls the water cycle on an integral and socially responsible way, based on the four values of safety, customer orientation, sustainability and innovation (Chiou, 2018). Driven by the principle of sustainability, it takes the lead in new collaborative platforms working across urban challenges, such as climate adaptation and circular economy. The company has the ambition to become climate-neutral, reducing $CO_2$-emissions as much as possible or offset them. It manages water not as an individual resource, but as a valuable source of many other resources and its main objective is to reclaim energy and raw materials from the water cycle and minimise waste through reuse. Waternet, which has been conducted an extensive research regarding the optimisation of the water system in Amsterdam, has a wide range of data which was willing to share to aid the conduction of this project.

#### 3.3.3. Recent researches as a foundation of this work

Sewage treatment plants have the potential to be at the core of resource recycling, with innovative technologies for sludge-to-energy conversion, phosphorus and nitrogen recovery. The city of Amsterdam leads the way for transition towards a circular economy participating in several research projects which investigate innovative options to make wastewater a source of added-value products and alleviate natural resource depletion. The last few years, researchers have started to examine the possibility of applying further phosphorus recovery from Amsterdam West's wastewater. van der Hoek et al. (2016) examined the possibility of reusing organic matter and phosphorus from Amsterdam West's wastewater, investigating several resource recovery measures based on the criteria of changes in material flows, recovered products and implementation horizon. Among all, phosphorus recovery from sewage sludge ash after incineration using the EcoPhos technology was one of the proposed strategies. Furthermore, a research carried by (van der Hoek et al., 2018a) promoted promising technologies for nitrogen recovery via the wastewater of Amsterdam, integrated in the existing configurations of the WWTP and without affecting to any other recovery methods. Their findings revealed that the application of THP for pre-treatment of WAS and air stripping from the digester reject water seems a promising strategy with high potential of nitrogen recovery and further energy production.

The main scenarios of this MSc Thesis were the object of study for previous research projects. Bailey et al. (2020) conducted a hydraulic modelling of the case study's water system which was relied on the stochastic drinking water demand and wastewater discharge and nutrients. The stochastic model (created by Mirjam Blokker) predicted the changes in the wastewater flow, temperature and nutrient mass (COD, TKN, TPH) in the sewer network after application of the tree water conservation scenarios of greywater reuse, rainwater use and installation of efficient water appliances. Alongside, a recent MSc Thesis investigated respective changes in the nutrient mass in case of disposal of food waste into the sewer system using kitchen waste grinders. The data availability constituted the appropriate supplies for a comprehensive investigation on the environmental feasibility of introducing integrated resource recovery-oriented options as a design parameter in the water system.

### 3.3.4. Site description

A typical Dutch household of a "real world" case study in Amsterdam was used as a research objective in this case. Prinseneiland neighborhood was selected as a case study due to the data availability from the local water company and previous researches conducted in the same study field. Prinseneiland-one of the three so called "Western Islands"-is a small district, approximately 3 ha (Mogoş, 2018), located in the northwest of Amsterdam's city centre. Its water transport system provides services for mostly businesses and residences. In the area there are 418 domestic households and 55 other premises (offices, ateliers, storage buildings) and is characterised by well-defined boundaries with simple sewage and drinking water systems. The sewer system is a looped and combined network, which lead to a pumping station (Bailey et al., 2020). There is a big dataset available from previous research and measurements on groundwater levels, sewer discharges, precipitation and potential evaporation (Rutten, 2013). The water service system of the catchment contains two flow meters for each water main supply drinking water to the island, providing live data recording of water demand, one overflow structure and one pump installation with pump flow and tank level readings recorded at the wastewater pumping station (Mogoş, 2018). The average occupancy per household is 1.7 persons, where single, dual occupancy and family households represent the 58%, 23% and 19%, respectively (Bailey et al., 2020).

# 4. Materials and methods

## 4.1. LCA methodology

### 4.1.1. Objective and procedure of LCA

The concept of circular economy conceives of a production and consumption system with minimal losses of resources and energy through extensive reuse, recycling and recovery and is gaining popularity around the world. However, closed loops are not always favourable from an environmental point of view (Haupt and Zschokke, 2017). There are many different sustainability tools, ranging from simple to complex and covering the various aspects of sustainability, such as environmental, social and economic (Brilhuis-Meijer and Goedkoop, 2015). The LCA is a well-established method which focuses only on the environmental side of sustainability. The objective of the LCA method is to deal with the complex interactions between a product or process with the environment, taking into account all the implications caused by the production, use, disposal of raw materials, as well as the avoided impacts from the resources offset accounting. LCA provides insights on technical factors that require further research and operational conditions with the highest potential for impacts reduction, ensuring that only the most promising technologies are pursued before lock-in occurs (Lam et al., 2020). Assessing a technology at its early development stage can provide opportunities to identify environmental impacts regarding resource use, human health and ecological consequences which can be potential barriers for its full-scale application. There are many opportunities of applying LCA at various scales to assess emerging technologies and integrated resource recovery systems, optimise individual recovery processes, and support technical decisions (Lam et al., 2020).

Numerous LCA studies have started, recently, focusing on resource recovery-oriented wastewater services due to their potential of closing water, energy and nutrient loops. Cope with water scarcity, mitigation of eutrophication and restoration of aquatic ecosystems, abatement of GHG emissions and reduction of global warming potential consist some of the promising environmental benefits. The LCA follows a transparent procedure and constitutes a standardised methodology with principles and framework provided by the International Organization for Standardization in the ISO-14040-1997 standard (Barrios et al., 2004). LCA is obtained by means of a systematic four-step procedure which is depicted in Figure 4.1) and listed below (Del Borghi, 2013):

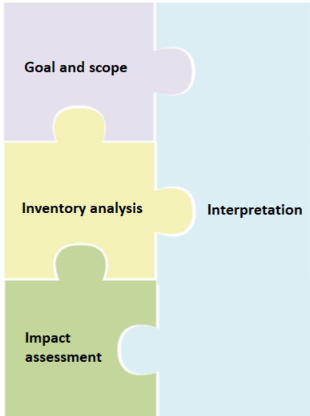



**Fig. 4.1:** The LCA phases (personal modification, (Barrios et al., 2004))

1. Goal and scope: Objective of the method, system boundaries, environmental scope, and functional unit of the process.
2. Inventory analysis: Collection and analysis of the foreground and background inventory data, all the environmental input and output.
3. Impact assessment: Application of the impact assessment method with classification of the environmental impacts and evaluation of their importance.
4. Interpretation: Drawing conclusions which are well-substantiated and adequately supported by the data and the followed procedures, identification of potential significant issues, sensitivity and/or uncertainly analysis.

### 4.1.2. Goal and scope

The goal of the study was to estimate the life cycle energy consumption, life cycle global warming potential, life cycle water consumption and life cycle freshwater eutrophication potential of several water conservation and resource recovery strategies within the urban water cycle compared to the baseline. The scope of this study included the energy and material input, and associated environmental releases during the construction and operation of the water and sanitation services starting from water extraction and ending with sewage sludge management. All the material input carried their own life cycle and overall resources that were used for their production. The environmental releases focused on the GHG emissions and nutrient releases. The implications of the end-of-life handling of system components were excluded from the scope, while the construction of the additional infrastructure needed for the in-house interventions (greywater/rainwater unit, kitchen grinder unit, additional pipelines) were explicitly included in the system's life cycle.

### 4.1.3. System boundary and functional unit

The scope of an LCA study is delimited by the system boundary. Boundary definition is a critical primary step in modelling analysis, inextricably interconnected with the goal of the study. The system boundary in the current project includes all the activities that are an essential part and are involved directly and indirectly to the current treatment processes, as well as those that are differentiated from the current water system (baseline) and surcharge the impacts. For instance, HCl generation is included (disregarding the location of production) while the energy use for water heating in the kitchen taps is not (kitchen tap hot water use remains the same across all cases), the materials for the construction of the greywater treatment system is included while the materials for manufacture of the water-efficient devices are not (similar construction with the conventional ones). The system boundary has been outlined broad enough to reflect all the sanitation activities that are involved in the water system, including the water supply services, domestic water end-use, wastewater and sewage sludge waste management, or could potentially be involved, such as the food waste stream.

All the activities contain foreground processes (e.g., operational activities for water/wastewater treatment, sewage sludge/food waste management) which reflect the main processes of the system under study and background processes (e.g., chemicals and materials production, electricity generation) which are 'hidden' behind each resource/material use. Regarding the air emissions, the most important direct emissions (e.g., $CH_4$, $N_2O$, excluding biogenic $CO_2$[3]) arising from the processes. Regarding the nutrient releases in the WWTP, nitrogen, phosphorus and COD discharges were taken into account.

Following a common practice in LCA of water and waste systems, the secondary products and services generated (recovered), alongside the water use and waste management, were credited by assuming substitution of the corresponding market products or services. On this basis, the energy recovered from biogas, sewage sludge waste and food waste management were accounted as substitution of the energy from the national grid. The recovered phosphorus and nitrogen were accounted as the substitution of the respective amount of phosphoric acid and ammonium sulfate fertiliser from the global market. The additional recovered materials (Aluminum chloride, Calcium chloride) from the phosphorus recovery technology from the sewage sludge ash (EcoPhos technology) were also taken into account assuming substitution of aluminum

[3] The $CO_2$emissions can either been from fossil or biogenic origin. Biogenic $CO_2$emissions belong to short carbon cycle. The biogenic emissions are not taken into account in national protocols as they are considered (by convention) as "carbon neutral" (global warming potential equal to zero) (Pradel and Reverdy, 2012).

alloy and de-icing agent from the global market, respectively. The overview of the system boundary is illustrated in Figure 4.2.

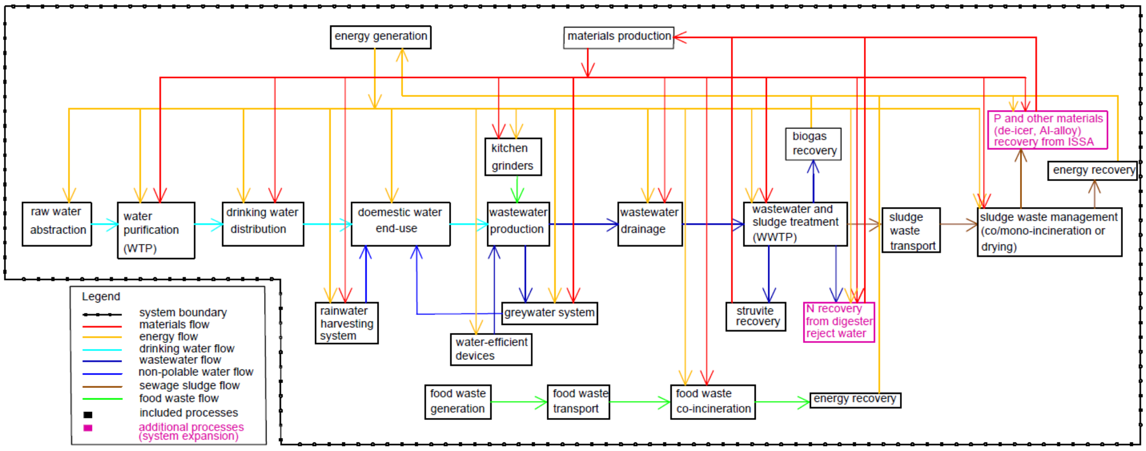



**Fig. 4.2:** Urban water system boundary and system expansion

Environmental impacts are the perturbations of natural cycles, induced by technical interventions of human activities.The quantification of the environmental impacts taking into account all environmental interventions associated with the life cycle of the system under study. However, all these interventions have to refer to the same functional unit (FU) to normalise the results in terms of per unit of service provided by a process. In the current research, the alternative scenarios contain various sanitation activities which serve different functions. The complexity of the functions led to the selection of a service-oriented FU which can include all the water and sanitation services of a household and allow a fair comparison between the alternative water systems and the baseline. The usage of the term "sanitation services" reflects all the services for disposal of sewage and solid waste which protect the public health. The FU was one household's water and sanitation demands in one year.

### 4.1.4. Inventory analysis

Three main kinds of data were used: i) site-specific data collected from reports, researches and personal communication with experts from the local industry, ii) global data from literature, iii) data from the LCA databases of the software. Appendix C presents the most relevant inventory data and assumptions used in the study.

#### 4.1.5. Impact assessment

A large amount of LCA-specific software programs have been developed, integrating extensive sets of data into a single software, but all with different focus and requirements. SimaPro is among the most used LCA software. The Dutch company PRé Consultants BV developed SimaPro to analyse complex products through their different life stages (i.e., from cradle-to-grave) (Barrios et al., 2004). SimaPro has been a world's leading sustainability software for nearly 30 years, containing up-to-date science-based methods and databases (SimaPro). The educational version SimaPro 9.1 was used in this study.

The scenario models was constructed starting with individual processes which were linked in assemblies. Each scenario was built as an aggregate of five main assemblies: the water supply, domestic appliances and units, wastewater management, sewage sludge management, food waste management, all of which composed of individual procedures. Each modular unit was represented as a network which was connected with background datasets in order to reflect the full supply chains and the embedded life cycle impacts. SimaPro has built-in background life cycle inventory data (e.g., regional electricity supply, chemical production, material production) which were used to specified the sub-processes. These background data include material, transport and energy input, output to the technosphere or nature (i.e., emissions, by-products), and recovered products (i.e., energy, fertilisers). In this case, the selected inventories was mainly concerned the local market of the Netherlands (NL) or the EU region (ReR), otherwise if the choices were restricted, the products related to the global market (GLO). Special attention requires the modelling of wastes: reuse, recycling, treatment or disposal methods, which have to be specified as to give to the software a more complete image of the material flows.

One of the checks of the output was the graphical representation (as a network) of the energy and materials flows of the life cycle of each scenario. This network representation is called gravity representation in SimaPro, since it assigns a thickness to connecting lines proportional to the contribution of its process/assembly on the environmental impact, presenting only the major contributors to the total life cycle of the system (Barrios et al., 2008).

The software allows the use of different methods for "valuation". The impact assessment methods (e.g., CML, ReCiPe, TRACI) obtain a single-score for each single unit process and material, reflecting the various impacts (SimaPro). This project mainly used the ReCiPe 2016 method but also the Cumulative Energy Demand (also called "primary energy consumption") to transform the life cycle inventory results into indicator scores. The former method was used to retrieve the results for the life cycle water consumption, global warming potential, freshwater eutrophication potential, while the latter was used to determine the life cycle energy consumption. Information regarding the local importance of the impact categories under study are given in Appendix D. The energy harvested approach "Cumulative Energy Demand" is a consistent approach, which quantifies the energy content of all different (renewable and non-renewable) energy resources. The impact category indicator results computed with this method reflect the safeguard subject "energy resources" but no other environmental impacts (Frischknecht et al., 2015). On the other hand, ReCiPe method transforms the long list of life cycle inventory results into a limited number of indicator scores. These indicator scores express the relative severity on an environmental impact category. Further information regarding the ReCiPe impact assessment method are given in Appendix E.

#### 4.1.6. Interpretation

The results from the inventory analysis and impact assessment are summarised during the interpretation phase (Cao, 2017). Life cycle interpretation starts with identification of the significant issues based on the results, continues with evaluation with qualitative checking of input data and quantitative analysis of any implication of changes in input data (sensitivity analysis) and ends up with conclusions, limitations recommendations (Barrios et al., 2008).

# 5. Results

Graphical representation of the results is depicted in this chapter illustrating the comparative environmental impacts of the alternative scenarios in comparison with the baseline, per assembly of processes. The impact categories include the life cycle energy consumption, the life cycle global warming potential, the life cycle water consumption and the life cycle freshwater eutrophication potential. A negative value means that the scenario has a lower environmental impact than the baseline, while positive value means a higher environmental impact. Details regarding the input data which were used in the modelling can be found in Appendix C.

## 5.1. Life cycle energy consumption

Figure 5.1 demonstrates the comparative results of the life cycle energy consumption, including both renewable and non-renewable energy sources.

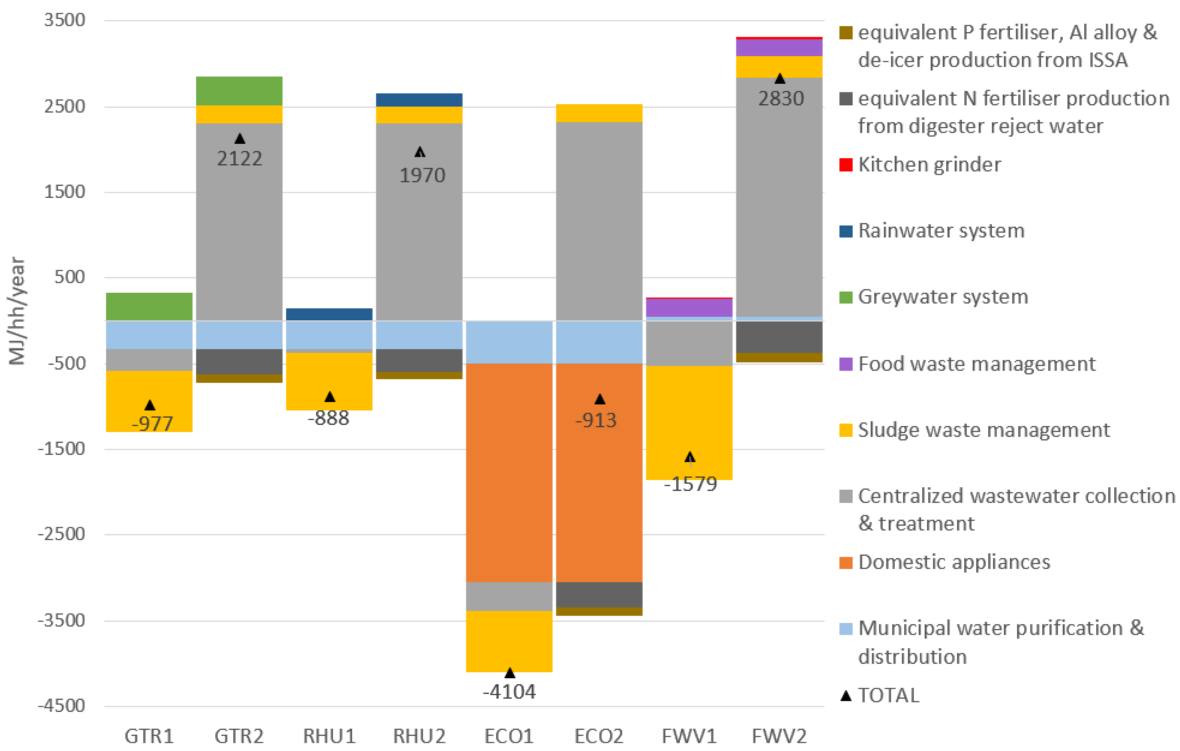



**Fig. 5.1:** Breakdown of life cycle energy consumption of the alternative scenarios relative to the baseline

‘ The two water conservation scenarios of GTR and RHU serve the same purpose; to provide greywater/rainwater to toilets and washing machines, and so they offset the same amount of energy for drinking water treatment and distribution. From Figure 5.1 it can be seen that the GTR is surcharged with more energy than the RHU because the greywater system, apart from the energy requirements for distribution of the recycled greywater, consumes quite significant energy for treatment operations (MBR system). In RHU, rainwater is not subjected to treatment, and the operation of the first flush diverters does not require energy, resulting in a low energy consumption only for rainwater distribution to the recipient appliances. GTR1 shows notable energy savings during the wastewater treatment mainly due to the reduced volume of wastewater under treatment coupled with increased biogas yield (energy offsetting) due to the increased inflow COD. In contrast, RHU1 presents a small energy saving attributed mainly to the slightly increased inflow COD, and by extension, increased biogas yield compared to the baseline. On the other hand, ECO1 shows the most considerable energy difference compared the baseline in the requirements for water heating in showers, dishwashers and washing machines. The water saving devices offset 54% of life cycle energy consumption attributed to the operation from these three devices (shower 37%, washing machine 9%, dishwasher 8%).

The results of FWV reveal a small energy burden for water purification and distribution due to the additional water use to flush the food scraps, as well as an insignificant energy excess for the kitchen grinder's operation. Regarding the food waste management, the reduced amount of food waste sent for incineration (5% of the total food waste produced) adds environmental burden compared to the baseline due to the energy recovery loss. On the other hand, the increased biogas generation and struvite recovery, which offsets high amounts of chemical fertilisers from the market, make the additional energy demand for water collection and treatment insignificant, leading to a considerable overall benefit in the WWTP.

The sludge waste drying process, despite the higher sludge transportation cost compared to the baseline, is superior to the co-incineration process in all the cases (GTR1, RHU1, ECO1, FWV1) and is differentiated across the scenarios based on the volume of the sludge waste produced. The superiority of this sludge management method is attributed mainly to the relatively high net energy recovered (small energy input, low waste generation for treatment). The fluctuations of the energy benefits of this category among the scenarios are based on the volume of the sludge waste produced. For instance, FWV1 which has the highest sludge waste production, results in the highest potential for energy recovery in the form of bio-fuels.

Contrarily, the sludge waste mono-incineration and phosphorus recovery technology surcharge the energy consumption of all scenarios (GTR2, RHU2, ECO2, FWV2) compared to the baseline. More than 50% of the energy burden is attributed to high sludge transportation cost (100 km distance) and demand on transport fuels (mainly petroleum), while a significant fraction is also attributed to the high energy requirements of phosphorus recovery (energy for operation of the technology and production of the chemicals used). The energy benefits arising from the recovery of phosphorus are not sufficient to counterbalance the energy burdens of this sludge management option.

The GTR2, RHU2, ECO2 and FWV2 demonstrate that thermal hydrolysis of sludge and air stripping for nitrogen recovery are the most energy intensive processes, which lead to an enormous energy burden. High amount of energy is required directly for the operation of these processes (60%) and indirectly for the production of sodium hydroxide (17%) and sulfuric acid (31%) which are used in air stripping process.

## 5.2. Life cycle global warming potential

The life cycle global warming potential is depicted in Figure 5.2.

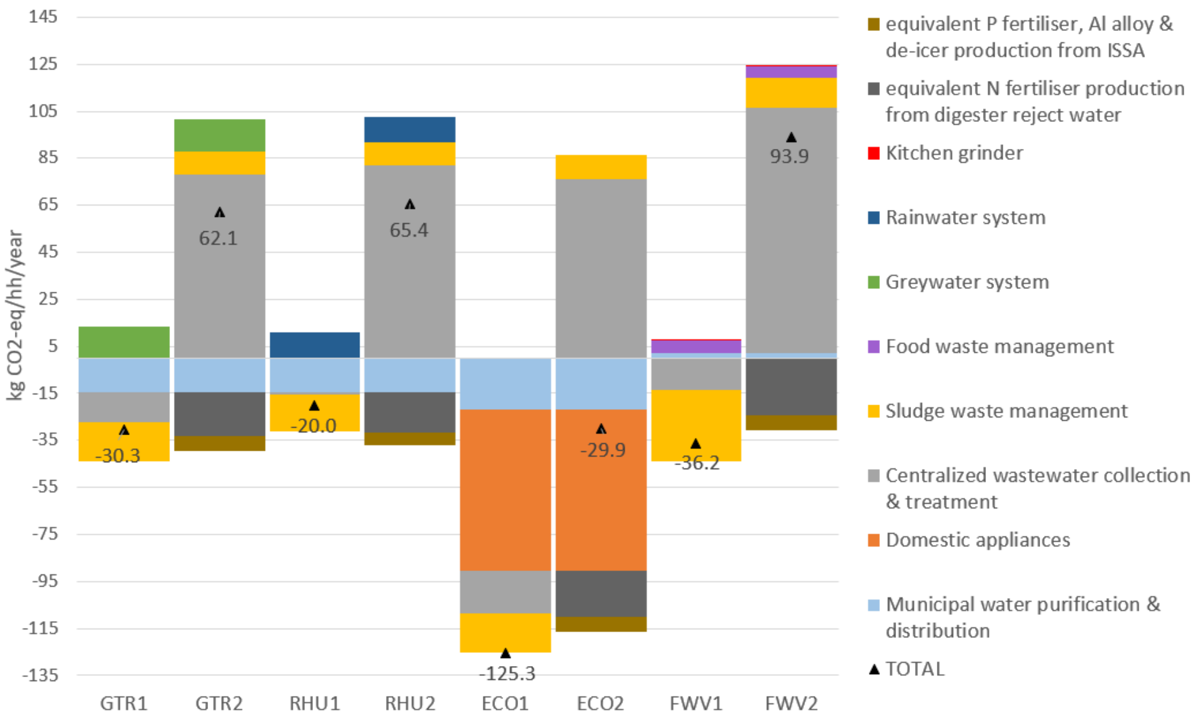



**Fig. 5.2:** Breakdown of life cycle global warming potential of the alternative scenarios relative to the baseline

Due to the high interconnection between production and use of energy and GHG emissions, the results of life cycle energy consumption and life cycle global warming potential follow similar trends. Even though the usage of mainly renewable energy sources in all scenarios (implementation of the scenarios after 2040-2050 where more green energy will be used) reduces the magnitude of the total life cycle global warming potential, the differences of this environmental metric between the scenarios are based on the requirements for combustion of fossil fuels for electricity, heat, and transportation. The most notable difference between life cycle energy consumption and life cycle global warming potential can be seen in GTR and RHU, where greywater system, despite the twofold energy burdens compared to the rainwater system (see Figure 5.1), produces more GHG emissions (by 22%) than the rainwater harvesting case (see Figure 5.2). This is attributed to two factors: firstly, the operation of water recycling systems is mainly based on renewable energy so the environmental burden of the GTR compared to the RHU from carbon intensity perspective is small; secondly, the greatest fraction of GHG emissions is attributed to the activities for production of construction materials for the recycling units (MBR system, rainwater harvesting tank) and the collection/distribution pipelines, activities which mainly rely on non-renewable energy sources.

## 5.3. Life cycle water consumption

The comparative results of the life cycle water consumption are depicted in Figure 5.3.

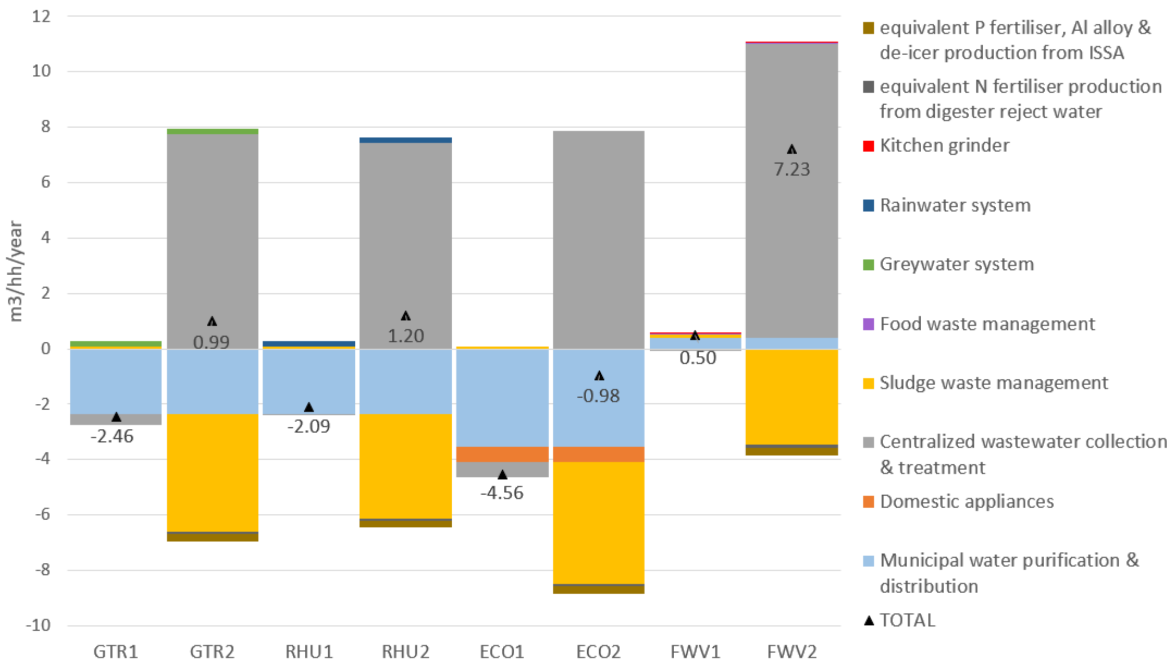



**Fig. 5.3:** Breakdown of life cycle water consumption of the alternative scenarios relative to the baseline

As it can be seen, GTR and RHU present high water savings regarding the municipal water supply sector. The water intensive activities, and mostly, the treatment of $FeCl_3$-rich sludge waste produced in coagulation phase (57% share), the energy use from biogas coming from wastewater treatment (16.5% share) and the usage of liquid oxygen[4] during the sand filtration process (11.5% share) are significantly limited due to the reduced demand for water purification. For same reason, ECO shows an even higher benefit due to the extremely low demand for potable water, while the FWV reveals a small environmental burden due to the additional water supply for flushing the food waste via the sewer system. The results, also, demonstrate that the manufacture of the greywater and rainwater systems require similar amounts of water. The environmental benefits of the two water conservation scenarios of GTR1 and ECO1, in the wastewater treatment sector, are mainly attributed to the high energy recovery from biogas which offsets part of the electricity from the national grid which is produced from wastewater treatment (anaerobic digestion of sewage sludge).

Since energy consumption is an indirect contributor to water consumption, in ECO scenario, the reduced energy requirements for water heating in the showers, dishwashers and washing machines result in reduction of life cycle water consumption. The main factors which lead to the indirect water savings are, in descending order, the use of biogas arising from the wastewater treatment, the electricity production from natural gas and the production of crystalline silicon which is used in photovoltaic technology. Regarding the FWV scenario, the insignificant energy requirements of kitchen grinders results in minimal environmental burdens from a life cycle water consumption perspective.

From Figure 5.3 it is visible that the sludge drying process for bio-fuels production adds a negligible burden from a water perspective compared to the baseline in all the cases (GTR1, RHU1, ECO1, FWV1). In contrast, the phosphorus recovery process from ISSA returns significant amounts of water to the environment, since great amounts of wastewater arise as by-products of this process. Moreover, the recovered products arising from the EcoPhos technology and the offsetting of chemical fertilisers, Al alloy and de-icer from the market add to the water savings.

[4]The continuous operation of slow sand filtration may induce serious operational problems. Algal growth and oxygen depletion account for the most common, which can lead to a premature clogging of the filter and anaerobic conditions in the filter bed due to the production of hydrogen sulphide and other taste and odour producing substances (Silva et al., 2012). This operation mode allows the diffusion of oxygen into the filter bed (oxygen content of the filter should be above 3 mg/L) during the draining phase to ensure that anaerobic conditions are avoided within the filter and organic matter may be easily degraded (ITACA, 2005).

The GTR2, RHU2, ECO2 and FWV2 reveal that the thermal hydrolysis and air stripping for nitrogen recovery from digester reject water are extremely water intensive processes, with most dominant parameter being the production of sulfuric acid (approximately 53% share of the overall burden from both processes) which is used in air stripping process. The greater the nitrogen mass in the wastewater, the higher the usage of chemicals (sulfuric acid, sodium hydroxide) and energy for the recovery of ammonium sulfate which require high amounts of water during their production phase.

## 5.4. Life cycle freshwater eutrophication potential

Figure 5.4 demonstrates the comparative results of life cycle freshwater eutrophication potential.

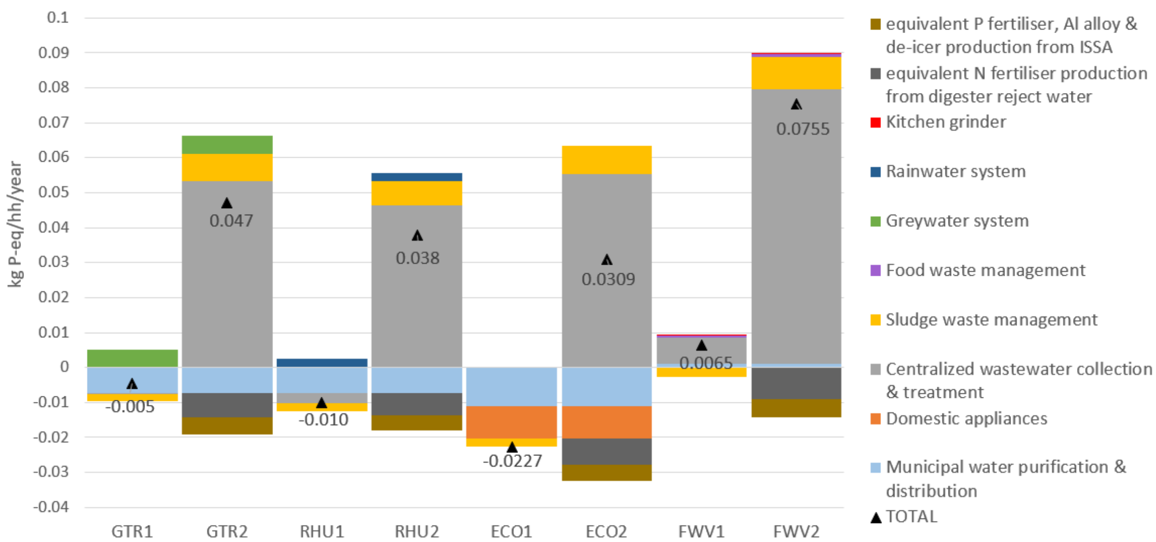



**Fig. 5.4:** Breakdown of life cycle freshwater eutrophication potential of the alternative scenarios relative to the baseline

The greywater system has higher potential for freshwater eutrophication compared to the rainwater system, since the MBR unit consists mainly of steel. Steel is an alloy of iron of which the production requires phosphorus and sulfur removal from the iron ore (Matsubae et al., 2015). The leaching of these substances leads to an increase in eutrophication. Alongside, as iron occurs only as iron oxides in the earth's crust, the ores must be converted using carbon of which the primary source is coking coal (World Coal Assosiation, 2020). The spoils of hard coal arising from the mining activities is one of the main parameters that affect the freshwater eutrohpication potential the most in this category (greywater/rainwater system), along with spoils of lignite and sulfuric tailings from copper mine. For the same reason, the energy savings from usage of water-efficient appliances (shower head, washing machine, dishwasher), in the ECO scenario, limits freshwater eutrophication potential.

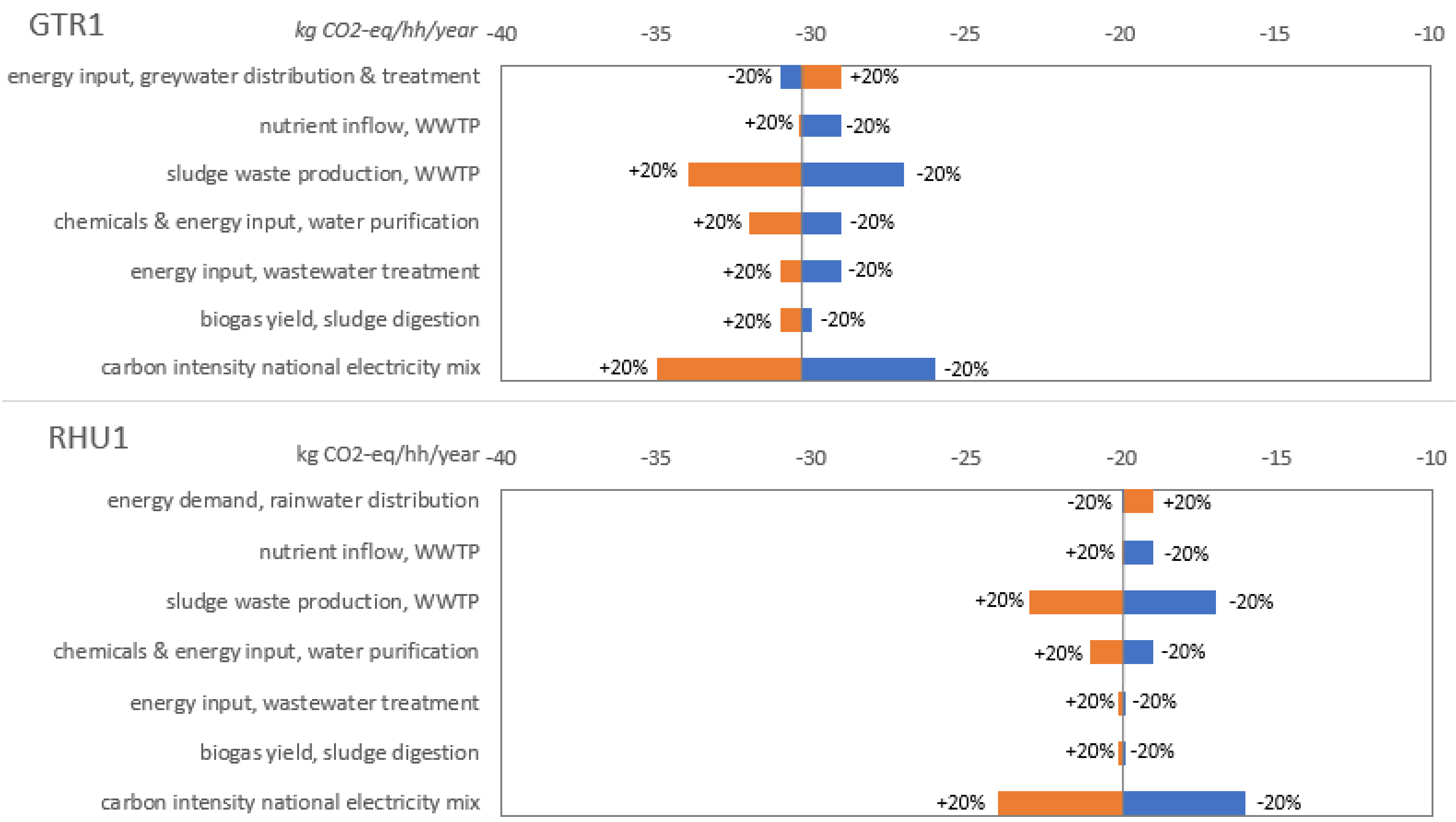
Water purification process requires significant amount of chemical use. This is why the three water conservation strategies present limited nutrient releases while FWV, which consumes slightly more drinking water at a dwelling level, presents a minimal increase. It is also observed that in the RHU1 there is environmental benefit from eutrophication perspective in the wastewater treatment process, which is attributed to the synergy of two reasons. The first reason is that the decreased TPH and TKN inflow in the WWTP leads to less direct nutrient releases in the water body (through effluent). The second reason is that the slightly increased COD inflow results in an increase in biogas yield which compensates energy from the grid reducing the potential for nutrient leaching arising from the mining activities. In the GTR and ECO, the increased biogas yield (offset electricity form the grid) counterbalances the increased direct nutrient releases to the water body via the effluent, resulting in zero comparative impact. On the other hand, in the FWV, the great biogas generation is not sufficient to counterbalance the substantial discharge of nutrients to the recipient water body.

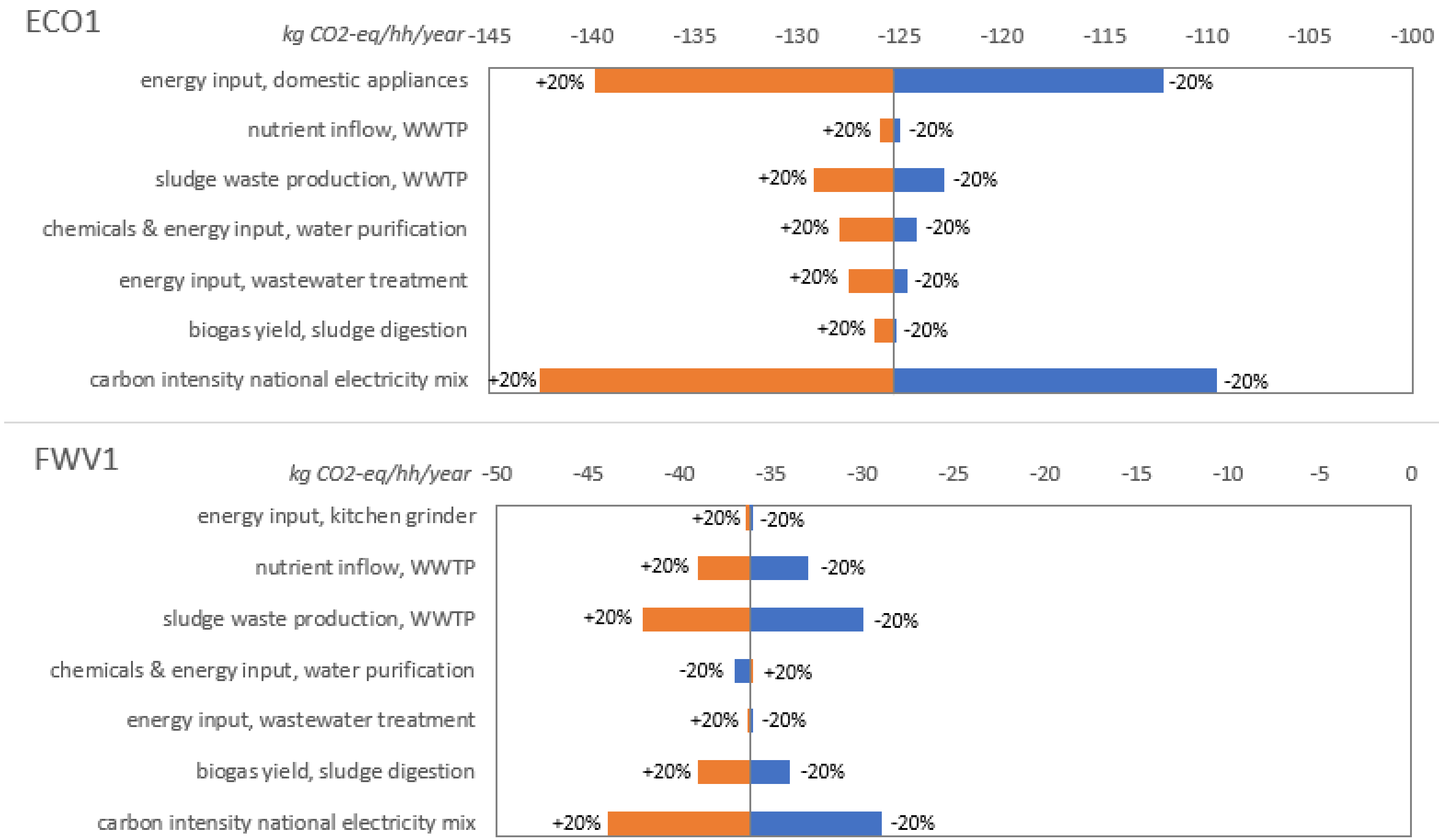
The high energy savings of the sludge drying process are mainly savings in renewable energy in all the scenarios. Consequently, the limited usage of fossil fuels, and thus, spoils from the mining activities makes the environmental benefits of the freshwater eutrophication potential not as significant. On the other hand, the sludge mono-incineration and the process of phosphorus recovery from ISSA reveal a significant environmental burden in all the scenarios. The wastewater which is produced as a by-product in the EcoPhos technology is the main parameter which increases the freshwater eutrophication potential, contributing more than 65% to the total burden of sludge management category. As a counterbalance, the recovered products from the EcoPhos technology (phosphoric acid, de-ider, Al alloy) reduce the possibility for freshwater eutrophication by approximately 60%.

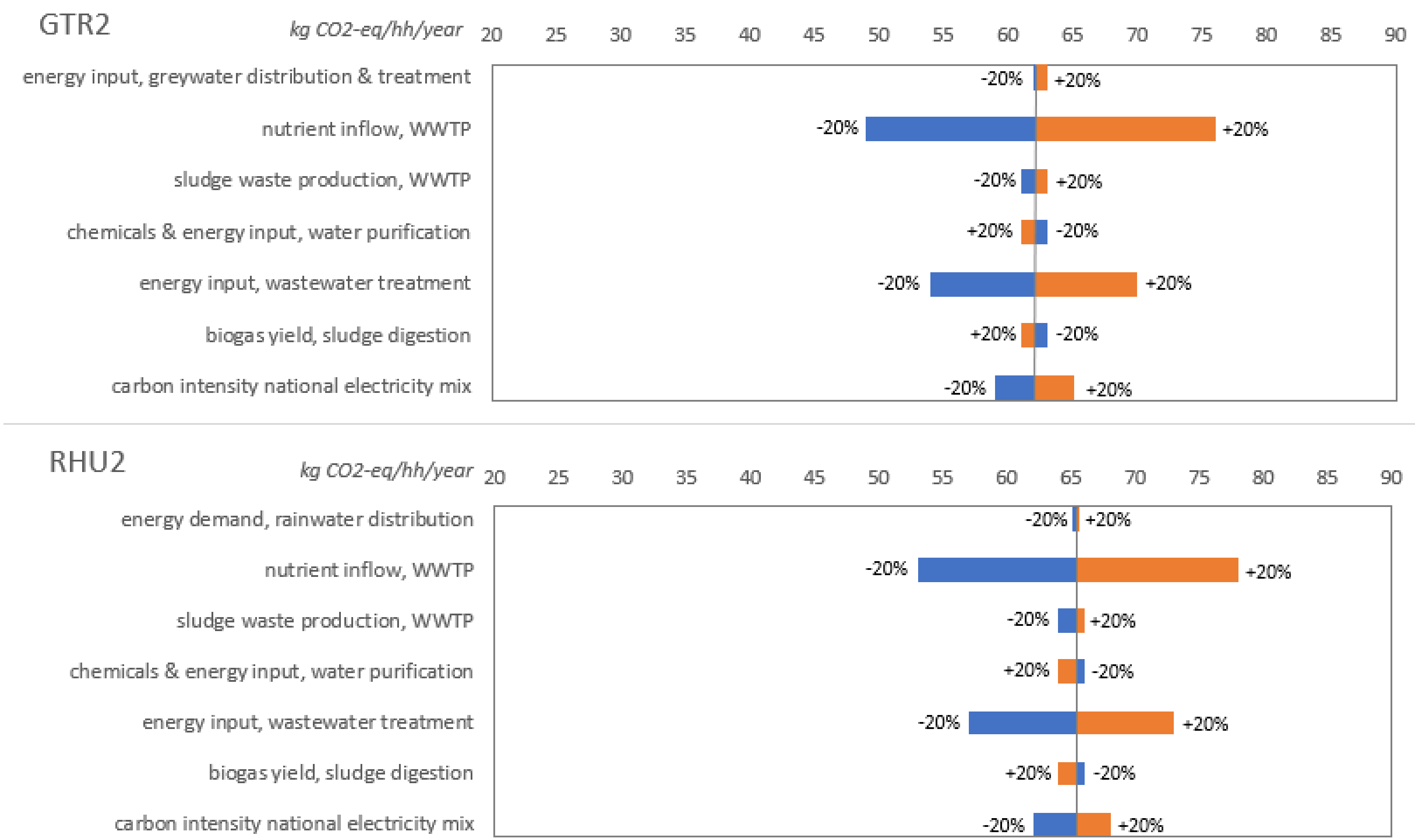
The processes of thermal hydrolysis and air stripping for enrichment and recovery of nitrogen from the reject water after sludge dewatering, apart from significant energy consumers and GHG releasers, they are dominant nutrient releasers. The usage of great amounts of energy in the WWTP, especially biogas from sewage treatment, increase the life cycle freshwater eutrophication potential. On the same time, the production of great amounts of sodium hydroxide (more than 30% contribution) and sulfuric acid (more than 20% contribution) which are used in air stripping process results in great leaching of nutrients in the recipient water bodies. These energy and chemical input of these two processes, which are proportionate to the retrieved ammonium sulfate (i.e., higher impact in the FWV2), vastly increase the comparative results. As it can be seen from Figure 5.4, offsetting equivalent amounts of ammonium sulfate fertiliser from the market is not sufficient to alter the extreme negative results of these specific recovery methods.

## 5.5. Sensitivity analysis

A sensitivity analysis was performed by perturbing (by +/-20%) each variable that is labelled on the y-axis by one at a time, while holding other variables constant at their reference case values to determine its influence on life cycle global warming potential. The sensitivity analysis results are presented as tornado graphs in the set of Figures 5.5.


GTR1
kg CO2-eq/hh/year
-40
-35
-30
-25
-20
-15
-10
energy input, greywater distribution & treatment
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix
+20%
-20%
RHU1
kg CO2-eq/hh/year
energy demand, rainwater distribution
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix


ECO1
kg CO2-eq/hh/year
-145
-140
-135
-130
-125
-120
-115
-110
-105
-100
energy input, domestic appliances
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix
+20%
-20%
FWV1
kg CO2-eq/hh/year
-50
-45
-40
-35
-30
-25
-20
-15
-10
-5
0
energy input, kitchen grinder
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix
+20%
-20%


GTR2
kg CO2-eq/hh/year
20
25
30
35
40
45
50
55
60
65
70
75
80
85
90
energy input, greywater distribution & treatment
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix
-20%
+20%
RHU2
kg CO2-eq/hh/year
energy demand, rainwater distribution
nutrient inflow, WWTP
sludge waste production, WWTP
chemicals & energy input, water purification
energy input, wastewater treatment
biogas yield, sludge digestion
carbon intensity national electricity mix

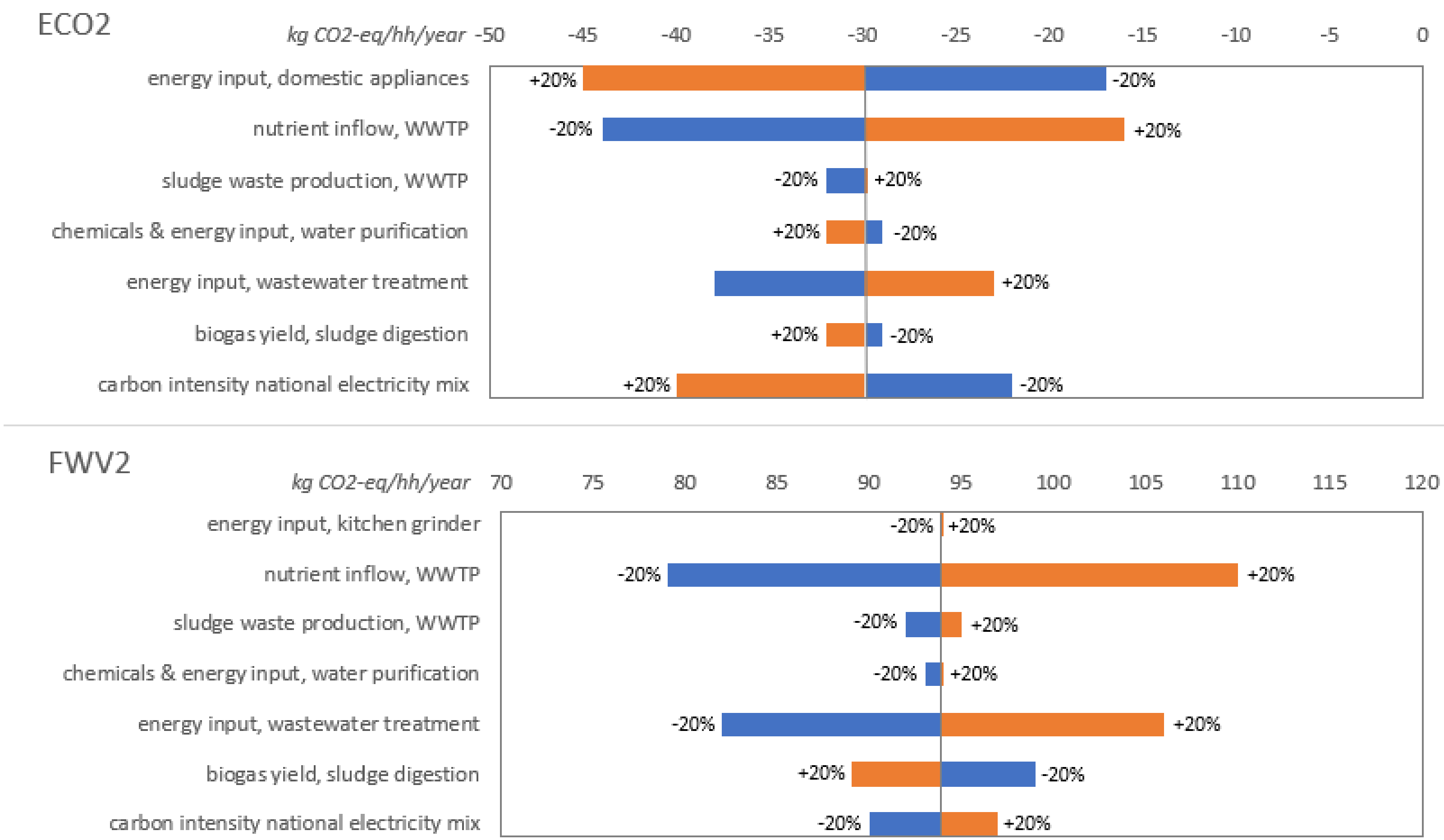



**Fig. 5.5:** Sensitivity analysis of the comparative results of life cycle global warming potential. The vertical lines represent the deterministic values and the horizontal bars the variation of the values associated with changes in the input parameters labelled on the left.

The deterministic results of life cycle global warming potential for each scenario are represented with a vertical line, and the variations in the values associated with changes in the input parameters labelled on the left side of the graphs are demonstrated with horizontal bars.

The profile of the most influential parameters changes between the two categories of scenarios. Among the investigated parameters, the carbon intensity of national electricity mix and the volume of sludge waste produced in the WWTP appear to be the two most influential variables for global warming potential in the GTR1, RHU1, FWV1 scenario. The national power grid provides electricity in all the involved energy users (households, WTE plants, WTP, WWTP), so changes on its carbon intensity significantly affect the overall results of global warming potential. Furthermore, changes in the amount of sludge waste produced also induce great changes in global warming potential mainly due to the high influence of the sludge transportation cost (transport fuels) and the equivalent energy production amounting to the fuel pellets produced from the dried sludge. Particularly, the variation of carbon intensity of electricity may result in a variation of up to +8/-7 kg $CO_2$-eq per household (hh) per day and the variation of sludge waste production up to +/-6 kg $CO_2$-eq/hh/day difference between the FWV and the baseline.

On the other hand, the results for the ECO1 scenario proves that the electrical energy demand of the domestic appliances for water heating (shower, washing machine, dishwasher) is the most dominant factor of influence of global warming potential on the entire urban water system. This fact is proven directly from the variation of the energy requirements input of these three devices, but also indirectly from the variations of the carbon intensity of the national electricity mix. Variations on the energy intensity of these devices result in up to +16/-17 kg $CO_2$-eq/hh/day difference between the ECO and the baseline.

In the GTR2, RHU2, ECO2 and FWV2 it can be seen that perturbations on the mass of the inflow nutrients in the WWTP, as well as on the energy requirements of the WWTP play dominant role in life cycle global warming potential. Since the resources input of the methods for further nutrient recovery is proportionate to the nutrient content of wastewater, and the methods of thermal hydrolysis and air stripping are extremely energy intensive technologies, the variations of these factors vastly change the overall results and make the rest of the parameters less significant.

# 6. Discussion

The research tried to address the following research questions:
I. Which interventions are the most resource efficient and promote an environmentally sustainable urban water system with regards to life cycle energy consumption, life cycle water consumption, life cycle global warming potential and life cycle freshwater eutrophication potential?; Which interventions shift burdens and induce more environmental challenges?
II. What are the key parameters that have the greatest influence on life cycle global warming potential of the urban water system?
III. What is the most efficient point of improvement in the entire urban water system and what measures need to be taken in order to adopt the most promising technologies into the implementation road map?

## 6.1. Interpretation of major outcomes

The comparative LCA based on the Prinseneiland community demonstrated that water conservation and resource recovery strategies embedded into the urban water system are not necessarily sustainable and resource efficient solutions, verifying the statement of Haupt and Zschokke (2017). The analysis pointed out technical characteristics and operational conditions that have the highest potential to reduce the environmental impacts and promote a resource efficient urban water system, that maximise resource utilisation with the least possible environmental impacts. The results also revealed technologies which although recover resources, they consume great amounts of resources (mainly water and energy) and increase the environmental emissions of GHGs and nutrients.

The results showed that water heating in showers, washing machines and dishwashers are, by far, the greatest energy consumers in the urban water system, followed by the energy requirements for drinking water supply (purification and distribution) and wastewater management (collection and treatment), as had been highlighted by Gerbens-Leenes (2016) too. In particular, they have a 90% share of the total energy consumption and 73% of the total GHG emissions of the entire water system. The comparative assessment showed that the usage of water efficient shower heads, dish washers and washing machines at a dwelling level can reduce by half (55%) all the environmental metrics under study (energy consumption, water consumption, global warming potential, freshwater eutrophication potential) arising only from the energy demand for water heating in showers (38%), washing machines (9%) and dishwashers (8%).

Among the water conservation scenarios, installation of water saving devices (low-flush toilets, efficient shower heads, waterless washing machines, waterless dishwashers) demonstrated the best environmental profile from energy and global warming perspective, followed by the greywater reuse and rainwater harvesting, fact that is in line with Racoviceanu and Karney (2010) and Xue et al. (2016) findings. This intervention, apart from the environmental benefits due to the reduced water heating demand in the household, reduced the GHG emissions and nutrient releases of the water purification process by up to 55%. Furthermore, in the WWTP, the biogas yield and struvite recovery increased and the GHG emissions decreased by 18 kg $CO_2$-eq/hh/year, without any effect in the freshwater eutrophication potential.

At a centralised sludge management level, sludge drying for bio-fuel production was the most attractive option. This method provided a tenfold increase of the energy savings and a tenfold decrease of the GHG emissions compared to the conventional sludge co-incineration process for heat and electricity generation. It also limited the life cycle freshwater eutrophication potential, while it slightly increased the life cycle water consumption.

Sludge mono-incineration for phosphorus recovery from the sludge ash is a debatable option based on the results. This method added more than 60% environmental burden compared to the baseline of sludge co-incineration with regards to life cycle energy consumption, life cycle global warming potential and life cycle freshwater eutrophication potential. Half of this burden is attributed to the high sludge transportation cost. The benefit from the recovered substances (phosphoric acid, aluminium alloy, de-icing agent) had a share of less than 40%. On the other hand, the phosphorus recovery technology (EcoPhos) returns great amounts of water to the environment, leading to up to 4.4 m3/hh/year water savings.

In line with the hypothesis, the implementation of thermal hydrolysis of sludge and air stripping as methods for nitrogen recovery from the digester reject water was deemed a weak resource recovery option which just shifts the burdens. The energy benefit from the equivalent N fertiliser production (ammonium sulfate) represented approximately 12%, while the remaining 88% of the impact was net energy burden. These technologies burdened the scenarios with up to 80 kg $CO_2$ and 0.07 kg P-eq/hh/year, demonstrating that the recovered nitrogen and the increased biogas yield from thermal hydrolysis application are insufficient to counterbalance the negative impacts. The sensitivity analysis showed that the energy requirements of these technologies highly affect the overall results of all environmental metrics. However, the assessment of the environmental impacts of these methods was based on inventory from pilot plants and laboratory experiments, which do not necessarily represent the environmental burdens which would be caused by a full-scale application. A system scale-up would probably limit the environmental burdens presenting a more efficient strategy with lower resource input and higher benefit from nitrogen recovery.

The sensitivity analysis showed that changes in the carbon intensity of the national electricity mix and in the volume of sludge waste produced can most influence the life cycle global warming potential in all scenarios. Furthermore, the results of ECO scenarios (ECO1, ECO2) revealed that the energy requirements for water heating in showers, dishwashers and washing machines are the driving forces of life cycle global warming potential in the entire water system. In case of integration of further phosphorus and nitrogen recovery technologies at centralised wastewater and sludge treatment facilities, the overall results present high sensitivity in the mass of inflow nutrients (TKN, TPH, COD) and in the energy demand for wastewater treatment. The driving force behind the variations in this case is the carbon intensive thermal hydrolysis and air stripping methods, of which the resources input are proportionate to the nutrient inflow and recovered equivalent substances.

The current analysis reflects some general trends and facts that are readily employable within both academic and industry environment and are valid to any community within the urban environment. However, the ability to draw general conclusions based on this work is, to some extent, limited since the results are case-specific and the values correspond to local topography, water resources quantity and quality, wastewater treatment methods, sludge management methods and climate conditions. The magnitude of the comparative results can vary from case to case and the current environmental burdens may be benefits in another occasion. For instance, the distance between WWTP and sludge mono-incineration plant is an important criterion for the selection of sludge mono-incineration for phosphorus recovery as a sludge management method. Even though this process is an attractive option to limit the Haber–Bosch process in WWTPs and offset significant amounts of energy from the wastewater treatment, it can only be an attractive resource recovery option in case of on-site application (mono-incineration plant adjacent to WWTP). Another example is the overall positive impact of disposing food waste into sewers compared to a baseline of food waste co-incineration which can be even greater when compared to cases with landfill disposal. Moreover, the magnitude of environmental benefits of water conservation strategies in the water purification sector would be lower in case of groundwater and not surface water (i.e., Rhine canal) management due to the lower requirements for treatment (energy/chemical input).

## 6.2. Limitations

Even though the research was well-supported by data (literature studies, simulations) regarding the drinking water supply sector and domestic water end-use, the next stages of the centralised wastewater and sludge management had significant gaps of quantified data. Lack of data from models regarding the WWTP operation and performance to different inflow wastewater qualities limited accuracy and increased uncertainty. The processes of the WWTP were modelled based on the assumption that the nutrient balance of TKN, TPH and COD remains the same (baseline level) with regards to the proportions of outflow categories (biogas, sludge waste, recovered substances, effluent). For more details regarding the nutrient balance see Appendix C.6. This assumption was, for instance, the reason of the increased eutrophication potential of WWTP in case of food waste valorisation via the sewer system. Another assumption was that the energy requirements for wastewater treatment are not affected by changes in treatment efficiency. This leads, for example, in underestimation of the energy benefit in biological treatment in case of nitrogen recovery from digester reject water. Furthermore, for the simulation of the sludge management methods, a combination of data from different literature studies was used with probability for overestimation of some of the resource output (e.g., airborne emissions, waste streams for treatment). Specific assumptions are also presented along Appendix C.

## 6.3. Recommendations for future research

Taking into account the outcomes and the limitations of this work, the following issues should be addressed in the future:

I. It is important to create integrated models which will simulate the entire urban water system, from drinking water supply until wastewater treatment, under conditions of water conservation and food waste valorisation via the wastewater streams. A complete assessment of such systems needs quantitative and qualitative data regarding the changes in the water end-use, wastewater generation, but also wastewater treatment operation and treatment efficiency.

II. Detailed information on materials and energy used and recovered, as well as the environmental emissions produced (e.g., airborne emissions, nutrient releases) during the sludge incineration and sludge drying processes have to be thoroughly investigated to refine the results of this study. Furthermore, modules simulating these specific processes should be included in SimaPro database to enhance the modelling done of these unit processes.

III. Future LCA studies should model the scenarios for different technology penetration rates to better reflect the possible diffusion of the strategies and give a more realistic dimension to the magnitude of the potential environmental impacts. Furthermore, greywater and rainwater recycling strategies could be investigated under different scales of implementation apart from a single household (e.g., community, neighbourhood).

IV. The combination of the LCA with a Life Cycle Cost Analysis would be ideal to portray a complete life cycle management which can support decision-making on a long-term basis.

## 6.4. Implementation road map and recommendations

Food waste valorisation via the sewer system is a very questioned approach. This strategy can provide high opportunity to maximise energy and nutrient recovery from wastewater and sludge waste streams. However, the WWTP operation needs to be revised to minimise the nutrient outflow via effluent (mitigate freshwater eutrophication potential) and make better use of the available nutrients in the sludge treatment stage. This strategy can also cause technical malfunctions and failures to the sewer network. The high sewer residence time due to the long distance that sewage has to travel between household and WWTP results in decomposition of the organic matter which can cause technical complications on the sewer network, such as biofilm formation, sulfide production and corrosion (Zan et al., 2019).

For every problem there is a solution, but the question is if these solutions are worth implementing. The high residence time can be minimised by applying decentralised wastewater treatment nearby the community. However, decentralised WWTPs require open space availability which is a major concern in densely populated urban areas. Furthermore, since there are no decentralised anaerobic digesters available yet (Tarpani and Azapagic, 2018), the bulky primary sludge has to be transported in centralised digestion facilities. In case of direct diversion of the food scraps to centralised WWTPs via the sewer, the network has to be redesigned and a separate infrastructure needs to be constructed to transport the food waste stream. Taking into account all of these challenges, it is advised this strategy to be implemented either in communities which are located close to centralised WWTPs or in new build communities with infrastructure adjusted to the hydraulic changes.

The low drinking water supply to the households under water conservation strategies and the increased concentration of wastewater due to low dilution in some of them (greywater reuse, water saving devices) hinder the water and wastewater transport. Changes in the hydraulic performance of the water system requires redesign of the water distribution and wastewater collection network (e.g., smaller diameters, changes in slope). Typically, large infrastructure systems, such as water transport systems, are dynamically stable and resist change (Agudelo-Vera et al., 2014a). Such changes can also be unfeasible and worthless from technical (and economic) perspective. For instance, increase of the slope in a sewer pipeline to allow better flow and prevent sedimentation and clogging can be impossible in low-lying areas (e.g., in the Netherlands). Moreover, the replacement of a relatively new network with pipelines of smaller diameter it may not worth implementing, and priority should be networks that have already reached their service life.

There is an environmental need to reduce global warming, and renewable energy cannot solely resolve the problem. Global warming is becoming more of a current focus and its impacts reduction requires simultaneous reduction of the energy consumption. The opportunity for high energy efficiency by the usage of water efficient devices at a household level, but also the availability of various devices in the market which do not disturb the users' behaviour, comfort and personal hygiene make the domestic water heating the most attractive point of improvement.

In this research, the scenarios were modelled for 100% technology penetration, implementation period of 2040-2050 and linear correlation with time. The smooth transition towards the adoption of water efficient appliances has to be supported by a concerted effort of actors (i.e., water authorities, drinking water companies, municipalities) and social groups, rather than held by the state or by an individual actor (Agudelo-Vera et al., 2014b). National guidelines and collaboration initiatives between the government, water companies, energy companies, manufacturers of innovations and other relevant market parties can enhance the adoption of water saving devices and slow down the increasing hot water use.

The diffusion of the strategy can start by arranging awareness campaigns, education of the public and advertising the innovations which aim to increase the comfort and lower water and energy bills. Changes in legislation and building code should also make the usage of water efficient devices (energy label class A) mandatory in every renovated or newly build household. At the same time, government has to support the use of such devices with subsidies, due to their high trade cost (e.g., waterless washing machines/dishwashers). Furthermore, as the rate of installation is relatively slow, water companies can monitor trends and identify key actors, drivers and barriers for steering the technological transition to guarantee reliable and sustainable drinking water systems (Agudelo-Vera et al., 2014a).

As a general conclusion, the primary focus has to be residential areas and not centralised sanitary infrastructures. The new building guidelines ought to be revised and include the installation of water saving devices as obligatory condition. Moreover, in new build communities, the water and sewer networks have to be designed as enablers of water conservation strategies to make the transition towards water-related energy efficiency a success story three decades from now.

# 7. Conclusions

The results revealed that resource recovery oriented solutions are not always resource efficient and a life cycle assessment is vital before lock-in measures occur. Based on the case-specific results of the analysis, the following conclusions can be drawn:

- The usage of water efficient devices (low-flush toilets, water efficient shower heads, waterless washing machines, waterless dishwashers) at a household level is the most effective intervention for transitioning towards resource efficient urban water systems. This intervention coupled with struvite recovery, biogas recovery and sludge waste drying for energy recovery at centralised wastewater and sludge treatment facilities promote a complete and robust urban water system which maximises the water, energy, nutrients utilization with the least possible environmental emissions. The adoption of this strategy can reduce the life cycle energy consumption by 79%, life cycle global warming potential by 72%, life cycle freshwater eutrophication potential by 22% and life cycle water consumption by 56%.

- Food waste valorisation via wastewater streams increases life cycle freshwater eutrophication potential due to the increased nutrient (TKN, TPH, COD) discharge via the effluent. On the other hand, it provides great opportunities for nutrient and energy recovery at a centralised wastewater and sludge treatment level.

- Greywater reuse outperforms rainwater harvesting in regards to life cycle energy consumption, life cycle global warming potential and life cycle water consumption, mainly due to lower demands for wastewater treatment, and higher efficiency of biogas and struvite recovery. However, rainwater harvesting presents a lower life cycle freshwater eutrophication potential, mainly due to lower nutrient inflow in the WWTP, and thus, lower nutrient discharge to recipient water bodies.

- Thermal hydrolysis of sludge and air stripping as methods of nitrogen recovery, despite offsetting N-based chemical fertilisers and producing a high biogas yield, vastly increase the environmental burdens, from every perspective, and demonstrate a weak solution of resource recovery. Since the environmental profile is sensitive to the energy requirements of these two methods and the selected inventory was based on pilot scale systems, a full-scale application might promote a more efficient method.

- The combination of sludge mono-incineration and phosphorus recovery from sludge ash can be a more efficient sludge management method compared to sludge co-incineration in regards to life cycle energy consumption, life cycle global warming potential and life cycle freshwater eutrophication potential, only in case of on-site application.

## References


F. A. Abdulrahman, A. H. Abdo, and I. Y. AL-FILALI. Waterless dish washer, 2011. URL https://patents.google.com/patent/US20140102485A1/en.

C. Agudelo-Vera and E. Blokker. How future proof is our drinking water infrastructure? *KWR, Nieuwegein, Netherlands*, 2014.

C. Agudelo-Vera, E. Blokker, C. Büscher, and J. Vreeburg. Analysing the dynamics of transitions in residential water consumption in the netherlands. *Water Science and Technology: Water Supply*, 14(5):717–727, 2014a.

C. Agudelo-Vera, E. Blokker, E. Pieterse-Quirijns, and W. Scheffer. Water and energy nexus at the building level. *REHVA European HVAC Journal*, pages 12–15, 2014b.

C. Agudelo-Vera, M. Blokker, J. Vreeburg, H. Vogelaar, S. Hillegers, and J. P. van der Hoek. Testing the robustness of two water distribution system layouts under changing drinking water demand. *Journal of Water Resources Planning and Management*, 142 (8):05016003, 2016.

S. Ahilan, P. Melville-Shreeve, Z. Kapelan, and D. Butler. The influence of household rainwater harvesting system design on water supply and stormwater management efficiency. In *International Conference on Urban Drainage Modelling*, pages 369–374. Springer, 2018.

A. Amann, O. Zoboli, J. Krampe, H. Rechberger, M. Zessner, and L. Egle. Environmental impacts of phosphorus recovery from municipal wastewater. *Resources, Conservation and Recycling*, 130:127–139, 2018.

AMS. New urban water transport systems. URL https://www.ams-institute.org/urban-challenges/circularity-urban-regions/new-urban-water-transport-systems/.

S. Antonini, S. Paris, T. Eichert, and J. Clemens. Nitrogen and phosphorus recovery from human urine by struvite precipitation and air stripping in vietnam. *CLEAN–Soil, Air, Water*, 39(12):1099–1104, 2011.

K. I. Ashley, D. S. Mavinic, F. A. Koch, et al. International conference on nutrient recovery from wastewater streams. In *International Conference on Nutrient Recovery from Wastewater Streams (2009: Vancouver, BC)*. IWA Publishing, 2009.

O. Bailey, T. Arnot, E. Blokker, Z. Kapelan, and J. Hofman. Predicting impacts of water conservation with a stochastic sewer model. *Water Science and Technology*, 80(11): 2148–2157, 2019.

O. Bailey, L. Zlatanovic, J. P. van der Hoek, Z. Kapelan, M. Blokker, T. Arnot, and J. Hofman. A stochastic model to predict flow, nutrient and temperature changes in a sewer under water conservation scenarios. *Water*, 12(4):1187, 2020.

C. Banks, S. Heaven, Y. Zhang, and U. Baier. Food waste digestion: anaerobic digestion of food waste for a circular economy. 2018.

J. C. Bare, P. Hofstetter, D. W. Pennington, and H. A. U. De Haes. Midpoints versus endpoints: the sacrifices and benefits. *The International Journal of Life Cycle Assessment*, 5(6):319, 2000.

R. Barrios, M. Siebel, A. Van der Helm, K. Bosklopper, and H. Gijzen. Environmental and financial impact assessment of two plants of amsterdam water supply. *Journal of Cleaner Production*, 2004.

R. Barrios, M. Siebel, A. van der Helm, K. Bosklopper, and H. Gijzen. Environmental and financial life cycle impact assessment of drinking water production at waternet. *Journal of Cleaner Production*, 16(4):471–476, 2008.

M. Blanky, Y. Sharaby, S. Rodríguez-Martínez, M. Halpern, and E. Friedler. Greywater reuse-assessment of the health risk induced by legionella pneumophila. *Water research*, 125:410–417, 2017.

Blue Mountain Co. First Flush Delta technology. URL https://rainharvesting.com.au/products/first-flush-diverters/first-flush-delta-diverters/first-flush-delta-post-wall/.

T. G. Bosklopper, L. Rietveld, R. Babuska, B. Smaal, and J. Timmer. Integrated operation of drinking water treatment plant at amsterdam water supply. *Water Science and Technology: Water Supply*, 4(5-6):263–270, 2004.

E. Brilhuis-Meijer and M. Goedkoop. LCA And C2C: Friends Or Foes? | PRé Sustainability, 2015. URL https://www.pre-sustainability.com/news/lca-and-c2c-friends-or-foes.

A. Campisano, D. Butler, S. Ward, M. J. Burns, E. Friedler, K. DeBusk, L. N. Fisher-Jeffes, E. Ghisi, A. Rahman, H. Furumai, et al. Urban rainwater harvesting systems: Research, implementation and future perspectives. *Water research*, 115:195–209, 2017.

C. Cao. Sustainability and life assessment of high strength natural fibre composites in construction. In *Advanced High Strength Natural Fibre Composites in Construction*, pages 529–544. Elsevier, 2017.

CBS Netherlands. GHG emissions Netherlands, 2019. URL https://www.cbs.nl/en-gb/news/2019/19/slight-drop-in-greenhouse-gas-emissions.

CBS Netherlands. CBS Netherlands, 2020. URL https://www.cbs.nl/en-gb/news/2020/12/household-and-farm-water-usage-surged-in-2018.

S. G. Center. Basic data on biogas. *Available Online, Accessed January*, 2012.

E. Chiou. An improved model of calcium carbonate crystallization: An improved kinetics-model for the calcium carbonate crystallization in the fluidized bed softening reactors at the weesperkarspel drinking water treatment plant. 2018.

City of Amsterdam. Amsterdam. URL https://www.amsterdam.nl/en/policy/sustainability/.

CIWM. Energy from waste. Technical report, 2005. URL http://www.dnrec.delaware.gov/SWMTWG/Documents/46fb136292274f30b8be70817244585d152NetherlandsWasteFromEnergyFacilities.pdf.

A. Clarke, N. Grant, J. Thornton, and E. Solutions. Quantifying the energy and carbon effects of water saving. *Environment Agency*, 2009.

A. G. Coche, J. F. Muir, and T. Laughlin. *Simple methods for aquaculture: management for freshwater fish culture ponds and water practices*, volume 21. Food & Agriculture Org., 1996.

E.-E. Commission et al. Disposal and recycling routes for sewage sludge, 2010.

A. Coudard. Food waste through the food-water-energy nexus lenses: A case study of amsterdam. 2019.

H. Cruz, Y. Y. Law, J. S. Guest, K. Rabaey, D. Batstone, B. Laycock, W. Verstraete, and I. Pikaar. Mainstream ammonium recovery to advance sustainable urban wastewater management. *Environmental science & technology*, 53(19):11066–11079, 2019.

L. De Clercq, E. Michels, E. Meers, J. Buysse, and A. Haumont. Legal framework of recovered phosphorus (struvite) as fertiliser in north-western europe. *Document number: BIOREFINE–WP5–A19–P1-D. Ghent, Belgium*, 2015.

R. De Jong. Governance of phosphate recovery from wastewater in amsterdam. 2017.

R. De Ruiter. The ecophos technology to close the p cycle and to safeguard the world's food chain. In *Abwasser–Phosphor–Dünger-Workshop inclusive Fachgespräch zum UFOPLAN-Projekt Klärschlammaschemonitoring, Berlin*, pages 28–29, 2014.

A. Del Borghi. Lca and communication: environmental product declaration, 2013.

E. Dijkgraaf and R. Gradus. Waste management in the netherlands. In *Handbook on waste management*. Edward Elgar Publishing, 2014.

A. M. Dixon, D. Butler, and A. Fewkes. Guidelines for greywater re-use: Health issues. *Water and Environment Journal*, 13(5):322–326, 1999.

P. D'Odorico, K. F. Davis, L. Rosa, J. A. Carr, D. Chiarelli, J. Dell'Angelo, J. Gephart, G. K. MacDonald, D. A. Seekell, S. Suweis, et al. The global food-energy-water nexus. *Reviews of geophysics*, 56(3):456–531, 2018.

S. Dominguez, J. Laso, M. Margallo, R. Aldaco, M. J. Rivero, Á. Irabien, and I. Ortiz. Lca of greywater management within a water circular economy restorative thinking framework. *Science of the Total Environment*, 621:1047–1056, 2018.

S. Donatello and C. R. Cheeseman. Recycling and recovery routes for incinerated sewage sludge ash (issa): A review. *Waste Management*, 33(11):2328–2340, 2013.

R. v. Duinen, T. Filatova, P. Geurts, and A. v. d. Veen. Empirical analysis of farmers' drought risk perception: Objective factors, personal circumstances, and social influence. *Risk analysis*, 35(4):741–755, 2015.

T. E. F. o. N. A. o. W. S. EurEau. The governance of water services in Europe, 2018. URL https://www.eureau.org/resources/news/1-the-governance-of-water-services-in-europe.

European Commission. Circular Economy Action Plan, 2020. URL https://eur-lex.europa.eu/resource.html?uri=cellar:9903b325-6388-11ea-b735-01aa75ed71a1.0017.02/DOC_1&format=PDF.

European Parliament. Directive (EU) 2018/851 on waste, 2018. URL https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:32018L0851.

T. D. Evans, P. Andersson, Å. Wievegg, and I. Carlsson. Surahammar: a case study of the impacts of installing food waste disposers in 50% of households. *Water and Environment Journal*, 24(4):309–319, 2010.

A. Fidar, F. Memon, and D. Butler. Environmental implications of water efficient microcomponents in residential buildings. *Science of the total environment*, 408(23): 5828–5835, 2010.

FLO-TEK. UPVC pressure pipes, a. URL https://flotekafrica.com/pvc-pressure-pipes/.

FLO-TEK. UPVC sewer pipes, b. URL https://flotekafrica.com/sewer-pvc-sewer/.

D. J. Flower, V. G. Mitchell, G. P. Codner, et al. Urban water systems: drivers of climate change? *Rainwater and Urban Design 2007*, page 274, 2007.

H. Fooij. Wastewater as a resource: strategies to recover resources from amsterdam's wastewater. Master's thesis, University of Twente, 2015.

W. E. Forum/PwC. Circular economy in cities evolving the model for a sustainable urban future. World Economic Forum Geneva, Switzerland, 2018.

J. Frijns, M. Mulder, and J. Roorda. *Towards a climate neutral water chain*. STOWA, 2008.

R. Frischknecht, F. Wyss, S. B. Knöpfel, T. Lützkendorf, and M. Balouktsi. Cumulative energy demand in lca: the energy harvested approach. *The International Journal of Life Cycle Assessment*, 20(7):957–969, 2015.

P. Gerbens-Leenes. Energy for freshwater supply, use and disposal in the netherlands: a case study of dutch households. *International journal of water resources development*, 32(3):398–411, 2016.

I. P. Geudens and J. van Grootveld. Dutch Drinking Water Statistics 2017, 2017. URL https://www.vewin.nl/SiteCollectionDocuments/Publicaties/Cijfers/Drinkwaterstatistieken-2017-EN.pdf.

M. Giezen. Shifting infrastructure landscapes in a circular economy: An institutional work analysis of the water and energy sector. *Sustainability*, 10(10):3487, 2018.

G. D. Gikas and V. A. Tsihrintzis. Assessment of water quality of first-flush roof runoff and harvested rainwater. *Journal of Hydrology*, 466:115–126, 2012.

M. Goedkoop, R. Heijungs, M. Huijbregts, A. De Schryver, J. Struijs, and R. Van Zelm. Recipe 2008. *A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level*, 1:1–126, 2009.

Government of the Netherlands. A Circular Economy in the Netherlands by 2050. Technical report, 2016. URL https://www.government.nl/documents/policy-notes/2016/09/14/a-circular-economy-in-the-netherlands-by-2050.

R. D. Graaf and A. v. Hell. Nieuwe sanitatie noorderhoek, sneek: Deelonderzoeken. *Amersfoort, The Netherlands. Rapport*, 48, 2014.

R. D. Gulati and E. Van Donk. Lakes in the netherlands, their origin, eutrophication and restoration: state-of-the-art review. In *Ecological Restoration of Aquatic and Semi-Aquatic Ecosystems in the Netherlands (NW Europe)*, pages 73–106. Springer, 2002.

H. Guven, O. Eriksson, Z. Wang, and I. Ozturk. Life cycle assessment of upgrading options of a preliminary wastewater treatment plant including food waste addition. *Water research*, 145:518–530, 2018.

M. Haupt and M. Zschokke. How can lca support the circular economy?—63rd discussion forum on life cycle assessment, zurich, switzerland, november 30, 2016. *The International Journal of Life Cycle Assessment*, 22(5):832–837, 2017.

J. Hofman and M. Paalman. *Rainwater harvesting, a sustainable solution for urban climate adaptation?* KWR Watercycle Research Institute, 2014.

A. Hospido, T. Moreira, M. Martín, M. Rigola, and G. Feijoo. Environmental evaluation of different treatment processes for sludge from urban wastewater treatments: Anaerobic digestion versus thermal processes (10 pp). *The International Journal of Life Cycle Assessment*, 10(5):336–345, 2005.

E. Iacovidou, J. Millward-Hopkins, J. Busch, P. Purnell, C. A. Velis, J. N. Hahladakis, O. Zwirner, and A. Brown. A pathway to circular economy: Developing a conceptual framework for complex value assessment of resources recovered from waste. *Journal of cleaner production*, 168:1279–1288, 2017.

I. E. A. IEA. Electricity generation in the Netherlands 2018, 2018. URL https://www.iea.org/countries/the-netherlands.

A. Iqbal, G. A. Ekama, F. Zan, X. Liu, H.-K. Chui, and G.-H. Chen. Potential for co-disposal and treatment of food waste with sewage: A plant-wide steady-state model evaluation. *Water Research*, page 116175, 2020.

ITACA. An introduction to slow sand filtration, 2005. URL https://www.itacanet.org/eng/water/Section%206%20Water%20treatment/ssfintro.pdf.

N. Jardin, D. Thöle, and B. Wett. Treatment of sludge return liquors: experiences from the operation of full-scale plants. *Proceedings of the Water Environment Federation*, 2006(7):5237–5255, 2006.

H. Jeong, O. A. Broesicke, B. Drew, and J. C. Crittenden. Life cycle assessment of small-scale greywater reclamation systems combined with conventional centralized water systems for the city of atlanta, georgia. *Journal of Cleaner Production*, 174:333–342, 2018.

A. Kalbusch and E. Ghisi. Comparative life-cycle assessment of ordinary and water-saving taps. *Journal of Cleaner Production*, 112:4585–4593, 2016.

S. J. Kenway. *The water-energy nexus and urban metabolism-connections in cities*. Urban Water Security Research Alliance Brisbane, 2013.

S. J. Kenway, R. Scheidegger, T. A. Larsen, P. Lant, and H.-P. Bader. Water-related energy in households: a model designed to understand the current state and simulate possible measures. *Energy and Buildings*, 58:378–389, 2013.

H. H. Khoo, T. Z. Lim, and R. B. Tan. Food waste conversion options in singapore: environmental impacts based on an lca perspective. *Science of the total environment*, 408(6):1367–1373, 2010.

I. Kirhensteine, V. Cherrier, N. Jarrit, A. Farmer, G. Paoli, G. Delacamera, and A. Psomas. Eu-level instruments on water reuse. final report to support the commission's impact assessment. *Luxemburgo: European Union*, 2016.

H. Kjerstadius, A. B. Saraiva, J. Spångberg, and Å. Davidsson. Carbon footprint of urban source separation for nutrient recovery. *Journal of environmental management*, 197:250–257, 2017.

Y. Kobayashi, N. J. Ashbolt, E. G. Davies, and Y. Liu. Life cycle assessment of decentralized greywater treatment systems with reuse at different scales in cold regions. *Environment international*, 134:105215, 2020.

J. Köhler. Detergent phosphates: an eu policy assessment. *Journal of business chemistry*, 3(2), 2006.

Y. Krozer, S. Hophmayer-Tokich, H. van Meerendonk, S. Tijsma, and E. Vos. Innovations in the water chain–experiences in the netherlands. *Journal of Cleaner Production*, 18 (5):439–446, 2010.

K. L. Lam, L. Zlatanović, and J. P. van der Hoek. Life cycle assessment of nutrient recycling from wastewater: A critical review. *Water Research*, 173:115519, 2020.

Y. Lechón, C. De La Rúa, and H. Cabal. Impacts of decarbonisation on the water-energy-land (wel) nexus: a case study of the spanish electricity sector. *Energies*, 11(5):1203, 2018.

H. Li, C. Jin, Z. Zhang, I. O'Hara, and S. Mundree. Environmental and economic life cycle assessment of energy recovery from sewage sludge through different anaerobic digestion pathways. *Energy*, 126:649–657, 2017.

S. LUCASSEN. Animal-based measures: A step towards rights for farm animals? *Society Register*, 3(3):159–166, 2019.

M. Lürling and M. Mucci. Mitigating eutrophication nuisance: in-lake measures are becoming inevitable in eutrophic waters in the netherlands. *Hydrobiologia*, 2020.

N. Marashlian and M. El-Fadel. The effect of food waste disposers on municipal waste and wastewater management. *Waste management & research*, 23(1):20–31, 2005.

L. F. Marmolejo, L. F. Diaz, P. Torres, and M. García. Perspectives for sustainable resource recovery from municipal solid waste in developing countries: Application and alternatives. *Waste Manag.–An integrated vision, Edition*, 1:153–166, 2012.

B. Martinson and T. Thomas. Quantifying the first-flush phenomenon: effects of first-flush on water yield and quality. In *14th International Rainwater Catchment Systems Conference*, 2009.

K. Matsubae, E. Webeck, K. Nansai, K. Nakajima, M. Tanaka, and T. Nagasaka. Hidden phosphorus flows related with non-agriculture industrial activities: A focus on steelmaking and metal surface treatment. *Resources, Conservation and Recycling*, 105:360–367, 2015.

W. McDonough and M. Braungart. *Cradle to cradle: Remaking the way we make things*. North point press, 2010.

F. Memon, Z. Zheng, D. Butler, C. Shirley-Smith, S. Lui, C. Makropoulos, and L. Avery. Life cycle impact assessment of greywater recycling technologies for new developments. *Environmental monitoring and assessment*, 129(1-3):27–35, 2007.

N. Mills, P. Pearce, J. Farrow, R. Thorpe, and N. Kirkby. Environmental & economic life cycle assessment of current & future sewage sludge to energy technologies. *Waste management*, 34(1):185–195, 2014.

Ministry of Economic Affairs and Climate. Energy Report Transition to sustainable energy, 2016. URL https://www.government.nl/documents/reports/2016/04/28/energy-report-transition-tot-sustainable-energy.

Ministry of Economic Affairs and Climate. Energy agenda: towards a low-carbon energy supply, 2017. URL https://www.government.nl/documents/reports/2017/03/01/energy-agenda-towards-a-low-carbon-energy-supply.

B. Mogoş. Prinseneiland, amsterdam: a water balance. Master's thesis, 2018.

M. Nagler, P. Aichinger, M. Kuprian, T. Pümpel, H. Insam, and C. Ebner. A case study for a cost-benefit-based, stepwise optimization of thermo-chemical was pre-treatment for anaerobic digestion. *Journal of Material Cycles and Waste Management*, 20(1): 266–273, 2018.

T. Nakakubo, A. Tokai, and K. Ohno. Comparative assessment of technological systems for recycling sludge and food waste aimed at greenhouse gas emissions reduction and phosphorus recovery. *Journal of Cleaner Production*, 32:157–172, 2012.

K. S. Ng, A. Yang, and N. Yakovleva. Sustainable waste management through synergistic utilisation of commercial and domestic organic waste for efficient resource recovery and valorisation in the uk. *Journal of Cleaner Production*, 227:248–262, 2019.

E. Nolde. Greywater recycling systems in germany—results, experiences and guidelines. *Water Science and Technology*, 51(10):203–210, 2005.

M. Oosterhuis, D. Ringoot, A. Hendriks, and P. Roeleveld. Thermal hydrolysis of waste activated sludge at hengelo wastewater treatment plant, the netherlands. *Water science and technology*, 70(1):1–7, 2014.

S. Parma. The history of the eutrophication concept and the eutrophication in the netherlands. *Hydrobiological Bulletin*, 14(1-2):5–11, 1980.

T. P. T. Pham, R. Kaushik, G. K. Parshetti, R. Mahmood, and R. Balasubramanian. Food waste-to-energy conversion technologies: Current status and future directions. *Waste Management*, 38:399–408, 2015.

P. Phothilangka, M. Schoen, and B. Wett. Benefits and drawbacks of thermal pre-hydrolysis for operational performance of wastewater treatment plants. *Water science and technology*, 58(8):1547–1553, 2008.

A. Pinasseau, B. Zerger, J. Roth, M. Canova, and S. Roudier. Best available techniques (bat) reference document for waste treatment. *European Commission Industrial Emissions Directive*, 75, 2010.

F. M. Pinkster and W. R. Boterman. When the spell is broken: gentrification, urban tourism and privileged discontent in the amsterdam canal district. *cultural geographies*, 24(3):457–472, 2017.

M. Pradel and A. Reverdy. Assessing ghg emissions from sludge treatment and disposal routes: the method behind gestaboues tool. 2012.

B. Prajapati. *Decentralized grey water reuse: Developing grey water treatment technology and mapping end-users' perceptions*. PhD thesis, Department of Geosciences and Natural Resource Management, Faculty of . . . , 2018.

F. M. Pulselli, F. Ciampalini, C. Leipert, and E. Tiezzi. Integrating methods for the environmental sustainability: The spin-eco project in the province of siena (italy). *Journal of environmental management*, 86(2):332–341, 2008.

A. I. Racoviceanu and B. W. Karney. Life-cycle perspective on residential water conservation strategies. *Journal of Infrastructure Systems*, 16(1):40–49, 2010.

F. research. Wood pellets calorific value, 2020. URL https://www.forestresearch.gov.uk/tools-and-resources/biomass-energy-resources/reference-biomass/facts-figures/typical-calorific-values-of-fuels/.

P. Roberts and N. Davies. The potential of food waste disposal units to reduce costs, 2012. URL https://www.local.gov.uk/sites/default/files/documents/potential-food-waste-disp-077.pdf.

K. Roest, P. Smeets, T. van den Brand, H. Cortial, and E. Klaversma. Tki loop-closure cleantech playground, 2016.

P. J. Rutten. The urban water cycle: A case study of the prinseneiland, amsterdam. 2013.

C. Ruz. The six natural resources most drained by our 7 billion people. *The Guardian*, 31, 2011.

S. Sengupta, T. Nawaz, and J. Beaudry. Nitrogen and phosphorus recovery from wastewater. *Current Pollution Reports*, 1(3):155–166, 2015.

J. Sijm, P. Gockel, A. Welle, and W. v. Westering. Demand and supply of flexibility in the power system of the netherlands, 2015-2050. key messages. *Policy Studies*, 2017: 2016, 2018.

A. Silva, E. Yogafanny, and S. Fuchs. Intermittent slow sand filtration for drinking water treatment in developing countries. 2012.

SimaPro. SimaPro | The World's Leading LCA Software. URL https://simapro.com/.

SimaPro. Updated Impact Assessment Methodology ReCiPe 2016, 2017. URL https://simapro.com/2017/updated-impact-assessment-methodology-recipe-2016/.

P. Simoës and H. Veldman. Ecostiler–cogeneration with biogas in the district heating system of amsterdam, netherlands. In *1st European conference on polygeneration, Tarragona, Spain*, 2007.

R. Sitzenfrei, J. Zischg, M. Sitzmann, and P. M. Bach. Impact of hybrid water supply on the centralised water system. *Water*, 9(11):855, 2017.

A. Smit, J. Van Middelkoop, W. Van Dijk, and H. Van Reuler. A substance flow analysis of phosphorus in the food production, processing and consumption system of the netherlands. *Nutrient Cycling in Agroecosystems*, 103(1):1–13, 2015.

R. Sobhan. An improved kinetic model and optimized configurations for pellet softening. 2019.

R. S. Srinivasan, W. W. Braham, D. E. Campbell, D. C. Curcija, and M. Rinker. Sustainability assessment frameworks, evaluation tools and metrics for buildings and its environment–a review. In *Proceedings of Building Simulation: 12th Conference of Intl Building Performance Simulation Association, 2011, Sydney*, 2011.

Statista. Average monthly rainfall in the Netherlands as of 2017, 2020. URL https://www.statista.com/statistics/802733/average-monthly-rainfall-netherlands/.

Stauffer, B. and Spuhler, D. Incineration (Large-scale), 2019. URL https://sswm.info/water-nutrient-cycle/wastewater-treatment/hardwares/sludge-treatment/incineration-%28large-scale%29.

M. Stavenhagen, J. Buurman, and C. Tortajada. Saving water in cities: Assessing policies for residential water demand management in four cities in europe. *Cities*, 79:187–195, 2018.

R. R. Z. Tarpani and A. Azapagic. Life cycle costs of advanced treatment techniques for wastewater reuse and resource recovery from sewage sludge. *Journal of Cleaner Production*, 204:832–847, 2018.

S. K. Teoh and L. Y. Li. Feasibility of alternative sewage sludge treatment methods from a lifecycle assessment (lca) perspective. *Journal of Cleaner Production*, 247:119495, 2020.

P. Tidåker, E. Kärrman, A. Baky, and H. Jönsson. Wastewater management integrated with farming–an environmental systems analysis of a swedish country town. *Resources, conservation and recycling*, 47(4):295–315, 2006.

T. Tjin-A-Tsoi. The Netherlands on the European scale 2019. 2019. URL https://longreads.cbs.nl/european-scale-2019/.

D. Tonini, A. Wandl, K. Meister, P. M. Unceta, S. E. Taelman, D. Sanjuan-Delmás, J. Dewulf, and D. Huygens. Quantitative sustainability assessment of household food waste management in the amsterdam metropolitan area. *Resources, Conservation and Recycling*, 160:104854, 2020.

S. Umapathi, M. N. Chong, and A. K. Sharma. Evaluation of plumbed rainwater tanks in households for sustainable water resource management: a real-time monitoring study. *Journal of Cleaner Production*, 42:204–214, 2013.

W. P. R. UN. Amsterdam population, 2020. URL https://worldpopulationreview.com/world-cities/amsterdam-population.

D. Đurđević, P. Blecich, and Ž. Jurić. Energy recovery from sewage sludge: The case study of croatia. *Energies*, 12(10):1927, 2019.

J. P. van der Hoek, H. de Fooij, and A. Struker. Wastewater as a resource: Strategies to recover resources from amsterdam's wastewater. *Resources, Conservation and Recycling*, 113:53–64, 2016.

J. P. van der Hoek, v. d. A. Struker, and J. De Danschutter. Amsterdam as a sustainable european metropolis: integration of water, energy and material flows. *Urban Water Journal*, 14(1):61–68, 2017.

J. P. van der Hoek, R. Duijff, and O. Reinstra. Nitrogen recovery from wastewater: Possibilities, competition with other resources, and adaptation pathways. *Sustainability*, 10(12):4605, 2018a.

J. P. van der Hoek, S. Mol, S. Giorgi, J. I. Ahmad, G. Liu, and G. Medema. Energy recovery from the water cycle: Thermal energy from drinking water. *Energy*, 162: 977–987, 2018b.

C. Van Dooren, O. Janmaat, J. Snoek, and M. Schrijnen. Measuring food waste in dutch households: A synthesis of three studies. *Waste Management*, 94:153–164, 2019.

R. van Duinen, T. Filatova, P. Geurts, and A. van der Veen. Coping with drought risk: empirical analysis of farmers' drought adaptation in the south-west netherlands. *Regional environmental change*, 15(6):1081–1093, 2015.

D. Vanham, T. Mak, and B. Gawlik. Urban food consumption and associated water resources: The example of dutch cities. *Science of the total environment*, 565:232–239, 2016.

A. P. Velenturf and J. S. Jopson. Making the business case for resource recovery. *Science of the total environment*, 648:1031–1041, 2019.

Voedingscentrum. Voedingscentrum. 2019. URL https://www.voedingscentrum.nl/Assets/Uploads/voedingscentrum/Documents/Professionals/Pers/Persmappen/Verspilling%202019/VC_Synthesis%20report%20on%20food%20waste%20in%20Dutch%20households%202019.pdf.

Water Board AGV. The Alkmaar household waste plant will process sewage sludge, 2020. URL https://www.agv.nl/nieuws/2020/juli/huisvuilcentrale-alkmaar-gaat-zuiveringsslib-verwerken/.

WaterWorld. Sludge drying plant, 2013. URL https://www.waterworld.com/international/article/16202292/sludge-drying-plant-uses-cooling-screw-technology.

World Coal Assosiation. Uses of coal, 2020. URL https://www.worldcoal.org/coal/uses-coal/how-steel-produced.

X. Wu and O. Modin. Ammonium recovery from reject water combined with hydrogen production in a bioelectrochemical reactor. *Bioresource technology*, 146:530–536, 2013.

S. Xiang, T. Zhang, and Y. Zhu. Waterless washing machine. URL https://patents.google.com/patent/CN105063960B/en?q=waterless+washing+machine{&}oq=waterless+washing+machine.

X. Xue, T. Hawkins, M. Schoen, J. Garland, and N. Ashbolt. Comparing the life cycle energy consumption, global warming and eutrophication potentials of several water and waste service options. *Water*, 8(4):154, 2016.

S. Zafar. Negative impacts of incinerationbased waste-to-energy technology. retrieved january 21, 2019, alternative energy news, and information about renewable energy technologies, 2018.

F. Zan, Z. Liang, F. Jiang, J. Dai, and G. Chen. Effects of food waste addition on biofilm formation and sulfide production in a gravity sewer. *Water research*, 157:74–82, 2019.

K. Zhou, C. Remy, C. Kabbe, and M. Barjenbruch. Comparative environmental life cycle assessment of phosphorus recovery with different generations of the airprex® systems. *International Journal of Environmental Science and Technology*, 16(5):2427–2440, 2019.

# Appendices

## A. Sustainability tools

The Ecological Footprint Analysis which is a method to measure the consumption of natural resources by the people of a region. Ecological footprint is the total area of productive land or sea required to produce all the materials that people consume on average and provide space to accommodate all the infrastructures (carrying capacity) (Pulselli et al., 2008). It basically represents the land that hosts the ecosystems that support the resource exploitation/extraction and guarantee the absorption of the emissions. This method examines the human consumption trends in relation to the potential availability of ecological goods provided by the area under study, expressed as the ratio between the bio-productive capacity of a certain region and the ecological footprint, determining the so-called ecological deficit or surplus.

The Life Cycle Assessment (LCA) is a well-established method which focuses only on the environmental side of sustainability (Brilhuis-Meijer and Goedkoop, 2015). It is a quantitative approach which considers both direct and indirect processes and their relative input resources, emissions and wastes output, as well as recovered resources aiming to quantify the potential environmental impacts of the whole life cycle of a process or a product, from "cradle to grave". This method leads the process development towards the goal to minimise the resources use, maximise the utilisation of a product and minimise the negative effects per utilisation unit of the product (Pulselli et al., 2008). It can also provide insights of potential trade-offs between different environmental impacts and/or economic performance, if integrated with Life Cycle Costing (Lam et al., 2020).

The Cradle to Cradle (C2C) is an viable alternative to the traditional "cradle to grave" manufacturing model presenting, instead, a model that reflects nature's cycle in which "waste equals food" (McDonough and Braungart, 2010). The method focuses on both social and environmental side of sustainability and uses qualitative criteria, such as health, material re-utilisation, renewable energy and carbon management, water preservation and social fairness (Brilhuis-Meijer and Goedkoop, 2015). When it comes to eco-design, C2C focuses on closing loops and creating added-value products/systems for the environment and society which are "more good, not less bad". It tries to identify what role a product/system can have to fit in this world, answering to the question "How can we provide personal transportation?" instead of "How can we design a better car", turning the focus into the product/system under design (Brilhuis-Meijer and Goedkoop, 2015).

## B. Future outlook

### Circular economy: Moving from theory to a standard practice

The European Commission's Circular Economy Action Plan (European Commission, 2020) sets out 54 ways to close the loop of product life cycles emphasizing on finding innovative means to move away from the current take-make-waste extractive culture, re-define growth and create positive society-wide benefits. The new Action Plan announces initiatives along the entire life cycle of products, targeting on design and promotion circular economy processes, foster sustainable consumption and extent the life cycle of the resources for as long as possible.

The Dutch government, which has always been ambitious when it comes towards sustainability, drew up "transition agendas" and new guidelines for 2050, in line with the EU's, to promote efficient use and reuse of raw materials with the least possible harmful emissions into the environment. The current regulations insufficiently target the transition because the focus much more aims at countering the adverse impacts of waste and emissions, rather than utilising the value of the raw materials. The government-wide program, in collaboration with a variety of stakeholders, aims to a transition from a linear to a circular economy, with an intermediate objective to reduce the use of primary raw materials (minerals, fossils, metals) by 50% by 2030 (Government of the Netherlands, 2016).

Within the endeavour to transform the parasitic cities into circular ones, researchers, public and private institutions have already started to investigate sustainable resource recovery-oriented practices which minimise process input such as water, energy, chemicals, avoid "side-wastes", and maximise recovery efficiency (Lam et al., 2020). As reported by the City of Amsterdam, the immediate plans for the next few years (by 2030) are 80% of the electricity for domestic use to come from solar and wind energy, the consumption of raw materials to be halved and the $CO_2$ emissions to be reduced by 55% compared to levels in 1990. A more ambitious goal for the $CO_2$ emissions has been set for 2050 with reduction by 95%. Amsterdam will no longer use natural gas by 2040, while it will be a fully circular and climate neutral city by 2050.

### Changes in the food chain

The transition to a circular economy is an international challenge and the Dutch government is in line with this. The government-wide program for a circular economy in the Netherlands by 2050 addresses many issues which are under the interest of the current project. Fostering of legislation and regulations is one of the main guidelines with ultimate goal to eliminate regulatory barriers and develop legal frameworks that encourage innovation and support investments. This new guideline focuses on five priorities that fit in with these of the European Commission, with biomass and food chains being among them. A new food system is promoted where raw materials, energy, water and nutrients are utilised efficiently and the natural capital is managed in a sustainable way. A policy to ensure sustainable production and consumption and minimise food waste through sustainable governmental assignments, raising public awareness and monitoring has already been outlined (Government of the Netherlands, 2016). The European Commission prioritises prevention measures which aim to reduce by half the food waste per capita, as well as promotion of the separate collection of the generated food waste and resource recovery (Tonini et al., 2020), (Government of the Netherlands, 2016).

In the last few years, the country is moving away from landfilling as a MSW disposal/treatment method, focusing on the recycling, composting and combustion for power generation (Tjin-A-Tsoi, 2019). However, Tonini et al. (2020) mentions that However, the 60-65% policy target set out for 2030-2035 on the amount of MSW which are sent for reuse or recycling is only likely to be met when incineration of food waste is avoided. While a general management hierarchy is proposed (European Parliament, 2018), the choice of the management scheme has to be situation-dependent to ensure the environmental, economic, and social sustainability at a local level.

**Recycling of raw materials from wastewater**

The "Business with biomass and bio-based gas" Green Deal urged the water authorities to extract and exploit valuable raw materials from wastewater. Although, there are serious concerns related with the use of sewage sludge waste in fertilisers production due to the high concentrations of wastewater in heavy metals, which can be transferred in the food chain. To that extent, the European commission has proposed much stricter controls on the composition of sludge used for spreading (Köhler, 2006). The opportunities for substituting phosphorus recovered from wastewater in fertiliser markets are already applied in a lot of countries for many years, but very recently in the Netherlands.

Changes have been made in the Dutch regulations starting with the Dutch Phosphate Value Chain Agreement that was signed in 2011 by more than 20 companies, knowledge institutes, NGO's and the Dutch Government (De Clercq et al., 2015). Therefore, from the first of January 2015 the category "recovered phosphates" had been added to the Dutch Fertiliser Law as officially published in the Dutch Statute book (De Clercq et al., 2015). The law promoted the creation of a market for recycled phosphorus enabling the usage of three recovered phosphates (i.e., struvite, magnesium phosphate and dicalcium phosphate) from wastewater and agricultural waste as a fertiliser in the Netherlands. On January 1, 2016 the Dutch Fertilisers Act allowed officially the use of struvite from wastewater as a fertiliser. According to Government of the Netherlands (2016), the Dutch government agreed with the proposal of the European Commission to review the Fertilisers Regulation, harmonise the trade requirements for artificial fertilisers to organic fertilisers and soil conditioners, and enable the EU trade of recovered fertilisers such as struvite without a waste label but with a CE marking. This proposal was expected to be adopted in 2018 by the the Netherlands.

**Towards green energy**

The Netherlands imports relatively high amounts of fossils and non-renewable materials (e.g., oil, gas and coal) to satisfy its energy requirements. Renewable energy in the Netherlands comes mainly from biomass, waste, wind, sun, while geothermal and hydro energy play only a minor role in the country. The relatively low share of renewable electricity (15%) in the overall electricity portfolio in 2018 (Tjin-A-Tsoi, 2019) indicates that green electricity has not been a top priority for the Dutch government and policies have been rather ineffective. Nevertheless, the Dutch have one of the most ambitious future targets for a successful participation in Europe's energy transition objectives and climate-change mitigation.

Moving from carbon intensive fossil fuels to more sustainable energy system is an area of major focus in the Netherlands. At the end of 2016, the Dutch government presented its "Energy Agenda", which indicates the policies that should lead to an almost carbon-neutral economy in 2050. Towards 2050, there will be many investments in capital goods with a long life, such as dwellings, electricity production and energy infrastructure. With regard to emissions from buildings, the two main policies are concern better insulation to reduce heat demand and replacement of natural gas by alternative fuels with lower GHG emissions (Ministry of Economic Affairs and Climate, 2016). The power generation will be based mostly on the use of solar and wind energy, smart grinds and stored energy (Ministry of Economic Affairs and Climate, 2017). On the same time, the energy requirements will be limited by adopting energy efficient technologies. For the power system this implies a larger share of electricity in total energy use, flexibility and system integration (Sijm et al., 2018).

Future energy sources of high-temperature heat will be electricity from renewable energy (e.g., wind, solar), electricity from smart grinds, biogas and carbon capture. Also, it is planned the exploitation of residual heat arising from large energy intensive businesses and industries by building public heat grids, steam grids, electricity connections, hydrogen infrastructure and $CO_2$ grids (Ministry of Economic Affairs and Climate, 2017). The Netherlands has a strong energy intensive industrial sector which accounts for nearly 25% of total $CO_2$emissions. The industrial sector can, therefore, play an important role in the transition to a low-carbon economy. Regarding the low-temperature heat, solar boilers, geothermal heat, heat pumps, double glazing, roof and floor insulation, residual heat from waste facilities, biogas/Green Gas from domestic organic waste, heat-cold storage will be some of the sources by 2050 according to (Ministry of Economic Affairs and Climate, 2017).

## C. Data collection and inventory assumptions

### C.1. Energy demand

Due to the future implementation of the scenarios (2040-2050), renewable-based energy sources of electricity and heat were assumed for all scenarios, according to the projections of the water company and the energy transition agenda of the Netherlands for 2050. The table below shows the input data regarding the energy mix.

Sources:*Assumptions based on personal communication with Ruijs Freya (Waternet, 2020) and the future national plans.**(Sijm et al., 2018) ***(Ministry of Economic Affairs and Climate, 2016))

| **Waternet energy mix (mean values)*** | | **National electricity mix (mean values)**** | | **High-temperature heat***** | |
|---|---|---|---|---|---|
| Projections 2050 | | Projections 2050 | | Projections 2050 | |
| Solar | 17.14% | Coal | 0.45% | National electricity grid | 50% |
| Biogas | 19.05% | Natural gas | 18.98% | Biogas | 30% |
| National electricity mix | 63.81% | Hydropower | 0.61% | Steam | 10% |
| | | Biomass | 0.61% | Heat pump (brine water) | 10% |
| | | Solar | 22.81% | | |
| | | Wind | 56.54% | | |

## C.2. Water supply and distribution

The inventory for both water pre-treatment and treatment process is presented on the table below. It has to be noted here that WTPs that produce their own ozone with air instead of pure oxygen, $N_2O$ is formed with the nitrogen from the air, with an average emission of 0.11 gr $N_2O$ (free) per $m^3$ of ozonated water (Frijns et al., 2008). Thus, this emission it was taken into account. The energy requirement for the water distribution was assumed 0.4 $MJ/m^3$ and the total energy for the UV disinfection, membrane filtration, flotation and aeration 4.29 $MJ/m^3$, according to (Gerbens-Leenes, 2016).

The purpose of the drinking water distribution system is to supply water of good quality at adequate pressure and flow. The four design parameters for a network are a minimal pressure, sufficient continuity of supply, meeting the actual drinking water demand and the fire flow demand (Agudelo-Vera et al., 2016). In case of high demand the pipe diameters have to be high enough to provide sufficient remaining pressure in the system to reduce friction losses and ensure that there is sufficient pressure, while in case of low demand, low diameters must ensure that the residence time in the water supply system is not too long to ensure water quality (e.g., bacterial growth with increasing water age, chlorine decay) (Sitzenfrei et al., 2017). The current layout of the water supply system in the area is a looped network with redundant capacity able to provide reliable water supply under critical conditions (pipe breaks, source failures, fire-fighting demand). Although, it has been measured that the current distribution network is oversized and the water-flow velocity is relative low with the water being stagnant most of the time. As a response to this issue and in combination to the low water demand in the three water conservation scenarios (GTR, RHU, ECO), the diameter of the main pipes were reduced from an average of 110 mm to 90 mm, assuming instead of looped, branched system. It is also proved that the reduced diameters were sufficient to transport the increased drinking water demand in the FWV scenario. As material of the pipelines was assumed PVC, with 100 years life expectancy. Data for the calculations of the mass of the PVC pressurised pipes were retrieved from (FLO-TEK, a), based on the total length of approximately 819 m of water distribution network. Based on the diameters of the ring and the average length of the network, they were calculated 6E-04 kg $PVC/m^3$ for the baseline, 7E-04 kg $PVC/m^3$ for the GTR/RHU scenarios, 10E-04 kg $PVC/m^3$ for the ECO and 4E-04 kg $PVC/m^3$ for the FWV scenario. The excavation activities were excluded from the modelling.

Sources: (Roest et al., 2016), modified based on recent operational changes in the treatment processes since 2016 (personal communication, Kramer Onno, (Waternet 2020))

| *Loenderveen pre-treatment plant* | | |
|---|---|---|
| | Normalised values | |
| **Raw water intake** | | |
| power consumption for raw water intake from canal | 0.011552 | kwh/$m^3$ |
| **Coagulation** | | |
| $FeCl_3$ (100%) | 0.038460 | kg/$m^3$ |
| transport $FeCl_3$ from Ibbenburen | 0.008460 | tkm/$m^3$ |
| production dry sludge | 0.034878 | kg/$m^3$ |
| transport dry sludge | 0.001290 | tkm/$m^3$ |
| **Lake reservoir** | | |
| - | | |
| **Pump from lake to sand filtration** | | |
| pump's energy consumption | 0.022320 | kwh/$m^3$ |
| **Sand filtration** | | |
| $H_3PO_4$ (100%) | 0.006404 | g/$m^3$ |
| transport $H_3PO_4$ from Ibbenburen | 0.000001 | tkm/$m^3$ |
| water (for backwash) | 0.002737 | $m^3/m^3$ |
| production of sludge | 0.013690 | kg/$m^3$ |
| transport of sludge | 0.000507 | tkm/$m^3$ |
| **Transport to Weesperkarspel** | | |
| pump's energy consumption | 0.027903 | kwh/$m^3$ |

| ***Weesperkarspel treatment plant*** | | |
|---|---|---|
| | Normalised values | |
| **Ozonation** | | |
| power consumption for $O_3$ production | 0.050523 | kwh/m$^3$ |
| **Softening** | | |
| power consumption | 0.159700 | kwh/m$^3$ |
| calcite pellets | 0.007275 | kg/m$^3$ |
| transport calcite pellets from North Holland | 0.000255 | tkm/m$^3$ |
| NaOH (50%) | 0.104303 | kg/m$^3$ |
| transport NaOH from Brussels | 0.022947 | tkm/m$^3$ |
| calcite pellets production | 0.100082 | kg/m$^3$ |
| transport calcite pellets to Ijmuiden | 0.004504 | tkm/m$^3$ |
| $CO_2$ usage | 0.011422 | kg/m$^3$ |
| NaCl (for ion exchange) | 0.000088 | kg/m$^3$ |
| transport NaCl | 0.000004 | tkm/m$^3$ |
| **Biological activated carbon filtration** | | |
| energy consumption (backwash) | 0.005670 | kwh/m$^3$ |
| activated carbon | 0.004434 | kg/m$^3$ |
| transport activated carbon from Hembrug | 0.000133 | tkm/m$^3$ |
| steam | 0.036600 | kg/m$^3$ |
| HCl (100%) | 0.000277 | kg/m$^3$ |
| transport HCl from Ibbenburen | 0.000058 | tkm/m$^3$ |
| **Slow sand filtration** | | |
| energy consumption | 0.024130 | kwh/m$^3$ |
| NaOH (100%) | 0.007480 | kg/m$^3$ |
| transport NaOH from Brussels | 0.001645 | tkm/m$^3$ |
| sand | 0.016120 | kg/m$^3$ |
| transport sand | 0.001612 | tkm/m$^3$ |
| sand discharge | 0.016120 | kg/m$^3$ |
| transport discharged sand | 0.001612 | tkm/m$^3$ |
| liquid oxygen | 0.003120 | kg/m$^3$ |
| transport liquid oxygen | 0.000156 | tkm/m$^3$ |

## C.3. Domestic water end-use

Every scenario presents different water end-use, from different sources (e.g., tap water, rainwater, greywater), and consequently different wastewater generation. Data regarding the baseline and the three water conservation scenarios (GTR, RHU, ECO) were taken from Bailey et al. (2020) and Agudelo-Vera and Blokker (2014), while the FWV scenario was based on the baseline with just an additional tap water consumption of 6.4 L/cap/day for flushing the food scraps according to Evans et al. (2010). Based on the drinking water demands and taken into account a water leakage of the water distribution network of 3.6%, the respective water purified in the WTP was calculated for each scenario. Respectively, the wastewater that ends up to the WWTP was calculated, taken into account an evaporation loss of 2.6%, the potential addition of external water resources (i.e., rainwater) or the water reuse/savings (i.e., greywater reuse, water-saving devices) and a groundwater infiltration of approximately 6.6% based on the distance between Prinseneiland neighborhood and Amsterdam West WWTP. The detailed tables regarding the diurnal water consumption (excluded the outside tap), as well as the annual drinking water consumption and wastewater production of one household are depicted below.

| | Baseline | GTR | RHU | ECO | FWV |
|---|---|---|---|---|---|
| | (L/cap/day) | | | | |
| | drinking water | drinking water | drinking water | drinking water | drinking water |
| bath | 0.8 | 0.8 | 0.8 | 0.8 | 0.8 |
| bathroom tap | 4.9 | 4.9 | 4.9 | 4.9 | 4.9 |
| dishwasher | 1.2 | 1.2 | 1.2 | 0.1 | 1.2 |
| kitchen tap | 8.1 | 8.1 | 8.1 | 8.1 | 14.5 |
| shower | 52.9 | 52.9 | 52.9 | 28.7 | 52.9 |
| | drinking water | greywater | rainwater | drinking water | drinking water |
| toilet | 29.7 | 29.7 | 29.7 | 5 | 29.7 |
| washing machine | 10.3 | 10.3 | 10.3 | 0.2 | 10.3 |
| | | | | | |
| total drinking water | 107.9 | 67.9 | 67.9 | 47.9 | 114.3 |
| total recycled water | - | 40 | 40 | - | - |

| | Baseline | GTR | | RHU | | ECO | FWV |
|---|---|---|---|---|---|---|---|
| | $m^3$/hh/year | | | | | | |
| water under purification in the WTP | 73.46 | 46.23 | - | 46.23 | - | 32.60 | 77.82 |
| domestic drinking water/recycled water use | 70.89 | 44.61 | 26.28 | 44.61 | 26.28 | 31.46 | 75.10 |
| domestic wastewater production | 69.07 | 43.46 | | 70.89 | | 30.65 | 73.16 |
| wastewater under treatment in the WWTP | 73.95 | 46.53 | | 73.95 | | 32.82 | 78.33 |

## C.4. Domestic water conservation

In the cases where greywater or rainwater recycling is applied the household appliances were held at a baseline water consumption. In the GTR scenario, part of the drinking water consumption was replaced with greywater, to supply the washing machine and the toilet flushing system with total water demand 40 L/cap/d, using as source the wastewater from the shower (52 L/cap/d), where the excess water were used as a back-up in the storage tank. The greywater treatment technology included membrane bioreactor (MBR) with screens and ultrafiltration modules. All the materials and energy input of the system's components were taken into account, as well as the $CH_4$ operational emissions of the MBR system, according to Jeong et al. (2018) and Kobayashi et al. (2020). The associated table with the inventory for the greywater system and the life expectancy of the materials is given below.

Sources: (Jeong et al., 2018) (input materials, energy, life expectancy); (Kobayashi et al., 2020) (direct operational gaseous emissions)

| **Greywater system** | | | |
|---|---|---|---|
| **Input (MBR)** | | | Life expectancy |
| Energy for treatment | 0.625 | kwh/m$^3$ | |
| steel (fine screen) | 1.97E-03 | kg/m$^3$ | 50 years |
| concrete (concrete pad) | 2.99E-02 | kg/m$^3$ | 50 years |
| steel (steel container) | 9.69E-03 | kg/m$^3$ | 50 years |
| steel (mixer) | 2.17E-03 | kg/m$^3$ | 10 years |
| PVC (aeration system PVC piping) | 9.32E-05 | kg/m$^3$ | 50 years |
| rubber (aeration system rubber piping) | 3.94E-04 | kg/m$^3$ | 50 years |
| steel (pump) | 1.05E-03 | kg/m$^3$ | 10 years |
| steel (MBR module steel housing) | 1.09E-03 | kg/m$^3$ | 50 years |
| PVDF (MBR module membranes) | 3.28E-04 | kg/m$^3$ | 10 years |
| steel (recycle pump) | 7.22E-04 | kg/m$^3$ | 10 years |
| cast iron (air blower) | 1.03E-03 | kg/m$^3$ | 15 years |
| polyester (controls/portable instruments) | 6.57E-05 | kg/m$^3$ | 25 years |
| sodium hypochlorite (membrane cleaning) | 4.9E-02 | kg/m$^3$ | |
| **Input (collection and distribution)** | | | |
| Energy for distribution | 0.164 | kwh/m$^3$ | |
| PVC (pipes) | 4.69E-02 | kg/m$^3$ | 25 years |
| steel (pump) | 1.2E-02 | kg/m$^3$ | 15 years |
| **Output** | | | |
| $CH_4$ (operational gaseous emissions) | 2.28E-05 | kg/m$^3$ | |

In the RHU scenario, rainwater was supplied to the toilets and the washing machines. For the rainwater harvesting we assumed a typical roof area of 90 m2 with 85% of that area connected to the collection system. In order to calculate the available rainwater, data on the average monthly rainfall patterns in the Netherlands for 2017 were used, according to Statista (2020). However, in order to ensure better water quality in the rainwater tank, it was assumed use of a first flush diverter alongside with the rainwater harvesting unit, installed at the down-pipe that supplies water to the tank. A revolutionary Delta chamber technology was used which consisted of 1m Delta High Volume chamber with diversion ability of 73 litres (Blue Mountain Co). Based on calculations, the collected water from the rooftop, after the subtraction of the diverted rainwater, was sufficient to cover the requirements for water in the washing machine and toilet, using a 3 $m^3$ rainwater storage tank. However, it was considered necessary the connection of the drinking water supply to the toilet and washing machine using a shut-off valve to allow water supply in case of rainwater unavailability (e.g., drought event). The associated data regarding the calculations of the rainwater collected and stored, the materials and energy input of the rainwater harvesting system and the life expectancy of the materials are presented in the table below.

Source: * Statista (2020)

| | Average rainfall in the Netherlands* | Collected rainwater from the rooftop | Stored water after first flush diversion |
|---|---|---|---|
| | mm/month | $m^3$/month/household | $m^3$/month/household |
| January | 65.1 | 4.98 | 3.86 |
| February | 45.4 | 3.47 | 2.69 |
| March | 61.3 | 4.69 | 3.63 |
| April | 47 | 3.60 | 2.79 |
| May | 54.9 | 4.20 | 3.25 |
| June | 64.6 | 4.94 | 3.83 |
| July | 69 | 5.28 | 4.09 |
| August | 65.3 | 5.00 | 3.87 |
| September | 67.1 | 5.13 | 3.98 |
| October | 70.6 | 5.40 | 4.18 |
| November | 76.5 | 5.85 | 4.53 |
| December | 75.6 | 5.78 | 4.48 |
| **total** | **63.53** | **4.86** | **3.77** |

Sources: * Material input for storage tank and life expectancy (Xue et al., 2016); Energy input and materials for the distribution network, and the respective life expectancy assumed the same as for the greywater distribution system (Jeong et al., 2018)

| **Rainwater system** | | | |
|---|---|---|---|
| **Input (storage tank)** | | | Life expectancy |
| concrete* | 2.28 | kg/$m^3$ | 50 years |
| steel* | 1.95E-05 | kg/$m^3$ | 50 years |
| cast iron* | 3.17E-05 | kg/$m^3$ | 15 years |
| bronze* | 7.9E-06 | kg/$m^3$ | 15 years |
| PVC* | 3.57E-04 | kg/$m^3$ | 50 years |
| **Input (collection and distribution)** | | | |
| Energy for distribution | 0.164 | kwh/$m^3$ | |
| PVC (pipes) | 4.69E-02 | kg/$m^3$ | 25 years |
| steel (pump) | 1.2E-02 | kg/$m^3$ | 15 years |

In this ECO scenario the ordinary plumbing devices got replaced with water-efficient ones. The scenario assumed use of 1 litre per flush in the toilet, usage of efficient shower head and 1 shower per person daily, and usage of nearly waterless washing machine and dishwasher (Agudelo-Vera and Blokker, 2014), (Bailey et al., 2020). Due to the fact that the water-efficient devices have very similar construction features with the conventional ones, the relative environmental cost from their manufacture was considered negligible and was not taken into account in the comparative LCA. The energy requirements of the washing machine and dishwasher are very important parameters and their contribution to the water system is significant. After a research in the available waterless devices in the market it was found that some waterless washing machines and dishwashers use the same or even less energy (up to 50%) than the conventional ones. In this work is was assumed that conventional and waterless devices consume the same energy per volume of operational water.

Based on research on the availability of commercial devices which are almost water-free, they were found only patented innovations of nearly water-free washing machines and dishwashers. The patented washing machine is a domestic unit, similar to the conventional one, including a rotary inner cylinder which is used for stirring clothes, a dirt removing outer cylinder for sucking grease and dust, a blower and a washing agent adding device (Xiang et al.). The patented dishwasher uses a combination of compressed air and blasting media to thoroughly remove grease and food particles, without hand tool scrubbing, manual rinsing, or use of soap, detergent or other chemicals (Abdulrahman et al., 2011).

The energy requirement of a conventional washing machine is approximately 10 $kWh/m^3$, while for a dishwasher 86.9 $kWh/m^3$ (Gerbens-Leenes, 2016). The energy requirement for water heating for the showers, which is the most dominant parameter in the whole water system, was calculated 105 $MJ/m^3$ of heated water, based on the equation for water heating obtained from Gerbens-Leenes (2016):

Energy for water heating = Vwg × sww × deltaT (J)

The parameter Vwg is the amount of water to be heated (g), sww is the specific heat of water (4.18 J/g·K), deltaT is the temperature difference between the tap water (15°C ambient temperature) and boiling water (assumed 40°C heating temperature (Kenway et al., 2013)).

### C.5. Domestic food waste valorisation

The food waste management was based on the assumption that food waste are sent for co-incineration in a distance of 8.2 km (Prinseneiland to WTE plant in Amsterdam). The annual food waste disposed via garbage were accounted 30.4 kg/person/year (Van Dooren et al., 2019). In the future, the food waste is expected to be minimised, based on the future goals of 2050 of the government (see Appendix B), but on the same time the population is expected to be increased. Therefore, it was assumed that the latter fact counterbalances the former resulting in the same food waste generation on average. In the FWV scenario, 95% of the food waste end up to the sink (Marashlian and El-Fadel, 2005), while the rest 5% is sent for co-incineration with the rest of the MSW. The kitchen grinder is a HDPE unit (6.842 kg HDPE/unit) with 15 years life expectancy, as reported by Graaf and Hell (2014). The unit operates with a 500 Watt motor, on average 2.4 times per day for 16 sec per use, and annual electricity consumption of 2–3 kWh/hh/year (Evans et al., 2010). It was used a calorific value of 37700 kJ/kg DM (6 8 kJ/kg food waste) and a mean DM content of 25% (Pham et al., 2015), which is in line with the data retrieved after personal communication with David Diepen, (AEB, 2020). The table below illustrates the inventory for the food waste incineration process.

Sources:* (Pham et al., 2015); ** (Khoo et al., 2010); The rest data retrieved from (Tonini et al., 2020)

| **food waste co-incineration (AEB)** | | |
|---|---|---|
| **Input** | | |
| Energy (for drying)* | 27920000 | kJ/ton DM |
| Electricity (for incineration)** | 70 | kWh/ton DM |
| CaO | 2.32E-03 | ton/ton DM |
| $CaCO_3$ | 4.54E-03 | ton/ton DM |
| NaOH (50%) | 3.84E-03 | ton/ton DM |
| HCl (30%) | 2.23E-04 | ton/ton DM |
| $Na_2S$ (12%) | 1.05E-04 | ton/ton DM |
| NaOCl (15%) | 6.43E-04 | ton/ton DM |
| $FeCl_3$ (40%) | 3.44E-05 | ton/ton DM |
| $NH_4OH$ (24.5%) | 3.58E-03 | ton/ton DM |
| Nitrogen | 1.03E-01 | $m^3$/ton DM |
| activated carbon in flue gas cleaning | 2.71E-03 | ton/ton DM |
| soap and foam-inhibitor | 3.03E-05 | $m^3$/ton DM |
| transport | 32.8 | tkm/ton DM |
| **Output** | | |
| Energy recovery* | 37700000 | kJ/ton DM |
| bottom ash | 1.8E-01 | ton/ton DM |
| fly ash | 9.9E-03 | ton/ton DM |
| other flue gas cleaning residues | 1.17E-02 | ton/ton DM |
| gypsum | 1.59E-03 | ton/ton DM |
| ferrous | 1.49E-02 | ton/ton DM |
| non-ferrous | 4.54E-03 | ton/ton DM |
| dust | 2.1E-03 | mg/ton DM |
| CxHy | 7.18E-04 | mg/ton DM |
| HCl | 3.95E-04 | mg/ton DM |
| CO | 1.9E-02 | mg/ton DM |
| $SO_2$ | 2.31E-03 | mg/ton DM |
| NOx | 1.32E-01 | mg/ton DM |
| heavy metals | 7.06E-06 | mg/ton DM |
| PCDD/F | 2.73E-05 | mg/ton DM |
| hydrogen fluoride | 8.88E-05 | mg/ton DM |
| $NH_3$ | 7.45E-04 | mg/ton DM |
| Cd | 2.53E-06 | mg/ton DM |

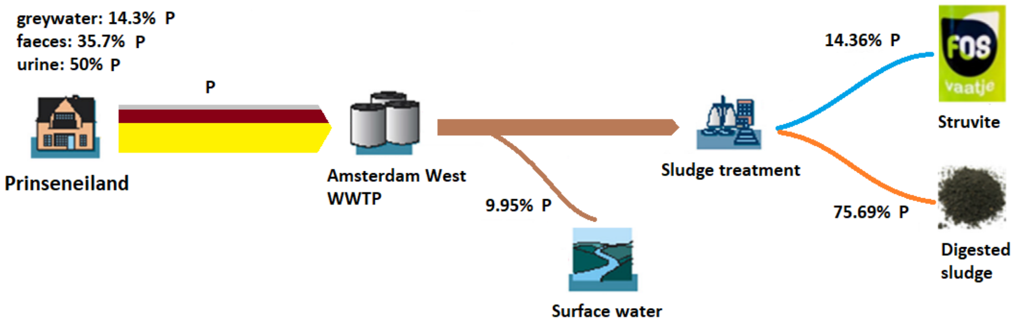
### C.6. Wastewater collection and treatment

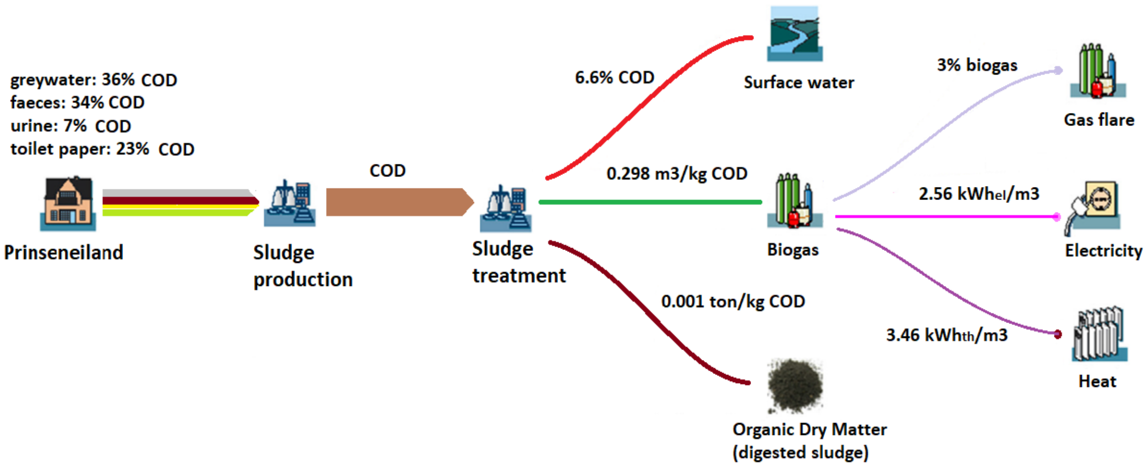
The data regarding the wastewater quality and the mass of the nutrient (TPH, TKN, COD) outflow from Prinseneiland catchment for the baseline and the three water conservation scenarios (GTR/RHU/ECO) were retrieved from Bailey et al. (2020), while the mass of nutrients for the FWV scenario were retrieved from a recent MSc thesis which examined the application of kitchen grinders in the same case study area. The table with the wastewater quality can be found in the table below. It is worth to note that the fluctuations in the nutrients mass of the wastewater across the scenarios were modelled by a stochastic model that predicts the nutrient changes in a sewer under water conservation scenarios.

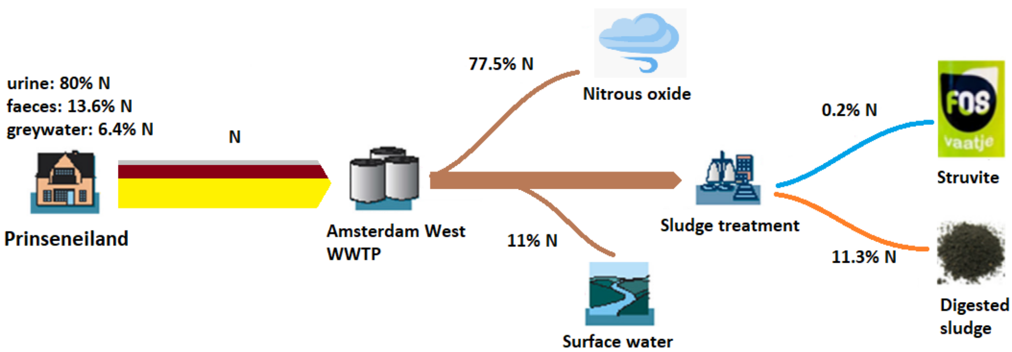
| | Baseline | GTR | RHU | ECO | FWV |
|---|---|---|---|---|---|
| | (g/person/day) | | | | |
| TKN | 8.92 | 9.55 | 8.75 | 9.92 | 12.42 |
| TPH | 0.88 | 0.94 | 0.85 | 0.98 | 1.03 |
| COD | 83.94 | 93.03 | 87.39 | 93.26 | 162 |

The current sewer system is a looped and partly combined network (i.e., stormwater and wastewater). Although, in the simulation of the wastewater flow it was assumed separate sewer system, using wastewater quality for dry weather flow (Bailey et al., 2020). As an average pipe diameter for the current wastewater collection network (baseline) it was assumed 250 mm. The sewer system in GTR and ECO scenario experiences a much narrower range of flow rates, which warrants smaller pipe diameters to enable the network to transport the solid particles. After modelling trials (within another MSc thesis project), it was found that a diameter of 160 mm (4 mm slope) performed satisfactory in all the cases (GTR, RHU, ECO, FWV). Therefore, PVC pipelines (100 years life expectancy) with 160 mm diameter were assumed for the calculation of the material's mass for all the alternative scenarios. The calculations for the mass of the PVC needed were based on information retrieved from FLO-TEK (b), taking into account a total length of approximately 683 m of wastewater collection network (Mogoş, 2018). The calculations revealed usage of 14E-04 kg PVC/$m^3$ in the baseline, 9E-04 kg PVC/$m^3$ in the GTR, 6E-04 kg PVC/$m^3$ in the RHU, 13E-04 kg PVC/$m^3$ in the ECO and 5E-04 kg PVC/$m^3$ in the FWV scenario. The excavation activities were excluded from the modelling.

Data regarding the nutrient balance of phosphorus, COD and nitrogen in the WWTP in the current situation were retrieved from Fooij (2015) and van der Hoek et al. (2018a), respectively. For the future scenarios it was assumed that the operation of the WWTP remains the same, and thus the shares of the nutrients that end up to the several streams (water body, struvite, sludge waste) remain the same. The nutrient balance in the WWTP is depicted in the Figures below.



The direct air emissions from the wastewater treatment process (waterline and sludge treatment) were accounted 4.3 $g/m^3$ $CH_4$ and 0.23 $g/m^3$ $N_2O$, according to Frijns et al. (2008). Although, the amount of nitrous oxide that is generated during wastewater treatment is still unclear (Frijns et al., 2008). The energy input for the wastewater collection were 1.05 $MJ/m^3$, while for the main wastewater treatment process 3.35 $MJ/m^3$, according to Gerbens-Leenes (2016). The additional input resources that were used for the struvite recovery process with AirPrex technology were 7.5 kWh electricity, 10.9-12.7 kg $MgCl_2$ and 60-70 $m^3$ air, normalised per kg of phosphorus recovered, as reported by Amann et al. (2018). The recovered resources from this process was 10 kg struvite (Berlizer Pflanze) per kg of recovered phosphorus (Amann et al., 2018), expressed as phosphoric acid ($P_2O_5$) fertiliser based on the 19.80% $P_2O_5$ concentration of struvite and the relation between P and $P_2O_5$ ($P=P_2O_5$ *0.44) (Nakakubo et al., 2012), (Tonini et al., 2020), (Coche et al., 1996).

Another waste stream that was taken into account in the modelling of the WWTP was the annual amount of solid waste withdrawn every year from screening process and sent for co-incineration in the energy company AEB. Approximately 2.6 kg solid waste/PE (personal communication with Peter Piekema, (Waternet, 2020)) are trapped annually in screens and co-incinerated with the rest of the MSW.

Regarding the nitrogen recovery from the digester reject water (GTR2, RHU2, ECO2, FWV2), two processes were added to the operational scheme of Amsterdam West WWTP; THP and air stripping. The former was implemented as a pre-treatment process of the WAS applying pressure of 19-21 bar at a temperature of 180°C, having as major side-effects, apart from the increase of N-load of the return water which was the main intention, the increase of biogas generation(by 30% (Li et al., 2017)) and the reduction of the sludge cake (by 25% (Phothilangka et al., 2008)). Subsequently, after the dewatering phase, air stripping was applied in the N-rich reject water where through air application, ammonia was removed from the wastewater to the gas phase. Based on the nitrogen mass on the digester reject water, which was calculated 26.6% of the nitrogen inflow, and 90% efficiency of the air stripping process, 24% of the total nitrogen flow is recovered (van der Hoek et al., 2018a). The recovered nitrogen was expressed as ammonium sulfate ($(NH_4)_2SO_4$) fertiliser based on the 21% N content of ammonium sulfate (Coche et al., 1996). The table below depicts the resource input which were used for both thermal hydrolysis and air stripping techniques.

Source: * (Mills et al., 2014); ** (Antonini et al., 2011)

| **Nitrogen recovery** | | |
|---|---|---|
| **Input (Thermal Hydrolysis)*** | | |
| energy | 0.537 | kWh/kg DM |
| energy (fuel) | 0.37 | kWh/kg DM |
| **Input (Air-stripping)**** | | |
| energy | 18.8-28.2 | kwh/kg (NH4)2SO4 rec |
| NaOH pellets | 2.35 | kg/kg (NH4)2SO4 rec |
| $H_2SO_4$ (96-98%) | 9.4 | L/kg (NH4)2SO4 rec |

The biogas production in the alternative scenarios was calculated based on the biogas yield per mass of COD (0.298 $m^3$ biogas/kg COD) that is valid in the current situation. Respectively, the sludge waste production was calculated based on the Organic dry matter (ODM) content per mass of COD (0.001 ton ODM/kg COD) that is valid in the current situation and taken into account that ODM is 68% of the total DM (Fooij, 2015). The table below presents the calculated energy recovery from the biogas and the WTE management methods (co-incineration or mono-incineration or drying) across the scenarios.

| | Baseline | GTR | | RHU | | ECO | | FWV | |
|---|---|---|---|---|---|---|---|---|---|
| | | first | second | first | second | first | second | first | second |
| **Biogas yield (CHP)** | | | | | | | | | |
| biogas production ($Nm^3$/PE) | 9.125 | 10.113 | 13.147 | 9.499 | 12.349 | 10.138 | 13.179 | 17.610 | 22.893 |
| electricity (kWhel/hh/year) | 38.5 | 45.2 | 58.8 | 42.5 | 55.2 | 45.3 | 58.9 | 78.7 | 102.3 |
| heat (kwhth/hh/year) | 52 | 61 | 79.3 | 57.3 | 74.5 | 61.2 | 79.5 | 106.3 | 138.1 |
| **Sewage sludge waste for treatment (incineration or drying)** | | | | | | | | | |
| sludge waste (kg DM/hh/year) | 44.2 | 51.9 | 38.9 | 48.7 | 36.5 | 52 | 39 | 90.3 | 67.7 |
| electricity from incineration (kWhel/hh/year) | 28.9 | - | 25.5 | - | 23.9 | - | 25.5 | - | 44.3 |
| heat from incineration (kwhth/hh/year) | 82.7 | - | 72.7 | - | 68.3 | - | 72.9 | - | 126.7 |
| energy from fuell pellets (drying) (GJ/hh/year) | - | 0.746 | - | 0.701 | - | 0.748 | - | 1.299 | - |

### C.7. Sewage sludge and food waste management

In baseline, the sewage sludge waste is sent for co-incineration in AEB energy company. It was assumed calorific value of 2.2 kWh/kg DM, 85% boiler efficiency and 35% electrical and 100% thermal efficiency Simoës and Veldman (2007) and personal communication with David Diepen, (AEB, 2020). The associated table with the inventory can be found below.

Sources:* (Đurđević et al., 2019); ** (Khoo et al., 2010); *** (Amann et al., 2018); The rest emissions from CIWM (2005)

| co-incineration (AEB) | | |
|---|---|---|
| **Input** | | |
| Heat (for drying)* | 1.8-2.2 | kWh/kg DM |
| Electricity (for drying)* | 0.1-0.3 | kWh/kg DM |
| Electricity (for incineration)** | 0.07 | kWh/kg DM |
| transport | 0 | tkm/kg DM |
| **Output** | | |
| Electricity recovery | 0.65 | kWh/kg DM |
| Heat recovery | 1.87 | kWh/kg DM |
| fly ash*** | 4.79E-01 | kg/kg DM |
| filter cake*** | 2.14E-02 | kg/kg DM |
| flue gas*** | 1.05 | kg/kg DM |
| $NH_3$ | 2E-06 | kg/kg DM |
| NOx | 2.71E-08 | kg/kg DM |
| $SO_2$ | 2.71E-08 | kg/kg DM |
| dust | 1.88E-06 | kg/kg DM |
| ash silos | 2.12E-07 | kg/kg DM |
| $CO_2$ | 8.47E-08 | kg/kg DM |
| CxHy | 2E-08 | kg/kg DM |
| HCl | 2.35E-06 | kg/kg DM |
| HF | 6.35E-07 | kg/kg DM |
| mercury | 2.47E-08 | kg/kg DM |
| cadmium | 2.24E-08 | kg/kg DM |
| heavy metals | 2.82E-08 | kg/kg DM |
| zinc | 4.24E-08 | kg/kg DM |
| doixins | 1.65E-08 | kg/kg DM |
| $N_2O$ | 1.78E07 | kg/kg DM |

In the GTR1, RHU1, ECO1 and FWV1 the sewage sludge is sent to the drying plant of Alkmaar (40 km distance), where the dried sludge is transformed into fuel pellets. For the production of fuel pellets it was assumed 0.85 kg fuel pellets per kg of sludge DM (WaterWorld, 2013). The calorific value is similar to wood biomass (wood pellets) with 10% moisture content (MC) and calorific value 17 GJ/ton fuel pellets (research, 2020). The inventory is depicted below.

Sources: * (Đurđević et al., 2019); ** (research, 2020); *** ecoinvent database

| **sludge drying to produce fuel pellets (Alkmaar)** | | |
|---|---|---|
| **Input** | | |
| Heat* | 1.8-2.2 | kWh/kg DM |
| Electricity* | 0.1-0.3 | kWh/kg DM |
| transport | 0.1538 | tkm/kg DM |
| **Output** | | |
| Energy recovery** | 0.0144 | GJ/kg DM |
| Sewage sludge, dried for clinker production*** | 1 | kg DM/kg DM |

In the GTR2, RHU2, ECO2 and FWV2 the sludge waste is sent for mono-incineration in Dordrecht (100 km distance) in order to apply phosphorus recovery from the ISSA. The data that were used regarding the mono-incineration process are presented in the table below. Tabular representation of the inventory for the phosphorus recovery process (EcoPhos technology) can be also found in this section. It has to be noted that apart from recovering of phosphoric acid as a fertiliser, there were retrieved Al chloride and Ca chloride which were accounted to offset de-icing agent and aluminum alloy from the global market.

Sources: * (Đurđević et al., 2019); ** (Amann et al., 2018); the rest emissions from CIWM (2005)

| mono-incineration (Dordrecht) | | |
|---|---|---|
| **Input** | | |
| Heat (for drying)* | 1.8-2.2 | kWh/kg DM |
| Electricity (for drying)* | 0.1-0.3 | kWh/kg DM |
| oil** | 2.14E-02 | kg/kg DM |
| lime** | 7.86E-02 | kg/kg DM |
| sand** | 1.43E-02 | kg/kg DM |
| NaOH** | 1.79E-02 | kg/kg DM |
| NH4** | 4.29E-03 | kg/kg DM |
| precipitants** | 7.14E-04 | kg/kg DM |
| $FeCl_3$** | 2.14E-03 | kg/kg DM |
| polyelectrolytes** | 5E-03 | kg/kg DM |
| HCl** | 1.43E-03 | kg/kg DM |
| water** | 6.57E-01 | kg/kg DM |
| transport | 0.3846 | tkm/kg DM |
| **Output** | | |
| Electricity recovery | 0.65 | kWh/kg DM |
| Heat recovery | 1.87 | kWh/kg DM |
| fly ash** | 4.79E-01 | kg/kg DM |
| filter cake** | 2.14E-02 | kg/kg DM |
| flue gas** | 1.05 | kg/kg DM |
| $NH_3$ | 2E-06 | kg/kg DM |
| NOx | 2.71E-08 | kg/kg DM |
| $SO_2$ | 2.71E-08 | kg/kg DM |
| dust | 1.88E-06 | kg/kg DM |
| ash silos | 2.12E-07 | kg/kg DM |
| $CO_2$ | 8.47E-08 | kg/kg DM |
| CxHy | 2E-08 | kg/kg DM |
| HCl | 2.35E-06 | kg/kg DM |
| HF | 6.35E-07 | kg/kg DM |
| mercury | 2.47E-08 | kg/kg DM |
| cadmium | 2.24E-08 | kg/kg DM |
| heavy metals | 2.82E-08 | kg/kg DM |
| zinc | 4.24E-08 | kg/kg DM |
| doixins | 1.65E-08 | kg/kg DM |
| $N_2O$ | 1.78E07 | kg/kg DM |

Source: (Amann et al., 2018)

| **EcoPhos** | | |
|---|---|---|
| **Input** | | |
| HCl (100%) | 5.4 | kg/kg P rec |
| steam | 8.4 | kg/kg P rec |
| resin | 0.003 | kg/kg P rec |
| water | 43 | kg/kg P rec |
| electricity | 0.42 | kWh/kg P rec |
| **Output** | | |
| phosphoric acid | 3.2 | kg/kg P rec |
| **by-products** | | |
| waste ash | 6 | kg/kg P rec |
| wastewater | 16 | kg/kg P rec |
| heavy metals | 27 | kg/kg P rec |
| Ca chloride | 3.8 | kg/kg P rec |
| Al chloride | 1.5 | kg/kg P rec |

## D. Importance of the selected impact categories

Water, energy and nutrients are essential parameters for the environmental assessment of a water system. Water systems are major energy users and nutrient releasers. Satisfying the demand for water and sanitation services currently requires significant amounts of energy to collect, treat, and deliver drinking water, and collect, treat and dispose of the generated wastewater. The ever increasing population at a global scale, which is also valid for the Netherlands and specifically for the city of Amsterdam (UN, 2020), increase the requirements for water and sanitation services, which in turn increase the energy demand. Meanwhile, the effluent from wastewater treatment facilities contains significant amounts of nitrogen and phosphorus and ranks as a significant contributor for riverine and coastal eutrophication. Freshwater eutrophication is also highly affected by the energy generation technologies. The phosphate effluents originated by the lignite and coal mining activities activities are significant contributors to eutrophication (Lechón et al., 2018). Therefore, the life cycle water consumption (also called "embodied energy"), the life cycle global warming potential, the life cycle water consumption and the life cycle freshwater eutrophication were selected as the environmental metrics under assessment in this work. These indicators are meaningful at the decision-making phase as they provide early warnings related to the environmental performance of modified water systems and identify hotspots for action and mitigation of the impacts.

There is high interconnection between energy generation and GHG emissions. Use and production of energy have a massive impact on the climate and vice versa. Increasing the supply of renewable energy allow to replace carbon intensive energy sources (e.g., fossil fuels) and significantly reduce global warming potential. The global climate deal to limit the increase of global average temperature to well below 2 °C aiming to 1.5 °C (Paris Agreement in 2015) cannot be achieved without a major overhaul of global energy production and consumption. The per capita GHG emissions in the Netherlands were 34% above the EU average in 2018 (CBS Netherlands, 2019). In the same year, GHG emissions amounted to 189.5 billion $CO_2$ equivalents, which was 2% less than the previous year, where The 75% of this reduction was related to lower $CO_2$ emissions by energy companies (CBS Netherlands, 2019).

Eutrophication most commonly arises from the oversupply of nutrients, usually nitrogen or phosphorus, in aquatic ecosystems, which leads to overgrowth of plants and algae and thus oxygen depletion (hypoxia). The increase of eutrophication is the result of human activities. Major sources of nutrients that induce eutrophication in freshwater and coastal ecosystems are the wastewater effluent discharges, excessive chemical fertilisers consumption and atmospheric deposition of nitrogen from burning fossil fuels[5]. A great fraction of the freshwater eutrophication is also caused by the spoils of coal or lignite or sulfidic tailings arising from the mining activities.

The Netherlands is one of the few countries in the EU in which urban wastewater treatment is in full compliance with the Urban Waste Water Treatment Directive, including 100% compliance for tertiary treatment targeted at the elimination of nutrients. According to Lürling and Mucci (2020), the average nutrient removal efficiency from municipal wastewater was 84.5% for nitrogen and 86.8% for phosphorus in 2016. Nonetheless over 99% of the wastewater being treated, the country has one of the poorest surface water qualities of the entire EU, with 40% of its lakes in a moderate ecological status and 60% having a poor or bad status (Lürling and Mucci, 2020). Eutrophication was a major issue in the Netherlands since 1950 (Gulati and Van Donk, 2002), but since 1975 has been recognised as an alarming pollution problem in the country (Parma, 1980). Recent climate change, habitat fragmentation and biotic exploitation of waters led to loss of resilience and thus acceleration of the problem.

[5]Once combusted the fossil fuels (e.g., coal, oil, natural gas) they discharge nitrogen oxides (NOx) into the atmosphere which return to rivers and lakes.

## E. ReCiPe methodology

ReCiPe method was first developed in 2008 in collaboration with the Dutch National Institute for Public Health and the Environment (RIVM), Radboud University Nijmegen (RUN), University of Leiden (CML) and PRé Sustainability (Goedkoop et al., 2009). The method has been given the name ReCiPe as it provides a “recipe” to calculate life cycle impact category indicators. ReCiPe determines indicators at two levels: 17 midpoint indicators and 3 endpoint indicators. Midpoint indicators are considered to be the links in the cause-effect chain (environmental mechanism) of an impact category, prior to the endpoint indicators, at which characterization factors or indicators can be derived to reflect the relative importance of emissions or extractions (Bare et al., 2000). Common examples of midpoint characterization factors include ozone depletion potentials, global warming potentials, and acidification potentials. Endpoint indicators show the environmental impact on three higher aggregation levels of human health, biodiversity and resource scarcity. An updated version, ReCiPe 2016, was developed to a better understanding of the environmental impact of goods, services and processes–up to date with the current scientific knowledge (SimaPro, 2017). The insights obtained can be used to reduce the environmental impact and to assess the extent in which a circular economy is beneficial for the environment. An overview of the impact categories that are covered in the ReCiPe 2016 method and their relation to the areas of protection is presented in the figure below (SimaPro, 2017).