Assessment of water reuse strategies in engineered water systems using urban metabolism and water-energy-pollution frameworks

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Declaration

‘I, Oriana Landa Cansigno, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.'

Dedication

To my parents.

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Abstract

Urban water metabolism (UWM) refers to various flows such as water, energy, materials and resources for water services and wastes and emissions to air, water and soil. However, the performance assessment of UWM has not yet considered the water-energy-pollution (WEP) nexus and the impacts of decentralised or centralised water reuse strategies within the framework.

This thesis develops an integrated UWM-WEP framework within an urban water system (UWS) to investigate sustainability performance assessment of various levels of water reuse strategies. A conceptual model was developed using the WaterMet2 tool and tested into Purisima and San Francisco del Rincon cities, in Mexico. WEP nexus were represented by six key performance indicators (KPI): urban water deficit, delivered potable water, energy, global warming potential, eutrophication potential and acidification potential. Nine strategies using greywater, domestic wastewater or centralised reclaimed water at three percentages of adoption (i.e. 20, 50, and 100%) were considered for various urban users and simulated over a planning horizon of 30 years (2015-2044). The KPI's of each strategy were compared against the ones obtained for a business-as-usual (BAU) strategy.

Results indicate centralised and decentralised strategies have different effects on the KPI. More specifically, centralised water reuse reduces energy while increasing acidification potential. However, maximising the centralised water reclamation is potentially possible in the case study. Decentralised domestic wastewater reduces acidification potential without affecting energy despite having an additional wastewater treatment. Decentralised water reuse is appropriate in new developments due to sewer modifications and the need for testing other technologies. The findings provide new evidence to create effective planning and water management policies, but the framework must be adapted for each specific context.

Impact statement

This thesis proposes a novel conceptual framework integrating two systemic approaches: the water-energy-pollutant and urban water metabolism to understand the impacts of operating water reuse strategies. The framework provides new insights for a comprehensive environmental assessment of decentralised and centralised water reuse strategies.

Results of the thesis provide evidence of the future water supply, energy consumption, global warming, acidification and eutrophication potential in an urban water system. This increases the understanding of the environmental impacts of different proportions of water reuse uptakes during the operation stage. Results can be useful to the water utilities to decide over the reuse strategies and water management plans, for example maximising water reuse uptakes, implementing decentralised strategies and developing local solutions to augment water supply.

This research project established professionals and strengthens academic and industrial collaborations with UCL. For the case study, three water utilities: SAPAF, SAPAP and SITRATA, all in Mexico, have provided data and were willing to support the research. The practitioners facilitated the exchange of information for research development. Secondly, this thesis led to a long-term collaboration between Dr Kourosh Behzadian from University of West London (UWL) and Dr Luiza Campos. These collaborations strengthen the capacities of Civil, Environmental and Geomatic Engineering Department, more specifically the research water group.

Finally, the communication of thesis findings to scientists and practitioners used different ways. Firstly, an open-access publication in the Journal of Environmental Science and Pollution Research can reach worldwide audiences. Secondly, three oral-presentations in international conferences in the USA, Portugal and the United Kingdom reached specific audiences and other Mexican students. Finally, multiple meetings on-site allowed the dissemination of the findings with local practitioners in Mexico.

List of publications

Some of the material in this thesis has been published and presented at conferences. The publications are:

Journal papers

Landa-cansigno, O., Behzadian, K., Davila-cano, D.I., and Campos, L.C. (2020). Performance assessment of water reuse strategies using integrated framework of urban water metabolism and water-energy-pollution nexus. *Environmental Science and Pollution Research*, *27*, 4582–4597.

Conference papers

Landa-Cansigno, O., Behzadian K., Davila-Cano, D. and L. C. Campos (2018) Water-energy-pollutant nexus assessment of water reuse strategies in urban water systems using metabolism based approach. In: Water Efficiency Conference 2018, 5-7 Sep 2018, Aveiro, Portugal.

Landa-Cansigno, O., Behzadian, K. and L.C. Campos (2017) Assessment of water-energy nexus in urban water reuse using a metabolic approach: a case study in Mexico. In: 11th IWA International Conference on water reclamation and reuse, 23-27 July 2017 Long Beach, California, USA.

Acronyms and abbreviations

AcP: Acidification potential
BAU: Business as usual or do nothing strategy
BOD: Biological oxygen demand
C: Centralised water reuse strategy
CAS: Conventional activated sludge
COD: Chemical oxygen demand
DEWAT: Decentralised wastewater treatment
DGw: Decentralised greywater reuse strategy
DW: Decentralised domestic wastewater reuse strategy
ENA: Environmental network analysis
EPA: Environmental protection agency in the USA
EuP: Eutrophication Potential
FU: Functional analysis
GHG: Greenhouse gases
GSA: Global sensitivity analysis
Gw: Greywater
GWP: Global warming potential
IO: Input and output
KPI: Key performance indicator
LCA: Life cycle analysis
LH: Latin Hypercube
MBR: Membrane biological reactor
MFA: Material flow analysis
NSE: Nash Sutcliffe efficiency
PR: Purisima del Rincon city
RMSE: Ratio of the roof mean square error
Rw: Reuse water subsystem
SB1: Subcatchment 1
SB2: Subcatchment 2
SDG: Sustainable development goals
SFR: San Francisco del Rincon city
TN: Total nitrogen
TP: Total phosphorus
TSS: Total suspended solids
UM: Urban metabolism
UWS: Urban water systems
WEC: Water energy carbon
WEP: Water-energy-pollution
WS: Water supply subsystem
WTW: Water treatment works
WWTW: Wastewater treatment work

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Chapter 1. Introduction

Chapter 1 Introduction

1.1 Background and motivation

Cities represent 0.5% of the Earth's total surface but sustain 54% of the world's population (United Nations, 2014). These places consume 75% of the world's natural resources and 80% of its total energy output (Peter and Swilling, 2012). Implementing integrated resource management is imperative for urban sustainability.

Of all the resources, water sits at the centre of all human activities and development. It is needed for drinking, cleaning, growing food, etc. yet water access is not guaranteed for every citizen. Around 10% of the global population (680 million) lack access to basic drinking-water services (WHO and UNICEF, 2019). The situation might worsen due to the expected rise in urban population from 54 to 60% by 2050 (United Nations, 2014). Meanwhile, water of the highest quality is leaked in distribution mains and misused for flushing toilets. Solving this problem is not an easy task because there are many attributable natural and anthropogenic causes. There is an uneven distribution where some regions are notably more prone to water scarcity due to their intrinsic arid conditions (UN-Water, 2015). Climate change scenarios increase freshwater vulnerability. The expected global rise in temperatures would enhance the risk of intensified rainfalls and droughts projecting higher uncertainties in arid regions (Faramarzi *et al.*, 2013). Likewise, the lack of treatment infrastructure causes 80% of raw wastewater to end up in freshwater bodies without having been treated (WWAP, 2017). Such discharges affect water quality, reducing its availability and are detrimental to the environment. Against this background, the water sector should respond in an integral way to promote water security while providing pollution control. This requires management strategies such as water reuse to ensure access to safe, sufficient and affordable water in cities.

The idea of reusing water dates back to many centuries (Jiménez-Cisneros, 2014b). However, reusing water is of increasing interest in cities within the context of a circular economy (Geissdoerfer *et al.*, 2017). A typical urban water sector is centralised and reactive to water pollution problems where a linear form of water management dominates, rather than one that is circular (Lundy *et al.*, 2013). This implies a "take-use-waste" approach in which drinking water is used and disposed of as wastewater[1] in nearby water bodies. Instead of this unsustainable approach, from a perspective of the circular economy, waste items are potential local resources for secondary use and can be used in the same city in a closed-loop system (Céspedes-Restrepo and Morales-Pinzón, 2018; Agudelo-Vera *et al.*, 2012).

Water reuse implementation is broadly categorised as centralised and decentralised. The former is a type of strategy that uses reclaimed water[2]. The latter strategy uses greywater from domestic effluent and the collection, treatment and reuse are all situated near the source (Suriyachan *et al.*, 2012). Water reuse can be used in cities in golf course and park irrigation, environmental replenishing, peri-urban agriculture, toilet flushing or drinking water treatment supply (Duong and Saphores, 2015; Lazarova *et al.*, 2012). Circularity in the water system can foster sustainability, for instance, greywater used for toilet flushing, gardening and laundry can reduce freshwater intakes by 20-40% (de Gois *et al.*, 2015; Matos *et al.*, 2014; Oron *et al.*, 2014; Domènech, 2011). To what extent decentralised or centralised strategies are advantageous from a circular point of view is a topic requiring further study.

1.2 Research problem and gaps

Urban water reuse implementation is far from reaching its fullest potential in water constrained countries despite the availability of treated wastewater. It is estimated that water reuse adoption does not surpass 50% of the total available treated wastewater in countries with water stress conditions such as

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[1] Raw wastewater is the water discharged from homes, business and industry without treatment. It contains coliform bacteria concentrations of $1x10^7$ CFU/100ml which is unsafe for public health (EPA *et al.*, 2012).
[2] Reclaimed water is the treated wastewater from municipal effluents of suitable quality for some specific reuse applications and often denote that wastewater has received at least secondary treatment (Jiménez-Cisneros, 2014b; EPA *et al.*, 2012; Leverenz and Asano, 2011; Raschid-Sally, 2010).

China (8%), Mexico (23%), Singapore (38%) and Qatar (47%) (FAO, 2016). Although this problem might be rooted in technical aspects, it is of general interest to raise awareness about water reuse benefits among stakeholders and decision makers. Therefore, there has been a growing interest in the investigation of the sustainable aspects of the integrated urban water systems (UWS).

A way of analysing the impacts due to implementation of water reuse is by the use of the water metabolism. This theoretical concept quantifies the energy and resources needed, as well as wastes produced in providing water services. Such services can be any, from delivery of potable water to the allocation of reuse water, but it can be interpreted as resource efficiency or in hydrological performance terms (Behzadian and Kapelan, 2015a; Farooqui *et al.*, 2016; Renouf *et al.*, 2017). The water metabolism derives from the urban metabolism theory, firstly introduced by Wolman (1965), to analyse exchanges of resources in a city. The metabolism analysis can specify the trajectories of raw material, energy, water, nutrients and pollutants using an analogy of biological process as anabolism (inputs) and catabolism (outputs) (Cui *et al.*, 2019). The UWM uses a wide range of independent methodologies for quantitative assessment such as material flows analysis, contaminant balance models (Sapkota *et al.*, 2018), substance flows analyses (to track down a specific nutrient)(Firmansyah *et al.*, 2016), while a hybrid tool integrating life cycle assessment (LCA) quantifies the environmental burden of delivering water (Goldstein *et al.*, 2013; Chester *et al.*, 2012). Various studies point out electricity as the main contributor to different impacts in the water sector and denote that metabolic patterns of nutrients are not fully understood (Farooqui *et al.*, 2016).

Significant amounts of energy are needed to sustain the water metabolism. The idea of the energy linked to water is described in the nexus theory. This describes the mutual dependencies of both sectors based on the premise that management decisions over one resource would affect the other (Smajgl *et al.*, 2016; Kenway, 2013). From a water-centric perspective, studies of the nexus focused mostly on a particular subsystem, for example, to compare the wastewater treatment technologies (Velasquez-Orta *et al.*, 2018; Kjerstadius *et al.*, 2017; Singh and Kansal, 2016). Assessments of the performance of decentralised and centralised water reuse, as part of the integrated UWS, are less common. Most of these studies compared freshwater savings, energy consumption and greenhouse emissions in the so-called water-energy-carbon (WEC) nexus. Previous studies showed contradictory evidence over the WEC. Some authors have stated that decentralised reuse has lower energy consumption demands and lower carbon emissions (Chang *et al.*, 2017; Opher and Friedler, 2016a), while others tend to favour centralised systems due to the benefits on sizing (Singh and Kansal, 2016). Even though the decentralisation of water reuse is becoming an important topic, these studies rarely paid attention to the interactions with the surrounding environment other than carbon emissions. For example, the nutrients released into water bodies can deteriorate the environment in the eutrophication process but specific treatment technologies would increase the energy consumption. The water-energy-pollution (WEP) nexus establishes the implications of energy, related to water consumption and air and water pollution (Kumar and Saroj, 2014). Up to now, there has not been any study on the WEP in the urban water metabolism of a real-world case study. Despite the existence of comparison between centralised and decentralised water reuse, none of previous studies have considered the implications of metabolic patterns to assess uncertainties on the fate of pollutants within the city boundaries. Such a gap needs to be assessed locally by using comprehensive metabolic frameworks of analysis.

1.3 Rationale

Water metabolism and water energy nexus analysis use a similar quantitative analysis. While the urban water metabolism depicts an inflow and outflow of resources, it does not show the internal relationships of sectors and subsystems (Zheng *et al.*, 2019). The WEN analysis advocates to illustrate such connections. Hence, the application of a hybrid framework would help to understand the sustainability aspects of the water sector.

Few hybrid frameworks have been tested, but none of them included the pollution aspects. Kenway (2013) focused on the nexus-metabolism of Australian cities from a water end-user perspective in households but they did not address water reuse holistically. Another study used an environmental network analysis and input-output framework to demonstrate the relationships among industries and wastewater discharges but it did not unveil the environmental impacts (Zheng *et al.*, 2019). A comprehensive framework using UWM based on material flow analysis and life cycle impact assessment was used to simulate the metabolism in the UWS of Oslo (Behzadian and Kapelan, 2015a). This study demonstrated decentralised reuse over temporal variations and environmental impacts, but the study did not test centralised reuse. Hence, a comprehensive framework is needed to provide information to identify hotspots of exchanges of energy and resource materials with full integration of the water reuse.

This thesis seeks to answer the following questions:

What are the main factors affecting the WEP nexus?

How does the metabolic performance of centralised and decentralised water reuse strategies affect the water-energy-pollutants nexus within an urban water system?

1.4 Aim and objectives

The overall aim of this study is to assess the performance of centralised and decentralised water reuse strategies within the UWS using a comprehensive modelling framework of urban water metabolism and the water-energy-pollutant nexus approach. The specific research objectives are:

Objective 1. Propose an integrated modelling framework using urban water metabolism and WEP nexus

Objective 2. Identify different centralised and decentralised configurations of water reuse strate gies. **Objective 3.** Evaluate the influence of centralisation and decentralisation level and adoption rates on the sustainability performance of the integrated UWS.

Objective 4. Provide recommendations on the management of water reuse strategies in the selected case study.

The novelty of this thesis lies within the hypothesis that integration of the urban metabolism and nexus approach can be used for performance assessment of water reuse strategies in real-world systems and provide crucial information to transform the consumption patterns by designing interventions within the urban water system.

The research adopted a quantitative and distributed approach for modelling the KPI of different water reuse strategies within the boundaries of the UWS. It focuses on the operation stage of the UWS, as the construction, maintenance, and demolition phases have a minor influence on the environmental impacts (Jeong *et al*., 2015; Lane *et al*. 2015; Machado *et al*. 2007). It uses a thirty year timeframe for simulations given the lifespan of water infrastructure. This research highlights the use of a systemic approach to assess the sustainable aspects of urban water reuse and provides evidence that can be used to create effective policies in the future to decentralise the water sector.

1.5 Thesis organisation

Following this chapter, which provided background on the research problem and motivations, **Chapter 2** presents a literature review on water reuse and the water-energy nexus and urban water metabolism approaches. **Chapter 3** describes the modelling framework developed and the general methodology adopted in this research. The chapter discusses the system boundaries, data sources model inputs, methods for calculation, model outputs, limitations and the model's applicability to the case study, the proposed framework was tested in a case study. To do so, a set of water reuse strategies were specifically formulated for the case study by considering information on the most common treatment technologies and the opinions of local experts on future adaptation and water reuse. **Chapter 4** presents the selected case study of San Francisco and Purisima del Rincon in Rincon, Mexico and describes the main characteristics of the UWS. It explains the process to develop the conceptual model, including a summary of the main data inputs and sources, the key assumptions, and calibration. It also enlisted the strategies to be assessed. **Chapter 5** gives the results of the model's application in the case study and discussion for each strategy at the UWS level and per component using a long-term simulation. A contribution and sensitivity analysis demonstrated the robustness of the model and the usefulness of the framework. **Chapter 6** draws conclusions and discusses future research.

Chapter 2. Literature review

Chapter 2 Literature review

2.1 Introduction

This chapter describes three fundamental aspects of the theoretical background used in this thesis. Firstly, it explains the urban water metabolism framework and the urban water metabolism models available. Secondly, it covers the key differences between the decentralised and centralised reuse strategies, including the characterisation of wastewater (i.e. greywater, municipal wastewater, domestic wastewater), the treatment technologies available and the regulatory aspects of the water reuse. Thirdly, it explores the water-energy nexus approach from the water sector perspective, it discusses the need to include pollutants as a sub-nexus in this approach and shows some quantitative methodologies to interpret the nexus.

2.2 Urban water metabolism approach

2.2.1 Urban metabolism and circularity

Urban metabolism is a theoretical framework assessing the sustainability of a city in terms of how inputs, outputs, storage and consumption of resources occurs in the urban ecosystem (Céspedes-Restrepo and Morales-Pinzón, 2018; Li *et al.*, 2018). In an earlier stage of the urban metabolism, Wolman (1965) based its work on the assumption that a city is not only a physical structure but similar to a living organism, that can demand resources to perform a human activity (Cui *et al.*, 2019; Kennedy *et al.*, 2007). In Wolman's analysis of UM in a hypothetical city, an input and output accounts for the water, food and fuel required and the resulting sewage and air pollutants. He pointed out the dominance of the water flows in the metabolism. A critique to this theory done by Golubiewsky (2012) is that it fails to present exact analogous metabolic processes in an organism (homeostasis, stability, etc). Instead, it focusses on cities as urban ecosystems and uses input and outputs within a city similarly to the anabolic and catabolic processes (Cui *et al.*, 2019).

Faraud (2017) explains that UM relies on a system theory to condemn the linearity occurring in cities and exposing wastefulness of urban management, appealing instead to circularity. Traditional water consumption patterns based on the linear "take-use-waste" approach do not match with the water scarcity situation. In this consumption, water and resources are used, while waste and emissions originated along the way are taken outside the city boundaries. In contrast, a circular approach envisions different options to close the resource loops, for example, minimise the external input flows (e.g. energy, food), or decrease outputs (e.g. wastewater, emissions) through recycling of waste as new sources for self-consumption (Novotny, 2013). Figure 2.1 compares two metabolic patterns, case city A relies on depletable materials while emissions and waste are no longer used. In city B, recycling is encouraged, which leads to reduce pollution and focuses on renewable sources.

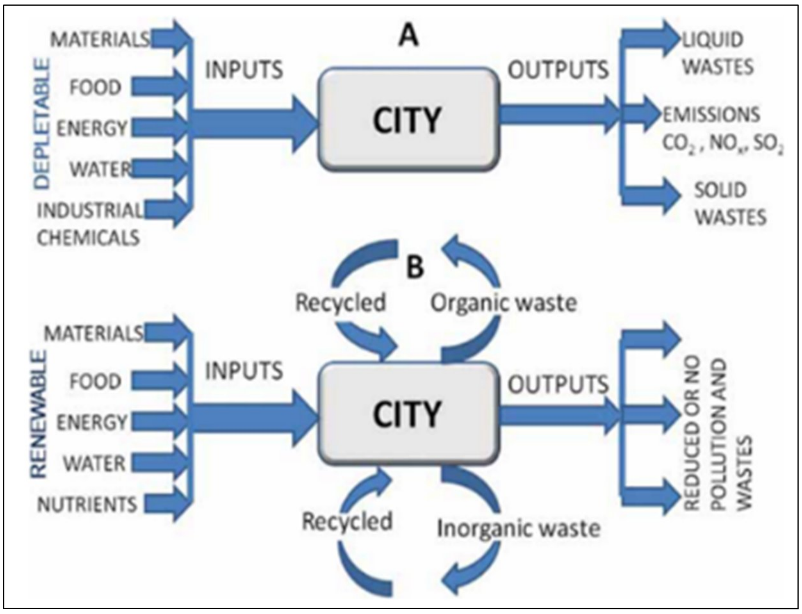

Figure 2.1 Linear (a) and circular (b) urban metabolism systems

Source: Novotny (2013), page 591

2.2.2 Urban water metabolism

The development of urban water metabolism (UWM) has been part of the reframing of the UM concept in the 21st century. In this approach, the urban water systems can be viewed as an independent system that connects the natural and anthropological water cycles through metabolic processes. Very few scholars attempted to define the concept, for example Huang *et al*. (2013)

defined UWM as an analysis of metabolic flux and socio-economic factors to identify the capacity and bottlenecks in the water system, to adopt mechanisms to cope with pollution, drought and flooding. They explain the UWM from a process-oriented point of view as a function of real and virtual water flows categories. Such flows include water directly available, water used by the ecosystem or humans and unavailable flows due to pollution or flood events.

Farooqui *et al*. (2016) define the UWM as the quantification of water exchanges between an urban entity and its supporting regions, both natural and managed, to generate indicators of metabolic performance. They used the UWM to investigate the efficiency of a water system based on indicators of hydrological performance and resource efficiency for water metabolism. Their framework uses two categories for the water flows: a) the anthropogenic flows such as potable water inflows, wastewater outflows, and rainwater harvesting and b) the natural hydrological flows such as precipitation, stormwater and runoff. Such a framework is an updated version of the hybrid model developed by Kenway (2013).

Behzadian and Kapelan (2015a) interpret the UWM as the required flows and fluxes to provide water services, which consequently generate some other fluxes and impact the social, economic and environmental systems. The UWM is demonstrated through a series of water flows, including potable water, green water, greywater, reuse water, wastewater and stormwater, all within the UWS. Their interpretation of UWM stated the need to understand associated fluxes such as energy, GHG emissions, and potential environmental impacts like eutrophication and acidification, and costs. Their framework proposed a series of urban sustainability indicators such as resilience, acceptability and costs. Although they analysed water reuse options in a city of Europe, their study is limited to rainwater harvesting and greywater reuse against conventional freshwater sources. Despite being one of the most comprehensive frameworks, the centralised reused was excluded. The aforementioned studies present a strong emphasis on the nature of water flows, whether they come from the natural or urban water cycles. As some of them have included other fluxes apart from water flows, they have become more complex. However, they have greater opportunity to be applied in the urban water sustainability to model the water reuse strategies.

2.2.3 Quantitative approaches

There are three schools of thought for urban metabolism assessments; the first focuses on “emergy” transformations, the second on flows exchanges within urban spaces, and a third one explaining the ecologic-metabolic relationships among them. In the first one, the “emergy” analysis measures the true value of the resources consumed during the metabolic cycle in terms of solar energy joule. However, this analysis is limited to suitable region-specific transformations and reliable data (Han *et al.*, 2018).

The second urban metabolism school is based on material flow accounting. In the 90’s, the concept UM adopted by the industrial ecology and urban planning areas emphasised tracking the resources/metabolites originated from human activities within the urban space (Cui, 2018; Kennedy *et al.*, 2007). The metabolic flows can be envisaged as mass, energy, water and other associated flows such as emissions. Under this approach, the information obtained aimed to identify hotspots of exchanges of energy and resource materials. For example, Zeller *et al.* (2019) use input and output to analyse waste efficiency in Brussels. The household sector stands out in the capacity to produce valorised waste. Yuan *et al*. (2011) used substance flow analysis (SFA) to track phosphorus flows in Chaohu city, China. It was found that fertilisers from agriculture sourced more phosphorus than any other sector studied (chemical industry, animal feeding and waste management). Villarroel *et al.* (2014) use an SFA in a slightly different way, to detect the suitable technologies to increase energy/nutrient recovery in the water sector. They suggested urine separation as the most suitable option in the context of the city of London. All these studies provide crucial information to transform the consumption patterns by designing interventions in different sectors. A specific approach for UWM is the MFA and energy flow analysis proposed by Farooqui *et al*. (2016). Farooqui’s framework compared different water management options in Australia, including water recycling and translated the flows in efficiency indicators. All quantitative approaches are of simple and efficient use, but none of them provides insights on the relationships among sectors (i.e. water, waste, food, energy) and lack the environmental vision, and specifically for the water sector, the nutrient metabolism is not yet explored.

In the last decade, there has been a proliferation of environmental analysis, which can be considered as the third surge of UM assessments although this is not yet recognised. This analysis is more comprehensive, trying to highlight ecological and sustainable aspects. Input-Output (IO) and ecological network analysis (ENA) stands out as a framework to illustrate interactions (i.e. mutualism, exploitation, competition) and flows among different sectors in the urban ecosystem. Previous analysis pointed out the industrial and transportation sectors as key factors in Beijing and Guangdong carbon metabolisms (Li *et al.*, 2018; Zhang *et al.*, 2015). In addition, another study on wastewater metabolism identifies the industry with the highest discharges of COD and ammonia (Zheng *et al.*, 2019). In these frameworks, IO quantifies the flows while ENA spatially determines the directions (positive or negative) of the interactions. However, this analysis did not address temporal variations, and pollutant metabolism was not considered.

Another sustainable framework is MFA-LCA (Goldstein *et al.*, 2013; Chester *et al.*, 2012). LCA is a process-based analysis that estimates the environmental burden of a product or service in different life stages, for example from cradle to gate (Lijó *et al.*, 2017). This assessment commonly informs about the potential environmental impacts such as global warming, eutrophication, acidification, material depletion and cumulative energy in current systems or as consequential analysis of future strategies implementation (Acero *et al.*, 2015). An UM–LCA model applied to five cities identified the key metabolic flows to impacts and estimated higher flows than those accounted for by direct quantifications. For example, Beijing embedded mass accounted for 60% of its total mass flow and Hong Kong’s embodied energy was 76% of total energy, both were underestimated by direct accounts by 2-3 and 2-4 times, respectively (Goldstein *et al.*, 2013). A disadvantage of using this framework is the extensive amount of data required.

Urban metabolism research comprises several quantitative approaches that align with different aspects of sustainability and are needed for policymakers. Water reuse, nutrients and pollutants from municipal wastewater are usually ignored in metabolism quantifications. The links between different urban metabolism components are complex, and most of the approaches described above look at the system as a black box and focus on flows exchanges outside the UWS boundaries. Therefore, they should be supported by complementary detailed approaches to simplify information for decision-makers.

2.3 Urban water reuse

2.3.1 Water reuse

Urban planned reuse is a human intervened system directly reusing water in different applications. There is a range of worldwide water reuse projects attending to local necessities. Groundwater recharge and golf irrigation in the USA (Jiménez-Cisneros, 2014a), urban and environmental applications in the Northern region of Europe, peri-urban agriculture in Southern Europe (Bixio *et al.*, 2006), drinking water reuse systems in Namibia and Singapore and other urban reuse cases reported in Australia, Japan, Middle-east and North Africa countries (Duong and Saphores, 2015).

The United Nations calls for a radical change of view of wastewater as a resource rather than waste that needs to be disposed of (WWAP, 2017). The capitalised view of wastewater as a resource encourage obtaining trade-offs through the control of water reclamation and reuse projects. Reuse in gardening, toilet flushing, and laundry in-household has a potential substitution of 30-50% of the domestic freshwater intakes (Chen *et al.*, 2013). Recovery of phosphorus through sludge application in agriculture is simple and low cost but due to potential cross-contamination of pathogens or heavy metals is restricted (Maaß and Grundmann, 2016; Linderholm *et al.*, 2012; Usman *et al.*, 2012). Urine separation is promising at the decentralised level because this effluent contains 80% of nitrogen and 56% of phosphorus discharge in domestic sources (Randall and Naidoo, 2018), but it requires fitting new pipelines. Also, the production of electricity from methane production (directly produced in anaerobic reactors) or hydrogen are options for energy recovery (Bdour *et al.*, 2009).

The Sustainable Development Goal (SDG) 6.3 aims to reduce the proportion of untreated wastewater and increase water reuse (United Nations, 2015). Nevertheless, there are some disparities between water reuse adoptions in different countries. Cyprus and Namibia directly use 95% of treated municipal wastewater, while other countries facing water pressures use less than 50%. Table 2.1 presents the percentage of water used estimated from the difference between the volume of wastewater treated and the volume directly reused in various countries. Data were estimated from AQUASTAT (FAO, 2016). Despite the efforts to concentrate global information on water reuse by the FAO, there is a vast majority of countries whose data is still unavailable.

Table 2.1 Water reuse percentage in selected countries and wastewater treatment and use flows

Country	Water reuse (%)	Treated municipal wastewater ($1x10^9$ m^3/year)	Direct use[3] of treated municipal wastewater ($1x10^9$ m^3/year)
Cyprus	99%	0.030	0.030
Namibia	97%	0.006	0.0058
Qatar	47%	0.204	0.095
Singapore	38%	0.511	0.194
Mexico	23%	3.897	0.898
Australia	21%	2.000	0.420
Saudi Arabia	16%	1.600	0.254
China	8%	49.31	3.860

Source: FAO, 2016

[3] The direct use of treated municipal wastewater is defined in the glossary of AQUASTAT database as “the treated municipal wastewater (primary, secondary, tertiary effluents) directly used, i.e. with any or little dilution with freshwater during most of the year” (FAO, 2016). The term does not explicitly refer to a particular use.

2.3.2 Centralised and decentralised strategies

Centralised or decentralised strategies have different configurations and operation. The centralised water reclamation and reuse is a common approach in cities with a typical configuration of combined stormwater and sanitary sewerage, a wastewater treatment work (WWTW) and a system to distribute reclaimed water back into the city. The main advantage is the economy of scale where these systems can treat large volumes of wastewater flows and increase their capacity at a small differential cost. However, it does require a substantial investment for a bigger infrastructure, pumping and electrical equipment. The networked pipeline or non-networked distribution system (e.g. tankers) can allocate reclaimed water elsewhere, as the WWTW in centralised schemes is far from the points of wastewater generation. The pipeline-related costs are nearly 40% of the cost of total project investment, which can be reduced if opted for local water reuse (Garrido-Baserba *et al.*, 2018). Such a reduction can be translated into economic terms and be especially favourable in new urban developments

Decentralised water reuse implementation considers separation and on-site use of different domestic wastewater flows, namely black, greywater, yellow and brown water. Greywater (Gw) is a mixture of effluents sourced from hand basins, showers (also light-greywater), washing clothes and optional kitchen effluents. Some authors refer to it as heavy GW if kitchen plus laundry effluents are included (Larsen *et al.*, 2016; Vakil *et al.*, 2014; Domènech, 2011). Toilet flushes are known as black water which is further separated into brown (excretes) and yellow (urine) water (Suriyachan *et al.*, 2012). The combination of brown, yellow and greywater is further referred to as domestic wastewater. Decentralised water reuse can be implemented in individual households, high-rise buildings, malls, or at clusters covering a portion of a city (Novotny, 2013). Completed decentralised strategies associated with areas with a complete lack of sanitation systems (e.g. rural areas). It is also employed in newly constructed residential clusters, buildings and households of densely populated cities because its modular design can cope with the space constraints.

A novel approach of water reuse at a semi-centralised scale is sewer mining, a hybrid configuration using centralised sewer combined with on-site wastewater treatment and reuse. In this, unused/untreated effluents are reinjected into the sewer again. This modality does not require piping separation but presents regulatory challenges which require further research (Makropoulos *et al.*, 2018). In all cases, a storage tank of sufficient capacity is essential to stock treated GW for a daily household supply without creating anaerobic conditions (Duong *et al.*, 2011). Regarding operation, a clear difference is the involvement of end-users in decentralised strategies, whereas public or private water utilities remain the total control of centralised strategies. This means that involvement, training and engaging a higher number of stakeholders are key to project success; otherwise, public health and environmental protection might be compromised.

2.3.3 Wastewater quality and quantity

Potable water supply varies from location and use, for example lower flows are in countries such as Israel and Greece and the highest in the USA (see Table 2.2). Municipal wastewater is produced from the used potable water that is discharged in the sewer system. It is composed mainly by the combination of domestic, commercial and industrial effluents. Greywater is constituted of 50% to 88% of domestic wastewater (Friedler and Lahav, 2006; Oteng-Peprah *et al.*, 2018). The quality of domestic wastewater, stormwater and municipal wastewater has been well studied in the past and average range is commonly known (Metcalf and Eddy, 2003). Table 2.3 summarised five parameters chosen to compare the wastewater quality: biological oxygen demand (BOD), chemical oxygen demand (COD), total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP). The table includes wastewater in different domestic appliances, as mixed greywater, domestic, stormwater and municipal wastewater.

Table 2.2 Total and disaggregated domestic water consumption in different countries (all values in litres per capita per day)

Household component	Brazil	Greece	India	Israel	UK	USA
Hand basin	24	9	17		12- 5	42
Kitchen sink[a]	32	12	37	27-33	25	-
Washing machine	8	21	33	8-10	27	45
Shower & bath	35	34	30	45-55	37	47
Toilet	50	59	25	50-60	55	35
Other	3	7	23	-	11	28
Total	151	142	165	130-158	169	197
Reference	Ghisi and Ferreira, 2007	Antonopoulou *et al.*, 2013	Mandal *et al.*, 2011	Oron *et al.*, 2014	Parker and Wilby 2013	Metcalf and Eddy

[a]May include drinking, cooking; [b]gardening, dishwashing, ablution, leakages, etc.

Table 2.3 Concentration in mg/L of main pollutants in effluents from domestic appliances, greywater, domestic and municipal wastewater

Component	COD	BOD	TN	TP	TSS	Reference
Handbasin	1171.0	568.0	14.3	2.3	-	Oktor and Çelik, 2019
	427.0	305.0	2.5	1.3	90.5	Noutsopoulos *et al.*, 2018
	208.1	101.0	5.1	3.3	18.5	Cardoso-Chrispim and Antunes-Nolasco, 2017
	653.0	265.0	-	-	-	Zipf *et al.*, 2016
	225.0	43.0	-	-	48.0	Vakil *et al.*, 2014
	335.0	-	-	-	61.0	Antonopoulou *et al.*, 2013
	1489.0	597.0	105.0	26.0	573.0	Halalsheh *et al.*, 2008
Kitchen	1119.0	831.0	5.5	2.7	319.0	Noutsopoulos *et al.*, 2018
	602.0	293.0	-	-	308.0	Vakil *et al.*, 2014
	775.0	-	-	-	299.0	Antonopoulou *et al.*, 2013
	26.0	536.0	11.4	2.9	134.0	Li *et al.*, 2009
	2050.0	1460.0	74.0	74.0	1300.0	
	2244.0	1100.0	51.0	18.3	644.0	Halalsheh *et al.*, 2008
	936.0	536.0	-	-	235.0	Mandal *et al.,* 2011
Washing machine	2072.0	1363.0	6.2	1.2	169.0	Noutsopoulos *et al.*, 2018
	274.1	77.0	4.3	2.3	32.7	Cardoso-Chrispim and Antunes-Nolasco, 2017
	824.0	269.0	-	-	1852.0	Vakil *et al.*, 2014
	231.0	48.0	1.1	0.0	68.0	Li *et al.,* 2009
	2950.0	472.0	40.3	171.0	465.0	Li *et al.*, 2009
	725.0	472.0	-	-	165.0	Mandal *et al.,* 2011

…continue Table 2.3 Concentration in mg/L of main pollutants in effluents from domestic appliances, greywater, domestic and municipal wastewater

Component	COD	BOD	TN	TP	TSS	Reference
Bath shower	654.0	385.0	10.6	1.7	-	Oktor and Çelik, 2019
	112.0	-	5.1	-	86.9	Mohamed *et al.*, 2018
	390.0	263.0	2.7	0.1	73.5	Noutsopoulos *et al.*, 2018
	272.8	123.1	50.3	5.3	155.8	Cardoso-Chrispim and Antunes-Nolasco, 2017
	461.0	81.0	-	-	148.0	Vakil *et al.*, 2014
	399.0	-	-	-	63.0	Antonopoulou *et al.*, 2013
	424.0	216.0	17.0	-	120.0	Mandal *et al.*, 2011
	100.0	50.0	3.6	0.1	7.0	Li *et al.*, 2009
	633.0	300.0	19.4	48.8	505.0	Li *et al.*, 2009
Toilet	5160.0	1245.0	492.9	86.0	3740.0	Molla, 2013
	900.0	300.0	100.0	20.0	-	Henze 1997 in Henze and Comeau, 2008
Mixed greywater	194	157	-	-	1.33	Patil and Munavalli, 2016
	244-284	56 -100	12-17.6	42.84 - 57.7	-	Mandal *et al.*, 2011
	100-700	47-466	25-183	1.7-34.3	0.11-22.8	Li *et al.*, 2009
	-	208-688	85-285	25-45.2	17.2-27	Gross *et al.*, 2007
	425-1583	215	-	17.2-47.8	5.7-9.9	Hernández Leal *et al.*, 2007
Domestic WW	250-800	110-350	20-70	4-12	120-400	Metcalf and Eddy, 2003
Stormwater	40-73	8-10	0.4-1	1-2	67-101	
Municipal WW	260-900	120-380	20-705	4-12	120-370	

COD is the most studied parameter. COD and BOD indicate the presence of organic matter from food waste and hygiene products. The kitchen effluents along with dishwasher and washing machine effluents source 50-90% of the organic load (Friedler, 2004). COD and BOD concentrations in mixed GW resemble those in municipal or industrial wastewater (Gross *et al.*, 2007; Hernández-Leal *et al.*, 2007). The concentrations of TN and TP in greywater vary with respect to the domestic appliance, up to 50mg/L in shower effluents for TN and TP (Cardoso-Chrispim and Antunes-Nolasco, 2017) and up to 74 mg/L of TN and TP in the kitchen (Li *et al.*, 2009). The highest concentrations are found in toilet flushing, which is separated from the GW inputs, but P content measured in some samples is similarly higher than the concentration of TP in municipal wastewater as 3.7-11 mg/L (Metcalf and Eddy, 2003). TP and TN sources are detergents and food waste, which in excess can cause adverse effects in water bodies due to the eutrophication process. Eutrophication is explained by Morelli *et al.* (2018) as an impact caused by the increase of nutrients loading in the ecosystems. As a result, the increase of TP and TN, which are typically the limiting nutrients, stimulates primary production of algae and cyanobacteria leading to their exponential growth. The latest phenomenon is also known as algal bloom. The death and microbial respiration of algae lead to the decrease in dissolve oxygen (DO), resulting in hypoxia, mortality of benthic organisms, and habitat compression. Accumulation of nutrients and organic matter hypoxic events are continuous and leads to long/term changes in the ecosystem. As a result, there is a negative change in the aquatic ecosystem and drinking water can present changes in taste, odour and accumulation of algal toxins.

From the literature reviewed, the mixed GW can be as polluted as municipal wastewater. High concentrations in mixed GW can be attributed to the absence of the diluting effect of stormwater where organic loads can accumulate. Raude *et al*. (2009) also explain that if water-saving techniques were applied, then such effluent reduction will lead to a major concentration in the effluent, which is a condition likely to occur in arid cities. This suggests the need for higher quality control treatments. The studies in Table 2.3 show a high variation due to location and sampling differences. Such studies sprawl worldwide including countries such as China, India, Israel, Greece, Netherlands, among others. In addition, the sample affects the concentrations, some studies reported the concentrations in shower/bath and kitchens alone or in any other combination, yet all of them were referred to as greywater. Others present the analysis per single household fitting component to represent a single house, school or building. There is not a city-level scale report. Such scale is necessary to represent the local characteristics, given that differences in lifestyle and economic level also cause variability (Oteng-Peprah *et al.*, 2018). Providing referential values of GW concentration at city level could serve as guidance to improve on-site sanitary treatment designs and re-define the requirements for reuse systems (Matos *et al.*, 2014). This would be especially useful in countries such as Mexico seeking to develop domestic reuse policies and where greywater quality characterisations are scarce.

2.3.4 Water reuse guidelines

Various guidelines on urban water reuse establish the discharge water quality criteria around the world. This is an important control to reduce the potential health risks associated and any impact the water can have because of recirculating any pollutant in the UWS. Table 2.4 presents a comparison of six guidelines all related to non-potable use of reclaimed and greywater: the guidelines for water reuse in the USA (EPA *et al.*, 2012), the NOM-003-SEMARNAT-1997 guideline in Mexico (SEMARNAT, 1998), which are both for centralised urban reuse; the British Standard 8525-1:2010 Greywater systems (BSI, 2010) and the Canadian Guideline for Domestic Reclaimed water for use in Toilet and urinal flushing (Ministry of Health Canada, 2010), both referred to domestic greywater use. It also includes the recommendations of water quality for irrigation and aquifer recharge in Europe, the guideline EUR 28962 (Alcalde-Sanz and Gawlik, 2017) and the excreta and greywater use in agriculture emitted by the World Health Organisation (WHO, 2006). Although agriculture reuse is beyond the scope of this thesis, these were included for the relevance of the authority that established the guidelines. The comparison used five key parameters, biological chemical demand (BOD), total suspended solids (TSS), faecal coliform (FC), *Escherichia coli* (*E. coli*) and residual chlorine (Cl_2).

All guidelines in Table 2.4 emphasise the quality of water in association with the end-uses. In centralised scale, these end-uses include urban irrigation, industrial, landscape or environmental protection; either direct or indirect contact with the user. On the decentralised scale, toilet flushing, gardening or car washing at household scale are the most common. Stricter regulations are expected in direct contact uses of water at decentralised schemes, however, this is not the case for all regulations. The Canadian guideline for toilet flushing is equivalent to the restricted uses with direct contact specified by the NOM-003 in terms of BOD, TSS and FC concentrations. Nevertheless, the US-EPA and the European Union propose the strictest concentrations in terms of BOD and TSS (<10 mg/L), while the former is for centralised use with direct contact with the user, the latter is for agriculture. The British Standard proposes a no detectable limit on the concentrations of bacteria *E. coli* for water used in washing machine and sprinkler gardening. This standard has a strong focus on other microbiological parameters such as *legionella pneumophila*, *intestinal enterococc*i and total coliforms. FC is a parameter with the highest variations. The FC permissible values for indirect contact uses in NOM-003 is five times higher than the EPA guideline (<1,000 MPN/100mL vs 200 MPN/100 mL). On the contrary, the WHO proposes only the use of greywater for agriculture. In terms of Cl_2, the strictest concentration standard (<0.5 mg/L) is proposed for gardening and the most relaxed one (<2 mg/L) is suggested for toilet flushing in the British Standard, and an intermediate concentrated (<1 mg/L) is proposed for centralised reuse in USA. The residual chlorine is not regulated for water reuse in Mexico. A homogenised set of basic parameters in guidelines would be necessary to facilitate comparisons at the international level.

Such variations then show that the quality appears not to be entirely correlated to the intended purpose but with the apparent contact with the user. Having a better effluent quality might increase the users' acceptance (Novotny, 2013; Mujeriego *et al.*, 2011; Zhang *et al.*, 2007). However, this could increase the cost associated with energy-intensive treatment processes which may be difficult to afford in several low-income countries and could risk the long-term operation of reuse systems. The challenge in treatment is to produce an appropriate effluent quality at a reasonable and affordable energy cost. This situation can be improved by creating ad-hoc standards for greywater reuse that are feasible to reach and keep at a minimum the health risk, along with increasing the number of guidelines to address decentralised reuse. Also, the review shows the need to unify the key parameters as primordial to facilitate international comparisons.

Table 2.4 Maximum pollutant concentrations in reclaimed water for urban non-potable uses and reclaimed water for domestic reuse.

Use		BOD (mg/L)	TSS (mg/L)	*E. coli* (CFU/100 mL)	FC (CFU/100 mL)	Residual Cl_2 (mg/L)	Country
Reclaimed water for municipal use	Unrestricted Non-potable uses with possible direct contact to the public	≤10	-	-	N.D.	1	USA[a]
		≤20	≤20	-	≤240	-	MX[b]
	Uses where public access is controlled or restricted by physical or institutional barriers such as fencing, advisory signage or temporal access.	≤30	≤30	-	≤200	1	USA
		≤30	≤30	-	≤1000	-	MX
	Use in construction soil compaction, dust control, washing aggregate, concrete	≤30	≤30	-	≤14	1	USA
	Industrial	≤30	≤30	-	≤200	1	USA
Greywater for domestic reuse	Toilet flushing*	≤20	≤20	-	≤200	≥0.5	Canada[c]
		-	-	250	-	< 2	UK[d]
	Garden watering (non-spray)	-	-	250	-	<0.5	UK
	Washing machine (non-spray)	-	-	N.D.	-	< 2	UK
	Spray application: Pressure washing, garden sprinkler use and car washing.	-	-	N.D.	-	< 2	UK
Reclaimed water for irrigation	Unrestricted irrigation (food/root crops consumed raw and food crops where the edible part is in direct contact with reclaimed water)	≤10	≤10	≤10			EU[e]
		-	-	$<10^4$	-	-	Worldwide[f]
	Restricted irrigation	-	-	$<10^5$	-	-	Worldwide

N.D.: No detectable. Adapted from [a]EPA et al., 2012, [b]SEMARNAT, 1997, [c]Ministry of Health Canada, 2010, [d]BSI, 2010, [e]Alcalde-Sanz and Gawlik, 2017 and [f]WHO, 2006

2.3.5 Wastewater treatment technologies used in water reuse

Technology treatment aims to produce a suitable effluent for the end-use and according to the guidelines for safety previously reported in Table 2.4. There are numerous technologies for water reclamation in decentralised and centralised systems use combined with secondary and tertiary treatment technologies. Commonly, secondary treatments in municipal wastewater are biological-based processes. The Conventional Activated Sludge (CAS) uses oxygen-supply bioreactors as well as primary and secondary settling tanks to collect suspended solids. The use of Membrane Biological Reactor (MBR) is deemed to have great potential in semi-centralised units because of its BOD removal efficiencies above 90%, compact size and minimal sludge production (Zhang *et al.*, 2009). This technology combines a biologically active sludge treatment and micro/ultra-filtration units. The main drawback in the operation of MBR is membrane fouling and chemical costs (Judd, 2017). Sequential Batch Reactor (SBR) is a step-batch control process based on the modified activated sludge treatment. It produces a high effluent quality in a very short time, using 60% less operational expenses than a CAS (Lijó *et al.*, 2017). The COD removal efficiency is high (~90%), but it performs in a very limited way with regards to nutrient removal efficiencies (11% of TN and 32% TP) (Hernández-Leal *et al.*, 2007). The use of RBC is an efficient method to treat heavy and light GW. Most of the case studies reported the use of RBC reactors coupled with a primary and a secondary unit to remove suspended solids (e.g. sand filter or settling tank). These configurations lead to efficiencies above 95% of BOD (Vakil *et al.*, 2014; Abdel-Kader, 2012). The disadvantages of RBC are the energy consumption and space needed.

2.3.6 Urban water models

Various models with different capacities, applications and scopes exist to assess water reuse in urban water systems. The urban water optioneering Tool (UWOT) is a decision tool that compares the combination of water savings options. It consists of the possibility to optimise water greywater and rainwater recycling, and water supply subsystems use and indoor demand profile. A recent development is the use of sewer mining (Makropoulos *et al.*, 2018; Rozos and Makropoulos, 2013). The UVQ (Urban Volume and Quality) tracks waterborne contaminants and their flows for alternative water service provision in the water cycle. The model considers water supply, stormwater and wastewater (Mitchell, 2005). The DMM (Dynamic Metabolism Model) is a model which calculates some performance indicators of the urban water system at a specific point in time. It was built to facilitate the strategic planning and preliminary design of sustainable UWS by comparing different management strategies (Venkatesh *et al.*, 2014). It was built in a MS-Excel interface and is easy to use although the modelling is limited to an annual scale. Water metropolitan-metabolism tool (WM2) is a distributed sequential time-step model for long-term strategic assessment of UWS performance. The model quantifies water but also the flows of energy, water and pollutants, emissions, and other impact categories while considering the economic and temporal changes (Behzadian and Kapelan, 2015b). Also, it considers scenarios of population growth, urbanisation and climate change. Similar to other water-balance models (UVQ, Venkatesh, UWOT, Aquacycle, SWAT), WM2 requires a low level of input data, tracking down some pollutants and simulating across different spatial scales (Behzadian *et al.*, 2014) . It offers a fundamental analysis over three of the sustainability dimensions: technical, economic and environmental. A key unique feature of WM^2 is the incorporation of specific impact categories, namely GHG emissions, eutrophication, and acidification, which are based on the principles of life cycle impact but constrained to the operational stage only.

2.4 Water-Energy nexus approach

The Water-Energy (WE) nexus approach is an analytical framework that highlights the direct and indirect interconnections of the water and energy sectors. Up to now, there is not a precise definition of the WE nexus and in practice, the approach can contain two or more elements (Manschatz *et al*., 2016). Many scholars focus on the two dimensions of the Water-Energy nexus as a supply/demand interaction (Kenway, 2013; Scott *et al.*, 2011). Defining the nexus across three dimensions is becoming more appropriate. The Water-Energy-Food (WEF) Nexus has received interest and attention since 2011, and definitively detonated the nexus studies in the last decade. From a water-centric perspective.

2.4.1 Water - energy nexus

The water sector uses around 7% of the total energy. The energy is required for the abstraction of fresh/groundwater sources, drinking water treatment, potable water distribution, household end-use (i.e. heating), treatment of wastewater and reuse. It includes electricity, which is the main form of energy used, and indirect energy which is embodied energy in chemicals, fuels, construction materials and even in their production inputs. Various studies quantify the energy inputs in different UWS cities and compare its performance as the kWh used to allocate/treat 1m^3 of water at a specific time (De Stercke *et al.*, 2018; Moredia-Valek, 2016; Wang and Chen, 2016; Venkatesh *et al.*, 2014; Kenway, 2013; Siddiqi and Anadon, 2011). Exhaustive reviews of energy in the water cycle of different case studies have been published in the last years (Mannan *et al.*, 2018; Lam *et al.*, 2017; Lee *et al.*, 2017; Wakeel *et al.*, 2016; Nair *et al.*, 2014). Authors of such reviews concluded that energy consumption is highly variable along the UWS (Figure 2.2) and depends on the geographic, distance, climatic and technological factors of each location. According to Mannan *et al.* (2018) groundwater abstraction and wastewater treatment are two of the most energy intense subsystems. Water demands are significantly sourced by groundwater around the world but the abstraction process (<0.8 kWh/m^3) depends on elevation, flow and pump type. Drinking water treatment of groundwater is typically a simple chlorination process and is considered less energy demanding than surface water sources. The treatment of surface water may need coagulation-flocculation and chlorination, while saline water requires reverse osmosis and ultrafiltration, which can use up to 5 kWh/m^3 (Burns, 2013).

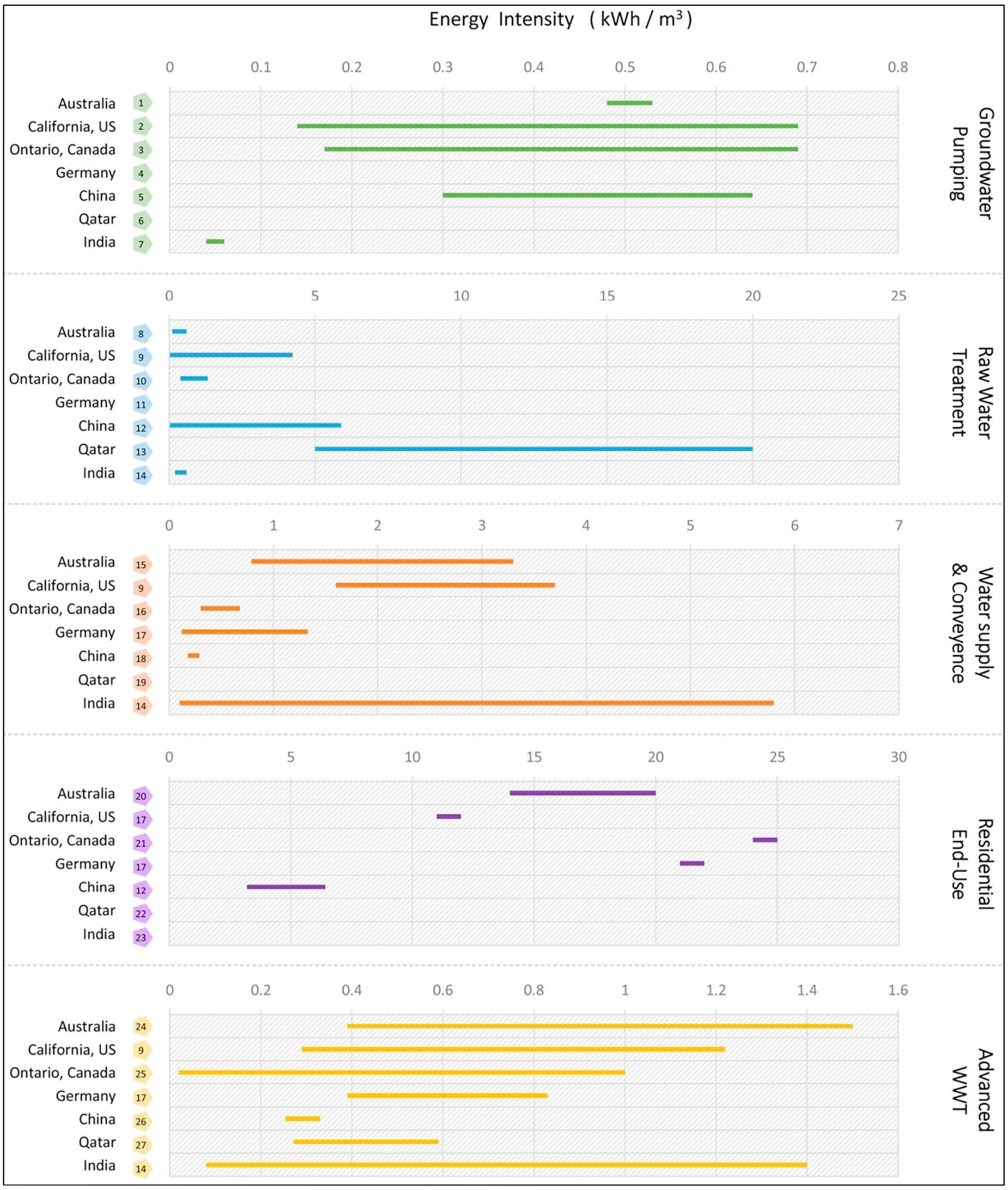

Figure 2.2 Energy intensities in various subsystems of the UWS in different countries

Source: Mannan *et al*. (2018) p. 305

Collection and treatment of wastewater require high energy according to the treatment level (Figure 2.3). Wastewater treatment is a key component of the UWS for energy and pollutant removal, electricity consumption and nutrient release. Some authors reveal that energy depends on local operation conditions such as scale, technology and efficiency (Su *et al.*, 2019; Velasquez-Orta *et al.*, 2018; Singh *et al.*, 2016).

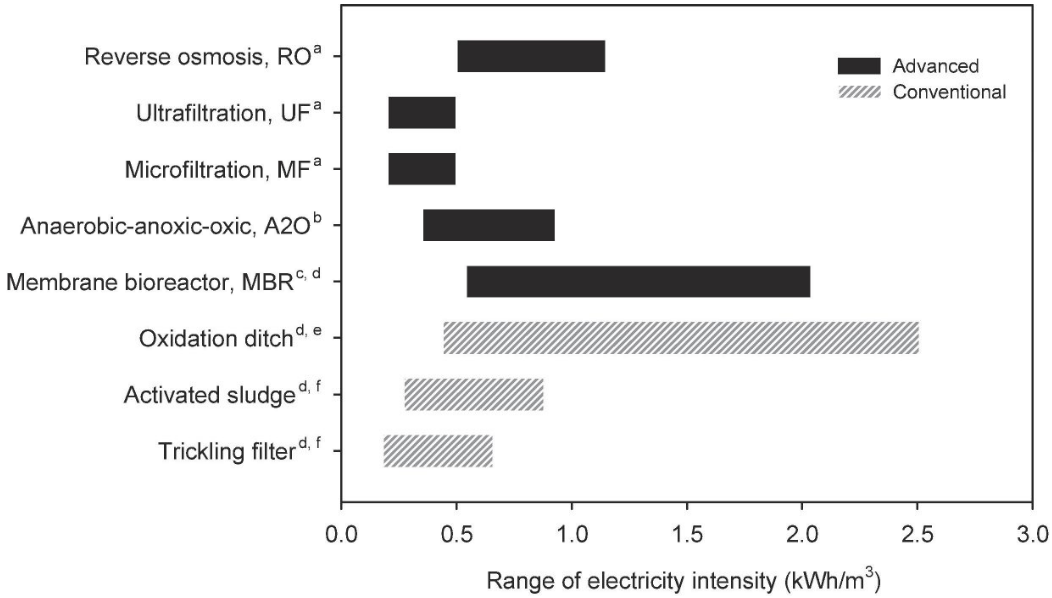

Figure 2.3 Electricity in different wastewater technologies
Source: Lee *et al*. (2017) p. 599

2.4.2 Water - energy – carbon nexus

The Water-Energy-Carbon (WEC) establishes the relationship between energy (production and consumption), water delivery and the release of greenhouse gases GHG. Such emissions, in CO_2 equivalent, result from direct and indirect energy uses and fugitive emissions (Lemos *et al.*, 2013). The WEC nexus analysis has given information about the hotspots for carbon emission reduction. Different water supply sources such as reclaimed water, greywater, rainwater and groundwater have been compared against conventional freshwater or transboundary inputs in terms of the potential water and energy savings. In some cases, the use of reclaimed water resulted in the most sustainable source. The study of Mo *et al.* (2014) compares a tertiary WW effluent, desalinated and freshwater supply in the regions of Tampa Bay (TB) and San Diego (SD), USA. Water reuse using reclaimed water achieved the best performance against desalination, saving 0.2% (5,550 TJ) and 1.3% (35,540 TJ) of the total electricity generation in TB and SD, respectively. Furthermore, 0.25 and 1.05 million metric tonnes of GHG emissions can be reduced annually in TB and SD, respectively, by using reclaimed water. This is because of the energy inputs required in the reverse osmosis and ultrafiltration, which are higher than the energy needed for water reclamation.

The application of water-saving techniques (e.g. efficient toilets) can achieve good reductions with less financial costs (Mo *et al.*, 2014). Increasing renewable energy sources in the regional electricity grid can reduce GHG emissions for UWS such as solar-powered cells, or biogas produced in the wastewater treatment work. Other energy conservation strategies related to materials are still poorly incorporated in practice, such as the increase of lifespan of water infrastructure, where it was demonstrated that investing in maintenance is more energy-efficient than constructing new facilities (Mo *et al*., 2014).

The WEC is dominantly the nexus to measure environmental impacts of the performance of decentralising the wastewater. Some studies concluded that decentralised greywater systems can reduce energy and emissions in comparison with centralised systems to some extent (Chang *et al.*, 2017; Opher and Friedler, 2016b; Singh *et al.*, 2016; Matos *et al.*, 2014) while others found contradictory results. An inventory of 50 WWTW in India and the UK showed that small scale plants consume twelve times the energy of large scale plants, 4.67 kWh/m^3 and 0.40 kWh/m^3, respectively. In addition, the carbon emissions of small plants (3.04 $kgCO_2eq/m^3$) is almost four times the emission in large ones (0.78 $kgCO_2eq/m^3$) attributable to operation capacity, technology and treated wastewater quality (Singh *et al.*, 2016). Silva-Vieira and Ghisi (2016) show a potential energy saving of 48% when decentralised-greywater was compared with centralised systems in a 20-year planning horizon in Florianopolis, Brazil. They found that greywater reuse energy consumption (0.5 kWh/m^3) is less than centralised (0.78 kWh/m^3) only when the demand is above 300 L/d and stated that benefits are directly correlated to the reduction of effluents into centralised systems rather than the source of water. Another study conducted by Opher and Friedler (2016a) reported that centralised systems are more negatively impactful due to the energy needs for pumping reclaimed water for large distances, which are higher than those required in clusters on the local scale. Finally, Duong *et al*. (2011) analyse reuse strategies at household and cluster scale, concluding that on-site wastewater treatment energy prevents household-scale systems from reducing as much energy as a cluster scale. All of these studies suggest that water reuse management requires the study consider local conditions. Even when it seems that water reuse is potentially suitable at semi-centralised scale rather than households; this might be notable on a bigger scale (Malinowski *et al.*, 2015).

2.4.3 Water – energy – pollution nexus

Kumar and Saroj (2014) bring up-front the Water-Energy-Pollution (WEP) in a city sustainability study. This nexus specifically establishes the implications of energy production, related to water consumption and environmental pollution (air and water). In their study in the city of Delhi, they also established that electricity generation dominates the climate emissions of 0.62 $kgCO_2$ eq emissions/ kWh. The water sector consumes 15% of the energy produced. The fuels burned in the road transport dominates the emissions to air ($PM_{2.5}$, SO_2, NOx, CO, VOC). The link between the energy and pollution was demonstrated through a mass balance of energy and considered the growth of population and increase in demands, hence a suitable approach for growing cities.

In contrast with the rest of the nexus, studies of WEP are minimal and none of the literature reviewed has used the water-energy-pollution approach to increase understanding in the water systems; in particular, because the water sector debates between the supply of non-conventional water sources at reasonable safeness and energy consumption. Among the various elements considered in the WEP nexus, the nutrients are ambivalent substances of resources (fertiliser) or pollution (eutrophication) making necessary their tracking and monitoring within the urban water system. For instance, it was observed that decreasing nitrogen concentration from 35 mg/L to 15 mg/L would lead to a reduction of 55% of marine eutrophication; nonetheless, it would increase the energy consumption by 50% (Lemos *et al.*, 2013).

2.4.4 Quantitative approaches for the WEN in the water sector

Various frameworks demonstrate the quantitative aspects of the nexus, some of these are presented in Table 2.5 and briefly summarised in this section.

Table 2.5 Aims and quantitative methodologies in different water-energy nexus analysis in the urban water sector

Method	Aim	Scale	Case study	Reference
Footprint	Quantify energy and emissions from different treatment systems	WWTW	China	Gu *et al.,* 2016
MFA	Conduct a comparative analysis of energy consumptions and GHG emissions	City	Toronto, Turin, Oslo and Nantes	Venkatesh *et al.*, 2014
	Accounted for energy consumption in the UWS	City	Mexico	Moredia-Valek *et al.*, 2017
	Assess potential energy savings due to implementation of rainwater, greywater in toilet flushing and laundry	Household	Brazil	Silva Vieira and Ghisi, 2016
	Account energy and water savings for greywater, rainwater and other IWM practices	Household - nation	20 countries	Wang *et al.,* 2015
SD	Investigate interactions between urban energy and water systems using an end-use perspective.	City	United Kingdom	De Stercke *et al.* 2018
Optimisa -tion	Uses a graphical representation from a network analysis perspective	Region and national	Spain	Tsolas *et al.,* 2018
IO + ENA	Accounted energy consumption in the UWS	Cities and region	China	Wang and Chen, 2016
IO + LCA	Compare energy, cost energy, and GHG emissions from greywater, freshwater, desalinated and reclaimed water	Cities Region	USA	Mo *et al.,* 2014
Hybrid UM	Estimates GHG, energy, water, nutrients in an UWS of different water sources, including rain and greywater.	City	Norway	Behzadian and Kapelan *et al.*, 2015a

MFA: Material flow analysis; IO: Input and Output; ENA: Environmental network analysis; LCA: Life cycle assessment, SD: Systems dynamics

MFA or IO frameworks focus on the long-term reliability of water systems, such as comparing different water supply portfolios. Common indicators are the potentials of water and energy savings in the UWS. Particularly, these methods in combination with scenario analysis compare the performance of implementing greywater, rainwater and desalinated water against conventional freshwater supplies (Silva Vieira and Ghisi, 2016; Duong *et al.*, 2011). However, these studies set up different boundaries on the analysis, mostly overlooking the sludge management and different spatial scales (e.g. city, region). Also in comparing the performance are the optimisation models, looking at objective functions to minimise costs of water supply (Tsolas *et al.*, 2018; Zhang and Vesselinov, 2016). Another set of studies have demonstrated this nexus framework through system dynamics by using the causal relationship of the water sector to energy and costs from a residential end-user perspective (De Stercke *et al.*, 2018).

There has been consistent growth in extending the WE nexus to the environmental assessments. The energy and carbon footprints are metrics of how much of these are used to produce a commodity or service. Footprints are used to evaluate the operation of different WWTW in practice. Also, the greywater footprint reduction has been suggested in China, a metric which correlates freshwater volume required to assimilate pollutants due to the WWTW (Gu *et al.*, 2016). This hypothetical volume becomes zero when the WWTW effluent meets or surpasses the water quality criteria. Footprint methods give an insight for comparing the WWTW operation, but the quantification is static and gives snapshots of the current situation. It does not show how different elements of the UWS interact with each other.

LCA is arguably the most widely spread methodology to complete the environmental impact assessment. Studies compared different environmental impacts of drinking water and water recycling options (Opher and Friedler, 2016a; Lane *et al.*, 2015; Muñoz *et al.*, 2010), comparisons of wastewater treatment technologies (Rahman *et al.*, 2016; Corominas *et al.*, 2013; Machado *et al.*, 2007) and nutrient recovery options (Bradford-Hartke *et al.*, 2015; Hasler *et al.*, 2015), although water reuse and sludge management are considered in limited approaches (Rahman *et al.*, 2016). Most of these studies are based on different functional units, which are referred to 1 m^3 of wastewater inflow or 1 m^3 of processed water, and established at different boundaries. Despite the differences in the methodology, many studies concluded that operation and maintenance stages present the major environmental impacts, whereas construction, assembling and demolition are negligible (Opher and Friedler, 2016b; Machado *et al.*, 2007). While LCA has many strengths, a disadvantage is that accessibility to impact factors depends on some databases which can be restricted, and the lack of uniformity in methodologies used across the literature makes comparison difficult. The above frameworks and models have limited practical applications to comprehend the implications of WEP within the integrated UWS by themselves. Therefore, it is necessary to combine two or more approaches to better understand the impacts of the various water reuse strategies and the implication of the WEP nexus.

2.5 Summary of the chapter

Urban metabolism is an approach to understand the exchange and efficiency of resources consumption at the city level. This analysis is helpful in tracking metabolic patterns (from linear to circular) to frame urban sustainability. The development of urban water metabolism (UWM) has been part of the reframing of the UM concept in the 21st century. Such a concept captures the inputs and outputs needed to provide water service.

Quantification of the urban water metabolism uses process-based methodologies such as MFA and IO, and there has been a proliferation of environmental analyses using ENA or LCA. In these hybrid frameworks, IO quantifies the flows while ENA spatially determines the directions (positive or negative) of the interactions or LCA to check the environment. These quantitative approaches are of simple and efficient use, but none of them provides insights into the interdependence among sectors. The links between different urban metabolism components are complex, and most of the approaches look at the system as a black box. Therefore, the urban metabolism components should be studied with complementary detailed approaches to simplify and support decision-making. Reuse of municipal wastewater is usually ignored in metabolism quantifications and focuses on flows exchanges outside the UWS boundaries. Urban water metabolism has not yet been employed substantially to analyse the performance and impacts of the water sector considering nutrients and pollutants metabolism. This increases in complexity but has greater opportunity for application in urban water sustainability and water reuse strategies assessments.

Water reuse dates back to many centuries, but is of increasing interest in circular economy and urban sustainability contexts. Circularity in the water system can foster sustainability, for instance, by greywater use. The capitalised view of wastewater as a resource encourages obtaining trade-offs through the control of water reclamation and reuse projects. Decentralised water reuse strategies are likely to grow in implementation and more evidence on trade-offs is needed. However, the literature reveals that data on greywater characterisations is scarce and disperse and requires appropriate tailored regulatory guidelines. It is necessary to generate water quality data at a city scale to further develop clear and specific greywater quality discharge guidelines. Additionally, contradictions on the capacity to reduce energy consumption concerning centralised reuse are frequently encountered. Therefore, performance assessment of centralised vs decentralised water reuse strategies is a topic that requires further investigation.

The WE nexus approach highlights the relationship between the water and energy sector. From a water sector perspective, a dominance on studies comparing energy inputs from a supply-demand perspective prevails. The WEC is dominantly the nexus to measure environmental impacts with concern about the performance of decentralising the wastewater. There is contradictory evidence that decentralised greywater systems can reduce energy and emissions in comparison with centralised systems. In addition, other aspects of pollution and environmental impacts are often dismissed (e.g. eutrophication). Such impacts can be captured in the water-energy-pollutant nexus; an analysis mostly overlooked in the nexus theory. The nexus results are strongly dependent on the spatial and temporal scale. Studies focused on WWTW technologies or comparison of supply from different water sources, but water reuse and sludge management was in some case omitted from the scope of these studies, lacking the integrated management perspective. Therefore, incorporating pollutants and environmental impacts in the analysis can provide a more holistic approach to compare water reuse at different scales.

Chapter 3. Methodology

Chapter 3 Methodology

3.1 Introduction

This thesis aims to assess the performance of water reuse strategies within the UWS of a real case study. In order to achieve that aim, a methodology was specifically developed. Figure 3.1 shows the sequence of steps followed to carry out the research and achieve the main aim.

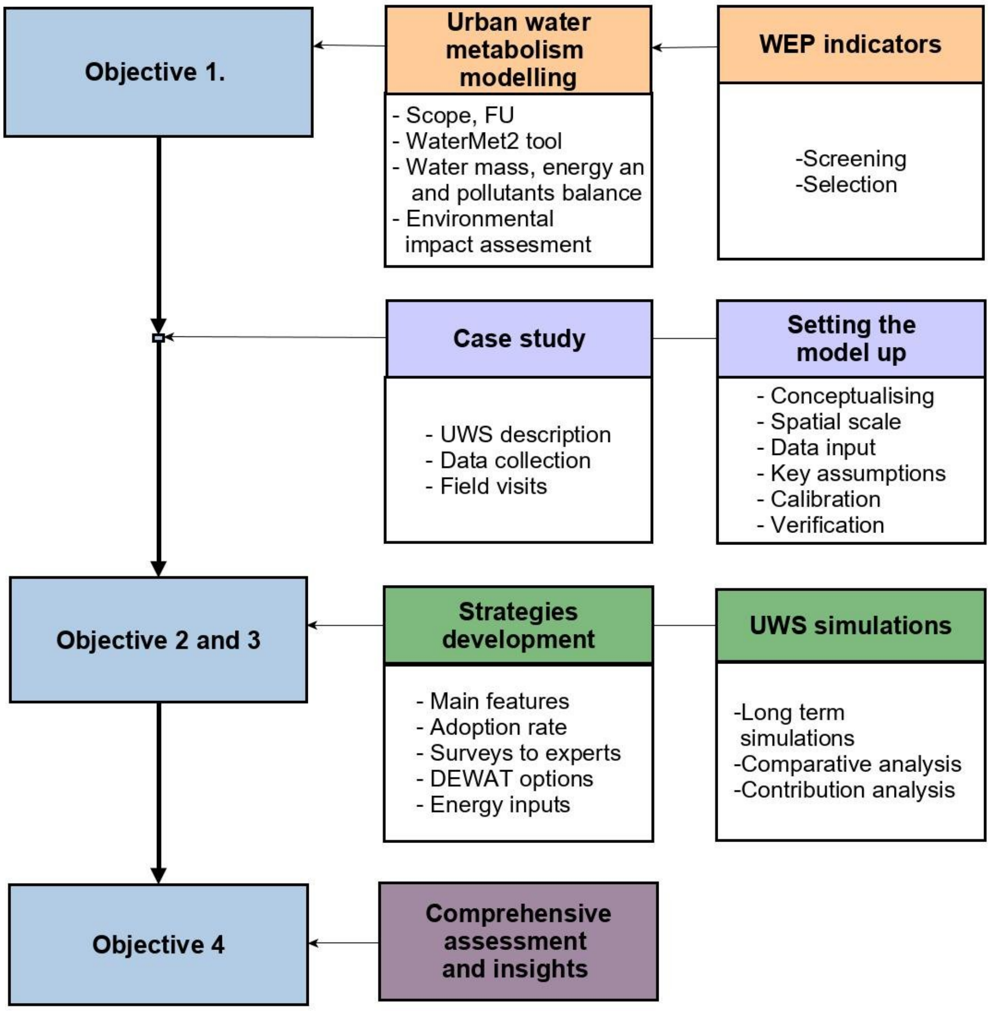

Figure 3.1 Diagrammatic representation of the research steps used in the methodology

The conceptual modelling framework definition (objective 1) requires a previous stage of the research problem, literature review, formulation of the research questions, and definition of the approach taken. All of these were presented in Chapters 1 and 2. Having screened several different methodologies, the strategy was to develop a hybrid method based on water metabolism and WE nexus. The proposed framework, built for the specific problem and gap identified, was tested in a real case study. To do so, a screening of possible case studies was undertaken to help define the case study. Then, a set of water reuse strategies were specifically formulated by considering the information on the most common treatment technologies along with the opinions of local experts on the future adoption of water reuse (objective 2). The model was set-up and the performance of the UWS was simulated for each strategy using the water metabolism-modelling tool, WaterMet^2 (WM^2). The results were presented for each strategy at UWS level, per component using a long-term simulation (objective 3). Finally, a contribution and sensitivity analysis identified the key contributors to the water-energy-pollution nexus elements. All this information was used to provide recommendations on the management of water reuse in the case study (objective 4).

3.2 Conceptual modelling framework

This conceptual framework is built into the use of two systemic approaches: urban water metabolism and the water-energy-pollutants nexus. Systemic approaches[4] show a global conception of the problem and an understanding of the interrelationships and interconnections within a system. A system is a set of ordered elements that interact among each other. The system thinking has to build up three requirements: purpose, description of the elements and characteristics, along with the interconnections that feed into the systems and relate the purpose and elements (Arnold and Wade, 2015).

The theory of urban water metabolism states that in order to perform the different functions involved in delivering water to urban areas, the UWS uses energy, chemicals and fuels, and produces pollutants, emissions, and waste.

[4] The term should not be confused with the systematic approach, which usually defines a model of how to do something.

It must be used to sharpen our understanding of resource/emission paths. In UM, the urban system is considered to be a black box, unable to unveil the links between metabolic processes within the systems and/or their subsystem's components (Huang *et al.*, 2018). On the other side, the nexus approach studies the interconnections among different elements or sectors and the subsequent consequences of decision-making; it specifically concerns the fact that the decisions required to solve a problem in one domain can cause difficulties in another. The contemporary nexus approach is dominated by resource management and security. It seeks to identify the different pathways that lead to resource security, how much stress is put on one sector to supply the demand in another sector, and the nature of the synergies and trade-offs obtained in the different domains of sustainability. The water-energy nexus considers the interconnections among the components as a set of key performance indicators (KPI) as the savings of all flows mentioned above. The present research introduces the pollutant-impacts on the nexus analysis in the water sector.

The application of these theories is demonstrated in this thesis through a conceptual model; this is defined as a simplified representation of a complex system composed by a problem statement, purpose responses or outputs, model inputs/experimental factors, the content of the model, and assumptions and simplifications made (Stewart, 2013). The model content is structured on the basis of what is modelled and how. The outcomes of the model are compared with the observations. A calibrated, validated model is then used for scenarios and the model in turn becomes a tool for explaining choices (Spijker *et al.*, 2010). Framing a real system into a conceptual model requires a level of abstraction in which the model fulfils the validity, credibility criteria within the data, and time constraints. Each discipline perspective conducts the boundaries and surroundings, as well as the level of detail of the conceptual models. In the context of the water sector, the conceptual models have an application and are part of a dependent entity, process, or spatial area, with or without interactions with the surrounding environment. This, thus links them more closely to the information provided by each theory. Figure 3.2 shows the main components of the framework used for the comprehensive assessment, while the description of elements will be outlined in the following subsections.

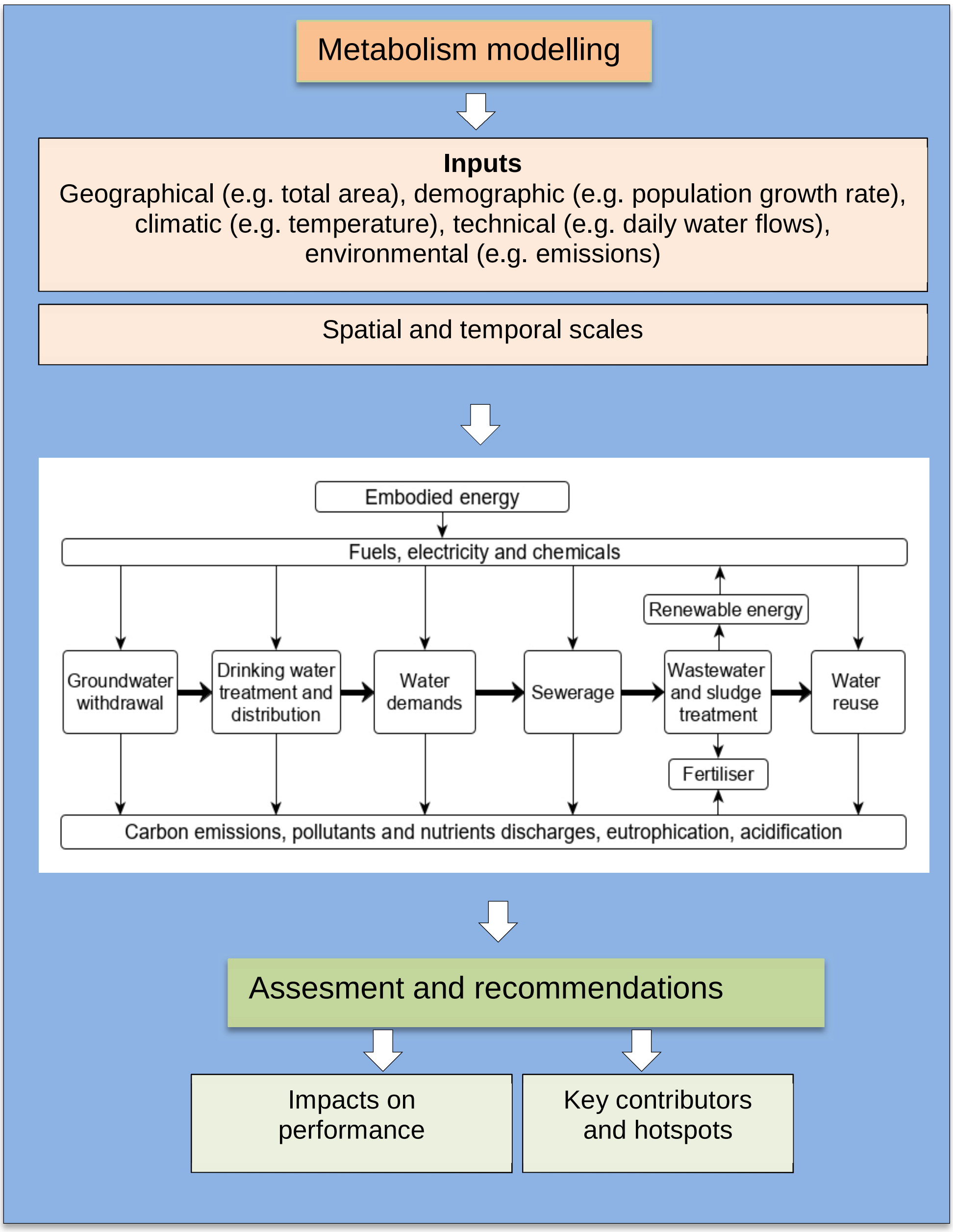

Figure 3.2 Simplified representation of the framework used in this thesis

3.2.1 Goal and scope

The goal of this assessment was to compare the performance of the long-term operation of suggested centralised and decentralised water reuse systems in the case study. The functional unit defined in this study is the water extracted, treated, delivered, and reused per year of operation (m^3/y), similar to other studies (Opher and Friedler, 2016a). This study normalised the results per water demand.

The scope of the analysis is the UWS operation stage, as the construction, maintenance, and demolition phases have a minor influence on the environmental impacts in comparison to the operation phase (Jeong *et al*., 2015; Lane *et al*., 2015; Machado *et al*., 2007). The operation stage includes the water withdrawals, demands, wastewater treatment and sludge, water reuse, and sludge disposal. It also considers the production and allocation of chemicals, electricity, and fuels used within the UWS, in addition to the recovery and replacement of by-products such as biogas and fertiliser replacement due to the use of sludge on agricultural land, described in the following sections. The study excludes the materials used in distribution and sewer networks, and the optional reclaimed water distribution due to lack of data (materials, age, size of pipelines) in the case study.

3.2.2 UWS and key components

The conceptual model is defined in a generic urban water system considered here as a set of five subsystems, shown in Figure 3.3. The main features of the subsystems are:

a) The water supply subsystem considers the number of sources and groundwater and/or freshwater flows, the drinking water treatment, storage, distribution mains and possible leakages up to the point of consumption.
b) The water demands subsystem specifies the different consumption points in the city, including domestic (e.g. toilet flushing, hand basin usage, washing clothes, shower, kitchen, and dishwasher), industrial, commercial, public (e.g. schools and hospitals), and urban irrigation uses. It is necessary, for this subsystem, to define the physical characteristics of local areas (e.g. permeable and impermeable areas).
c) The sewerage subsystem refers to the wastewater generated at each demand point minus the loss per lack of connection to the sewer

network. The sewer system is sequentially connected between different catch basins associated with different sub-catchments based on the gravity of stormwater/wastewater collection systems. The sewer can be a separated or combined network. The latter accounts for stormwater and loss due to evaporation and impermeable space according to local characteristics.

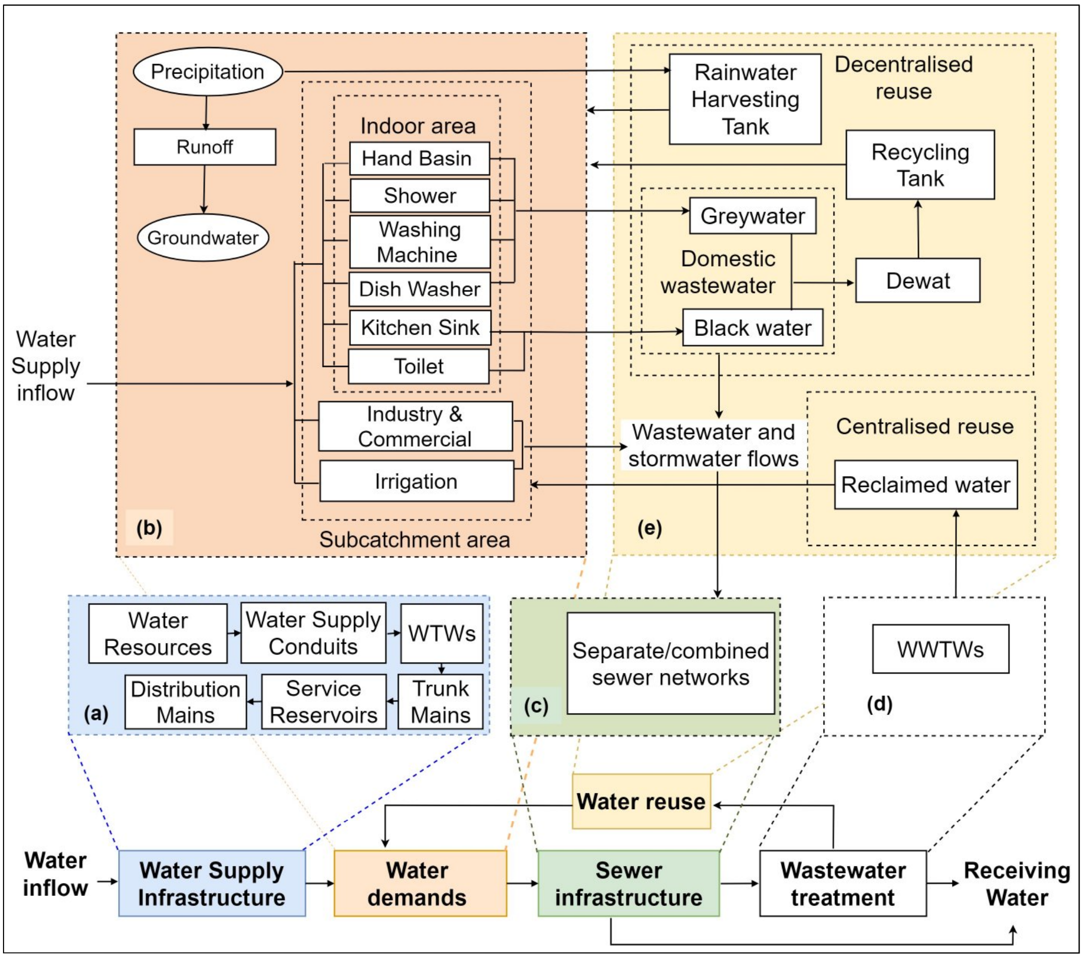

Figure 3.3 Urban Water System and subsystem components, (a)Water supply, (b) Water demands, (c) sewer, (d) wastewater treatment and (e) water reuse. Source: Modified from Behzadian *et al.* (2015a)

a) <u>The wastewater treatment subsystem</u> is the water reclamation process. It uses a mass balance to determine the effluent quality. It is important to define the possible energy recovery (as electricity) and different options for sludge management such as landfill disposal, use in agriculture or incineration.

b) The water reuse subsystem contains centralised and decentralised water reuse. Centralised water reuse includes the distribution of reclaimed water obtained from the WWTW to the demand points. Decentralised water reuse in this thesis is constituted by any combination of domestic effluents generated in one or all household fitting components, typically greywater from washing machines, showers, and hand basins (composed by greywater, yellow and black water). It also includes a decentralised wastewater treatment (DEWAT) facility, a storage recycling tank, and the infrastructure to allocate the treated water at the consumption points. The remaining domestic wastewater effluents that are not considered for reuse, overflows, and any sludge produced in a DEWAT facility are directly discharged into the sewer networks and eventually reach the centralised WWTW. A further description can be found in Behzadian *et al*. (2014).

3.2.3 WaterMet2 tool

This research uses the WaterMet^2 (WM2) modelling tool. This is a conceptual mass balance based model that tracks down the main flows and fluxes such as water, energy, materials, chemicals, and pollutants among others in a UWS (Behzadian and Kapelan, 2015b). The WM2 is based on the urban water metabolism approach, with the ability to study urban water systems and interventions for resilience and sustainability. Various applications of the model include: a comparison of rainwater harvesting, greywater recycling, and desalination scenarios around population growth in the Galapagos Islands and Ecuador (Reyes *et al.*, 2017); the implementation of rainwater harvesting to minimise water demand and local floods in Oslo (Behzadian *et al.*, 2018), and the decision to increase water sources or pipeline rehabilitation for resilience and sustainability in a European city (Behzadian and Kapelan, 2015b; Morley *et al.*, 2013).

The WM2 is a time-step and spatial distributed model. It simulates the operation of UWS by aggregation of stocks and main water flows (potable, wastewater, green, and greywater) at daily steps simulations for a user-defined period (up to 40 years). Sequential modelling, using the water flows, tracks down different associated fluxes such as energy consumption, GHG, pollutant mass loads entering the system, eutrophication, and acidification potential The model operates at four spatial scales, namely the indoor, local area, subcatchment, and city (Behzadian *et al.*, 2014). The indoor area is the smallest spatial scale representing a single household property without any surroundings; typical water demands include toilet, shower, washing machine, and kitchen (Behzadian and Kapelan, 2015b). The local area contains any number of similar indoor areas with similar water demands and surroundings consisting of pervious and impervious surfaces. The key elements defining the local areas are the number of inhabitants, surface occupied, and the characteristic of permeable and impermeable surfaces. Any group of local areas forms a subcatchment. The city is the largest spatial scale aggregating all previously mentioned elements.

3.3 Modelling

This section presents the most relevant flows, components and equations of the model adopted by WM2 presented in Behzadian *et al.* (2018).

3.3.1 Water flows

WM2 model takes a demand approach, similar to other frameworks (Kenway, 2013). The water model is developed first, with the subsequent inclusion of the energy, pollutant load and environmental impacts associated with such water flows. The main water flows calculated are potable water, referred to as water that has had treatment to render it drinkable; water supply flow, which is the sum of potable water; any other water delivered; plus, water leakages. The simulation assumed a daily time step approach. Firstly, the daily water demand in the local area was calculated, secondly, the release to consumers in constraints by conveyance infrastructure capacity and storage components (Behzadian and Kapelan, 2015b). The water demanded depends on the water use and is affected by population growth, industrial expansion and the season. Precipitation is either collected as stormwater in a combined sewer, percolated through the impervious area, or transformed into green water in the rainwater harvesting tank (Behzadian and Kapelan, 2015b).

3.3.2 Wastewater flow and quality

Wastewater flow is calculated by WM2 using a simplified approach modelling the key elements: wastewater as a percentage of the water demand, stormwater (denoting the combined/separate draining routes to the sewer and loss through percolation and evaporation); the wastewater treatment work and the final points of receiving treated water.

Water quality is defined in sewerage, wastewater treatment and water reuse subsystems. The pollutants and their concentrations should be defined at household scale in terms of the toilet, shower, kitchen, washing machine, and hand basin, at local scale as industrial and commercial, and in stormwater. Such inflow concentrations were obtained through the literature using the references listed in Table 2.3. Five pollutants were defined: chemical oxygen demand (COD), total phosphorus (TP) and total nitrogen (TN), total suspended solids (TSS) and biochemical oxygen demand (BOD). Pollutant load in kg/d is the product between pollutant concentrations in kg/m^3 multiplied by their flow rate in m^3/d. The model assumes a complete mixed reactor without any dispersion, diffusion, decay or growth of pollutants. No biogeochemical functions are embedded; hence, only a percentage of the reduction efficiencies is required (Behzadian and Kapelan, 2015b). The wastewater quality is expressed by Eqs. 1 and 2.

$$\boldsymbol{L1_{it} = L0_{it} + V_{it} \times C_{it}} \quad \textbf{Eq. 1}$$

$$\boldsymbol{L1_{it+1} = L1_{ijt} \times (1 - R_{it}/100)} \quad \textbf{Eq. 2}$$

where $L1_{it}$, $L0_{it}$= load of contaminant i at WWTW and day t after and prior to mixing with inflow, respectively (kg/day); c_{it}= concentration of contaminant i for inflow to WWTW at day t (mg/L), R_i=removal percentage of contaminant i for wastewater in WWTW (%/100); $L1_{it+1}$ = load of contaminant i at WWTW at day $t+1$ (kg/day).

3.3.3 Sludge

Sludge generation is calculated by Eq. 3, based on the percentage removal of total suspended solids:

$$\boldsymbol{Sl_t = (TSS_a \times TSS_{rmv}) \times (\frac{1}{1000}) \times TWW_t} \qquad \textbf{Eq. 3}$$

where $Sl_t$is the total sludge generated at time step t (kg/day); TSS_a is the total suspended solids concentration in the influent wastewater (mg/L); TSS_{rmv} is the total removal in the WWTW and sludge process (%/100); 1/1000 is the conversion factor to kg/m^3; TWW_t is the treated wastewater at time step t (m^3/day).

The final disposal of sludge through landfill, agriculture, or incineration is assumed in the model as a percentage of the total sludge mass, which is inputted by the user. The model assumes credited scenarios for fertiliser replacement equivalents to urea and single super phosphate. Such assumptions are consistent with other models (Foley *et al.*, 2010; Lundie *et al.*, 2004). The amount of potential replacement of such by-products in each time step is calculated according to Eqs. 4 – 7.

$$\boldsymbol{SSP_t = SP_t \times SF \times 18/100} \qquad \textbf{Eq. 4}$$

$$\boldsymbol{SP_t = PC \times TWW_t \times P_{rmv}} \qquad \textbf{Eq. 5}$$

where SSP_t is superphosphate generated at time step t (kg/day); SP_t is phosphorus sludge generated at time step t (kg/day); SF is the percentage of sludge converted to fertiliser (%/100); 18% is the amount of P_2O_5 in the composition of fertiliser; PC is total phosphorus concentration of untreated influent wastewater [(mg/L)/1000]; P_{rmv} is total phosphorus removed (%/100).

$$\boldsymbol{U_t = SN_t \times SF \times \frac{28}{60}} \quad \textbf{Eq. 6}$$

$$\boldsymbol{SN_t = NC \times TWW_t \times (N_{rmv} - N_{rls})} \quad \textbf{Eq. 7}$$

where U_t is the urea generated at time step t (kg/day); SN_t is the nitrogen sludge generated at time step t (kg/day); NC is the total nitrogen concentration of untreated wastewater/influent [(mg/L)/1000]; N_{rmv} is the total nitrogen removed (%/100); N_{rls} is the nitrogen released to the atmosphere (%/100) and 28/60 is the fraction of urea 45%. Note that P_{rmv}, N_{rmv} and N_{rls} are the input data in WaterMet[2].

3.3.4 Biogas

Biogas generated by sludge treatment is calculated by Eq. 8:

$$\boldsymbol{BG_t = DB \times BTW \times TWW_t} \quad \textbf{Eq. 8}$$

where BG_t is the biogas generated at time step t (kg/day); DB is the density of biogas (kg/m^3); BTW is the biogas generated per unit volume of treated wastewater (m^3/m^3); TWW_t is the treated wastewater at time step t (m^3/day).

3.3.5 Total net energy

The energy flows result from the net balance of direct energy consumption (e.g. fossil fuels and electricity) or indirect use (e.g. embodied energy in chemicals and by-products). Different energy sources identified in the urban water system are summarised in Table 3.1.

Table 3.1 Source of energy in UWS

Consumed		Avoided
Direct	**Indirect**	**WWTW**
Electricity from grid Fuels	Embodied in chemicals	Electricity from biogas Energy in SSP[b] Urea

Total energy balance is the result of caused minus avoided energy. Total caused energy, as a result of transport and groundwater treatment, wastewater and reclaimed water flows, is calculated using Eq. 9:

$$\boldsymbol{En_{wnt}} = \left(\boldsymbol{EnU_{wn}} + (\boldsymbol{EnFU} \times \boldsymbol{FU} \times \boldsymbol{DFU}) + \left((\boldsymbol{EnCh_p} + \boldsymbol{EnCh_{tr}}) \times \boldsymbol{ChU}\right) - \boldsymbol{EnB}\right) \times \boldsymbol{QW_{nt}} \qquad \textbf{Eq. 9}$$

where En_{wt} is the energy caused at time step t due to the water flow wn (kWh/d); wn is either water supply, wastewater or water reused; EnU is the electricity used per wn flow (kWh/m^3); $EnFU$ is the energy of relevant fuel used (kWh/kg); FU is transport fuel used (litre/m^3); DFU is density of fuel (kg/litre); $EnCh_p$ and $EnCh_{tr}$ are the embodied energy of the production and transport of the relevant chemical used (kWh/kg); ChU is the chemical used (kg/m^3); EnB is the electricity from biogas (kWh/m^3); QW_{nt} is the flow of water n at time step t (m^3/d).

The avoided energy is caused by the replacement of electricity and fertilisers for sludge treatment, as in Eq. 10.

$$\boldsymbol{EnE_{avt}} = \boldsymbol{En_B} - ((\boldsymbol{EnSSP_p} + \boldsymbol{EnSSP_{tr}}) \times \boldsymbol{SSP_t})) - ((\boldsymbol{EnU_p} + \boldsymbol{EnU_{tr}}) \times \boldsymbol{U_t})) \qquad \textbf{Eq. 10}$$

where EnE_{avt} is the energy avoided at time step t (kWh/d); En_B is the electricity produced in the biogas chamber (kWh/d); $EnSSP_p$ and $EnSSP_{tr}$ are the embodied energy of production and transport of single superphosphate produced (kWh/kg); SSP_t is the superphosphate generated at time step t (kg/day); EnU_p and EnU_{tr} are the embodied energy of production and transport of single superphosphate produced (kWh/kg); U_t urea generated at time step t (kg/day).

The embodied energy is calculated in both production and transport of chemicals, fuels or by-products by Eq. 11.

$$\boldsymbol{EnE_i} = \sum \boldsymbol{En_i} \times \boldsymbol{m_i} \qquad \textbf{Eq. 11}$$

Where EnE_i is the embodied energy of i (kWh/m^3), *i* is either chemicals, fuels or fertilisers; En_i are the characterisation factors of cumulative energy demand (kWh/kg_i) in Ecoinvent3 database (Werner *et al.*, 2016); m_i load is the quantity of *i* used in the specific flow (kg/m^3).

3.3.6 Environmental impact assessment

Environmental impact categories analysed through WM2 are the potential impacts of global warming, eutrophication and acidification. These are calculated based on the adapted methodology of life cycle impact assessment (ISO 14044). This standard establishes four steps: goal and scope definition, inventory analysis, impact assessment and interpretation. The quantitative inventory is the amount/load of classified inputs (i.e. electricity, chemicals, fuels, by-products) according to the functional unit. The inventory also states their characterisation factors of the potential environmental impact value. In the impact step, all inventoried inputs are multiplied by the characterisation factor. In WM2, such factors correspond to those established for a CML[5] 2001 method (Heijungs *et al.*, 1992) or reported in the literature. The generic form to calculate an impact is presented in Eq. 12.

$$\boldsymbol{Impact_j = \sum_j CF_j \times m_j} \qquad \textbf{Eq. 12}$$

where the impact; *j* can be global warming, eutrophication or acidification potential; CF_j are the coefficient factors of the potential impact related to the substance *i* ($kgCO_2$ eq/kg or $KgPO_4$ eq/kg or $kgSO_2$/kg); m_i is the quantity/mass of the substance *i* inventoried (kg), *i* is either chemicals, fuel, sludge, biogas or fertiliser.

In WM2 a distributed and daily time step calculation allows for the temporal analysis of each environmental impact. This will be shown in the following quantifications, where the variables changing over average daily time steps in

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5 CML is a method developed in The Netherlands; it takes its name from the Dutch abbreviation of Leiden University Institute of Environmental Sciences.

this equation are water flows that result from each impact in the corresponding time step. Other parameters are constant over the average daily time step.

a) Global warming potential (GWP)

Global warming is a phenomenon caused by the anthropogenic greenhouse gas emissions that enhance radiative forcing (i.e. heat radiation absorption), causing the temperature at the earth's surface to rise. Although this is also a natural phenomenon, the GWP impact estimates the GHG emissions released from anthropogenic origin into the atmosphere (Guinee *et al.*, 2002). GWP accounted emissions resulted in the operation of UWS emitting to air such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O); the latter two have higher GWP potentials, 28 $KgCO_2/kg\text{-}CH_4$ and 265 $kgCO_2/N_2O$, respectively (IPCC, 2014). All of these are expressed in kg CO_2 equivalent. Such emissions result from the entire urban water system and some are assumed to be credited or avoided. The GWP of the entire system is calculated as the balance of the global warming potentials in the water supply, drinking water treatment, wastewater treatment, sludge management, and biogas chamber. A summary of sources of GHG emissions in the water system is shown in Table 3.2.

Table 3.2 Source of global warming potential in UWS

Caused GHG Emissions[a]			Avoided GHG emissions
CO_2 caused	**CH_4 caused**	**N_2O caused**	**CO_2 avoided**
• Electricity used • Fossil fuel used • Embodied in chemicals	• Biogas released • Biogas incomplete combustion • Fugitive emissions from landfill • Fugitive emissions from fertiliser	• Fugitive Landfill • Fugitive Fertiliser	• Electricity from biogas • SSP[b] • Urea

[a] Caused CO_2 are in water supply, WWTW and water reuse; CH_4, N_2O and CO_2 avoided are in WWTW only. [b]SSP: single superphosphate

Caused CO_2 emissions, both direct and indirect, are present in the water supply, wastewater and water reuse subsystem. Direct emissions originate due to the use of electricity from the grid and the burning of fuels to transport water. Indirect emissions (also known as embodied) comprise CO_2 emissions from producing and allocating chemicals and fuels (Eq. 13). Carbon content released in the biological reactors from the wastewater treatment process is assumed to be biogenic in origin and thus excluded from this accounting.

$$\boldsymbol{GWP_{wnt} = \Big((GEn \times EnU) + (GFU \times FU \times DFU) + \big((GCh_p + GCh_{tr}) \times ChU\big)\Big) \times QW_{nt}} \qquad \textbf{Eq. 13}$$

where GWP_{wt} is the resulted global warming potential at time step *t* due to the water flow *n* (kg CO_2-eq/d)*;* *n* is either water supply, wastewater, or water reused; GEn is the GHG emissions of electricity used (kg CO_2-eq/kWh); EnU is the electricity used (kWh/m^3); GFU is the GHG emission of relevant fuel used (kg CO_2-eq/kWh); FU is transport fuel used (litre/m^3); DFU is density of fuel (kg/L); GCh_p and GCh_{tr} are the embodied GHG emissions of production and the transport of relevant chemical used (kg CO_2-eq/kg); ChU is the chemical used (kg/m^3); QW_{nt} is the flow of water n at time step *t* (m^3/d).

In addition, the wastewater subsystem has two additional managements: biogas and sludge. The directly caused and avoided CH_4 emissions are due to the incomplete combustion of biogas or credited as a result of the production of renewable electricity. Caused GWP for biogas management is shown by Eq. 14.

$$\boldsymbol{GWP_{Bt} = Gm \times MB \times \big((1 - BC) + (1 - DC) \times BC\big) \times DB \times BTW \times TWW_t - ((GE \times EB) \times TWW_t)} \qquad \textbf{Eq. 14}$$

where GWP_{Bt} is the global warming potential in the biogas chamber, Gm are the GHG emissions of methane release (kgCO_2-eq/kg); MB is the methane percentage in biogas (%/100); BC is percentage of biogas captured (%/100); DC is degree of biogas combustion (%/100); DB is density of biogas (kg/m^3); BTW is biogas generated per unit volume of treated wastewater (m^3/m^3), GE

is the GHG emissions of relevant substituted electricity (kg CO_2-eq/kWh); EB is the electricity generated from biogas (kWh/m^3); TWW_t is the treated wastewater at time step t (m^3/day).

Sludge management causes fugitive CH_4 and N_2O emissions from sludge disposal in landfills and agriculture. Avoided emissions are due to fertiliser replacement, which is assumed to be urea and single super phosphate and is consistent with other accounts (Foley *et al.*, 2010; Lundie *et al.*, 2004). The balance of caused and avoided emissions is expressed by Eq. 15.

$$\boldsymbol{GSl_t = GM \times S_t \times (FMF \times FS + FML \times LS) + GN \times S_t \times (FNF \times FS + FNL \times LS) - ((GSSP_p + GSSP_{tr}) \times SSP_t)) - ((GU_p + GU_{tr}) \times U_t))} \qquad \textbf{Eq. 15}$$

where GSl_t is GHG caused at time step t (kg CO_2-eq/day); GM is GHG emissions of methane (kg CO_2-eq/kgCH_4); S_t is the sludge generated at time step t (kg/day); FMF are the fugitive methane emissions from fertiliser (kg/kg of sludge); FS is the percentage of sludge converted to fertiliser (%/100); FML are the fugitive methane emissions from landfill (kg CH_4/kg of sludge); LS is the percentage of sludge converted to landfill (%/100); GN are the nitrous oxide emissions (kg CO_2-eq/kg of methane); FNF are the fugitive nitrous oxide emissions from fertiliser (kgN_2O/kg of sludge); FML are the fugitive nitrous oxide emissions from landfill (kgN_2O/kg of sludge); $GSSP_p$ and $GSSP_{tr}$ are the emissions of SSP production and transport, respectively; SSP_t is the emission superphosphate generated at time step t (kg/day); GU_p and $GU_{tr}$are theGHG emissions of substituted urea for production and transport (kg CO_2-eq/kg); U_t urea generated at time step *t* (kg/day). Emissions in units other than CO_2 are converted to the equivalent using 28 KgCO_2/kg-CH_4 and 265 kgCO_2/N_2O values (IPCC, 2014).

A good proportion of GHG emissions is related to heating water at household levels (e.g. between 4 and 5% of the UK's emissions; Chisholm *et al.*, 2013). The reduction of GHG emissions related to water heating is out of the direct control of the water industry and its regulator, therefore these emissions are not considered in this research. The research specifically focuses on direct and indirect emissions by water companies during operational phases over the planning horizon

b) Eutrophication

Eutrophication potential is the impact produced by the presence of macronutrients, nitrogen (N) and phosphorus (P), in bodies of water. Such nutrients cause elevated biomass production in both aquatic and terrestrial ecosystems and its decomposition may lead to decrease oxygen levels in bodies of water (Morelli *et al.*, 2018). The additional consumption of oxygen in biomass decomposition is measured as BOD and accounted in the CML impact method as COD; such biomass is also considered under the category of eutrophication (Guinee *et al.*, 2002). COD is the amount of oxygen required to oxidise the organic compounds in a water sample to carbon dioxide and water utilising a strong chemical oxidant (Guinee *et al.*, 2002).

CML is an early model and does not consider the spatial difference for eutrophication emissions released to soil, air, and water. Hence, the CML method does not distinguish freshwater and marine eutrophication. The nutrient fluxes considered in terms of eutrophication potential by this thesis are the discharges of phosphorus (P), nitrates (NO_3), and chemical oxygen demand (COD) in effluents, the indirect eutrophication caused by electricity, fuels, chemicals, and sludge disposal (Table 3.3). The emissions of ammonia (NH_3) released through sludge and biogas are considered due to the contribution to N deposition in rivers and lakes (Zhang *et al.*, 2017). The characterisation factor is kg PO_4 eq/kg. The equivalent factors are 1 kgPO_4, 0.35 kgPO_4/NH_3, 0.022 kgPO_4/COD, and 3.06 kgPO_4/P (Heijungs *et al.*, 1992). Such equivalences use the stoichiometric relationship of the Redfield ratio (Morelli *et al.*, 2018). This ratio describes a generic elemental composition in biomass, $C_{106}H_{263}O_{110}N_{16}P$, used to develop an equivalency between elements and nutrient forms (Heijungs *et al.*, 1992).

Table 3.3 Source of Eutrophication in UWS

Caused Eutrophication					Avoided Eutrophication
PO_4 caused	**TP caused**	**NH_3 caused**	**NO_3 caused**	**COD caused**	**PO_4 avoided**
• Electricity • Fossil fuel • Embodied	• TP discharge to water	• Release of ammonia in incomplete biogas combustion • Fugitive landfill • Fugitive fertiliser	• TN discharge to water	• COD discharge to water	• Electricity • SSP • Urea

Eutrophication accounted in the water supply, wastewater, and water reuse subsystem for transport and treatment of water flows is calculated in Eq. 16.

$$\boldsymbol{Eu_{wnt}} = \Big((\boldsymbol{EuEn} \times \boldsymbol{EnU}) + (\boldsymbol{EuFU} \times \boldsymbol{FU} \times \boldsymbol{DFU}) + \big((\boldsymbol{EuCh_p} + \boldsymbol{EuCh_{tr}}) \times \boldsymbol{ChU}\big)\Big) \times \boldsymbol{QW_{nt}} \qquad \textbf{Eq. 16}$$

where Eu_{wnt} is the Eutrophication ($kgPO_4$ eq/d); EuEn is the eutrophication factor electric energy ($kgPO_4$ eq/kWh); EnU is the electricity used (kWh/m^3); Eu_{FU} is the eutrophication factor of fuel used ($kgPO_4$ eq/kg_{FU}); Eu_{Ch} is the eutrophication factor chemical "n" ($kgPO_4$ eq/kg_{Chn}); FU is transport fuel used (litre/m^3); DFU is density of fuel (kg/litre); $EuCh_p$ and $EuCh_{tr}$ are the embodied acidification emissions of the production and transport of the relevant chemical used (kg CO_2-eq/kg); ChU is the chemical used (kg/m^3); QW_{nt} is the flow of water n at time step *t* (m^3/d).

Eutrophication in discharges are calculated by Eq. 17.

$$\boldsymbol{Eu_{WDt}} = (\boldsymbol{MFWwO_{P_i}} \times \boldsymbol{EU_{P_i}} \times \boldsymbol{WW_{sw}}) + ((\boldsymbol{MFTWw_{P_i}} - \boldsymbol{MFRw_{Pi}}) \times \boldsymbol{EU_{P_i}} \times \boldsymbol{TWWD_t} \qquad \textbf{Eq. 17}$$

Eu_{WDt} is the eutrophication in water discharges at time step *t*; $MFSwO_{Pi}$ is the mass pollutant flow in sewer overflows of pollutant "i" (Kg/d); P*i* is COD, TP, NO3, TP; Eu_{Pi}: Eutrophication of pollutant "i" ($kgPO_4$ eq/kg_{Pi}); WW_{sw} is wastewater (m^3/d); $MFWwO_{Pi}$ is the mass flow "i" in wastewater overflows (kg/d); $MFTWw_{Pi}$ is the mass flow of treated wastewater of pollutant "i" (kg/d); $MFRw_{Pi}$: Mass flow of reuse water (kg/d); $TWWD_t$ treated wastewater discharged considering the reuse water flow at time step t (m_3/d).

Eutrophication in sludge is expressed by Eq. 18.

$$\boldsymbol{Eu_{Slt}} = \left(\boldsymbol{MFSl_{TN}} \times \boldsymbol{FEm_{NH_3}} \times \boldsymbol{EU_{NH3}} \times (\boldsymbol{A}+\boldsymbol{L})\boldsymbol{S_t}\right) - ((\boldsymbol{EuSSP_p} + \boldsymbol{EuSSP_{tr}}) \times \boldsymbol{SSP_t})) - ((\boldsymbol{EuU_p} + \boldsymbol{EuU_{tr}}) \times \boldsymbol{U_t})) \quad \textbf{Eq. 18}$$

$\boldsymbol{Eu_{Slt}}$ is the eutrophication caused in sludge management (kg PO_4-eq/d); $MFSl_{TN}$ is the mass flow of TN in sludge (Kg_{TN}/d); FEm_{NH3} are the fugitive emissions NH_3 ($kgNH_3$/KgTN); Eu_{NH3} is the eutrophication factor of ammonia ($kgPO_4$ eq/$kgNH_3$); A and L are the fractions of sludge (%/100) sent to agriculture or landfill; S_t is the sludge generated at time step t (kg/day); $EuSSP_p$ and $EuSSP_{tr}$ are the emissions resulting from SSP production and transport, respectively; SSP_t is the superphosphate emission generated at time step t (kg/day); EuU_p and EuU_{tr} are the emissions of substituted urea for production and transport (kg PO_4-eq/kg); U_t is urea generated at time step *t* (kg/day).

The balance of eutrophication for the biogas chamber is the result of ammonia release for incomplete combustion and credited for the use of renewable electricity, as shown by Eq. 19.

$$\boldsymbol{EuP_{Bt}} = \boldsymbol{Eu_{NH3}} \times \boldsymbol{NH3} \times \left((\boldsymbol{1}-\boldsymbol{BC}) + (\boldsymbol{1}-\boldsymbol{DC}) \times \boldsymbol{BC}\right) \times \boldsymbol{DB} \times \boldsymbol{BTW} \times \boldsymbol{TWW_t} - ((\boldsymbol{EuE} \times \boldsymbol{EB}) \times \boldsymbol{TWW_t}) \quad \textbf{Eq. 19}$$

where EuP_{Bt} is the eutrophication potential in the biogas chamber, Eu_{NH3} is the eutrophication factor of ammonia ($kgPO_4$ eq/$kgNH_3$); NH_3 is the ammonia percentage in biogas (%/100); BC is percentage of biogas captured (%/100); DC is degree of biogas combustion (%/100); $DB$is density of biogas (kg/m^3); BTW is biogas generated per unit volume of treated wastewater (m^3/m^3), EuE is the eutrophication of relevant substituted electricity (kg PO_4-eq/kWh); EB is the electricity generated from biogas (kWh/m^3); TWW_t is the treated wastewater at time step t (m^3/day).

c) Acidification

Acidification takes into account the increasing concentration of acidifying substances in the lower atmosphere that are then converted into acid rain (Hasler *et al.*, 2015). Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, ecosystems and materials (buildings). Examples include fish mortality in Scandinavian lakes, crumbling of building materials and forest decline (Guinee *et al.*, 2002). The major emissions of acidifying pollutants are SO_2, NO_2 and NH_3. Acidification potential is expressed in kg SO_2 equivalent. 1 kgSO_2/kg-SO_4 is equivalent to 0.7 kgSO_2/kgNO_2 and 1.88 kgSO_2/kgPO_4/NH_3 (Heijungs *et al.*, 1992). The sources of acidification in the entire water systems are listed in Table 3.4.

Table 3.4 Source of acidification in UWS

Caused Acidification			Avoided Acidification
SO_2 caused	**NH_3 caused**	**NO_2 caused**	**SO_2 avoided**
• Electricity • Fossil fuel • Embodied • Hydrogen sulphide in biogas combustion	• Fugitive landfill • Fugitive fertiliser • Release of ammonia in incomplete biogas combustion	• Biogas combustion	• Electricity • Single Superphosphate • Urea

Acidification potential in transport and treatment in main water flows (water supply, wastewater and water reuse) is calculated in Eq. 20.

$$\boldsymbol{AcP_{wnt}} = \Big((\boldsymbol{APEn} \times \boldsymbol{EnU}) + (\boldsymbol{APFU} \times \boldsymbol{FU} \times \boldsymbol{DFU}) + \big((\boldsymbol{APCh_p} + \boldsymbol{APCh_{tr}}) \times \boldsymbol{ChU}\big)\Big) \times \boldsymbol{QW_{nt}} \qquad \textbf{Eq. 20}$$

where AP$_{wt}$ is the resulted global warming potential at time step *t* due to the water flow *n* (kg SO_2-eq/d)*;* *n* is either water supply, wastewater or water reused; GEn are the acidification emissions of electricity used (kg SO_2-

eq/kWh); EnU is the electricity used (kWh/m^3); $APFU$ are the acidification emissions of relevant fuel used (kg SO_2-eq/kWh); FU is transport fuel used (litre/m^3); DFU is density of fuel (kg/L); $APCh_p$ and $APCh_{tr}$ are the production and transport emissions of relevant chemicals used (kg SO_2-eq/kg); ChU is the chemical used (kg/m^3); QW_{nt} is the flow of water n at time step *t* (m^3/d).

Acidification in biogas is expressed by Eq. 21.

$$\boldsymbol{AcP_t = (AcP_{NH3} \times AB \times (1 - DC) + AN \times AB \times DC \times \frac{46}{17} + AS \times HB \times DC \times \frac{64}{34}) \times DB \times BTW \times TWW_t} \qquad \textbf{Eq. 21}$$

where AC_t is the acidification caused in the biogas chamber at time step t (kg SO_2-eq/day); AcP_{NH3} is the acidification of ammonia gas (kg SO_2-eq/kg); AB is the ammonia percentage in biogas (%/100); DC is the degree of biogas combustion (%/100); AN is the acidification of nitrogen dioxide (NO_2) gas (kg SO_2-eq/kg); AS is the acidification of hydrogen sulphide (H_2S) gas (kg SO_2-eq/kg); HB is the hydrogen sulphide (H_2S) percentage in biogas (%/100); DB is the density of biogas (kg/m^3); BTW is the biogas generated per unit volume of treated wastewater (m^3/m^3); TWW_t is the treated wastewater at time step t (m^3/day).

Caused acidification in sludge is calculated by Eq. 22.

$$\boldsymbol{Ac_{Slt} = AcP_{NH3} \times S_t \times FAF \times FS - ((AcSSP_p + AcSSP_{tr}) \times SSP_t)) - ((AcU_p + AcU_{tr}) \times U_t)} \qquad \textbf{Eq. 22}$$

where Ac_{Slt} is acidification caused by sludge management at time step t (kg SO_2-eq/day); AcP_{NH3} , acidification of ammonia gas (kg SO_2-eq/kgNH_3); S_t =sludge generated at time step t (kg/day); FAF =fugitive ammonia gas emissions from fertiliser (kg NH_3/kg of sludge); FS= percentage of sludge converted to fertiliser (%/100); $AcSSP_p$ and $AcSSP_{tr}$ are the emissions of SSP production and transport, respectively (kg SO_2-eq/kgSt); SSP_t is the emission superphosphate generated at time step t (kg/day); AcU_p and ACU_{tr} are the substituted urea emissions from production and transport (kg SO_2-eq/kg); U_t urea generated at time step *t* (kg/day).

3.3.7 Data input

The model requires a different type of information. Table 3.5 shows the data required in seven categories. Such data are a combination of empirical and secondary data.

Table 3.5 Summary of input data required for modelling the framework

Category	Sub-category	Data
Location	Total area	Geographic coordinates
Area of study	Climatic data	Total area (ha) Permeable and impermeable area (%)Daily recorded data of rainfall Maximum, average, and minimum temperature (^oC) Monthly average vapour pressure Relative humidity
Population	Demographic data	Total population (inhabitants) Number of households Occupancy Annual population growth rate (%)
Water supply subsystem	Drinking water treatment	Number and type of drinking water treatment plants Daily water extraction (m^3/d) Energy consumption (kWh/m^3) Type and chemicals dosage Leakages Proportional distribution
Water demand subsystem	Demands	Daily domestic and public water supplies per capita (L/c.d) Daily industrial and irrigation water demand (m^3/d)
Sewerage subsystem	Capacity	Type and capacity (m^3/d) % conversion from potable water to wastewater Proportion of wastewater lost (%) Pollutants concentration (mg/L)

…continue table 3.6 Summary of input data required for modelling the framework

Category	Sub-category	Data
Wastewater subsystem	Wastewater treatment	Number and type of WWTW Daily wastewater inflow and outflows (m^3_{ww} /d) Type and chemicals dosage kg/m^3_{ww} Remotion efficiency (%)
	Energy	Energy consumption (kWh/m^3_{ww}) Energy production (kWh/m^3_{ww}) Fuels (kg/m^3)
	Sludge	Monthly sludge production Biogas production (m^3_{biogas}/m^3_{ww}) Biogas use and release (%) Disposal type
Water reuse subsystem	Characteristics of strategies	Type of reuse Start year of operation Sources of treated wastewater and end use Recycling water tank capacity (m^3) Energy in retrofitting (kWh/m^3_{wr})
	DEWAT	Energy in operation Type and chemicals dosage kg/m^3_{gw}
Environmental impacts	Equivalent factors	GHG emissions kg CO_2eq/kg; eutrophication kgPO_4eq/kWh; Embodied energy kWh/kg
	Characterisation factors	GWP, EuP, AcP and energy demand characterisation factors

3.4 Key performance indicators

The performance assessment of water reuse strategies comprised a set of five KPIs, as shown in Table 3.7, selected from the three angles of the WEP nexus to undertake a performance assessment of water reuse strategies.

Table 3.7 Description of key performance indicators

Nexus	KPI	Unit
Water	Deficit of water supply	%
	Potable water savings	m^3/year
Energy	Energy savings	kWh/m^3/year
Pollution	Global warming potential	kg CO_2eq/m^3/year
	Eutrophication potential	Kg PO_4eq/m^3/year
	Acidification potential	Kg SO_2eq/m^3/year

Deficit of water supply is defined as one minus the ratio of the total water supplied to the total water demand over the planning horizon. It is expressed in percentage (Behzadian and Kapelan, 2015a). Hence, fully supplied water demands have a deficit of zero and any other deficit will have a value above zero. Potable water is defined as the amount of potable water supplied from conventional water resources.

3.5 Case study and data collection

3.5.1 Selection of case study

The case study must comply with a number of requirements and criteria in order to test the model. Three relevant considerations were: a) availability of data. The UWS has enough information on the main data of historic water withdrawals, energy consumption, and chemicals used in addition to other descriptions related to the UWS (see Table 3.4). Access to other databases and information or the possibility of using other sources of information was also important; b) Collaboration agreements. The possibility to access primary data from water utilities and establish a collaboration agreement to support this thesis; and c) Research interest in urban water reuse. The area under study is water reuse.

3.5.2 Desk research

Desk research was carried out to compile the use of relevant literature for factor emissions, CML factors from CML-IA baseline V3.01 / World 2000 and cumulative energy demand factors in the Ecoinvent database (Werner *et al.*, 2016), and other publications. Official requests to access public data of monitoring climate stations in the study (e.g. daily temperature, daily precipitation, wind speed (m/s), relative humidity (%), vapour pressure (hPa) and daily average of sunshine hours). Public information consulted included maps. These included urbanised plots (polygon), land use (polygon), infrastructure and industries (polygon), political boundaries per municipality, and the Digital Elevation Model.

3.5.3 Field visits

Three field visits were scheduled to the case study area during the period 2016-2018, with a follow-up visit in 2019. Each visit had a specific goal and different activities were performed during the average two-week duration of each visit. The first aimed to establish the collaboration and collect a first set of data. Activities planned included meeting with water utility directors, key government representatives from the local areas in order to inform them of the interest in collaboration, outline the list of data required, visit the area, and collect data on water and energy. The second visit aimed to perform a survey of local experts in order to better understand the local water reuse practice. The aim was to complete and update the information required. The third visit was used to disseminate the partial results and update decision makers with the progress made. They also provided the researcher the latest data.

3.5.4 Surveys to experts

A complementary analysis used the views of an expert panel opinion on water reuse perspective. The responses were only informative for configuring the water reuse strategies and for the researcher to understand the general context of the case study. This model does not take a mixed qualitative-quantitative approach. A panel consisting of 12 practitioners and eight researchers shared their views about the possible strategies in the future growth of water reuse. An online survey was specifically designed to gather their opinions. The survey took the form of a structured questionnaire in which numeric open questions asked about the possible increase in the adoption rate of water reuse (both at centralised and decentralised levels). Using Opinio 7.6.4 software (2017), the questionnaire was set up online. Beforehand, every participant received via email an informative sheet explaining the aim of the survey, instructions and the use of their responses. Whenever possible, the panellist received personal assistance from the researcher. An English translation of the original Spanish questionnaire is in Appendix 1. The responses were anonymised referring to each expert with a random ID to protect personal data. The survey complied with the ethics and Data Protection

Act 1998 in the UK and was registered as Z6364106/2015/06/69 at the Finance and Data Protection office at UCL. The analysis used descriptive statistics, via SPSS (V.20). Multiple choice questions were presented with a histogram of frequencies to determine the popularity of the strategies. The numeric questions used the mean values, complemented with frequency histograms. The possible interventions were presented to a group of representatives from each water utility to discuss improvements.

3.6 Setting up the conceptual model

3.6.1 Typology and characteristics of the UWS

The step to set up the model refers to defining the urban water system in the conceptual model within the modelling tool. To do that it is necessary to make some assumptions about the typology of the simplified UWS into the elements predefined by the modelling tool. It is also necessary to specify the characteristic of each component, determine the constant values, run and simulate the model. The first specification is the definition of the UWS subsystems, if an integrated UWS is required.

3.6.2 Local area definition

The local areas comprise a number of similar indoor areas (households), combined with industrial/commercial sectors and outdoor areas. Local areas were assumed to represent a portion of a subcatchment with different sizes and user demand. The local areas also specified the parameters for rainfall/runoff. Local areas are embedded into subcatchments.

3.6.3 Model calibration and validation

The calibration and validation of water supply was compared to the delivered potable water flow data from the water utility for two subsequent years with those obtained through the simulation. One-year data was set up for calibration and subsequently for validation. This trial and error sequence uses various water parameters such as the monthly coefficients of water demand, the seasonal temperature contribution to water demand, imperviousness and rainfall-runoff coefficients.

Outcomes of model calibration and validation were evaluated using three indicators widely applied in previous studies: the Nash-Sutcliffe efficiency (*NSE*), percent bias (*PBIAS*), and the ratio of the root mean square error (*RMSE*) to the standard deviation of measured data (*RSR*) (Golmohammadi *et al.*, 2014; Moriasi *et al.*, 2007). The *NSE* describes the relative magnitude of the residual variance compared to the measured data variance (Eq. 23). It indicates how well the plot of observed versus modelled data fits the 1:1 line. *NSE* ranges from (-∞, 1], with 1 being a perfect fit, the negative values observed are better than the predicted ones (Krause *et al.*, 2005). In this study, a satisfactory range of [0.5, 1] was adopted for *NSE* values. The *PBIAS* measures the average tendency of the modelled data to be larger or smaller than their observed counterparts (Eq. 24). The optimal value of the *PBIAS* is 0, whereas positive values indicate overestimation bias, and negative values indicate underestimation bias. *PBIAS* is satisfactory for ≤ ±10% Moriasi *et al*. (2007b). *RSR* measures what is considered a low *RMSE* based on the standard deviation observations (sd_{obs}) (Eq. 25). *RSR* varies from the optimal value of 0 to a large positive value. A satisfactory range of [0, 0.7] was adopted for *RSR* in this study.

$$NSE = 1 - \left[\frac{\sum \left(Q_{obs,t} - Q_{mod,t} \right)^2}{\sum \left(Q_{obs,t} - \overline{Q}_{obs} \right)^2} \right] \quad \textbf{Eq. 23}$$

$$PBIAS = \left[\frac{\sum \left(Q_{obs,t} - Q_{mod,t} \right) \times 100}{\sum Q_{obs,t}} \right] \quad \textbf{Eq. 24}$$

$$RSR = \frac{RMSE}{sd_{obs}} = \frac{\sqrt{\sum \left(Q_{obs,t} - Q_{mod,t} \right)^2}}{\sqrt{\sum \left(Q_{obs,t} - \overline{Q}_{obs} \right)^2}} \quad \textbf{Eq. 25}$$

where $\overline{Q}_{obs}$ is the average value of the observed data, m^3/s.

3.7 Configuration of water reuse strategies

The strategies of water reuse are configured through different arrangements of the various elements within the urban water system. These include the wastewater inflow sources, the end use of treated wastewater, additional sources (energy and chemicals) and, in the decentralised case, a treatment system and storage tank. It also requires the definition of temporal aspects, such as the planning horizon and the time to start the operation (Behzadian *et al.*, 2014).

3.7.1 Layout and wastewater source

Three types of reuse strategies were tested in the model: (1) centralised reuse using reclaimed water, (2) decentralised using treated domestic wastewater and (3) decentralised using greywater. All strategies are considered to have been implemented in the current system of the case study, represented as BAU.

Centralised reuse is the type of reuse strategy modelled at city scale. It uses reclaimed water from the centralised WWTW, which was originally sourced from municipal wastewater. This means that all effluents within the city, such as homes, business, industry, public services, and runoff water will be inputs. This wastewater is collected in the sewer, treated in the central WWTW, and returned to users. The remaining treated effluent is discharged into the receiving water bodies, as well as the untreated overflows produced in the sewer and WWTW. Figure 3.4 presents the general layout of centralised reuse, indicating all water sources. It is understood that irrigation demand will not produce an effluent and that potable water is a simplification of the water supply subsystem described previously (e.g. water sources are not presented). Water for reuse has different uses either at city scale in irrigation or industries or at local scale (e.g. toilet flushing), although the latter is less common.

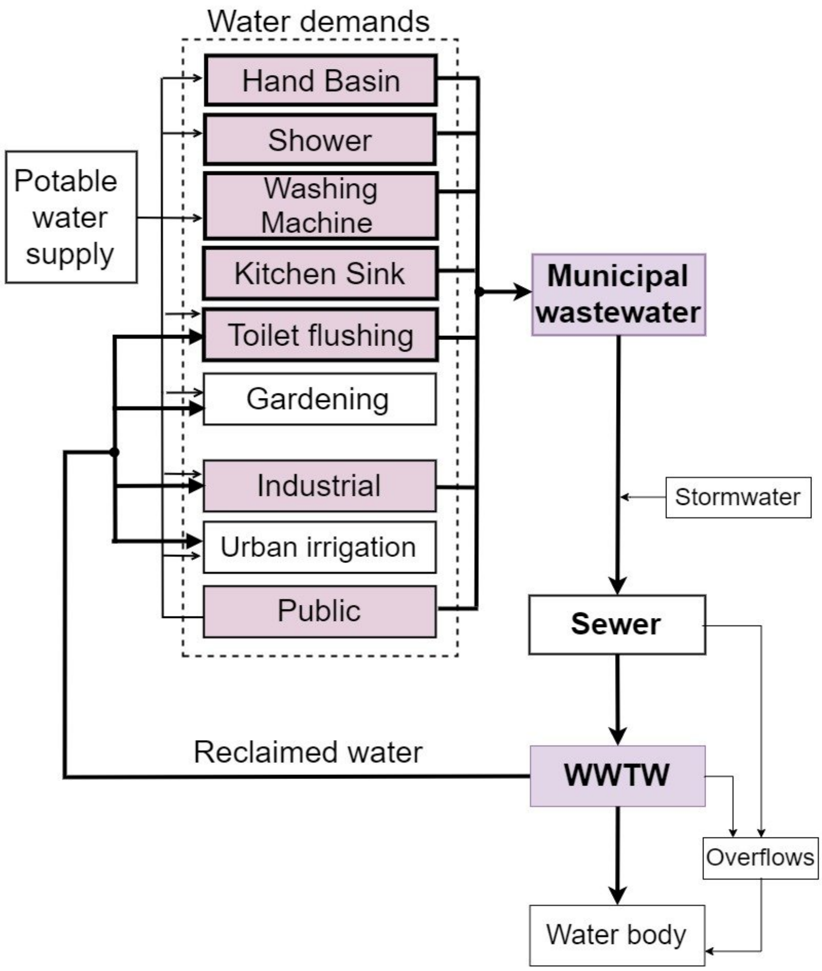


Figure 3.4 Layout of the centralised domestic wastewater strategies in the UWS. Purple cells indicate the sources of municipal wastewater, all discharged into the sewer.

Decentralised reuse can be envisaged as a total decentralised unit when a self-contained unit supplies water at the same point of origin, for example in a household or building. It can also be semi-centralised with supplies to one or various local areas near the point of origin. Another categorisation depends on the wastewater type; previous studies have focused on different mixes of greywater, but using domestic wastewater is possible (see Figure 3.5). Basically all household effluents are collected on-site and are separated from stormwater, industrial, and public effluents. Such domestic wastewater is treated in a DEWAT system and then sent for reuse. Note that not all elements are defined for the water supply subsystem.

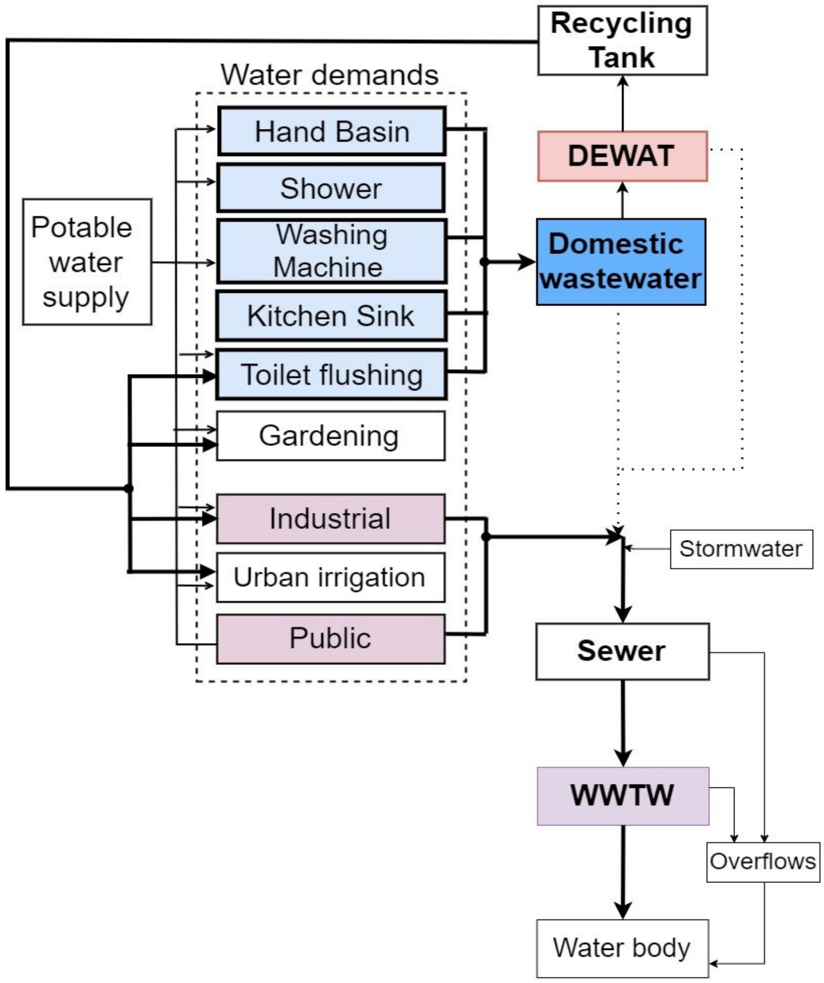


Figure 3.5 Layout of decentralised domestic wastewater strategies in the UWS. Blue cells indicate the inputs to the domestic wastewater, purple effluents are assumed to be discharged into the sewer.

Figure 3.6 presents a decentralised greywater strategy. The three effluents forming GW are the hand basin, shower, and washing machine. GW enters a DEWAT system and after treatment is stored in the recycling tank. It is assumed that sludge and overflows from the DEWAT are sent into the central WWTW. In addition, two effluents (kitchen, toilet) are assumed to be transported separately as black water and mixed up with the remaining effluents from the city (industrial and public demand). All of these effluents are combined with stormwater in the sewer and treated in the central WWTW.

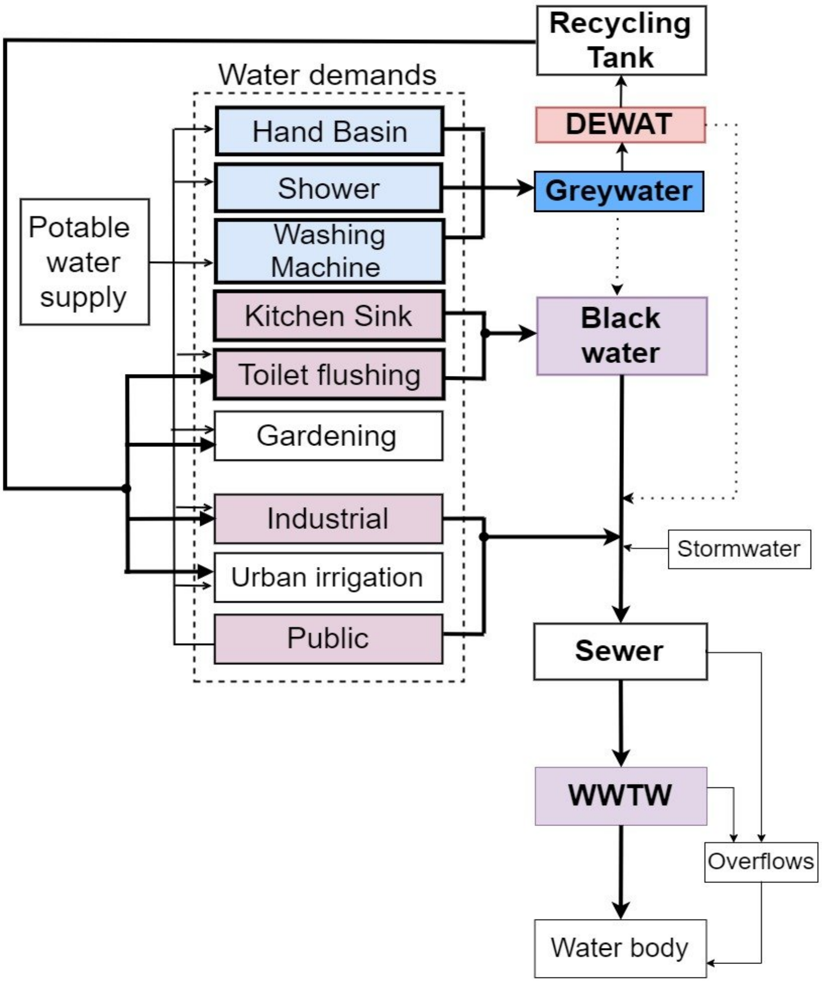


Figure 3.6 Layout of decentralised greywater strategies in the UWS. Cells coloured with blue are inputs to the greywater, purple effluents are assumed to be discharged into the sewer as municipal wastewater.

In summary, the wastewater composition for each strategy is as follows:

$$\mathbf{DG = HB + Sh + WM} \qquad \textbf{Eq. 26}$$

$$\mathbf{DW = DG + WM + To} \qquad \textbf{Eq. 27}$$

$$\mathbf{CM = DW + Ind + Pu + Sw} \qquad \textbf{Eq. 28}$$

where *DG, DW,* and *CM* are the wastewater inflows, greywater flow, domestic wastewater, and centralised municipal wastewater, respectively; *HD* is hand basin effluent, *Ki* is kitchen effluent, *WM* is washing machine effluent *Sh* is shower effluent; *To* is toilet effluent, *Ind* is wastewater from industry, *Pu* is wastewater from public services, and *Sw* is the stormwater flow, all in m^3/d.

3.7.2 Decentralised treatment and storage

All decentralised reuse uses a Decentralised Wastewater treatment facility (DEWAT). It was assumed that decentralised strategies use MBR strategies. Their selection was based on a desired quality effluent concentration of 30 mgBOD/L and 30 mgTSS/L. The treatment processes assume a screening and degritting, followed by disinfection using NaOCl. All sludge and overflows from a DEWAT system are assumed to be transported to the central WWTW. The DEWAT effluent is required to be stored in a recycling tank. The tank's function is to store and distribute the treated water to local areas. It is necessary to define the volume. Five tank capacities: 0.5, 1, 1.5, 2, 2.5m^3 were tested prior to running the simulations in order to determine at which point the equilibrium will be reached.

3.7.3 Allocation and energy for distribution

It is assumed that only a central retrofitting pipeline retrofits the water to six allocation sites. Transportation distances and level differences were estimated based on the digital elevation model (ArcGis 10.2) and land use maps in the case study. The energy required and the pipeline head loss based on the Hazen-Williams equation assume that recycled water has a continuous flow with a velocity of 1m/s, pump efficiency of 0.80 and uses PEAD 40 material.

3.8 Analysis of results

3.8.1 Comparative and contribution analysis

The total value of KPI serves to analyse the performance of strategies in comparison to BAU. This analysis aims to compare the difference between the seven key performance indicators obtained through reuse strategies and BAU. This will estimate the effect of the implementation scale and reuse adoption. Additionally, the KPI are analysed through their decomposition within the urban water subsystems in order to highlight the impact hotspots. A further contribution analysis aims to identify the main element influencing each KPI.

This analysis is made through comparisons of the percentage of each element over the estimated total.

3.8.2 Sensitivity analysis

A sensitivity analysis explores the model behaviour and studies how uncertainty in the model output is related to uncertainty in its input (Saltelli, 1995 in Saltelli, 2010). Two typical sensitivity analyses exist: local and global. The former considers the output variability around a specific value, while the latter studies variations within the entire space of variability (Pianosi *et al.*, 2016). A local sensitivity analysis is obtained by a one at a time method, varying one parameter while keeping other parameters constant (Saltelli *et al.*, 2010). This method is a simple measure with a straightforward interpretation, although this does not analyse the interactions among parameters.

On the other hand, global sensitivity examines the change response by varying all parameters in two forms, one-at-a-time or all-at-a-time. Although the term "local sensitivity" has been also used interchangeable in referring one-at-a-time studies, the difference is in the study of an specific and not the entire possible space. Global analyses are sampling-based techniques and so in order to create a random number of parameter samples, the distribution probability function (i.e. normal, lognormal, uniform, etc.) must be defined. Among a broad range of sampling techniques (Gan *et al.*, 2014), Monte Carlo and Latin hypercube are two typical random sampling techniques. Monte Carlo algorithms randomly sample the parameters independently from the respective probability density functions. Latin hypercube is a stratified sampling without replacement technique, where the parameters are randomly distributed into N equal probable intervals and then sampled (Marino *et al.*, 2008). A sampling point exists only once when projected to any single dimension and requires a sample size (N) of at least M+1, where M is the number of parameters varied (Gan *et al.*, 2014).

The sensitivity contribution of the parameters to the model output can be quantified by various measures based on the linear and monotonicity of the model (Pianosi *et al.*, 2016; Saltelli *et al.*, 2010; Marino *et al.*, 2008; Schumacher *et al.*, 2016). Sensitivity index is the simplest to use. Regression and correlation-based analysis measures the correlation between the input parameter and output value (i.e. Pearson, Partial correlation, standard regression) in linear models and their rank-transformed values (e.g. Spearman rank correlation, partial rank correlation) in a non-linear monotonic model. In the case of non-linear and non-monotic models, variance decomposition methods are used such as Sobol and FAST; however, these are computationally expensive.

Environmental pollutants can be very uncertain and thus, a global approach was chosen to identify the highest and non-relevant parameters. Sensitivity analysis is applied in the form of one-at-a-time to 30 pollutants from all household appliances (toilet, kitchen, shower, etc.) and industrial inputs within a range concentration from minimum to maximum values, as specified in Table 3.7. Sensitivity analysis were tested for global warming, eutrophication, and acidification potential impacts. The sampling technique chosen was Latin hypercube (LH) because it can achieve the same accuracy with a smaller number of samples than Monte Carlo sampling. It was assumed that the pollutants follow a uniform distribution as per similar studies (Mo, 2012).

A one-at-a time approach was used, creating 50 runs samples within the range presented in Table 3.7 for GWP, eutrophication and acidification evaluated in BAU. The sensitivity index (SI) was calculated through the relative difference between the covariance of outputs (y) and inputs (x) following Eq. 29.

$$\boldsymbol{SI} = \frac{\boldsymbol{CV_y}}{\boldsymbol{CVx}} \qquad \textbf{Eq. 29}$$

The coefficient of variation, CV_y, is defined by Eqs. 30-32.

$$\boldsymbol{CV_y} = \frac{\boldsymbol{100}}{\sqrt{\boldsymbol{n_0 - 1}}} \times \frac{\boldsymbol{S_y}}{\boldsymbol{\bar{y}}} \qquad \textbf{Eq. 30}$$

$$\boldsymbol{S_y} = \sqrt{\frac{\boldsymbol{1}}{\boldsymbol{n_0 - 1}} - \sum_{\boldsymbol{i=1}}^{\boldsymbol{n_0}} (\boldsymbol{y_i} - \boldsymbol{\bar{y}})^{\boldsymbol{2}}} \qquad \textbf{Eq. 31}$$

$$\bar{y} = \frac{1}{n_0}\sum_{i=1}^{n_0} y_i \qquad \textbf{Eq. 32}$$

where S_y is the standard deviation, $\bar{y}$ is the mean value of y and n_0 is the number of data points. CV_x, S_x and $\bar{x}$ are calculated in an analogous way for parameters x.

Table 3.8. Minimum and maximum pollutant concentration from household fittings selected for sensitivity analysis

No.	Parameter	Min –Max (mg/L)	Mean
1	BOD_{HD}	0.0 – 597.0	313.2
2	BOD_{Ki}	293.0 – 1,460.0	792.7
3	BOD_{WM}	48.0 – 1,363.0	451.9
4	BOD_{Sh}	50.0 – 385.0	202.6
5	BOD_{To}	300.0 – 1,245.0	772.5
6	$BODI_{nd}$	200 – 1,000.0	500.0
7	COD_{HD}	208.1 – 1,489.0	611.8
8	COD_{Ki}	26.0 – 2,244.0	1136.5
9	COD_{WM}	231.0 – 2,950.0	1202.2
10	COD_{Sh}	98.0 – 654.0	380.8
11	COD_{To}	900.0 – 5,160.0	2,235.0
12	$CODI_{nd}$	375 –1,500.0	750.0
13	TP_{HD}	1.3 – 26.0	8.3
14	TP_{Ki}	2.7 – 74.0	24.5
15	TP_{WM}	0 – 171.0	43.7
16	TP_{Sh}	0 – 49.0	11.2
17	TP_{To}	15.0 – 86.0	36.6
18	TP_{Ind}	20.0 –80	40.0
19	TN_{HD}	2.5 – 105.0	31.8
20	TN_{Ki}	5.5 – 74.0	35.5
21	TN_{WM}	1.1 – 40.3	12.9
22	TN_{sh}	4.0 – 50.0	14.1
23	TN_{To}	100.0 – 492.9	235.8
24	TN_{Ind}	25.0 –100.0	50.0
25	TSS_{HD}	18.5 – 573.0	175.0
26	TSS_{Ki}	134.0 – 1,300.0	483.0
27	TSS_{WM}	32.7 – 1,852.0	419.9
28	TSS_{Sh}	7.0 – 505.0	152.9
29	TSS_{To}	450.0 – 3,740.0	1,706.7
30	TSS_{Ind}	200.0 –1,000.0	300.0

BOD: Biochemical oxygen demand; COD: chemical oxygen demand, TN: Total nitrogen; TP: total phosphorus and TSS: Total suspended solids; HB: hand basin; Ki: Kitchen; W: washing machine; Sh: shower; To: toilet; Ind: industrial

3.9 Summary

This chapter has presented the theoretical framework and addressed the approaches selected. The water metabolism approach defined the UWS and limited the spatial boundaries to a set of five subsystems: water supply, water demand, sewerage, wastewater treatment, and water reuse. The WE nexus approach is built on the premise of how interconnected elements affect one another. The pollutants and their implications in the environment are the key aspects in the proposed framework. The indicators selected for the analysis were water supply deficit, potable water, total energy, renewable energy, global warming potential, eutrophication potential, and acidification potential.

The metabolism modelling uses the WaterMet2 tool, which is a conceptual and time-step mass balance model that tracks down the main flows and fluxes in the UWS. The functional unit defined in this study is the water extracted, treated, delivered, and reused per year of operation (m^3/y), similar to other studies (Opher and Friedler, 2016a). This study normalised the results per water demand. The scope of the analysis is the operation stage of the UWS, as the construction, maintenance, and demolition phases have a minor influence on the environmental impact. Different primary and secondary data are required, such as the general physical characteristics of the site under study, the water, energy, and other operational features within the UWS, and the environmental data to model the impact. To set up the conceptual model it is necessary to make some assumptions about the typology of the simplified UWS into the elements predefined by the modelling tool. It is also necessary to specify the characteristics of each component, determine the constant values, run, and simulate the model. The model is then calibrated and validated. Configuring the strategies requires the definition of various elements including wastewater inflow sources, the end use of treated wastewater, additional sources (energy and chemicals) and, in the decentralised context, a treatment system and storage tank. Three types of reuse strategies can be tested by the model. The results are analysed using a comparative of all strategies against BAU and a contribution analysis of each element to the KPI.

Chapter 4. Case study

Chapter 4 Case study and model development

4.1 Introduction

The cities of San Francisco del Rincon and Purisima del Rincon Guanajuato State, Mexico were chosen as a case study. Selecting this case study was a difficult task in light of the data constraints. Previous attempts to collaborate with other water utilities failed because they lacked or restricted access to fundamental potable water data input. Another problem encountered was the different approach of the water utilities for the water reuse (i.e. industrial rather than urban) which do not match the research gap and objective. In light of these problems two choices were made: a) to use a modelling tool with minimal historical potable water input required (see 3.2.3 WaterMet2 tool) and b) to gain access to a smaller case study through professional connections for previous research project collaborations. These were rational and adequate choices because the case study complies with the previously established criteria for selection.

a) **Availability of data**. The UWS has enough information on the main data of historic water withdrawals, energy consumption, chemicals used as well as other descriptions related to the UWS. The access to other databases and information, for example, climatic variables, census data and land uses were accessible in Mexico through public institutions. The main limitation was on information about sewage, more specifically on materials. Nonetheless, securing the access to drinking water and wastewater data was fundamental for developing the model. This is because preliminary screening and other authors (Loubet *et al.*, 2016) found that drinking water production can be treated as a confidential source and would not be able to apply the framework.
b) **Collaboration agreements.** These cities are located adjointly to each other and because of their proximity the integrated urban water system sprawls into the two cities. Then, UWS is operated by three utilities: SAPAF for the potable and sewerage operation in San Francisco del Rincon, SAPAP in Purisima, and SITRATA to operate the wastewater treatment and reuse in both cities. These water utilities use water reuse

and promptly established a collaboration agreement to support this thesis.

c) **Interest in urban water reuse**: In the case study, the mainstream "Turbio River" is heavily polluted from untreated industrial wastewater effluents. Thus, surface water is unsuitable for potable consumption. Instead, the cities rely on groundwater for potable water supply. The domestic and industrial water demands are continuously increasing due to population growth and industrialisation in the area, mainly from automobile assembling parts and shoe manufacturing. As a result, the water supply decreased by 20% (204 to 178 L/p/d) during 2010-1014 (CEAG, 2014). The wastewater treatment work in San Francisco and Purisima del Rincon, namely "San Jeronimo", produce an effluent suitable for urban non-potable reuse. Water reuse is part of an integrated water management plan in the metropolitan area to preserve the aquifer and improve the river water quality in the basin.

The above characteristics provide a rich and feasible context in which to examine the WEP nexus. In fact, it was found that part of the water utility belongs to a consortium of water and wastewater companies for climate mitigation (WaCCliM). This is an international collaboration on water-energy-carbon nexus aiming to reduce emissions by improving operation. It only focuses on San Francisco city and does not have a modelling analysis (WaCCLim, 2017).

4.2 Water situation overview

Mexico is one of the most populated countries in the world (107 million inhabitants, INEGI, 2010). The urban population is 86.3 million inhabitants, almost four times more than in 1950, whereas 26.1 million inhabitants live in rural areas (CONAGUA, 2008). This population growth and concentration in urban areas in the last 60 years have resulted in a constant pressure over water resources. The water availability was 4,4160 m^3/inhab/y in 2006 (CONAGUA, 2008). This is far from being categorised as water scarce (1,000 m^3 per capita) or extremely scarce (500 m^3 per capita per year (Qadir *et al.*, 2007). In reality, water scarcity is related to the uneven spatial and temporal distribution of water at the national level. The Southern part of the territory is considered humid and rainy (2000 mm/y), while the centre and northern are arid or semi-arid (600 mm/y). Rainy season lasts four months during summer (June-September) while the dry season occurs during the rest of the year. The arid and semi-arid region is in the central and northern part covering an extension of 1.3 x 10^6 km^2 or 66% of the territory and supporting 75% of the total population and 85% of the economy.

Guanajuato state is located in the centre of Mexico, a region facing rapid industrialisation. Like many municipalities in the world, the municipalities of Guanajuato face the challenge to provide a water supply to their growing population, in competition with other economic sectors, relying on finite supplies. Unlike other parts of Mexico, Guanajuato state shows one of the lowest potable water consumption in comparison with the national average. The urban potable water demand in the capital, Leon city (116 L/c.d) is one of the lowest at national level. For example, Tijuana city consumes 176 L/c.d and Mexico City 220 L/c.d.

4.3 Study area

The cities of San Francisco del Rincon and Purisima del Rincon are located in the central part of Mexico adjacently to each other (Figure 4.1). In addition, the area is a semi-arid region with limited water resources. The annual average temperature is 18˚C, but it fluctuates between -2 ˚C in December up to 33 ˚C in April and May. The annual average precipitation is 600 mm, occurring mainly from June to September (wet season). The dry season (April and May) is critical for water supply as it naturally lacks rainfall and the stakeholders experience the highest annual temperature (SMN, 2017).

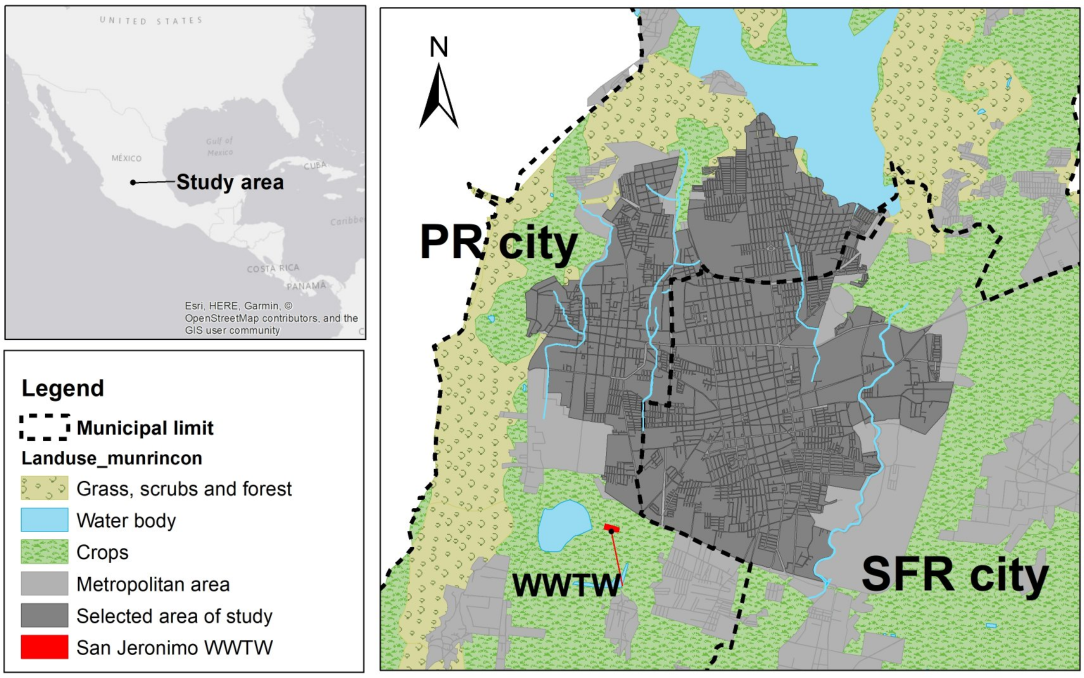


Figure 4.1 Location of San Francisco and Purisima del Rincon cities. Source: Landa-Cansigno *et al*. (2020)

4.4 Urban water system description

4.4.1 Water resources and water supply

In Mexico, around 76% of water intakes supply water for agriculture (66.05 x10^9 m^3/y) and 14.5 % (12.58 x10^9 m^3/y) is used for urban uses, among which 7.36 x10^9 m^3/y are supplied from groundwater and 5.22 x10^9 m^3/y (CONAGUA, 2017). At state level, the water supply is sourced mainly by groundwater, through 688 wells. Table 4.1 shows the volume of water extracted per each source in the state, more than 95% of the total volume is groundwater.

Table 4.1. Total water extraction per year per source in Guanajuato State

	Volume (m^3)	Percentage (%)
Groundwater	269,888,537	96.9
Surface water	8,656,282	3.1
Total	278,544,819	100

Source: CEAG, 2014

The case study uses Turbio River groundwater to supply potable water. The water table varies from 14 up to 137 m (Static level) from the ground surface and 70-300 m dynamic levels (SAPAF and Consultores, 2016). There are 22 boreholes, each one connected to a pump which uses on average 0.42 kWh/m^3. The groundwater quality is considered optimal for human consumption, and hence the only drinking water treatment is chlorination. The doses of residual chlorine (Cl_2) have to reach a concentration of 0.2-1.5 mg/L as stated in the Mexican guideline for potable water consumption NOM-SSA-127-1994 (Secretaria de Salud, 1994). The extracted water flow in SFR is 4,449,525 m^3/y while in PR is 2,377,290 m^3/y (CEAG, 2017). All withdrawn water is stored in 23 elevated tanks and then distributed by gravity to the consumers. A major problem in the distribution network is the water lost through leakages, 40% in SFR and 53% in PR of the total water withdrawn (CEAG, 2014).

4.4.2 Composition of urban water demands

There are four main demands: domestic, industrial and commercial, irrigation and public services (e.g. schools, hospitals). Domestic water demand is the average volume of water consumed per capita in household uses, while domestic water supply considers the losses in the system. On average, each inhabitant receives a daily supply of 239 litres and a water demand of 161 L (CEAG, 2014). The total population is 114,651 inhabitants, distributed as follows: 71,139 inhabitants (62%) in San Francisco and 43,512 inhabitants (38%) in Purisima. Nearly 3% of households lack potable water, electricity and sewer services (INEGI, 2010). Hence, the population served by the UWS is 111,600. Industrial and commercial demands vary in composition. The main industries are manufacturers of automobile assembling parts or shoe components. Commerces are hotels, restaurants, shopping malls, and also small stalls (e.g. hairdressers, small restaurants, clothes selling; DENUE 2015). Urban irrigation demand is periodically performed to maintain the vegetation in the green central reservations, open spaces in parks and sports facilities. There is no urban irrigation during the rainy season (July-early September) or when there are severe droughts. Public demand is the water consumed in hospitals, health centres, schools and firefights. This demand can decrease during holidays, as students do not attend school in December, summer and Easter breaks.

The demands in the area were calculated multiplying the annual water supply per the proportion of water usage. Guanajuato's State Water Commission (CEAG, 2014) reports the Guanajuato State water (domestic, industrial-commercial and public-irrigation) at the municipal level. It was assumed that a subcatchment has the same water usage patterns as the municipality. The irrigation demand (*WD_i*) was calculated as:

$$\boldsymbol{WD_i = A_i \times I_f \times a \times 0.001} \ \left(\boldsymbol{m^3/L}\right) \qquad \textbf{Eq. 33}$$

where A_i is the area of irrigation (m^2), I_f is the water per square metre in the area (5 L), *a* is a correction factor assuming irrigation is undertaken once every three days (0.5). The public water demand was calculated as the difference between the obtained flow of public-irrigation minus WDi. Indoor water demand for all local areas is assumed to be 32% for toilet flushing, 22% for showers, 16% for washing machines, 15% for kitchen, 9% for hand basin and 6% for gardening according to the reference values of Parker and Wilby (2013). There is no data available for household water consumption in the case study.

4.4.3 Sewerage

There is a combined (storm and wastewater) sewer in each of the cities. The network capacities are 73,440 m^3/d in total, 41,861 m^3/d SFR: and 31,579 m^3/d in PR. There is no storage nor CSO structure and there is no available data on the total length or lifespan inventory of the sewer network. In SFR it is estimated a total of 83.9% of sewer coverage (SAPAF and Consultores, 2016).

4.4.4 Wastewater treatment

The WWTW *San Jeronimo* (sp.), built in 2013, treats the wastewater effluents produced in both San Francisco and Purisima cities. Design flows are as follows, maximum flow during stormwater is 500 L/s (43,200 m^3/d), the overflows are considered above 350 L/s (30,240 m^3/d) and average operational flow is 250 L/s (21,600 m^3/d). The pre-treatment units consist of a coarse screen (12 mm), a fine screen (6mm), a degriter using a vortex turbine, and a pumping station. The pumps supply wastewater to two parallel circular primary settling tanks of 125 L/s (10,800 m^3/d) each one has removal efficiencies of 50-60% TSS and 30-40% BOD. The secondary treatment uses conventional activated sludge (CAS) deployed as the most popular technology in Mexico (CONAGUA, 2017). Then the water goes to two circular secondary settling tanks. These two units can be observed in Figure 4.2. Disinfection system consists of UV-chlorination but in practice only chlorination is used.

Figure 4.2. Secondary treatment in San Jeronimo Wastewater treatment work, (left) primary settling tanks and (right) activated sludge bioreactor

The WWTW produces an effluent <30 mg/L BOD according to the non-potable water reuse quality stated in the Mexican standard NOM-003-SEMARNAT-1997 (See Table 4.2). In addition, in the plant foams and sludge (primary and secondary) go to the sludge thickeners (gravity) before the reactor. An anaerobic digester (high rates, SRT 20 days, 35°C, 2,500 m^3) stabilises the sludge, destroying 50% SSV. The use of iron chloride ($FeCl_3$) reduces sulphur hydrogen (H_2S) by oxidation beforehand the anaerobiosis process. The sludge is then passed through a press band filter and produces a dry sludge of 3,500-4,500 TSS/d. Sludge is then deposited into agricultural fields nearby as a soil conditioner, except for sludge from pre-treatment which is deposited in landfills. In this digester, biogas is produced at rates of 47.5 m^3/h (GIZ, 2016) or 0.06 m^3_{biogas} per cubic metre of wastewater inflow. A pressure container stores the entire biogas produced, but only 45% of the total volume is used in the electricity generator. Such a generator produces 0.3 kWh/m^3

supplies 40% of the electric self-operation demand (SITRATA, 2015). Excess of biogas is burned before being released into the atmosphere. Figure 4.3 shows both the anaerobic reactor and the biogas storage chamber.

Table 4.2 Average concentrations of key pollutants in inflow and outflows, and respective percentage removal for 2017 and 2018

Parameter	Inflow (mg/L)	Outflow (mg/L)	% removal
BOD	376	27	93%
COD	691	56	92%
TN	52	13	75%
TP	16	2	87%
TSS	240	8	97%

Figure 4.3. Sludge management system, (left) anaerobic reactor and (right) biogas storage chamber

4.4.5 Water reuse

In Mexico, there is a recent interest in non-potable planned urban water reuse albeit having implemented irrigation reuse over 100 years ago. The use of both treated and untreated wastewater has promoted crop production, economy and reduced water shortages in the agricultural sector (Duran-Alvarez and Jimenez-Cisneros, 2014). In contrast, urban water reuse is practiced on a relatively small scale, Mexico reuses 78,000 m^3/d at national level for lawn irrigation, public parks, recreational lakes and car washing (Jiménez, 2008).

There is a growing interest in non-potable urban water reuse driven by depletion of groundwater sources, increase in potable water supply and sanitation rates.

This case study is one of the few successful urban planned and controlled water reuse. The reclaimed water is mainly used by local water utilities in urban irrigation, and by housing developers who use water for soil stabilisation. Users need to transport the water using tankers of 20 L per journey from the WWTW to parks and construction sites.

Figure 4.4 Water reuse in the case study, left application in green lanes in San Francisco city and right irrigation of gardens in the WWTW "San Jeronimo"

The Table 4.3 specifies the total reuse flows and the distribution per user. The total reuse water flow in 2014 was 27x10^3 m^3 while in 2015 there was a slight decrease to 24 x10^3 m^3. These reuse flows reached only 1% of the treated wastewater in 2015 (SITRATA, 2015). This estimation might increase due to the future construction of a distribution network of reclaimed water to Purisima del Rincon.

Table 4.3 The total flow used reused (m^3/y)

Year	Construction industry	Sewer cleaning	Urban irrigation	Total
2014	11,447	175	15,971	27,593
2015	12,184		12,230	24,414
Grand Total	23,631	175	28,201	52,007

4.4.6 Receiving water

Untreated wastewater overflows from the sewage and the WWTW (e.g. during a storm event) are discharged into Veneros Stream and Turbio River. The treated wastewater which is not recycled is also discharged into the same river.

4.5 Conceptual model and key assumptions

4.5.1 Main data

Primary data were obtained directly by the water utilities during the period 2016-2017. These data are mainly the databases of operation, containing water flows and energy consumption. These databases were collected during the researcher's visits to the case study. Secondary sources of information were obtained by different sources of information, such as the National Water Commission (CONAGUA), Meteorological National Service and relevant literature. Main input data are listed in Table 4.4 and the main characteristics of the case study are summarised in Table 4.5.

In addition, a set of historic data on daily temperature (max, medium and minimum, in ^{o}C), and daily precipitation (mm) series of historic data from 01/01/1980 – 31/12/2016 were obtained from Guanajal station located in San Francisco del Rincon (Latitud 21.03N –Longitud 101.85W, elevation 1,767 masl). Data were sourced from the national climatic database CLICOM (CICESE, 2016) and from CONAGUA for the period of 2012-2016. Missing data from specific months were replaced by the average for that period. Monthly average data series wind speed (m/s), relative humidity (%) and vapour pressure (hPa) series from 1980-2011 were obtained from Guanajuato Observatory station (SMN, 2017). Daily average sunshine hours in San Francisco del Rincon City were calculated between the difference in sunrise and sunset times obtained in a database (Manatechs, 2016).

Table 4.4 Main input data per category topic and location in this thesis

Category	Data and source	Location
Demographic data	Total population per city XI Census 1990 , XII Census 2000, XIII Census 2010; Population counts 1995 and 2005 from the National Institute of Statistics and Geography (INEGI, 1990, 1995, 2000, 2005, 2010). Population forecast from 2015 to 2044 are 1 to 3%	Appendix 2
Pervious and impervious area	Total area 2,844 ha. SB1 1615 ha, pervious 26%, roofs 58%, roads 16%. SB2 pervious 31%, roofs 56%, roads 13%.	Appendix 3
Potable water subsystem	Water withdrawals and energy consumption (2015-2016) in San Francisco and Purisima del Rincon cities, Mexico (SAPAF and SAPAP, 2017).	Appendix 4
Wastewater inflow and effluent	Monthly wastewater inflows, electricity and biogas production during 2015-2016 (SITRATA, 2017).	Appendix 5
Emissions inventory	Inventory analysis and characterisation factors for direct and fugitive emissions (various sources)	Appendix 6

Table 4.5 Summary of the main characteristics of the case study

Domain	Characteristic	Total	SFR	PR	Reference
Location	Longitude	-	101˚51.28	101˚52.45	INEGI, 2010
	Latitude	-	21˚01.06	21˚01.50	
	Elevation (masl)	-	1,756	1,767	
Area	Area (ha)	2,844	1,615	1,229	INEGI, 2016
	Pervious area (%)	-	26	31	
	Impervious area (%)	-	58	56	
	Roof area (%)	-	16	13	
Population	Total population	114,651	71,139	43,512	INEGI, 2010
	Population served (inhabitants)	111,600	69,169	42,431	
	Households with access to potable water (number)	24,960	15,614	9,344	
	Average occupancy per household	-	4.4	4.5	
Water supply	Water boreholes (Qty.)	22	12	10	SAPAF, 2017; SAPAP, 2017)
	Annual withdrawals in 2015 (x$10^6 m^3$/y)	9.48	6.11	3.37	
	Energy (kWh/m^3)	-	0.40	0.44	
	Leakages (%)	-	40	53	CEAG, 2014
Water demand	Total water demand (×$10^6 m^3$/y)	5.45	3.84	1.61	CEAG, 2017 SAPAF, 2017 SAPAP, 2017
	Domestic demand (×$10^6 m^3$/y)	4.17	2.73	1.43	
	Public use and irrigation (×$10^6 m^3$/y)	0.25	0.21	0.04	
	Industrial demand (×$10^6 m^3$/y)	1.03	0.	0.13	
Sewer	Transportation capacity (×$10^3 m^3$/d)	73.64	41.86	31.58	SITRATA, 2017
Wastewater treatment	Plant capacity (×$10^3 m^3$/d)	21.60	-	-	
	Energy use (kWh/m^3)	0.38	-	-	
	Biogas production (m^3/h)	47.5	-	-	
	Energy production (kWh/m^3)	0.03	-	-	
Water reuse	Total flow (×10^3/y)	52.00	-	-	

The main chemicals in the system inventoried were chlorine gas, NaOCl, iron chloride, fuels used in the plant is diesel, and the electricity is provided by the grid. Main emissions related to global warming potential, eutrophication, acidification and embodied energy from substances and electricity are shown in Table 4.6.

Table 4.6 Summary of characterisation factors for climate change, eutrophication, acidification and embodied energy

	GWP ($kgCO_2/kg$)	Eutroph ($kgPO_4eq/kg$)	Acidification ($kgSO_2eq/kg$)	Emb energy (kWh/kg)
Chlorine (gas)	1.82	0.003	0.0124	6.97
NaOCl	1.14	0.0021	0.0079	4.63
Iron chloride	1.21	0.208	0.0089	4.87
Diesel	3.13[a]	0.0004	0.004	12
Electricity	0.458[a]	0.00016	0.00119	-
SSP	0.548	0.00941	0.00248	0.575
Urea	1.52	0.004	0.025	4.81

Sources: Werner *et al*. (2016) and [a]SEMARNAT (2016)

The fugitive emissions related to sludge disposal are in Table 4.7. This table presents the methane, ammonia and nitrous oxide emissions related to sludge application in farm lands and in landfills.

Table 4.7 Fugitive emissions from sludge management

	Fertiliser	Landfill
Methane	0.0143 $kgCH_4/dkg$[a] (Liu *et al.*, 2013)	0.0606 $kgCH_4/dkg$ (Liu *et al.*, 2013)
Ammonia	0.2 $KgNH_3$-N/KgN biosolids (Foley *et al.*, 2010)	0
Nitrous Oxide	0.00085 KgN_2O/dkg (Liu *et al.*, 2013)	0.001kgN_2O/dkg (Liu *et al.*, 2013)

[a] dkg: Total dry solids in sludge

4.5.2 UWS set up

The UWS was set up using the WM2 modelling tool completing the specifications for each component according to the layout defined in Figure 4.5 and data obtained in Section 4.5.1. The UWS is briefly defined in a sequential manner in this section. The model requires the definition of the spatial boundaries and main features of the UWS described in Section 3.2.2. As aforementioned, these subsystems are independent within each city except for the WWTW which is shared by both cities. For simplicity, it was assumed that each city represents one subcatchment (SB), San Francisco del Rincon city is SB1 and Purisima del Rincon city is SB2. Both cities represent the whole UWS despite being managed by different water utilities.

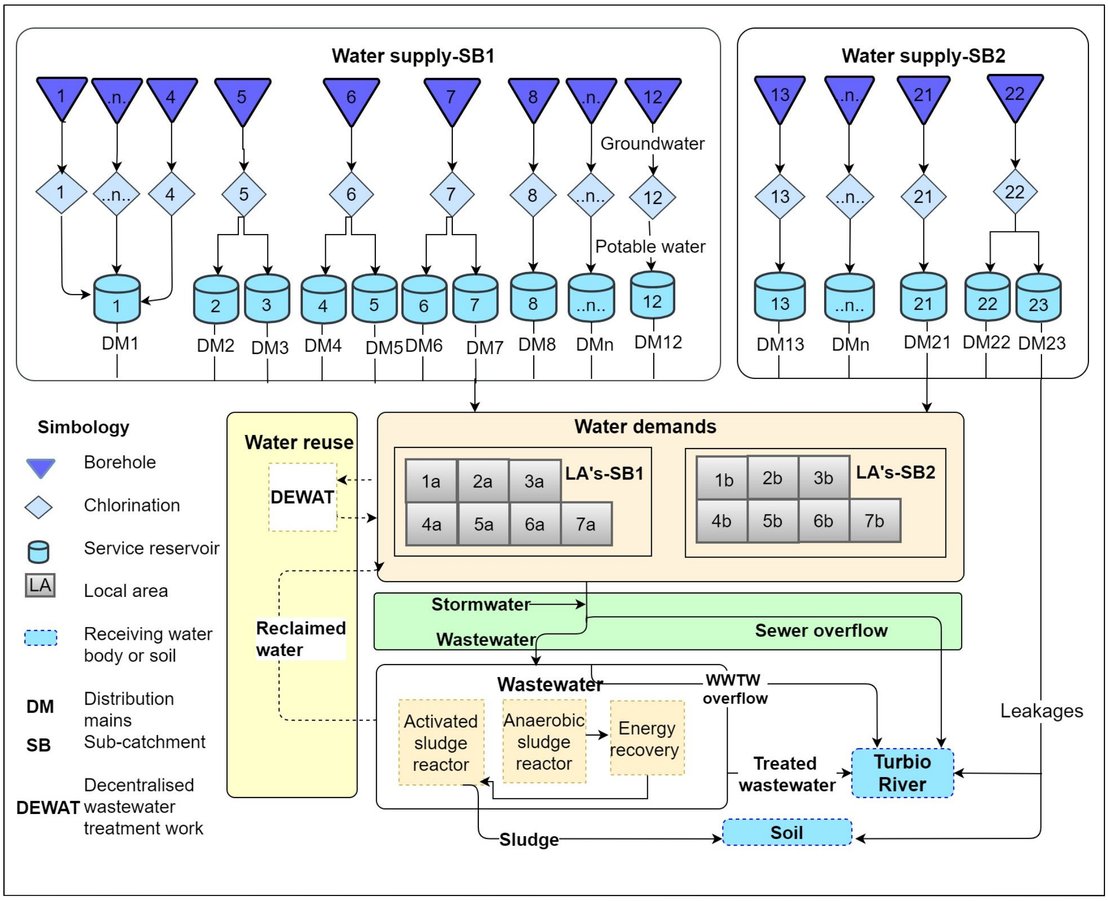


Figure 4.5 Schematic layout of the UWS of the cities of Rincon, Mexico

The UWS was defined as a set of five subsystems: Potable water, water demands, sewage, wastewater treatment and reuse according to the description in Section 3.2.2. The water supply subsystem is assumed to have 12 boreholes in SB1 and 10 boreholes in SB2. Each one has an on-site chlorination system as the only drinking water treatment. Hence, there was considered a total number of 22 water treatment works (WTW's) with no split of water flow. Water supply conduits connect the water sources with their respective WTW, as this is not the case for the case study, supply conduits were defined as dummy elements with a capacity of $1x10^6$ m^3/year. Water resources and WTW are connected through the water supply conduits, each one defined per the maximum capacity extraction of each borehole defined by the permission of extraction. WTW connects to reservoir tanks to store potable water. From the UWS of the case study the number of storage tanks is different according to the borehole. Here a percentage of split was deduced from the layout and flows.

The distribution mains connect the service reservoir to the demands. A connection of each water source per each WTW was considered with no split of water. The predefined number of storage tanks identified were 12 tanks in SB1 and 11 tanks in SB2. Although the majority of tanks are connected to each borehole, there are some exceptions where one tank receives water from four boreholes or where one borehole supplies two tanks. The volume of each borehole is defined by the split into the total capacity and the number of storage tanks. Arrangements can be seen in Appendix 7.

4.5.3 Local areas

It was proposed to divide the water demand subsystem into seven local areas with various specifications defined for each city (Figure 4.5). Table 4.8 shows the specific characteristic of size and demand users per local area.

Table 4.8 Specifications of the subcatchments (SBs) and local areas (LAs)

SB No.	No. of LAs	Area (ha)	Inhabitants	No. of households	Industrial demand (m^3/d)	Irrigation demand (m^3/d)
SB1 (SFR)	2	81	3,458	781	120	15
	2	162	6,917	1,561	240	30
	2	323	13,835	3,123	480	60
	1	485	20,750	4,684	720	90
SB2 (PR)	2	62	2,121	467	15	5
	2	123	4,242	934	30	10
	2	246	8,489	1,870	60	20
	1	368	12,727	2,803	90	30

SFR: San Francisco del Rincon city; PR: Purisima del Rincon city

The variations of water demands consider the influence of annual, monthly and daily influence. For example, the 1-3% annual population growth, and assumes equivalent industrialisation and public irrigation growth. The monthly variations to irrigation and public demands due to seasonal stakeholders' behaviour and the daily variations due to temperature influence. All of these are adjusted for calibration purposes.

The existing sewer system is a combined sewer of stormwater and wastewater with a capacity of 41,861 m^3/d in SB1 and 31,579 m^3/d in SB2. It was assumed that 80% of the potable water converts into wastewater discharge in the sewer (Comision Nacional del Agua, 2007). Stormwater inflow (runoff) is estimated from the daily time series of weather data required by the model. For calculating runoff over the entire planning horizon it was necessary to gather data from 30 years before. It was assumed to have an infiltration coefficient of 0.5 and a runoff coefficient of 0.4. Sewer has no storage structure.

The specifications of the WWTW refer to the operation and hydraulic capacities, such as wastewater flows, storage flows, treatment efficiencies and resource recovery. Each subcatchment discharges the entire wastewater flow into San Jeronimo WWTW, considering 60% of SB1 and 40% of SB2. The two receiving water bodies of both treated and untreated wastewater are Veneros stream and Turbio River.

The recovery subsystem includes the use of biogas for energy production with a capacity to collect 100% of biogas produced and a combustion efficiency of 80%. The nutrients off-settings due to the use of sludge in agriculture. This is assumed to replace urea and single super phosphate (SSP). Urea $CO(NH_2)_2$ is a N-based fertiliser with a 46% of N. Single super phosphate $Ca(H_2PO_4)_2$.

4.6 Model calibration and validation

The model calibration required a trial and error evaluation of observed vs simulated monthly water supply values in 2015, while validation uses values observed in 2016. This calibration process adjusted the water demand to 102 L per capita water consumption in SB1 and 80 litres per capita in SB2. This is aligned with information provided by SAPAF water utility: 99 L/c.d and 169 L/c.d of water consumption and supply respectively. The public water demand was adjusted to 3.7 L/c.d and 4.0 L/c.d in SB1 and SB2, respectively. This demand had a reduction during public holidays, specifically during April (-30%), July and August (-50%) and December (-30%). The irrigation demand is null during July, August and September. The calibration includes also the contribution of temperature to irrigation 30% in SB1 and 60% in SB2. Figure 4.6 presents the final calibration and validation results for the water withdrawn in 2015. The model was slightly overestimated during May to September (dry season).

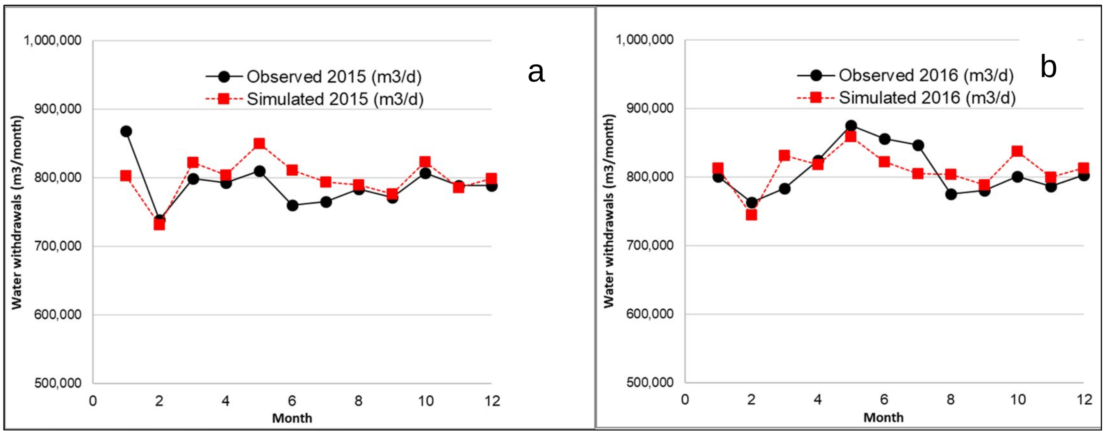


Figure 4.6 Calibration (a) and validation (b) of the UWS

The results of NSE, Pbias and Pearson coefficients are in Table 4.9. According to Moriasi *et al*. (2007b), a model is considered calibrated for flow if monthly NSE ≥ 0.65, PBIAS ≤ ±10% and RSR ≤ 0.60. If NSE is 1 it is considered that the model fits perfect, negative values the observed is better than predictor (Krause *et al.*, 2005)In general, the validation of the model agrees with the Pbias of the system.

Table 4.9 Results of statistics parameters related to the model calibration and validation

	NSE	Pbias	RSR
Calibration	0.0906	-1.2312	0.95
Validation	0.3748	-0.4373	0.79
Acceptable values	(>0.5) Range 1 to -∞	<+-10	<0.7

Regarding the water quality, a general comparison for the average load of pollutants per year was performed. These values corresponded to wastewater quality measurements as mixed inflows in the WWTW. For the case of the model, these are the disaggregated values per household fitting component.

4.7 Water reuse strategies selected

4.7.1 Experts' opinion on water reuse

Identifying possible tendencies of future adoption proportions uses a local expert assessment (see Section 3.5.4). The survey distributed to experts aimed to collect their view on current water reuse practice within the region of the case study. A total of 19 experts fully answered the questionnaire. The experts defined themselves within the areas of expertise of wastewater treatment (33%), water supply (20%), water reuse (19%), sludge management (6%), and other (14%). As practitioners (60% of respondents), they work in water utility companies located at, or near, the case study. Researchers formed 40% of the respondents and they work in universities, research centres, or cooperation agencies. All know the context of water in Guanajuato State.

Asked to identify the top three strategies to cope with climate change and water scarcity in the cities of Guanajuato, the participants indicated that repairing leakages in the potable mains water distribution was the most preferred strategy, followed by centralised reuse and household savings appliances (see Figure 4.7). These are interventions in the water supply, water reuse, and water demand subsystems. Rainwater and greywater reuse, as well as urban irrigation, were the least preferred options. In between were the options to intervene in industry or reduce irrigation water usage in public spaces. Similarly, the study of Moredia-Valek (2016) found that local experts in Mexico City prefer leakage control to water reuse as a strategy to improve the water sector. They argued that an improvement of 20% in leakages would significantly reduce the water import to the city. Such responses indicate an awareness of the leakage problem encountered all over the country, where water loss can reach up to 40% of available drinking water (CONAGUA, 2018).

At the same time, it highlights that water reuse is not yet seen as an integral strategy to cope with water pollution and climate change targets. Furthermore, it is well known that stakeholders might prefer indirect forms of water reuse and may reject direct contact (Bichai *et al.*, 2018; Friedler and Lahav, 2006). Future studies might try to highlight the barriers to stakeholder acceptance in the local context. This will be important because in the long term it is necessary to build public awareness of the benefits associated with water reuse, enforcement of policies, and public health regulations for successful implementation.

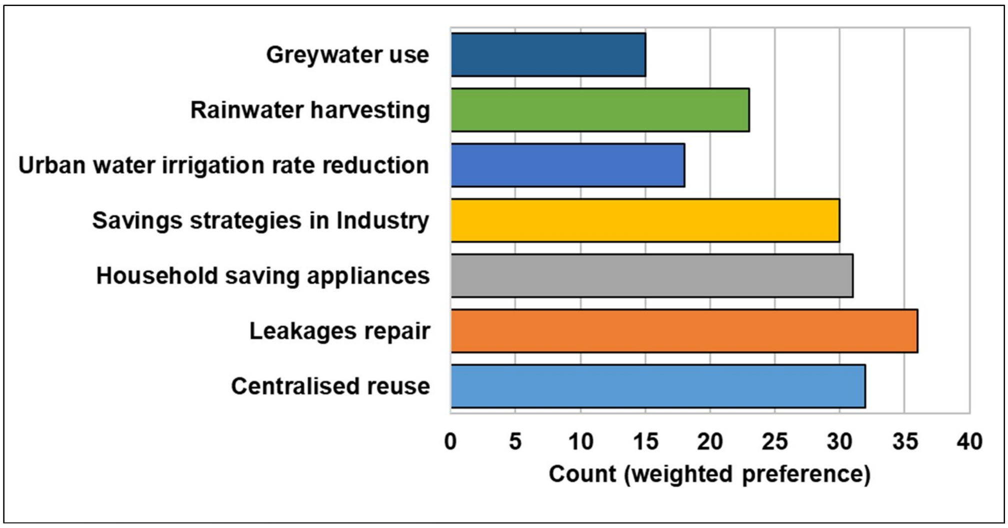


Figure 4.7 Experts' choice on water management strategies

Finally, an open-ended question asked about the possible reuse adoption perspective. Reuse adoption refers to the amount of water fulfilling a specific percentage of a demand. The question asked the respondent to identify the projected percentage of water reuse in ten and twenty years in the whole area. Responses vary (number of valid questionnaires, 16). Within ten years, centralised water reuse might increase up to 25%, while decentralised could rise up to 15%. The tendency is less evident when projecting a change within the next twenty years. Either it was in the range of 7-25% or 75-100% in centralised reuse and 5-15% or 46-75% in decentralised systems. The specific choice of 100% reuse adoption was only seen for centralised reuse and not for greywater (see Figure 4.8). Other studies have projected an urban water reuse potential of 5% (Almeida *et al.*, 2013). Such differences might be explained by the decision to use a model that examines scarcity, demographic density, and treatment rate functions in a fuzzy logic model. In this thesis, the potential for adoption of water reuse was limited to the wastewater treatment rate using the opinion of local experts. However, the decision-making process is highly complex and also requires the consideration of economic and political aspects. In the case of Mexico, the water reuse strategy is very unclear as are the statistics of water reuse used to estimate a growing rate. The National Water Commission reports a current indirect reuse of 78.8m^3/s against a direct reuse of 39.8m^3/s, from the former only 8.6m^3/s can be substituted for drinking water (CONAGUA, 2018). Without further detail it is difficult to perceive a national tendency and thus, considering only the expert views, the adoption reuse chosen estimates were 20%, 50%, and 100%, to match the full range of opinions. Other studies based on social factors have also considered optimistic scenarios of 90% adoption in households (Kandiah *et al.*, 2019). Further research is needed to project the adoption reuse in more detail in Mexico.

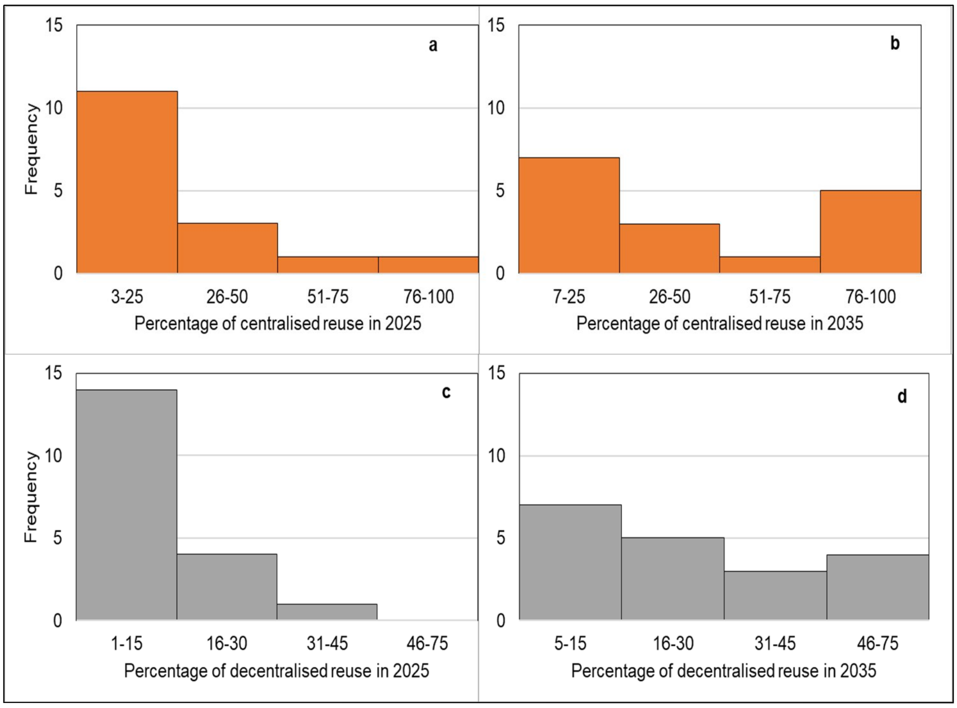


Figure 4.8 Frequency histogram of future reuse adoption in centralised reuse in a) 2015 and b) 2035, and in decentralised in c) 2025 and d) 2035. Graphs presents the adoption proportions and Y-axis the counts according to local experts

4.7.2 Strategies proposed

It was considered to compare three types of strategies: centralised (C), decentralised using domestic wastewater (DW) and decentralised using greywater (DG). Three decentralised proposed the use of greywater (DG20, DG50 and DG100) and three using domestic wastewater (DW20, DW50 and DW100). All of them are assumed to use a MBR reactor. Three others used centralised reclaimed water (C20, C50 and C100) for alternative uses of reclaimed water produced in the current WWTW. It was assumed that treated wastewater will be used in urban irrigation, toilet flushing and industrial uses. Interventions were set in two steps, one in 2024 and the other in 2034. The nine strategies chosen in this research represent a spectrum of different options of centralised and decentralised configurations, all presented in Table 4.10. The strategies analysed are thought to be in conjunction with the current installed wastewater treatment and water reuse systems, which involves the use of an activated sludge WWTW and 1% of water reclaimed in urban irrigation and construction.

Table 4.10 Summary of key features of each strategy

Strategy	Name	Treatment
BAU	Business as usual	Existing system
C20	Centralised, reclaimed water, 20% of reuse adoption	Centralised WWTW, uses reclaimed water, already installed
C50	Centralised, reclaimed water, 50% of reuse adoption	
C100	Decentralised, reclaimed water, 100% of reuse adoption	
DW20	Decentralised domestic, wastewater, 20% of reuse adoption	Decentralised, uses domestic wastewater, uses a MBR DEWAT
DW50	Decentralised domestic, wastewater, 50% of reuse adoption	
DW100	Decentralised domestic, wastewater, 100% of reuse adoption	
DG20	Decentralised domestic, greywater, 20% of reuse adoption	Decentralised, uses greywater, uses a MBR DEWAT
DG50	Decentralised domestic, greywater, 50% of reuse adoption	
DG100	Decentralised domestic, greywater, 100% of reuse adoption	

4.7.3 Treatment and storage tank

It was assumed that the DEWAT system is composed of a screening, sand filter, MBR, and chlorine disinfection. After modelling the tank capacity (Figure 4.9), it can be seen that a tank of 0.5 m^3 is enough for the demands.

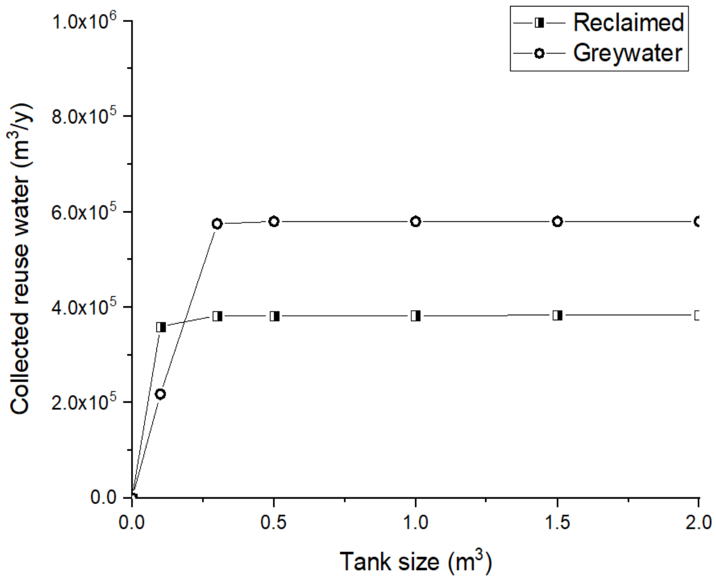


Figure 4.9 Storage greywater tank capacity

4.7.4 Energy usage

Treatment energy consumption was assumed equivalent to the disaggregated values in a WWTW of 5,000 m^3/d capacity reported in Longo *et al.* (2016). These energies are 0.047 kWh/m^3 for pre-treatment (pumping, screening and de-gritting), 0.0071 kWh/m^3 for primary settling, 0.63 for an MBR reactor. Additionally, it was assumed that the plant includes a sand filter of a consumption of 0.25 kWh/m^3 (Singh *et al.*, 2016). Thus, the total energy required for a decentralised treatment was considered to be 0.93 kWh/m^3.

In addition to energy used in treatment, it was necessary to include the energy required for the transportation of treated wastewater for water reuse strategies which was estimated based on the physical level difference and pipeline head losses between the WWTW, or DEWATs and the six local area tanks (three for each subcatchment) where water reuse is transported and used. Energy inputs per cubic meter of water reused are shown in Appendix 8.

Table 6. Energy demands in water reuse facilities in kWh per cubic meter of water reused

Strategies	Treatment*	Transportation	Total
C20	0.380	0.145	0.525
C50	0.380	0.204	0.584
C100	0.380	0.223	0.603
DW20	0.930	0.123	1.050
DW50	0.930	0.162	1.089
DW100	0.930	0.221	1.148
DG20	0.930	0.136	1.063
DG50	0.930	0.181	1.108
DG100	0.930	0.214	1.141

*Energy demands in centralised strategies corresponds to that in the WWTW

4.7.5 Summary of the chapter

The UWS of the case study is an integrated system sprawl into two adjacent cities of the cities of San Francisco and Purisima del Rincon, Mexico. The system has a serving population of 116,300 inhabitants, as well as industries, commerce and public uses. The water withdrawn is 9×10^6 m^3/y supplied from groundwater form Turbio River Aquifer. The system has an average of 40% of water losses. The WWTW, based on an activated sludge system, was designed to serve both cities and aimed to supply reclaimed water for reuse, although only 1% of the treated wastewater is used. The system recovers biogas to produce electricity and use the anaerobic digested sludge into agriculture. By reviewing the current water context in Mexico and by considering the water reuse status in Guanajuato State it was found that the case study is suitable to demonstrate the framework application. It can be highlighted that water reuse is part of the region’s water management strategy. With advances and efforts already taken in place in the area, it was identified the need to assess different types of water reuse as a portfolio of options and the willingness to collaborate in this research.

Exhaustive data gathering stage includes desk research, field visits and meetings with water utilities' personnel. Data obtained from primary and secondary sources include the daily, monthly and annual water flows and energy consumption in the years 2015-2016, type and quantity of by-product production, climate series of different variables over the last 30 years. Other data was calculated, for example the population growth rate resulted in 2-3% over the planning horizon, the update of the emissions inventory according to the chemicals, energy and fuels used in the case study.

The conceptual model was developed based on the typology of the UWS in the case. To elaborate the model, it was assumed that each city is a sub-catchment composed by seven local areas of various sizes and water demands. Then, the key components characteristics of each of the UWS were defined in WM2 from the data gathered. The model calibration required a trial and error evaluation of observed vs simulated monthly water supply values in 2015, while validation uses values observed in 2016. This calibration process adjusted the water demand to 102 L per capita water consumption in SB1 and 80 litres per capita in SB2. The calibration and validation showed a good agreement with the PBIAS indicator.

Finally, nine water reuse strategies were set up among three categories: centralised, decentralised using domestic wastewater, decentralised using greywater, depending on the wastewater inflow and scale level. Each strategy category varies the reuse adoption in 20%, 50% and 100% according to the ranges proposed by the experts. Additional electricity required in water reuse resulted in 0.2- 0.3 kWh/m^3 for water transportation from the WWTW to three distribution sites in centralised reuse, and for decentralised the energy for DEWAT.

Chapter 5. Strategies assessment and overall discussion

Chapter 5 Strategies assessment and overall discussion

5.1 Introduction

This chapter presents the results and a discussion of the WEP nexus performance in the water metabolism of nine strategies and BAU in the Rincon cities case study. The results present six assessment criteria: water supply deficit, potable water delivered, total energy, global warming potential, eutrophication potential and acidification potential. The chapter includes the results of the studied KPIs (see Appendix 9), the contribution analysis of each component to identify the key sources affecting the WEP nexus, and the comparisons of centralised and decentralised strategies. The discussion includes the implications of diversify the water reuse strategies in the case study and offers recommendations for decision makers.

5.2 KPI’s

5.2.1 Urban water supply deficit

The urban water supply deficit provides the magnitude of supply-demand gaps and when these are expected. In the case study, the deficit in BAU is equivalent to 24 x10^3 m^3/y and is noticeable at the end of the simulation period during 2040-2044. The strategies C20 and DW20 have a deficit equivalent to 3 x10^3 m^3/y, which is smaller than the value found in strategy DG20 (4.5 x10^3 m^3/y). Figure 5.1 shows the monthly simulation of water demand vs water supply, and it can be observed that implementing water reuse delays the occurrence of deficit in comparison to BAU, as this will be expected to occur in one year lapse during 2044. This indicator is useful to understand urban water deficit risks and to derive policy lessons that can limit the vulnerability of the population.

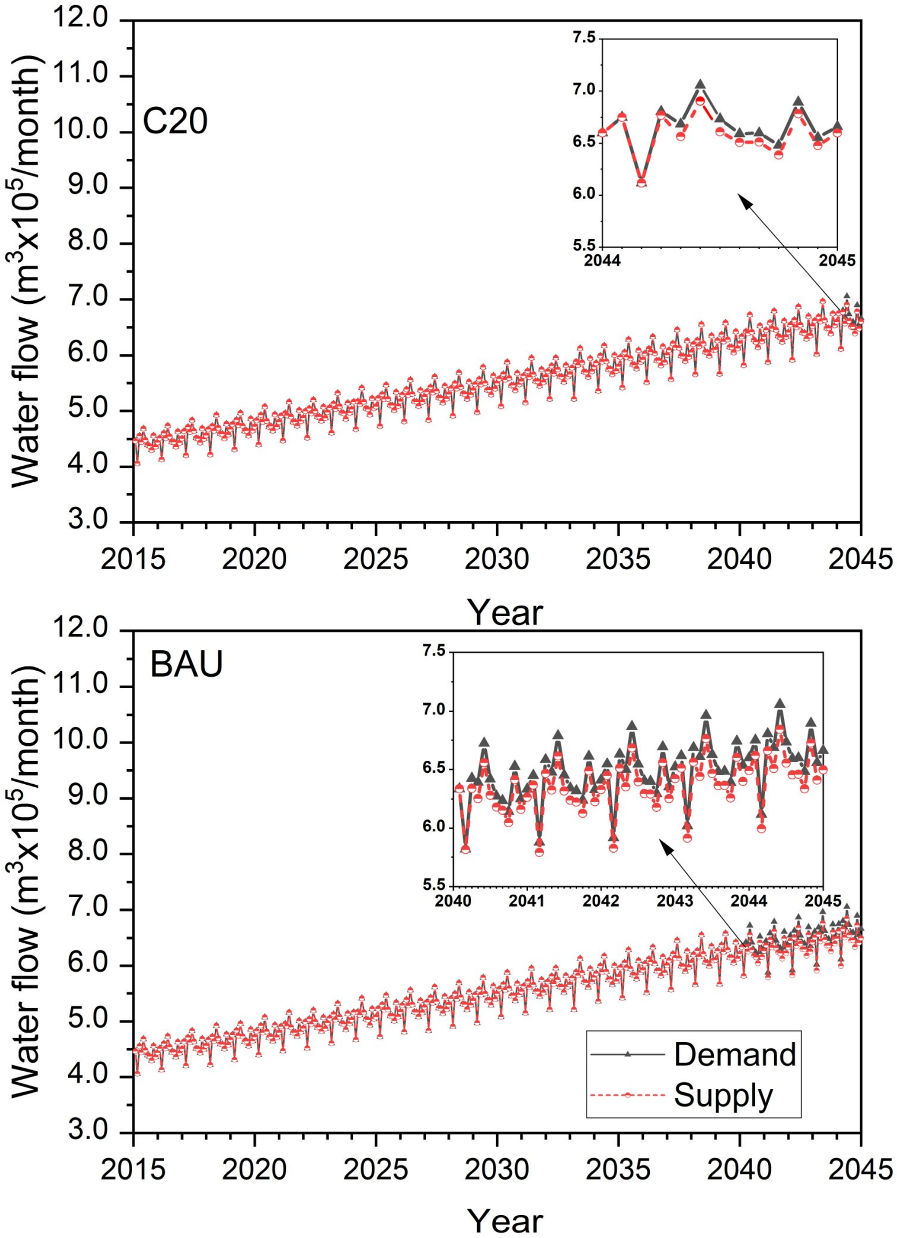
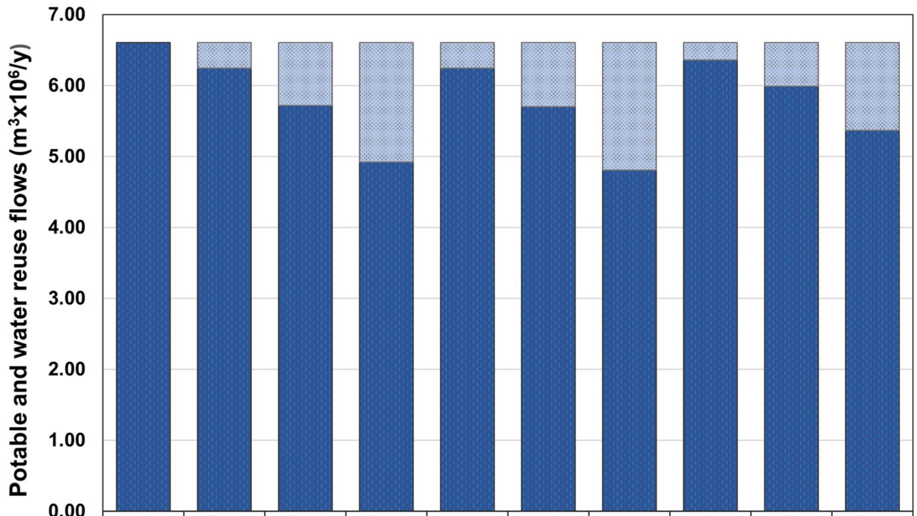


Figure 5.1 Monthly water demand and water supply in BAU (bottom) and C20 (top) along the simulation period (2015-2044).

5.2.2 Potable water savings

Potable water savings indicate the system's capacity to supply the water demand using treated wastewater instead of potable water. Potable water is referred to as drinking water from conventional groundwater sources and treated wastewater can be either reclaimed water, domestic wastewater or greywater. The average annual demand in all BAU and strategies is 6.6 x10^6 m^3/y. The water savings vary according to strategy type and reuse adoption. Both C and DW strategies seem to have relatively similar water savings (0.4

x10^6 m^3/y in 20% adoption reuse and 0.9 x10^6 m^3/y in 50% adoption reuse), although DW is slightly higher for the 100 adoption (1.8 vs 1.6 x10^6 m^3/y). These savings are in the order of 5% to 26%. DG strategies showed less potential to save water (0.3 x10^6 m^3/y in DG20 up to 1.2 x10^6 m^3/y in DG100) and DG savings are 3 to 18%. As expected, higher reuse adoption can save greater quantities of water, nearly three and five times when implemented at 50% and 100% reuse adoption, respectively. Figure 5.2 presents the annual average supply flow in the period 2015-2044 per source, the dark blue bars represent potable water and light blue the reused water.



	BAU	C20	C50	C100	DW20	DW50	DW100	DG20	DG50	DG100
Reuse	0.00	0.36	0.89	1.69	0.36	0.90	1.80	0.25	0.62	1.23
Potable	6.60	6.24	5.72	4.92	6.24	5.70	4.80	6.36	5.99	5.37

Figure 5.2 Water supply sources per strategy and BAU. Annual average values are indicated below the bars.

Figure 5.3 shows the percentage distribution per water demand for all nine assumed strategies. Demands are 75% domestic, 20% industry, 3% public and 2% irrigation, with the maximum savings seen in domestic and industrial reuse. Greywater results from hand basins, laundry and showers, representing 53% of the domestic wastewater or 39% of the total potential water supply. The potential demand of toilet flushing is 24% of the total water demand.

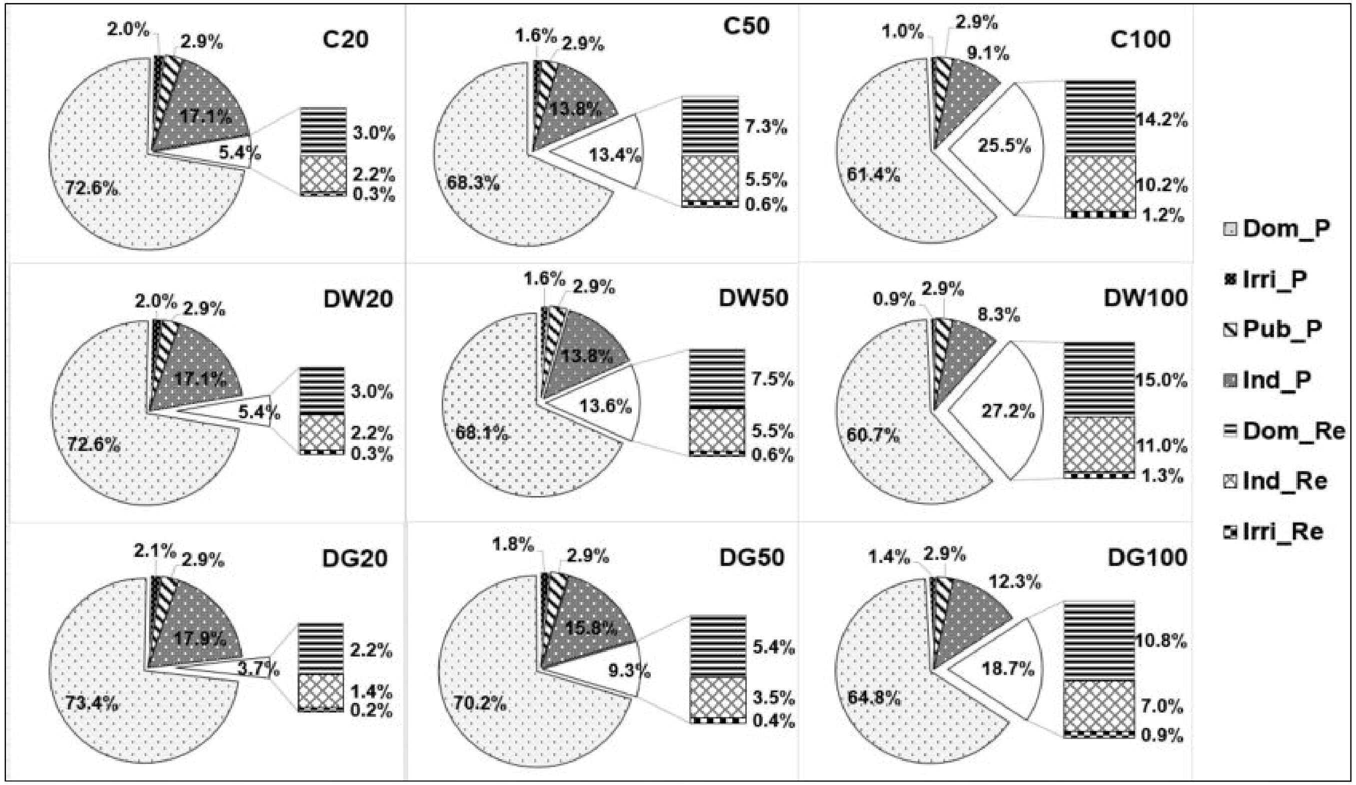


Figure 5.3 Percentage of water demand supplied by potable and reuse water sources in each of the nine strategies. Largest percentage represents potable supply for domestic use, bar in graphs represent all uses using reused water.

Dom: Domestic demand; Irri: Irrigation; Pub: Public demand; Ind: Industrial; P: Potable water source; Re: reused water

The results of water saving potential for greywater in this thesis are smaller than in other studies (see Table 5.1).

Table 5.1 Comparisons of water savings with other studies

Water saved (%)	Type of reuse	Country	Reference
3-18%	Toilet flushing, public irrigation, and industrial in 100% of demand	Mexico	This thesis
6-32%	Landscape and toilet flushing in MFZ	USA	Jeon *et al*., 2018
17-49%	Landscape and toilet flushing in SFZ		
19-31%	Toilet flushing and garden irrigation in buildings	Israel	Friedler *et al*. 2008
31%	Landscape and toilet flushing	Korea	Jang *et al.*, 2010
40%	Toilet flushing and irrigation	Israel	Duong *et al*., 2011
48%	Toilet flushing and agricultural gardens household; estimations at a neighbourhood of 800 inhabitants	India	Mandal *et al*., 2011
52%	Greywater for industry and Industrial water 10% in the city	Netherlands	Agudelo-Vera *et al*., 2012

These differences may be due to the assumptions regarding greywater supply/demand and the boundaries of the method of accounting. Unlike the study by Friedler *et al*. (2008), this thesis considers the total demand rather than focusing on just the domestic, which gives a higher perception of savings. Another key difference is the capacity to supply greywater within the function of the domestic water supply. This case study shows a low water demand per capita (110 L/c.d in SFR and 88 L/c.d in PR) than assumed in other research (161 L/c.d in Israel, Friedler *et al*., 2008; and 220 L/c.d in Atlanta, Jeong *et al*., 2018). Furthermore, the toilet flushing demand is also assumed to be higher in other accounts (e.g. 45% in Atlanta vs 32% in this case study). Another possible cause is the water supply deficit, as the model accounts the balance it is assumed that the water reused will fulfil the initial requirement and then start accounting savings. In the other quantifications, the model does not consider whether the demand varies over the period of study. These differences would result in higher potential demands than this thesis has found.

5.2.3 Total net energy

Total net energy is the balance between caused and avoided energy in both direct and indirect forms along the UWS. Figure 5.5 shows the net energy balance of BAU and the nine strategies in a bar chart; the total energy value in kWh/m^3 is above the bars and the contribution per each subsystem is within the bars in percentage value. The results indicate that from the water-energy point of view, all reuse strategies have neutral or positive effects on energy consumption. BAU consumes 1.15 kWh/m^3. The implementation of centralised strategies would use less energy in C50, with a total of 1.08 kWh/m^3 (-6%) and C100 is 1.02 kWh/m^3 (-11.5%).

In centralised strategies, increased adoption of reuse reduces the energy consumption, being the strategy with the highest proportion of uptake (C100) and the most energy efficient (Figure 5.4). This is because the energy savings in water supply (for groundwater replacement) increase with reuse adoption. Implementing centralised reuse might increase the energy needed for pumping back the reclaimed water to the city, as stated by Opher and Friedler (2016a). However, in the case study, this does not surpass the savings by reducing the energy involved in groundwater pumping. This case study found that to increase centralised reuse a distribution system is required, which seems feasible given that some pipelines were implemented in Purisima City in 2018.

Decentralised reuse types, DG and DW, have the same energy consumption as BAU. This is unexpected given the fact that implementing a DEWAT system might increase the total energy required (MBR using 1.1 kWh/m^3). This is a result of energy savings due to reductions in groundwater withdrawals (12 % in DW100), plus the savings due to the reduction in wastewater inflows (6 % in DW100), and a minimal saving due to reductions in the use of chemicals (<1%). These energy savings, as well as those by the off-setting of fertiliser due to the application of sludge reuse in agriculture, seem to be enough to compensate for the increase in DEWAT treatment flows, even in the case of DW100 in which the highest water reuse flow is yielded (1.8 x10^6 m^3/y).

However, the modelling was limited to one technology (MBR). Further studies must consider the differences in energy inputs if other technologies are used, for example, technologies based on wetlands would not use electricity and savings will be observed without changing the efficiencies. On the other hand, other technologies would increase energy usage and, consequently, decentralised treatment would consume more energy than BAU. These technologies may have different efficiencies and must be studied under the nexus approach.

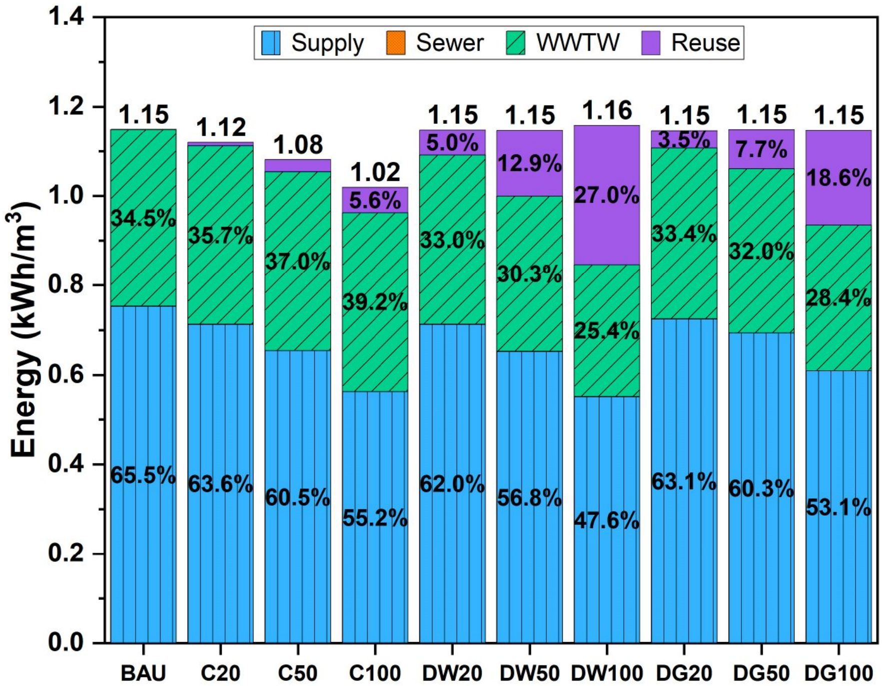


Figure 5.4 Total energy per strategy and contribution percentage per subsystem component to the energy consumption. Each bar represents the total energy per strategy, values in kWh/m^3; the relative contribution per subsystem is shown within the bars in percentage.

Energy is consumed differently along the integrated water system. The water supply subsystem is the highest energy contributor, followed by the WWTW and finally, water reuse. The water supply subsystem uses 66% of the total energy consumed in BAU. Once reuse is implemented, the energy spent on water supply decreases to 62-63% in C20, DW20, and DG20, but higher reductions were found at higher adoption rates. For example, the energy of the water supply represents 55%, 47%, and 53% of the total energy consumption in C100, DW100, and DG100, respectively. The wastewater subsystem is the second most intense energy subsystem and it was found that this type of strategy affects energy consumption in this subsystem. For example, centralised strategies increase energy use at higher reuse adoptions, from 35% in C20 to 39% in C100. In addition, the energy decreases in DW20 and DG20 from 33% to 25-27%. This may be explained by the fact that the wastewater inflow in the centralised system tends to increase as an effect of the decreasing water deficit. Decentralised strategies reduce the wastewater inflow to the already implemented central WWTW. Consequently, the energy in WWTW is reduced. However, the reduction is small in the decentralised greywater reuse, because the DEWAT treats less flow separately. The water reuse subsystem contributes the least amount of energy and increases its consumption in all strategies at higher proportions in the decentralised system.

The findings presented by this thesis are in disagreement with other reports that state that decentralised reuse is less energy intense than centralised. For example, Silva-Vieira and Ghisi (2016) found a potential saving of 48% in energy consumption when efficient fixture and grey water were implemented when compared with centralised systems. Also in Chang *et al*. (2017), decentralised WW reuse designed for non-potable domestic use has a comparable energy demand than centralised WW reuse, but less than the conventional water supply. These differences are partly due to the boundaries selected, functional unit, flow, technology selected, and all specificities are inherently local. Hence, this thesis cannot support an extrapolation of results towards other case studies.

The contribution analysis identifies the energy per specific input (i.e. electricity, chemicals, fuels, biogas and in by-products recovery). Table 5.2 presents the relative amount of energy spent in chemicals, electricity, and fuels in BAU and three strategies (all at 100% adoption reuse). The results indicate that electricity used in pumping groundwater is the principal contributor to energy in all strategies and BAU. It is generally acknowledged that electricity in the water supply is the highest contributor in urban water systems for both water abstractions and potable treatment (Moredia-Valek *et al.*, 2017; Lemos *et al.*, 2013). In this thesis, the energy for drinking water treatment and distribution are negligible. This is because groundwater systems have a better raw water quality and are only considered to require a simple treatment (chlorination). In comparing two systems in the USA, Mo (2012) found that a system based on groundwater uses 31% less energy than a similar one using freshwater due to the presence of fewer coagulants and disinfectants in the drinking water process. The use of gravity distribution pipelines also reduces the energy intake and the highest energy consumed is attributed to pumping groundwater. Local conditions play an important role as the energy used in pumping depends on the local topography, flow, and dynamic levels (Chang *et al.*, 2017; Wakeel *et al.*, 2016).

Table 5.2 Breakdown of energy contributors to the operation per subsystem for three strategies and BAU, values in kWh/m^3 and percentage in parenthesis

		BAU		C100		DW100		DG100	
WS	Elec (+)	0.74	64.7%	0.56	54.5%	0.54	47.0%	0.60	52.5%
	Ch (+)	0.01	0.76%	0.01	0.7%	0.01	0.6%	0.01	0.6%
WWTW	Elec (+)	0.36	31.6%	0.37	36.0%	0.27	23.3%	0.30	26.0%
	Ch (+)	0.03	2.8%	0.03	3.2%	0.02	2.1%	0.03	2.3%
Reuse	Elec (+)	0.00	(0%	0.06	5.6%	0.31	27.0%	0.21	18.6%
Total		1.15	100%	1.02	100%	1.16	100%	1.15	100%

Ch: chemical, Elec: electricity, FF: fossil fuels contribution is <0.01% all cases

The embodied energy in chemicals does not surpass 3%, having little influence on the system. An energy contribution of 5% and 9%, respectively, has been estimated by another study in relation to the treatment process that transforms groundwater and freshwater for human consumption (Mo, 2012). In this study, it can be observed that the influence of chemicals to embodied energy are less significant than direct energy consumption.

The electricity demand can be offset by integrating renewable sources. In this case study, however, the contribution of renewable energy in total energy use is almost negligible (<1%). The generation of renewable electricity in the WWTW between the BAU and centralised strategies is relatively similar, although it is slightly larger in centralised strategies due to the increase in water supply to achieve 100% in reliability. Decentralised facilities reduce renewable energy generation in WWTW and this can have a negative impact on the energy performance of decentralised strategies. In particular, DW100 would experience the highest reduction of renewable energy generation (i.e. 26%) as the wastewater inflow to the WWTW decreases by 25% in relation to the BAU. Although the total amount of renewable energy generation with respect to the entire UWS in all strategies is minimal (< 1%), at wastewater treatment level it can substitute 45% of conventional fossil-fuel based electricity in the WWTW (SITRATA, 2017). The electricity produced from biogas is the most adopted technology in the existing WWTW, and some interventions such as optimisation of anaerobic digestion, coagulation, and flocculation, and co-digestion with other high organic wastes (e.g. food waste) can increase the biogas yield and energy recovery (Gu *et al.*, 2017). These options are also feasible in this case study.

In addition, producing more clean energy is in line with international commitments for climate change mitigation and adaptation. In addition, with reduced wastewater inflow to the WWTW due to the implementation of decentralised strategies, the proportion of renewable energy produced onsite is thus minimal using these strategies.

Overall, onsite energy generation provides the most direct way to achieve energy recovery under the current conditions; however, the embodied energy and carbon footprint offsets that can be achieved are limited. The WWTW cannot completely offset the direct energy use supplied by the grid. Other arrangements are suggested in this case study to reduce the pumping of water, such as leakage repair.

5.2.4 Global warming potential

The GWP counts all CH_4, N_2O and CO_2 emissions as direct, embodied, and fugitive emissions in the UWS. Figure 5.5 shows the total GWP in $kgCO_2$-eq/m^3 and the percentage contribution per each UWS subsystem within the bars. The GWP values of BAU (0.84 $kgCO_2$-eq/m^3) and C20, C50, DG20, and DG50 are very similar (0.8-0.83 $kgCO_2$-eq/m^3), while some reuse strategies such as C100 and DW100 are smaller (i.e. 0.78 and 0.75 kWh/m^3, respectively). The same figure shows that the water supply subsystem causes 41% of GWP in BAU, 32-39% in centralised reuse strategies, and 33-39% in decentralised strategies. The WWTW subsystem generates 60% of GHG emissions, with a slight increase of 60-64% in centralised systems, and reductions in DW (57-47%) and DG (58-53%). This might be due to the fact that WWTW inflows are decreased in decentralised strategies and hence the energy consumption in this subsystem component is reduced. Opher and Friedler (2016a) also found that the contributions of decentralised strategies were lower than the centralised (2.9×10^3 Ton CO_2-eq/y vs 3.6×10^3 Ton CO_2/y) due to the reduction of untreated wastewater in the central WWTW and a drop in organic loads.

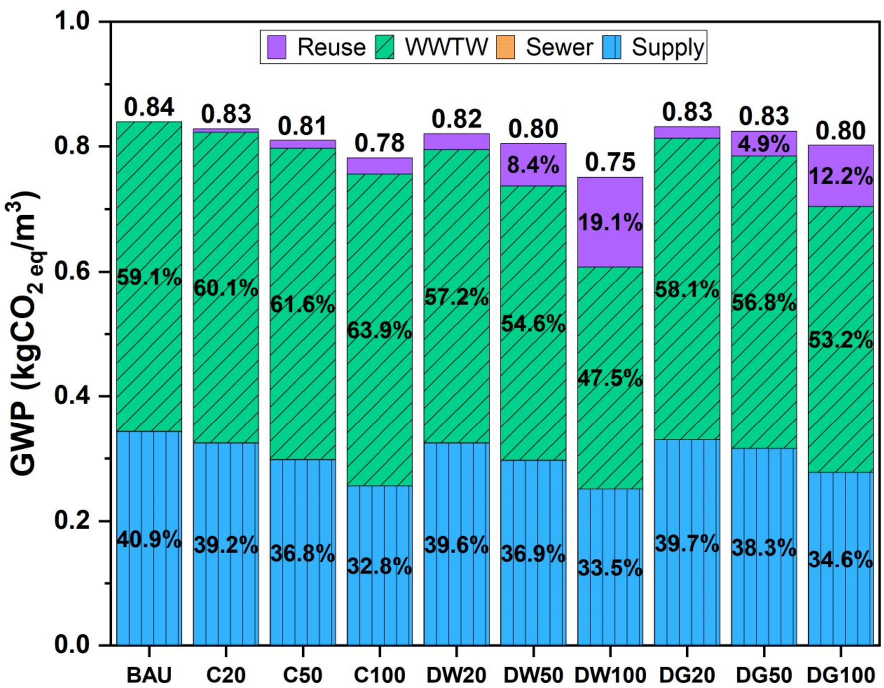


Figure 5.5 Total global warming potential per strategy and contribution per subsystem component. The percentages omitted in centralised strategies are <5%

The sewer subsystem has no input in GWP emissions because in the case under study a gravity system without electricity is used. This is local specific, as in other cases like Mexico City where wastewater flow is not entirely treated but conveyed to the disposal site, the sewerage has 8% more emissions than

WWTW (Moredia-Valek, 2016). Also, the quantifications used in this thesis excluded sewerage data from the analysis as there were no data related to infrastructure age, length, or material inventory. Future work might improve the sewer analysis by including these data. Water reuse is the third component causing GWP, mainly due to the energy for treatment in DEWAT and the transportation of reused water.

A contribution analysis explores in detail the influence of each input for the three strategies and BAU see Table 5.3. All strategies are in Appendix 11. The dominant contributor to GWP is electricity; more specifically pumping groundwater contributes 40% in BAU and 30-55% to GWP in all strategies. In wastewater treatment, electricity is responsible for another 15-20% of GWP in centralised reuse and 0-20% in DEWAT. This is not surprising, as it has been found that consumption of fossil-based electricity contributes to GWP in the Israeli water system (Opher and Friedler, 2016a). Similarly, grid electricity in Mexico is mainly sourced by fossil fuels: 87% gas and coal and 13% from other sources (Santoyo-Castelazo *et al*., 2014).

The second contributors to GWP are biogas and sludge management, both part of the WWTW. The model used in this thesis assumed that treatment by-products are proportional to wastewater inflow (Behzadian *et al.*, 2014), hence biogas and sludge increase in centralised strategies and decrease in decentralised systems. Biogas chambers release methane from the incomplete combustion, contributing 18.4% to GWP in BAU, 19.8% in C100, and ~15.5% in DW100 and DG100 (Table 5.3). Biogas contribution to GWP cannot be neglected, as methane is 28 times higher than CO_2 (IPCC, 2014). For example, in the hypothetical case of half of the biogas being utilised and the rest released, the total GHG emissions would increase to 1.457 kg CO_2/m^3 and CH_4 contribution will increase to 64% (Landa-Cansigno *et al.*, 2020).

Table 5.3 Breakdown of the GWP contributors to the operation per subsystem for three strategies and BAU, values in kg CO_2/m^3 and percentage in parentheses

Source		Emission	BAU		C100		DW100		DG100	
WS	Electricity	CO_2	0.34	(40.6%)	0.25	(32.6%)	0.25	(33.2%)	0.28	(34.4%)
	Chem	CO_2	0.002	(0.2%)	0.002	(0.2%)	0.002	(0.2%)	0.002	(0.2%)
WWTW	Electricity	CO_2	0.17	(19.8%)	0.17	(21.4%)	0.12	(16.3%)	0.14	(17.1%)
	Fuel	CO_2	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
	Chem	CO_2	0.01	(1.0%)	0.01	(1.1%)	0.01	(0.9%)	0.01	(0.9%)
	Biogas	CH_4	0.15	(18.4%)	0.15	(19.8%)	0.11	(15.2%)	0.13	(15.8%)
	Renewable electricity	CO_2	- 0.01	-(1.6%)	- 0.01	-(1.7%)	- 0.01	-(1.3%)	- 0.01	-(1.3%)
	Sludge-landfill	CH_4	0.06	(7.0%)	0.06	(7.6%)	0.04	(5.3%)	0.05	(6.8%)
	Sludge-landfill	N_2O	0.01	(1.1%)	0.01	(1.2%)	0.01	(0.8%)	0.01	(1.1%)
	Sludge-fert	CH_4	0.08	(9.9%)	0.08	(10.7%)	0.06	(7.6%)	0.08	(9.6%)
	Sludge-fert	N_2O	0.05	(5.5%)	0.05	(6.0%)	0.03	(4.2%)	0.04	(5.4%)
	SSP	CO_2	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)
	Urea	CO_2	- 0.02	-(1.9%)	- 0.02	-(2.2%)	- 0.01	-(1.5%)	- 0.02	-(1.9%)
Rw	Electricity	CO_2	0.00	(0.0%)	0.03	(3.3%)	0.14	(19.1%)	0.10	(12.2%)
UWS	Total	CO_2	0.84	(100%)	0.78	(100%)	0.75	(100%)	0.80	(100%)

Specifically in wastewater treatment, the dominant emission is methane (see Table 5.4). This is produced by the incomplete combustion of biogas (19% in BAU) and by sludge disposal (10%), as can be seen in Table 5.3. By increasing the percentage of biogas capture and efficiency in the electricity generator, these emissions might be reduced. Sludge disposal contributes another 20% to GWP, 10% from methane and nitrous oxide emissions (<10%) in BAU, 23% of GWP in BAU, 26% in C100, 18% DW100 and 22% in DG100. The study carried out by Daelman *et al*. (2013) found significant contributions to nitrous oxide emissions (78%) from an activated sludge plant and anaerobic digestion sytem. They explained that the greater nitrogen load in sludge would lead to an increase in N_2O release. However, they included only the wastewater treatment sytem and not the entire system, as in this research. In another study, the contribution of these N_2O emissions to GWP were similar to this thesis (4-7%), supporting the small contribution to this environmental impact (Lane *et al.*, 2015). Nitrous fugitive emissions are less studied because these are incorporated in sludge management, which is often excluded from analysis.

Table 5.4 Emissions per type in WWTW for all strategies, all values in $KgCO_{2eq}/m^3$

Strategy	CO_2	CH_4	N_2O
BAU	0.146	0.296	0.056
C20	0.146	0.297	0.056
C50	0.145	0.297	0.056
C100	0.145	0.298	0.057
DW20	0.138	0.279	0.052
DW50	0.130	0.261	0.049
DW100	0.108	0.210	0.038
DG20	0.140	0.289	0.055
DG50	0.134	0.281	0.054
DG100	0.117	0.258	0.052

The avoided GHG emissions due to by-products valorisation (such as fertiliser and electricity from biogas) cannot be undermined because even at a small percentage of contribution (<4%) these can neutralise the emissions produced by the use of chemicals. Mexico is considered one of the world’s top 10 emitters of greenhouse gas emissions, contributing 110.4 $MtCO_2eq$ or 0.25% of global greenhouse gas emissions (Ge *et al.*, 2014). It ratified the Paris Agreement mitigation contributions in 2015 and developed long-term plans to decarbonise its economy. Thus, the strategies used, even if minimal, are focused on achieving the national reduction targets.

From the previous results, all strategies reduce GWP, however decentralised ones lead to a greater reduction in GHG emissions due to the reduction of both groundwater and wastewater flows. This effect can be seen by decreasing the electricity used in pumping, WWTW, the fugitive emissions in sludge disposal, and incomplete burned biogas released. The highest reduction can be obtained from decentralised wastewater at 100% adoption reuse (DW100), which reduces GWP by 10%. However, this is only valid when the DEWAT energy requirement does not surpass the savings, if the energy requirements change (due to alterations in technology, head loss, efficiency) the results might be different. Future work can be undertaken to compare different technologies with more or less energy requirements such as RBC, filters, and wetlands. These findings support the theory that increasing water reuse is a good way to reduce the global warming potential and is beneficial for the case study.

5.2.5 Eutrophication potential

Eutrophication potential accounts for the direct nutrients discharged to water, embodied indirect sources, and resource recovery offsets. Total Eutrophication in gPO_4/m^3 is shown in Figure 5.6 and reveals that BAU resulted in 22.65 gPO_4eq/m^3 and all reuse strategies resulted in less than BAU. Eutrophication potentials range of 18.9-21.9 gPO_4eq/m^3in centralised strategies, 16.2-21.4 gPO_4eq/m^3 in DW, and 20.4-22.2 gPO_4eq/m^3 in DG, being those with the highest reuse adoptions (e.g. DW100) and lowest eutrophication values. The results of this thesis confirm that implementing water reuse, of any type, is beneficial in order to reduce the eutrophication potential. This is explained by the fact that eutrophication is directly affected by the nutrient loads in discharged effluents (Lijó *et al.*, 2017). If nutrient loads are diverted to reuse, then the loads released into the water bodies are minor.

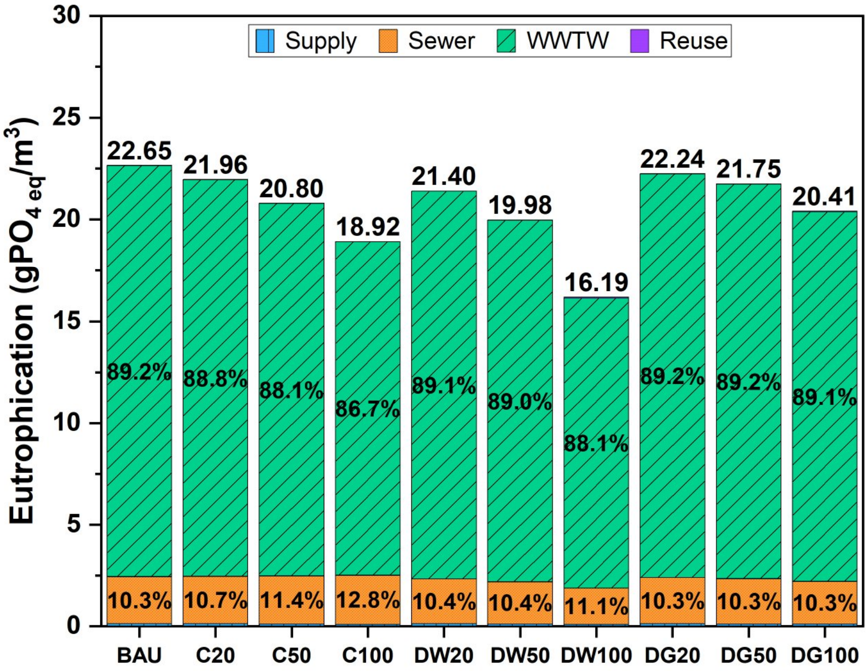


Figure 5.6 Total eutrophication and percentages per subsystem component

The specific contribution disaggregated per input is shown in Table 5.8 for the BAU, C50, DW50, and DG50 strategies; others can be seen in Appendix 12.

Table 5.5. Eutrophication potential of each source in the UWS for BAU and selected strategies. All values in PO_{4eq}/m^3 and percentages in parentheses

Source		Emission	BAU		C100		DW100		DG100	
WS	Electricity	PO_4	0.11	(0.5%)	0.08	(0.4%)	0.08	(0.5%)	0.09	(0.4%)
	Chemicals	PO_4	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
SW	Overflow	TP	1.66	(7.3%)	1.74	(9.2%)	1.27	(7.8%)	1.49	(7.3%)
		NO_3	0.02	(0.1%)	0.02	(0.1%)	0.01	(0.1%)	0.02	(0.1%)
		COD	0.65	(2.9%)	0.65	(3.4%)	0.51	(3.1%)	0.60	(2.9%)
WW	Electricity	PO_4	1.08	(4.8%)	1.08	(5.7%)	0.80	(4.9%)	0.89	(4.4%)
	Chemicals	PO_4	0.05	(0.2%)	0.05	(0.3%)	0.04	(0.2%)	0.04	(0.2%)
	Treated and untreated Discharge	TP	15.97	(70.5%)	12.36	(65.4%)	11.28	(69.7%)	14.27	(69.9%)
		NO_3	0.26	(1.1%)	0.20	(1.0%)	0.18	(1.1%)	0.25	(1.2%)
		COD	1.33	(5.9%)	1.06	(5.6%)	0.93	(5.7%)	1.24	(6.1%)
	Renewable electricity	PO_4	- 0.004	(0.0%)	- 0.004	(0.0%)	-0.003	(0.0%)	-0.004	(0.0%)
	Sludge landfill	NH_3	1.60	(7.1%)	1.62	(8.6%)	1.10	(6.8%)	1.49	(7.3%)
	SSP	PO_4	- 0.01	-(0.1%)	- 0.01	-(0.1%)	-0.01	-(0.1%)	-0.01	-(0.1%)
	Urea	PO_4	-0.04	-(0.2%)	- 0.05	-(0.2%)	-0.03	-(0.2%)	-0.04	-(0.2%)
Rw	Elec	PO_4	0	(0.0%)	0.009	(0.0%)	0.05	(0.3%)	0.03	(0.2%)
UWS	Total	All	22.65	100%	18.92	100%	16.19	(100%)	20.41	(100%)

Eutrophication is caused mainly, at a level of 89%, in the wastewater treatment work due to overflows of untreated wastewater and treated wastewater discharges; among which the phosphorus discharged into receiving water accounted for >65% in reuse strategies and 70% in BAU. These findings only confirm that WWTW is the major hotspot for controlling eutrophication by monitoring water quality in treated effluent (Jeong *et al.*, 2018; Lijó *et al.*, 2017). Other pollutants, such as COD and ammonia contribute to 9 and 7%, respectively. In addition, other sources of eutrophication, such as chemicals, electricity, and offsets due to sludge management and renewable energy replacement seem negligible. The different accounting methods, approaches, and reference units limit the capacity to compare total eutrophication (see Table 5.6). TRACI 2.1 and ReCiPe (midpoint H) are two other methods found in similar studies. TRACI 2.1. considers the effects of TN, TP, COD and atmospheric deposition of substances emitted to air (Rahman *et al.*, 2016), similar to the CML method used in this thesis. ReCiPe accounts for TN and TP as measures of freshwater and marine eutrophication, respectively. Both methods (TRACI and ReCiPe) report eutrophication in nitrogen equivalents.

Table 5.6 Comparison of methods to quantify eutrophication

Reference	Method	Boundary and FU	WWTW technology	EuP	Main finding
Lijó *et al.*, 2017	ReCiPe Midpoint H	WWTW; treated wastewater in a city	AnMBR	3.42 gN/m^3	57-99% of the impact is caused by discharge
Jeong *et al.*, 2018	TRACI 2.1	UWS; $1m^3$ of water reused	MBR	0.5 gN/m^3	82% of eutrophication is attributed to WWTW
Rahman *et al.*, 2016	TRACI 2.1	WWTW; $1m^3$ of treated wastewater	27 technologies, Level 1: conventional	16-18 gN/m^3	Nutrient removal can reduce 70% of total eutrophication
This study	CML 2001, world	UWS; $1m^3$ of water extracted, treated, and reused in the city	MBR	22 gPO_4/m^3	89% is caused in WWTW; reuse strategies reduce eutrophication

Interestingly, greywater reuse does not reduce as much eutrophication as DW strategies. This can be explained through a detailed analysis of pollutants within the water system (see Table 5.7 for phosphorus example). DW recycles all organic load in DEWAT and sends the light greywater into the WWTW. The TP and TN discharged are reduced to a minimum, so too are their mass flows. DG sends all organic load in toilet flushing and kitchen wastewater into the WWTW, so not only the concentration of phosphorus would be higher but so too is the domestic wastewater inflow, both increasing the wastewater inflow. DW100 has a minor flow mass along these four elements of the UWS because the diversion is prior to entry to the sewer. Even a slight increase is observed at the end of the period in the sewer and WWTW. Once the water is treated, a portion of the water will be diverted before being released in the river.

Table 5.7 TP mass flow in each subsystem component, all values in Tonne/y

Component		BAU	C100	DW100	DG100
Sewer	Inflow	113.6	122.4	80.0	101.5
	Overflow	3.6	3.8	2.74	3.2
WWTW	Inflow	110.0	118.66	77.3	98.4
	Overflow	2.1	2.2	1.5	1.8
	Outflow	32.4	34.9	22.7	28.9
Receiving Water	Treated	32.4	24.5	22.7	28.9
	Untreated	5.7	5.9	4.3	5.1
	total	38.0	30.4	26.9	34.0

Further reductions in eutrophication might be obtained through interventions in the WWTW. An option is to treat the wastewater effluent to a higher quality by increasing removal efficiency. A 70% reduction on the average eutrophication potential was found when increasing the removal of nutrients from TN = 8 mg/L; TP = 1 mg/L to higher levels (i.e. TN = 3 mg/L and TP = 0.1 mg/L) (Rahman *et al.*, 2016). Although this is good from the eutrophication side, adapting and implementing tertiary treatment technologies might lead to further energy requirements. This would affect other aspects of the nexus and, as discussed in previous KPI (i.e. energy consumption and global warming), electricity was a key element. Another option is to implement central phosphorus recovery. Phosphorus-based products (calcium phosphate, struvite, white phosphorus, phosphoric acid and phosphate salts) can be recovered from wastewater, sewage sludge, dewatering rejected streams, and sewage ashes to be reused in other sectors. The struvite ($MgNH_4PO_4{\cdot}6H_2O$) can be produced from anaerobic digested liquor, sludge before anaerobic digestion, leachates of sludge after dewatering, or digested sludge after thickening. Struvite can be obtained through precipitation using $MgCl_2$ and NaOH to adjust pH; the process can recover 90-98% of phosphorus in lixiviates. Struvite recovery is recommended for small-medium size plants due to the operational cost involved (Linderholm *et al.*, 2012). Sludge ashes also contain phosphorus that can be recovered through acid hydrolysis using nitric acid (HNO_3), hydrochloric acid (HCl), phosphoric acid (H_3PO_4), or oxalic acids. However, in practice, these technologies are rarely implemented at full scale. Exploring this nutrient recovery potential is important in the context of Mexico, which is an agriculture-based economy and is largely dependent on fertiliser imports. In 2018, the country imported 1.8 million tonnes of urea and 0.9 million tonnes of SSP from China, Iran, and USA (FAO, 2018). Using home-produced materials would help the economy and the circular metabolism of the nutrients. It is necessary to further explore the implications of such recovery technologies and impacts in the UWS.

5.2.6 Acidification potential

The acidification potential impact results from ammonia, sulphur dioxide, and sulfhydric acid in the environment. The average annual acidification in BAU is 10.38 gSO_2eq/m^3, as are the centralised strategies and DG100. The remainder have minor acidification potential, DW from 7.6 to 9.8 gSO_2eq/m^3, DG 20 9.7 gSO_2eq/m^3, and DG50 10.1 gSO_2eq/m^3. Acidification is sourced by 90% in the wastewater treatment, 8-9% in water supply and <5% water reuse in all cases (see Figure 5.7). Acidification decreases in decentralised strategies due to the replacement of groundwater electricity and reduction in wastewater inflow as opposed to centralised wastewater inflow, as explained before for GWP.

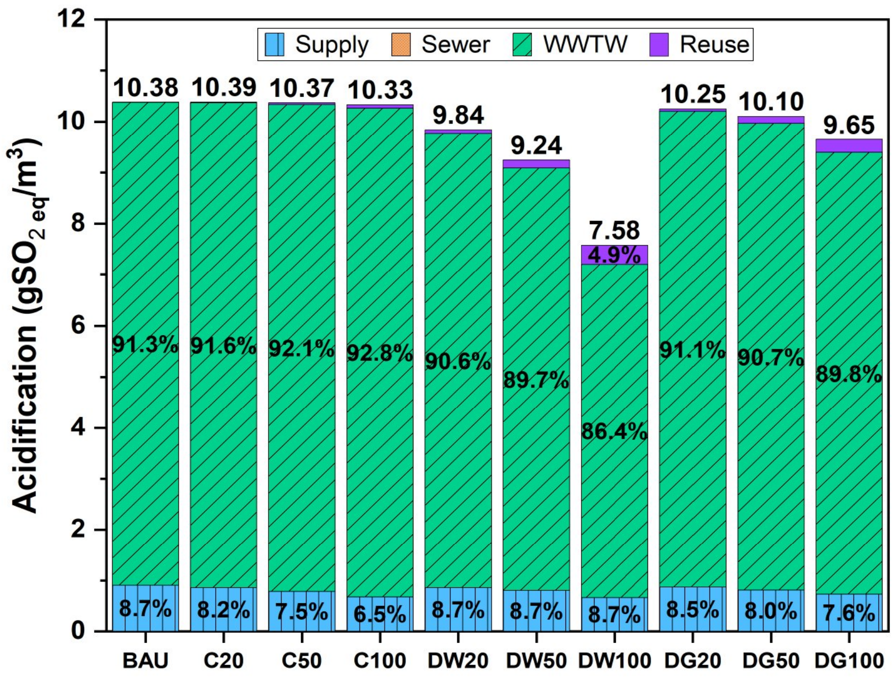


Figure 5.7 Total acidification and percentages per subsystem component

The contribution analysis in Table 5.8 shows that sludge management produces at least 80% of the total emissions released to the atmosphere in the form of NH_3. Rahman *et al*. (2016) found that sludge management from 15 different wastewater technologies account for up to 20% of total acidification. They attributed acidification to fossil fuel combustion and acidification substances during chemical production. In a greywater system in Atlanta (USA), 61% of acidification was attributed to electricity (Jeong *et al.*, 2018), and 90% was attributed to electricity for the water system in Aveiro, Portugal (Lemos *et al.*, 2013). In contrast to earlier findings, no evidence of electricity as the main cause of acidification was detected in the case study. In the municipalities of Rincon, the energy used to pump groundwater in the water supply subsystem is 8% and is calculated at 4% in the central WWTW. Energy from DEWAT makes a minimum contribution to acidification. A probable explanation for these differences is in the accounting boundaries and factors assumed for sludge disposal. Where this thesis considers the final disposal of sludge, others did not include it as part of the water system.

Table 5.8. Acidification potential of each source in the UWS for BAU and selected strategies. All values in gSO_{2eq}/m^3 and percentages in parentheses

Source		Em	BAU		C100		DW100		DG100	
WS	Elec	SO_2	0.89	(8.53%)	0.66	(6.40%)	0.65	(8.56%)	0.82	(8.45%)
	Chem	SO_2	0.02	(0.15%)	0.01	(0.11%)	0.01	(0.15%)	0.01	(0.13%)
WW	Elec	SO_2	0.43	(4.17%)	0.43	(4.20%)	0.32	(4.21%)	0.36	(3.68%)
	Chem	SO_2	0.06	(0.59%)	0.06	(0.59%)	0.04	(0.59%)	0.05	(0.52%)
	Biogas	H_2S	0.69	(6.67%)	0.69	(6.72%)	0.51	(6.73%)	0.57	(5.89%)
	Ren EE	SO_2	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.29%)
	Sludge	NH_3	8.59	(82.79%)	8.73	(84.44%)	5.89	(77.69%)	7.98	(82.71%)
	SSP	SO_2	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)
	Urea	SO_2	-0.26	-(2.54%)	-0.28	-(2.75%)	-0.19	-(2.48%)	-0.26	-(2.66%)
Rw	Elec	SO_2	0.00	(0.00%)	0.07	(0.66%)	0.37	(4.91%)	0.25	(2.63%)
UWS	Total	SO_2	10.38	(100%)	10.33	(100%)	7.58	(100%)	9.65	(100%)

Additionally, the acidification factor for electricity chosen was 0.00119 $kgSO_2/kWh$ but the selected value for NH_3 release from sludge is much larger at 0.2 $kgNH_3/kgN$, and considering that the acidification potential of NH_3 is 1.8 times than SO_2 it is not surprising to observe a higher proportion (see Section 4.5.1). A recommendation that can be made is to generate factors of fugitive emissions tailored to local conditions. Avoided emissions formed by renewable electricity and offsets of industrial fertilisers are calculated at less than 3%. The complete analysis for all strategies can be consulted in Appendix 13.

5.3 Comparative of strategies on the WEP nexus

The results of the analyses of the water reuse strategies are summarised in Figure 5.8, presented as the savings made by five KPIs; the indicator 'water supply deficit' was omitted because it has values <1%.

All strategies studied may save the delivered potable water from groundwater sources, and could reduce eutrophication and GWP. The biggest differences in the performance of centralised and decentralised strategies are the energy and acidification savings. The highest energy savings are found to be related to centralised strategies (i.e. from 2.7% in C20 to 11.5% in C100), whilst decentralised strategies, of any kind, have almost negligible energy savings (0.1-0.3%) or even more energy use (i.e. -0.8%. in DW100). This is due to potable water replacement and the water deficit effect, which was discussed in Section 5.2.1. The acidification potential is significantly reduced in DW reuse compared to DG and centralised reuse (for example, up to 27 % in DW100 relative to 7% in DG100 and 0.4% in C100). The results might suggest that among the nine studied strategies, the most influential are DW100 and C100. DW100 provides the greatest reduction for potable water supply (27%), GWP (17.8%), the EuP (28%), and AcP (27%) but does not reduce energy use. C100 performs relatively well in potable water supply (25%) and En (11%), and on average in GWP (7%) and EuP (16%), but does not reduce acidification. Having said this, it is important to consider the implications of implementing one type of reuse over others examined by the case study.

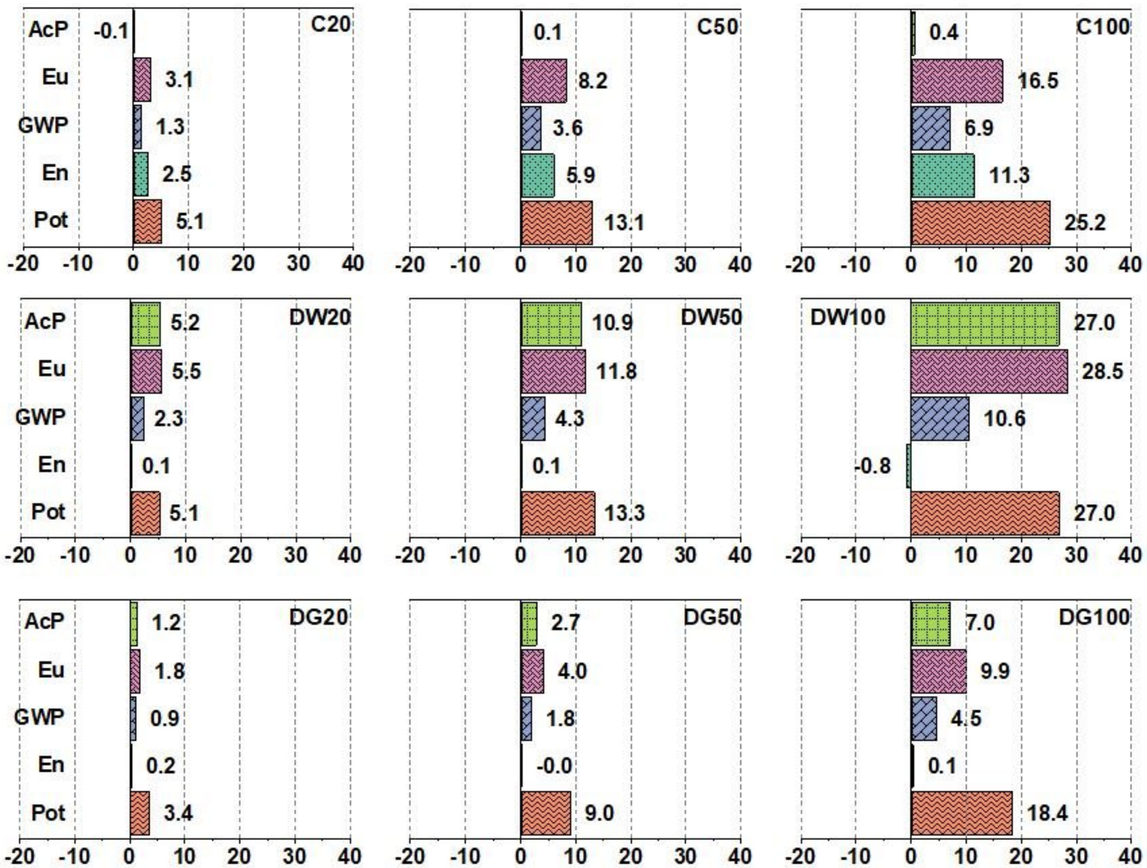


Figure 5.8 Percentage of savings of all KPI's for all nine strategies. AcP:Acidification potential, GWP: Global warming potential, EuP: Eutrophication potential; En: Energy; Pot: Potable water from groundwater sources

The dichotomy of centralisation vs decentralisation is currently at the centre of the debate in the water sector (Libralato *et al*., 2012). Planning with a decentralised perspective considers the change in infrastructure and the regulatory and social contexts. Implementing DW strategies requires an independent sewer network for domestic wastewater and others for industrial effluent and stormwater. Historically, the collection and treatment of municipal wastewater has been set up to be managed centrally in combined sewers and WWTW, while wastewater source separation in general is easier in areas with no sewage. Reconfiguring sanitary sewers might be very difficult in densely populated areas, as in the case study, but could be an interesting option in new developments where domestic wastewater sewers can be planned ahead of construction. By doing so, it would be necessary to modify the local urban plans in addition to land use and incorporate the views of local neighbourhoods.

The lack of energy saving in DW strategies operating in the case study can be translated as an economical drawback. For example, in 2014 electricity for groundwater cost £2,7000 per year[6], which was 10% of the income required to operate the water system in SFR (£2.7 million per year; CEAG, 2014). Centralised reuse can save up to 11% of energy, which would absorb the cost of pumping groundwater, an advantage unseen in decentralised reuse. Besides, energy in Mexico is currently suffering from a recent crisis due to the drop in oil prices, which is directly related to the electricity service (Melgar, 2020). Future studies must include a cost-benefit analysis on the investment in infrastructure and economic savings.

On the other hand, strategic planning to increment central water reuse is advantageous in the case study. It requires infrastructure to supply reclaimed water; currently only one city has a line, leaving the other region without one. In regard to water use, a preferred option to follow may be to increase the adoption of industrial reuse, which might save 15% of water (see Figure 5.2).

[6] Total annual income was Mx$69,250,304 and expenditure of electricity in groundwater was Mx$6,801,924 in 2014 (CEAG, 2014). The exchange rate selected was £1= Mx$25

Some industries, however, would need to probe and adapt the industrial processes to integrate an additional pipeline. Industrial reuse is currently being undertaken by 92 tanning industries in the city of Leon, Guanajuato, very close to the location of the case study (SAPAL, 2018). The water utility in Leon has implemented economic incentives (e.g. lower price for reclaimed water vs drinking potable water) and demonstrated safety in the use of reclaimed water in water reuse projects. Similar strategies can be carried out in the case study region; in particular, tailoring demonstrations to the local industry. Increasing urban irrigation reuse is the easiest method, as it only requires an increased number of trips. However, this will only add 1% of reclaimed water usage. The last reuse type studied in this thesis was toilet flushes. This strategy may require social and behavioural change interventions as opposition is normally encountered in direct public contact reuse (Bichai *et al*., 2018; Friedler and Lahav, 2006). Previously identified interventions to overcome community rejection involve the engagement of stakeholders during the planning and operational lifetime of the project, education programmes for residents, and the dissemination of information to developers on best practice concerning water (Sharma *et al*., 2012). Even if all reused water is demanded, there will be enough water for other usage, such as agriculture. This sector demands 75% of water resources (CONAGUA, 2018), meaning that the water can be used on crops grown for non-human consumption in the peri-urban area. Hence, urban and agricultural reuse can co-exist and strengthen the integrated water management. Finally, derived from the results of acidification in centralised strategies, a key point is to convert sludge into valuable products. For example, mixtures of 5% sewage sludge and ceramic can produce good quality bricks, creating surplus, and avoiding deposition in agricultural land and the consequent emissions (Martínez-García *et al*., 2012).

In all cases, it is needed to maintain cooperation among decision-makers in both cities. So far, the three water utilities partnership has been shown to tackle some of the institutional barriers identified in other cases; for example, the uncoordinated institutional framework, unclear roles and responsibilities, minimal monitoring, and poor strategic vision (Brown and Farrelly, 2009). In addition, a good relationship between the local governments of both cities in the case study is necessary to maintain the successful operation of the system in long-term projects.

In this section some recommendations were given to promote water reuse in San Francisco and Purisima del Rincon, so the findings cannot be extrapolated to other cases and the ultimate decision relies on decision-makers in these locations.

5.4 Sensitivity analysis results

The sensitivity coefficient was used to identify the sources of wastewater pollution that have the greatest influence on global warming, eutrophication, and acidification in BAU. The analysis assumed that the parameters were independent of one another. A total of 50 runs of each of the 30 parameters were conducted in the range of values presented in Table 3.8. Parameters were randomly sampled using a Latin hypercube technique considering a uniform distribution. One parameter was varied while the rest were held constant.

The sensitivity index used has a scale of 0 to ∞. The closer to zero, the less influence. For the GWP (Fig. 5.9a), the most important parameter was the total suspended solids sourced by the toilet (SI=0.15). Others with a minor influence (<0.1) were TSS in washing machines, industry, and kitchens. For eutrophication (Fig. 5.9b), the four pollutants showing greatest sensitivity were TSS in toilet and phosphorus in toilets, washing machines, and industry. In the case of acidification (Fig. 5.9c), the top four pollutants were practically the same as in GWP. COD and BOD in all household fittings and industry effluents have zero influence in GWP and acidification. BOD in hand basins, kitchens, and washing machines is found to have no impact on eutrophication. The remaining pollutants have a minimal influence (see Appendix 18).

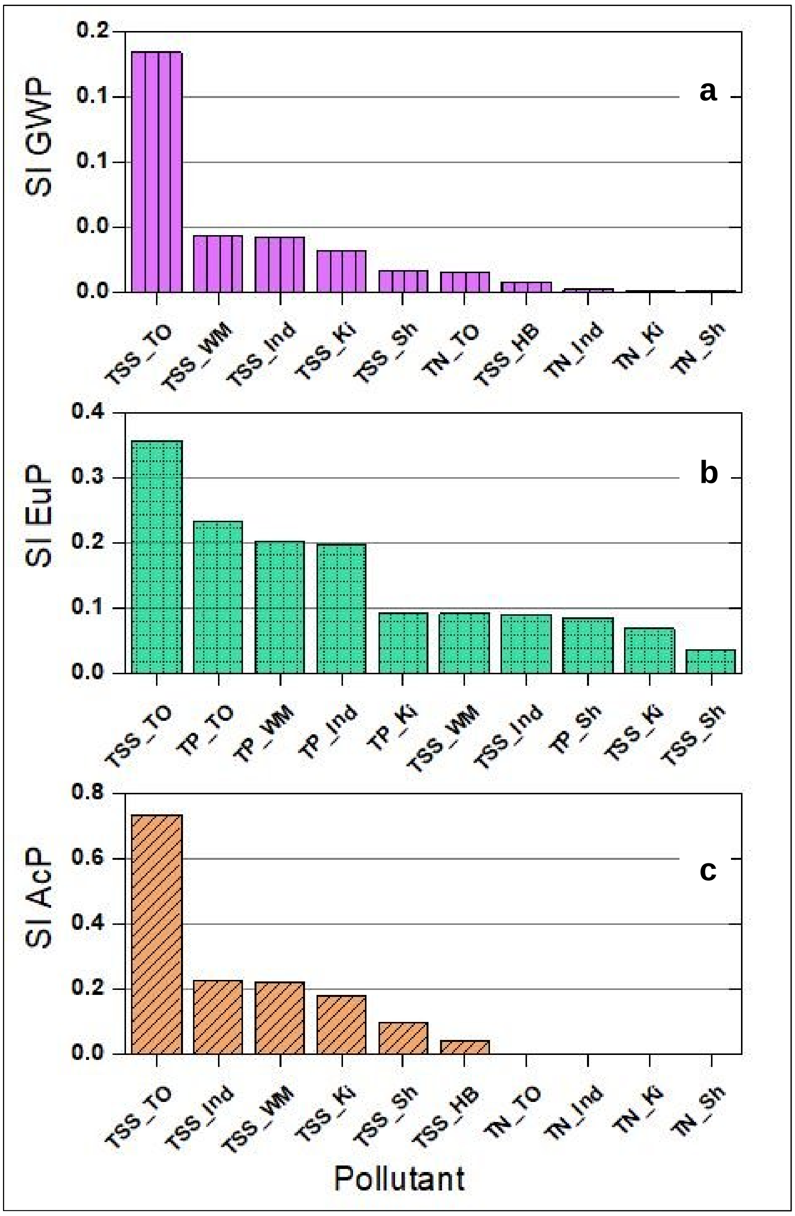


Figure 5.9 Top ten sensitivity index of a) Global warming, b) Eutrophication and c) Acidification changes in BAU using one-at-a-time approach.

TSS: Total suspended solids; TN: Total nitrogen, TP: Total phosphorus. HB: Hand basin; ki: kitchen; WM washing machine; Sh shower, To: toilet: Ind: industry

The relatively low sensitivity index found in relation to global warming suggests that other parameters in the operation may have a stronger influence in this impact. For example, Cornejo (2015) found that carbon footprint was the most sensitive to methane emissions resulting from BOD in wastewater. In the case of the model used in this thesis, emissions are calculated based on a rate of biogas production per volume of wastewater, and hence changes in concentration are not evident in the analysis. However, the results previously obtained (see Section 5.2.4) suggest that the direct electricity and biogas use rate may be more sensitive to GWP. Therefore, this analysis can be improved by testing other operational parameters.

TP was the largest contributor to eutrophication potential, accounting for 70% of the total, with sludge management being the second contributor. The results highlight why eutrophication is sensitive to different sources of phosphorus and TSS. Sludge management causes 90% of acidification due to fugitive emissions. Acidification is very sensible to TSS sourced by toilets, which is also the main component of sludge (see Eq. 3).

In summary, the sensitivity analysis showed that total suspended solids and total phosphorus in toilet flushes have a great influence on global warming, eutrophication and acidification impacts in BAU. Other parameters such as COD and BOD were negligible, suggesting that these can be excluded from the model. Future work should focus on considering other parameters of interest, for example electricity, and complete the analysis for at least one type of each reuse strategy (C50, DW50 and DG50).

5.5 Summary of the chapter

The results obtained in this chapter reveal that the metabolic performance of both centralised and decentralised strategies would affect the KPIs positively upon almost all water-energy-pollution nexus elements within the water system.

Energy balance shows that centralised strategies are more energy efficient than decentralised under the assumptions made for the case study. The energy in all six decentralised strategies is equivalent to BAU (1.15 kWh/m^3), while the energy in the three centralised is minor (i.e. 1.02 kWh/m^3 in C100). The reduction in groundwater use leads to a decrease in the electricity used within the water supply subsystem in all strategies. These savings in the decentralised system are enough to compensate for the additional energy use in the DEWATs, and are higher than the energy needed for centralised reuse (only the transportation of reclaimed water). With regard to renewable energy, it is affected by the sewer inflow; however, in the end it only represents <1% of total energy consumed in the water system.

The GWP of BAU is 0.84 $kgCO_2$-eq/m^3. While C20, C50, DG20, and DG50 are very similar to BAU, the C100 and DW100 strategies reduce emissions to 0.8 and 0.78 $kgCO_2$-eq/m^3, respectively. The WWTW is a key subsystem in the impact of GWP, generating around 60% of the impact because of the electricity and biogas combustion. In addition, the electricity used in groundwater withdrawals and wastewater treatment emits nearly 40% of all emissions alone. The best performing strategy is DW100 (10% savings). This is because the electricity used in pumping, WWTW, the fugitive emissions in sludge disposal and biogas will decrease, as well as the GHG emissions. However, this is only valid when the DEWAT energy requirement does not surpass the savings and whether the energy requirements change (due to a change in technology, head loss, efficiency) further studies might be required.

Implementing water reuse resulted in a decrease in the eutrophication in all strategies, regardless of the scale level. However, higher EuP reductions were found in DW, then centralised and, finally, greywater. These were as follows from 22.65 gPO_4eq/m^3 to 16.19 gPO_4eq/m^3 in DW100, 18.92 gPO_4eq/m^3 in C100 and DG is 20.41 gPO_4eq/m^3. For phosphorus discharged into receiving water, at least 65% is sourced from treated effluent of the WWTW and 10% from sewer overflows. Other pollutants contribute to 9% of COD and 7% of ammonia. A further reduction in eutrophication might be obtained through phosphorus recovery in WWTW by testing technologies to reduce phosphorus concentration in sludge or recover it as struvite; however, further studies are needed as some of these technologies are not tested at larger scales. The offsets (due to sludge management and renewable energy replacement) do not influence more than 1% of total eutrophication.

The average annual acidification in BAU is 10.38 gSO_2eq/m^3 with a general decreasing tendency in decentralised strategies, both DW and DG.

Centralised strategies C20 and C50 are equivalent to BAU and just a small saving is observed in C100. Acidification is sourced by 90% in wastewater treatment, 8-9% in water supply and <5% water reuse in all cases, more specifically due to ammonia in sludge management. These findings highlight the importance of sludge management.

Although all strategies would show positive savings in all KPIs, it is arguable that the most influential water reuse strategies for the case study are DW100 and C100.

Chapter 6. Conclusions and future work

Chapter 6 Conclusions and future work

The main objective of this thesis was to assess the performance of centralised and decentralised water reuse strategies within a UWS. This objective has been achieved by proposing a comprehensive framework based on the urban water metabolism and the water-energy-pollutant nexus approaches. Its application was tested in the real case study of Rincon cities in Mexico.

6.1 Contribution to knowledge

The contribution to knowledge of this thesis is summarised by three key points. Firstly, this is a pioneer study addressing the water metabolism under a water-energy-pollution nexus perspective to analyse water systems. The novel framework used highlights the complexity of the interactions among pollutants, energy, and the environmental impacts to the integrated UWS. This analysis provides information around the implications of reusing water on six KPI in the integrated water system, including groundwater extraction, wastewater treatment, biogas, and sludge management. Using this approach, the analysis can contribute to create more circular and sustainable water systems.

Secondly, studies within the Mexican context are very rare and, to the knowledge of the author, only three others address water-energy nexus. Two studies focused on describing the UWS and simulating rain-water harvesting in Mexico City (Moredia-Valek *et al.*, 2017; Valdez *et al.*, 2016). The closest study is that reported by the WaCLiM project in the case study (WaCCLim, 2017). However, that study focused on recommendations for WWTW operation and only covered a portion of the system (Purisima city was excluded). This thesis includes the entire system comprising both cities, assessing water reuse strategies as part of the integrated UWS, which has not been done previously.

Thirdly, it contributes to the current debate about whether or not to decentralise the water sector. The results intend to support the decision making process by providing information about environmental impacts and trade-offs when centralised and/or decentralised water reuse strategies are implemented in the case study. The thesis findings support the maximisation of water reuse to achieve the highest savings in all KPIs and highlight the benefits of a centralised or decentralised system over the business as usual system.

6.2 Conclusions

The key findings per objective are briefly summarised.

Objective 1. A conceptual framework for quantifying the main flows and fluxes of water, energy, pollutants and other environmental impacts was developed. This objective was accomplished by critically reviewing state-of-the-art urban water, the energy-nexus framework, and the urban water metabolism approach. By comparing the key elements and features of the system under analysis, the methodology described a set of five subsystems within the UWS. It also establishes clear spatial and temporal boundaries for comparison and explains the unit of analysis, as these have been highlighted as key characteristics in defining the system and affecting the model. The modelling approach of urban water metabolism was considered to first integrate all main UWS components of the water supply, stormwater, and wastewater subsystems before incorporating the influence of the intervention strategies on other components of the urban water cycle. Six performance assessment criteria were selected to represent the WEP nexus. These were: deficit of water supply, energy consumption, total eutrophication potential, total acidification potential, and global warming potential. All of these were found to be feasible and viable to consider. A previous comparison of different modelling tools identified WaterMet2 as one of the most complete modelling tools with a limited data requirement and complete urban water model and environmental impact assessment. Consequently, WaterMet2 was selected as the modelling tool.

Objective 2. Nine different water reuse strategies were defined. Three key features established the selection of the strategies: wastewater source and adoption rate. The three sources of wastewater were: reclaimed water from centralised WWTW, domestic treated wastewater from a DEWAT, and greywater from a DEWAT. All of these strategies were set up for the real case study of San Francisco and Purisima del Rincon in Mexico.

Objective 3. Using the assessment and conceptual model developed, the adoption of reuse and centralisation levels were evaluated. All strategies perform better or relatively similar to BAU, but they behave differently to each KPI. All strategies in this study were found to save potable water, eutrophication and GHG emissions.

The energy in all six decentralised strategies and BAU are equivalent (1.15 kWh/m^3). Centralised strategies are more energy efficient (i.e. 1.02 kWh/m^3 in C100) than decentralised under the assumptions made. The reduction in groundwater use leads to a decrease in the electricity used in the water supply subsystem in all strategies, but in decentralised systems these savings are not enough to compensate for the additional energy use in the DEWAT systems. With regard to renewable energy, it represents just <1% of total energy consumed.

The WWTW is a key subsystem in the impact of GWP, generating around 60% of the impact due to electricity and biogas combustion. In addition, the electricity used in groundwater withdrawal and wastewater treatment emits nearly 40% of all emissions alone. DW100 strategy produces the lowest GWP of 0.78 kgCO_2-eq/m^3. This is because the electricity used in pumping and WWTW will decrease, but so will biogas and fugitive emissions in DW100. The results may differ if using other DEWAT technology. Future work should test different technologies.

Implementing water reuse resulted in a decrease in the eutrophication in all strategies, regardless of the centralisation level. However, higher EuP reductions were found in DW100 from 22.65gPO_4eq/m^3 in BAU to 16.19 gPO_4eq/m^3. C100 produced 18.92 gPO_4eq/m^3 and the result for DG was 20.41 gPO_4eq/m^3. The phosphorus discharged into receiving water causes 75% of this impact, COD is responsible for 9%, and ammonia just 7%. Phosphorus recovery, before releasing the water into the river, might benefit the environment and economy - a topic requiring further research. The offsets (due to sludge management and renewable energy replacement) do not influence more than 1% the total eutrophication.

Decentralised strategies decrease acidification and centralised strategies has no effect on this KPI. Most acidification is caused by the ammonia released in sludge management, 8-9% in water supply, and <5% in water reuse in all cases. These findings highlight the importance of sludge management.

A sensitivity analysis shows that from all thirty pollutants, two have the highest influence in the model output in BAU: total suspended solids and total phosphorus, both sourced by toilet flushes.

Objective 4. The most influential water reuse strategy is DW100, a decentralised type using domestic wastewater; installing DW reuse may require the installation of a separate domestic sewer along with DEWAT facilities, which could be costly. Hence, this option seems more appealing in new urban developments. Further analysis of the socio-economic aspects of decentralised treatment and testing of other treatment technologies would complete this recommendation.

The second recommendation is to maximise the use of centralised reuse, especially the industrial type. This would require an infrastructure for distributing the water to users, but it will reduce 11.5% of energy consumption with respect to BAU if C100 is implemented. To further reduce acidification and eutrophication would require improvement to the nutrient recovery system in the WWTW and a reduction of nutrients in sludge.

Urban greywater reuse is increasingly being adopted worldwide, however this strategy did not surpass the performance of DW or centralised strategies. In any case, social interventions to overcome rejection, institutional cooperation to maintain the success of long-term water reuse projects, and an improved legal framework are needed.

Other recommendations to water utility companies are based on improvements in biogas combustion and maximising the production of renewable energy, including nutrient reduction in treated effluent through phosphorus recovery, and reduction of sludge disposal in agriculture through the valorisation of this resource in other industrial chains.

Overall, the conclusions from addressing both research questions are:

The results show that the interaction between the WEP nexus elements can be quantified as a result of the metabolic performance simulation of integrated UWS. Consequently, the assessment of water reuse strategies with respect to WEP nexus criteria can unveil the direct and indirect influences between the nexus elements (i.e. water, energy and pollution) that are either difficult to recognise or unexpected due to the complexity of the integrated UWS. The opposite influences could occur due to the complex and indirect interaction that may exist between UWS components and the overall system. Wastewater treatment work is a key subsystem in the UWS for the nexus, more specifically wastewater inflows, the electricity used in the WWTW, and sludge management are the three main contributors to the nexus. The second most important is the water supply subsystem, more specifically the electricity used in pumping groundwater.

It must be acknowledged that the methodology chosen for this thesis used a typical real-world case study to explore the capabilities of the suggested framework, thus the findings obtained here cannot be extrapolated to other UWS. For example, the adoption/uptake percentages of water reuse options must be tailored based on socio-economic factors in the UWS. Hence, more test cases should be conducted on other real-world case studies in order to extract some general outcomes with respect to the water-energy-pollution nexus for water reuse strategies.

6.3 Future work

The following topics are suggested for future work:

Modelling the effect of nutrient management options in both sludge management and wastewater treatment. It was found that NH_3 from sludge management in disposal sites largely contributes to acidification (90%), and TP from effluents contributes to eutrophication (30%). It would be very beneficial to explore various options of sludge management apart from agriculture reuse, in addition to being useful to compare further options such as incineration or biological treatment. With regard to phosphorus in effluent, it might be interesting to model the effect of TP recovery along the wastewater treatment work in the form or struvite, for example. In such a way, it would reduce the concentration of this pollutant in both the sludge and discharge. Although few studies have been undertaken in the past comparing sludge management options, it is very rare to find them for the purposes of comparing centralised and decentralised water reuse.

Investigating the influence of different DEWATs. The finding that decentralised reuse has the same energy as BAU was obtained under the assumption of installing an MBR technology. Other technologies tested in different case studies (e.g. RBC, constructed wetland, combined sand filters) would modify energy consumption and efficiency. The comparison of such technologies under the developed framework would provide new insights over decentralised reuse.

Local characterisation of greywater pollution. The proposed conceptual model requires a detailed quantitative data of greywater quality in each of the household components. To the author's knowledge, no study has characterised the GW in the level of detail required in Mexico. This was solved by using other sources of information in other countries. This, however, could introduce uncertainties in the results. It is suggested that the quality of greywater may be adjusted to the local characteristics. Producing local data on greywater quality through sampling and analysis in the case study would help to reduce uncertainties. It is also suggested to expand the study for other pollutants, such as metals, plastics and endocrine disruptors.

Adapting decentralised quality guidelines in Mexico. Water reuse policies in the context of the case study, Mexico, are largely formulated for centralised scale (NOM-003). Such regulations are based on the end use, mostly for agriculture. Some international policies address decentralised reuse, but are limited. The insights of this research, from the water-energy-pollution approach, discovered that decentralised reuse using domestic wastewater and greywater can outperform centralised reuse in most of the KPI studied. Given the worldwide interest in urban greywater, it is natural to think that decentralised water reuse will soon be adopted in Mexican cities. To this end, it is necessary to adapt the regulatory frames and policies to control, implement, and operate decentralised systems.

Completeness of sewerage data. It is worth noting that data on sewerage were constrained due to the lack of a reliable inventory of materials, size and age of pipelines, hence embodied energy for this sector was left out. In respect to energy usage, data regarding household energy consumption was out of the scope. This can increase embodied energy in the sewer and be the main element contributing to energy from this subsystem. Thus, a future study might address the lack of these data.

Consideration of other types of water reuse. In the future, it might be necessary to consider water reuse in agriculture. As the results show, treated wastewater exceeds the urban non-potable demand, thus presenting a potential application outside the city area. Scaling up the analysis to a region would also require the consideration of a groundwater system. In addition, it will be important to include rainwater harvesting as another decentralised strategy, which may have some seasonal impact on the case study.

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Appendix

Appendix

Appendix.1 Surveys

Thank you for participating in this PhD research at University College London. This survey aims to identify future interventions of water reuse, and to score key performance indicators of water reuse and nutrient recovery interventions. The ultimate goal is to understand the effects of such interventions in the Water-Energy-Nutrient nexus of the urban water system in Purisima and San Francisco, Guanajuato, Mexico.

The estimated time to complete the survey is 10 minutes. Please answer the questions the best as you can, your participation is very important. If necessary, you can leave some questions in blank or drop out of the survey at any time.

Results are anonymous and your private data is confidential. If you have any question or comment please contact the email oriana.cansigno.13@ucl.ac.uk

I accept the terms to participate in the survey

Part I. Implementation scenarios

a) What is your working sector?

(1) Research and innovation
(2) Public sector (local operation)
(3) Public sector (Regional management)

b) What is your area of expertise?

(1) Water sources and supply
(2) Sewage
(3) Wastewater treatment
(4) Water reuse
(5) Sludge management
(6) Biofuels and energy
(7) Management of water utilities

c) In your opinion, what strategies are crucial in urban areas of Guanajuato to cope with climate change and water scarcity? Please tick all that apply and then select the three most important.

• Implementation of consumer-saving technologies in household consumption (saving keys, low-energy baths, efficient washing machines)
• Central-level water reuse
• Implementation of Rainwater
• Leakage reduction
• Reuse of greywater at the house-level
• Water conservation strategies in the industry
• Saving practices in urban irrigation of parks and gardens
** This question is to introduce the topic, confirm and consider water in the region and classify unconventional water sources

The study area has a population equivalent to 115,000 inhabitants. The wastewater treatment plant produces a useful effluent for urban reuse according to NOM-003-SEMARNAT-1997. In this study, urban reuse considers irrigation of parks, industries and washes. In the following questions assume that the localities of the study area could be developed similarly and do not take into account the variations by population increase.

d) In 2015, the urban water reuse from the centralised wastewater treatment plant in the urban area in the study was 1%. In the next decades, how do you expect this total percentage to change?

In 10 years, centralised water reuse will increase from 1% to ______ % in the study area.
In 20 years, water reuse will increase at a total percentage of __% in the study area

* Scenario questions only include increase because BAU includes reuse and is calibrated thus, cannot be estimated to stop reuse).

* Restrict responses to greater than or equal to 1%

e) In this research, greywater considers the effluents of showers, washing machine, and hand-basins. What would be the potential uses of greywater in urban applications in Guanajuato? Please rank them from 1 to 4, where 1 is the most suitable and 4 the least. Use N/A to state this use is not suitable for the area.

(1) Urban irrigation of parks
(2) Toilet flushing
(3) Industry (e.g. construction)
(4) Other indoor uses (shower, kitchen, etc.)

f) Currently, the use of light greywater is zero. In the next decades, how do you expect this total percentage to change?

In 10 years, greywater reuse will increase from 0% to ______ % in the study area.
In 20 years, greywater reuse will increase at a total percentage of __% in the study area.

g) What would be the potential uses of rainwater harvesting in a city in Guanajuato? Please rank them from 1 to 4, where 1 is the most suitable and 4 the least. Use N/A to state this use is not suitable for the area.

(1) Urban irrigation of parks
(2) Toilet flushing
(3) Industry (e.g. construction)
(4) Other indoor uses (shower, kitchen, etc.)

h) In the area, rainwater harvesting is zero. In the coming decades, how much do you expect this percentage to change?

In 10 years, the use of greywater will increase from 0% to ______% in the study area.
In 20 years, the use of greywater will increase to a total percentage of ___% in the area of study.

Appendix.2 Demographic data and population forecast

This estimation used and arithmetic method presented in following formulae:

$$Pf = Pc + At$$

Where, Pf is the future population in the future (inhab), Pc is the current population current (hab), A is the growth rate (hab/y) and t is the design period (number of years). The growth rate was calculated as the average of the difference between the population in the last 20 years. The population per city in the period 1990-2019 were obtained from XI, XII and XIII census in the municipality (INEGI 1990, 2000, 2010) and the population counts of 1995 and 2005 (INEGI 1995 and 2005). These are shown in Table A.1.

Table A. 1 Historic population in San Francisco and Purisima cities in the period 1990-2010

Year	San Francisco	Purisima	Source
1990	52291	12486	INEGI, 1990
1995	64577	15885	INEGI, 1995
2000	65183	25274	INEGI, 2000
2005	68282	33825	INEGI, 2005
2010	71139	43512	INEGI, 2010

Regression coefficient to this model was above 95%, R^2SB1:0.9911 and R^2SB2: 0.9989

Table A.2 Population variation per year

Year	Population SFR	Rate	Population Purisima	Rate
2015	75851	1.066237	51268.5	1.178261
2016	76793.4	1.012424	52819.8	1.030258
2017	77735.8	1.012272	54371.1	1.02937
2018	78678.2	1.012123	55922.4	1.028532
2019	79620.6	1.011978	57473.7	1.02774
2020	80563	1.011836	59025	1.026991
2021	81505.4	1.011698	60576.3	1.026282
2022	82447.8	1.011562	62127.6	1.025609
2023	83390.2	1.01143	63678.9	1.02497
2024	84332.6	1.011301	65230.2	1.024361
2025	85275	1.011175	66781.5	1.023782
2026	86217.4	1.011051	68332.8	1.023229
2027	87159.8	1.010931	69884.1	1.022702
2028	88102.2	1.010812	71435.4	1.022198
2029	89044.6	1.010697	72986.7	1.021716
2030	89987	1.010583	74538	1.021255
2031	90929.4	1.010473	76089.3	1.020812
2032	91871.8	1.010364	77640.6	1.020388
2033	92814.2	1.010258	79191.9	1.019981
2034	93756.6	1.010154	80743.2	1.019589
2035	94699	1.010052	82294.5	1.019213
2036	95641.4	1.009952	83845.8	1.018851
2037	96583.8	1.009853	85397.1	1.018502
2038	97526.2	1.009757	86948.4	1.018166
2039	98468.6	1.009663	88499.7	1.017842
2040	99411	1.009571	90051	1.017529
2041	100353.4	1.00948	91602.3	1.017227
2042	101295.8	1.009391	93153.6	1.016935
2043	102238.2	1.009303	94704.9	1.016653
2044	103180.6	1.009218	96256.2	1.01638
2045	104123	1.009133	97807.5	1.016116

Appendix.3 Pervious and impervious area

Pervious area refers to the total area occupied by parks, sport facilities, green lanes, peri-urban irrigation plots and water bodies. The impervious area is divided into two, a) Roofs which include the surface urbanised by households, health centres, schools, markets, temples, industries, and b) roads: streets and paved surfaces. Areas were estimated through a Geographic information system supported in ArcGis 10.2. The maps used were the urbanised lots (polygon), land use (polygon), infrastructure and industries (polygon), political boundary per municipality and Digital Elevation Model, all sourced from the National Statistics Institute Information (INEGI, 2014).

Table A.3. Pervious and impervious areas in the case study

	SB2 area	SB1 area
Total area	1,229.00 ha	1,615.00
Pervious sub-total	377.22 (30.7%)	424.79 (26.3%)
Green lane	2.61	4.99
Parks	3.27	5.64
Sport facilities	-	1.67
Water bodies (Streams in the city)	13.34	16.49
Peri-urban irrigation	358.00	396.00
Impervious sub-total	850.00	1219
Household roof	690.4 (56.2%)	938.7 (58.1%)
Industry		25.90
Roads	161.4 (13.1%)	(15.6%)

Appendix.4 Potable water subsystem database Water withdrawals

Table A. 4 Summary of water withdrawals and energy in the UWS in 2015

Borehole	Groundwater withdrawals (m^3x10^3/y)	Energy (MWh/y)	kWh/m^3
1	0.00	0.16	n.d
2	282.36	73.90	0.262
3	121.44	36.17	0.298
4	1977.71	695.76	0.352
5	906.15	348.32	0.384
6	468.40	314.88	0.672
7	587.69	191.68	0.326
8	407.72	197.98	0.486
9	542.01	300.40	0.554
10	669.27	241.44	0.361
11	68.69	41.87	0.609
12	73.90	37.10	0.502
13	428.96	85.80	0.200
14	907.04	246.86	0.272
15	492.22	201.30	0.409
16	147.50	117.23	0.795
17	493.90	296.72	0.601
18	282.44	232.68	0.824
19	130.54	44.51	0.341
20	193.71	67.42	0.348
21	228.55	162.39	0.711
22	67.53	38.56	0.571
Grand Total	3372.38	2772.69	0.822

Table A. 5 Summary of water withdrawals and energy in the UWS in 2016

Borehole	Groundwater withdrawals (m^3x10^3/y)	Energy (MWh/y)	kWh/m^3
1	256.20	142.40	0.556
2	435.02	85.71	0.197
3	530.69	120.05	0.226
4	1428.73	518.43	0.363
5	771.93	308.96	0.400
6	446.26	272.00	0.610
7	588.36	197.28	0.335
8	428.25	217.16	0.507
9	508.41	264.64	0.521
10	674.72	254.96	0.378
11	101.42	64.45	0.635
12	86.87	46.53	0.536
13	392.72	86.96	0.221
14	851.13	230.69	0.271
15	476.60	185.10	0.388
16	141.00	82.99	0.589
17	604.44	303.44	0.502
18	297.74	230.11	0.773
19	183.68	54.84	0.299
20	204.77	66.96	0.327
21	223.49	178.86	0.800
22	61.43	37.12	0.604
Grand Total	3436.99	1457.06	0.424

Appendix.5 Wastewater, electricity and biogas consumption

Table A.6 Monthly wastewater inflows, electricity consumption and biogas production in the UWS in 2015 and 2016

Month	Wastewater (m^3/month)	Electricity (kWh/m^3_{ww})	Biogas (m^3/m^3_{ww})
Jan-15	363,279	-	-
Feb	319,741	0.001	0.08
Mar	347,531	0.01	0.07
Apr	383,604	0.03	0.07
May	371,032	0.02	0.08
Jun	337,968	0.02	0.06
Jul	313,052	0.01	0.04
Aug	368,033	0.01	0.06
Sep	354,677	0.02	0.05
Oct	424,054	0.02	0.05
Nov	437,008	0.02	0.06
Dec	424,049	0.03	0.08
Jan-16	478,590	0.03	0.08
Feb	401,083	0.03	0.08
Mar	413,389	0.03	0.08
Apr	399,004	0.03	0.08
May	430,622	0.06	0.08
Jun	382,707	0.05	0.07
Jul	490,902	0.03	0.05
Aug	514,366	0.01	0.03
Sep	411,998	0.03	0.05
Oct	460,395	0.02	0.04
Nov	412383	0.04	0.06
Dec	408050	0.04	0.05
	Annual total	0.03	0.06

Appendix.6 Emissions inventory

Table A. 7 Summary of carbon emissions for electricity and diesel in Mexico

Input	Description	Value	Unit	Reference
Electricity in Mexico	Carbon emission, value Mexico in 2016	0.458	TonneCO_2/MWh	Semarnat, 2016
		0.458	KgCO_2/kWh	
Diesel	Carbon emissions for Diesel RP Leon	3.138	KgCO_2/kgfuel	INE, 2014
	Carbon emission for Diesel RP Leon	2.602	kgCO_2/litre	INE, 2014
	Density of diesel in Leon, Mx	0.829	kg/litre	INE, 2014

Table A.8 Main chemicals and dosage in the UWS

Chemical	Dosage (kg/m^3)	Description
Chlorine gas	0.0007	1 kg Chlorine, gaseous {RoW}| chlor-alkali electrolysis, mercury cell | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)
Sodium hipochlorite		1 kg Sodium hypochlorite, without water, in 15% solution state {RoW}| sodium hypochlorite production, product in 15% solution state | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)
$FeCl_3$	0.0043	1 kg Iron (III) chloride, without water, in 40% solution state {RoW}| iron (III) chloride production, product in 40% solution state | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)
Sodium hipochlorite	0.0006	1 kg Sodium hypochlorite, without water, in 15% solution state {RoW}| sodium hypochlorite production, product in 15% solution state | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)

Table A.9 GWP, AcP, EuP and embodied energy in the production of electricity, chemicals and fuels

Input	GWP (kgCO2eq/fu)	Acidification (KgSO2eq/fu)	Eutrophication (kgPO4eq/fu)	Embodied energy	
				KWh/kg	MJ-eq
Electricity	0.485	0.001192051	0.000165661	0.83	3.000286
Diesel	3.138	0.004079401	0.000424531	11.99	43.15
Chlorine gas	1.405584703	0.010463507	0.002496691	5.19	18.69672
NaOCl	0.957889510	0.007020337	0.001935107	3.67	13.21975
Iron chloride	1.021345374	0.008046269	0.002572284	3.92	14.10305

Source: Ecoinvent database, SEMARNAT 2016 and INE, 2014

Table A.10 GWP, AcP, EuP and embodied energy in the allocation of electricity, chemicals and fuels

Production site[a]	Distance (km)[b]	Transport[c]	GHG emissions[d] $kgCO_2eq$		Acidification $kgSO_2eq$		Eutrophication $kgPO_4eq$		Embodied energy[e] kWh	
Chlorine gas			Factor	Estimated	Factor	Estimated	Factor	Estimated	Factor	Estimated
Monterrey, Mex	737.6	1 tkm Transport, freight, lorry 3.5-7.5 metric ton, EURO3 {RoW}| transport, freight, lorry 3.5-7.5 metric ton, EURO3 | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	0.504	372.026	0.002	1.749	0.001	0.424	2.148	1584.441
Leon-SFR	23	1 tkm Transport, freight, light commercial vehicle {GLO}| market for | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	1.949	44.833	0.009	0.196	0.002	0.053	8.243	189.589
Total per kg				0.417		0.002		0.0005		1.774
Sodium hipochlorite			Factor	Estimated	Factor	Estimated	Factor	Estimated		
Ecatepec, Mex	357.6	1 tkm Transport, freight, lorry 3.5-7.5 metric ton, EURO3 {RoW}| transport, freight, lorry 3.5-7.5 metric ton, EURO3 | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	0.504	0.180	0.002	0.001	0.001	0.000	2.148	768.162
Leon-SFR	23	1 tkm Transport, freight, light commercial vehicle {GLO}| market for | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	1.949	44.833	0.009	0.196	0.002	0.053	8.243	189.589
Total per kg				0.045		0.000		0.000		0.958
$FeCl_3$			Factor	Estimated	Factor	Estimated	Factor	Estimated		
Ecatepec, Mex	357.6	1 tkm Transport, freight, lorry 3.5-7.5 metric ton, EURO3 {RoW}| transport, freight, lorry 3.5-7.5 metric ton, EURO3 | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	0.504	0.180	0.002	0.001	0.001	0.205	2.148	768.162
Leon-SFR	23	1 kg Polyethylene, low density, granulate {RoW}| production | Alloc Def, S (of project Ecoinvent 3 - allocation, default - system)	1.949	44.833	0.009	0.196	0.002	0.053	8.243	189.589
Total per kg				0.045		0.000		0.000		0.958

a: Assuming Chlorine is produced in Monterrey oil Refinery, Mexico (data from IQUISA)
b: Distance was calculated using app SCT (mappir) for terrestrial distance
c: Chlorine gas is transported in 900 kg by road and then distributed in 68 kg cylinders by truck; ClO is transported in liquid from Mexico state to SFR
d: CML-IA baseline V3.01 / World 2000 from Ecoinvent
e: Conversion factor 1 MJ to 0.2777 kWh

Appendix.7 Typology of the potable water system

Table A.11 Connections and splits in the water supply system

Borehole	MaxCapacity (L/s)	WTW	Connection SR-WTW	Split_SR-WTW	NewID Tank	Reservoir flow	IDSubCat-SR	Split_SubCat-SR
1	0	1	1-1	0.000	1	153	1-1	0.39
2	25	2	1-2	0.163	1			
3	41	3	1-3	0.268	1			
4	87	4	1-4	0.569	1			
5	37	5	2-5	0.500	2	18.5	1-2	0.07
5	37	5	3-5	0.500	3	18.5	1-2	0.07
6	32	6	4-6	1.000	4	32	1-4	0.08
7	43	7	5-7	1.000	5	43	1-5	0.10
8	21	8	6-8	1.000	6	21	1-6	0.07
9	50	9	7-9	0.500	7	25	1-7	0.05
9	50	9	8-9	0.500	8	25	1-8	0.05
10	50	10	9-10	0.500	9	25	1-9	0.06
10	50	10	10-10	0.500	10	25	1-10	0.06
11	19	11	11-11	1.000	11	19	1-11	0.01
12	17	12	12-12	1.000	12	17	1-12	0.01

…continue Table A.11 Connections and splits in the water supply system

Borehole	Max Capacity (L/s)	WTW	Connection SR-WTW	Split_SR-WTW	Storage Tank	Reservoir flow	IDSubCat-SR	Split SubCat-SR
13	55.7	13	13-13	1.000	13	55.7	2-13	0.13
14	45.4	14	14-14	1.000	14	45.4	2-14	0.28
15	29.1	15	15-15	1.000	15	29.1	2-15	0.15
16	8.8	16	16-16	1.000	16	8.8	2-16	0.05
17	25.3	17	17-17	1.000	17	25.3	2-17	0.15
18	21.1	18	18-18	0.500	18	10.55	2-18	0.04
18	21.1	18		0.500	19	10.55	2-19	0.04
19	38.5	19	19-19	1.000	20	38.5	2-22	0.01
20	10.5	20	20-20	1.000	21	10.5	2-21	0.06
21	11.0	21	21-21	1.000	22	11	2-22	0.07
22	28.0	22	22-22	1.000	23	28	2-23	0.02

Appendix.8 Energy inputs in the transportation of reused water

Energy in water reuse transportation was calculated assuming the distribution of water reuse to three principal areas in each city. Zone A, B and C are located in SB1 and zones D, E, F in SB2. Distance and elevations were estimated from the maps using ArcGis 10.2. It was assumed that strategies at 20% only transport the water to the nearest zones (A and D), strategies at 50% transport the water to zone A, B, D, F and strategies at 100% transport to the six zones.

The head loss was calculated using the Hazen-Williams equation for head loss and then calculating the power.

$$hf = \frac{10.58 \times L \times Q^{1.85}}{C^{1.85} \times d^{4.87}}$$

Where hf is the head loss in metres (water) over the length of pipe, L is length of pipe (m), Q is the volumetric flow rate (m3/s), C is the pipe roughness coefficient (PEAD), d is the diameter of the pipe (m).

$$Power = \frac{d * g * h\ * Q}{n * 1000}$$

Where d: density in kg/m^3, g: gravity (m/s^2), n: pump efficiency, Q: flow (m^3/s) H: Head (m)

Table A.12 Head loss and energy calculation for reuse strategies

	C20	C50	C100
Distance from WWTW to reuse area			
Zone A	2,500.00	2,500.00	2,500.00
Zone B	0.00	0.00	1,500.00
Zone C	0.00	0.00	2,000.00
Zone D	2,700.00	2,700.00	2,700.00
Zone E	0.00	2,500.00	2,500.00
Zone F	0.00	0.00	1,500.00
Total distance (m)	2,580.00	3,580.00	6,280.00
Reuse flow			
m3/y	539,401.26	1,328,254.78	2,512,869.93
m3/h	123.15	202.17	382.48
m3/s	0.0342	0.0562	0.1062
Head Loss			
Flow (m^3/s)	0.034	0.056	0.106
Length (m)	2580.00	3580.00	6280.00
Diameter (m)	0.199	0.248	0.354
Diameter (in)	8	10	14
Coefficient (Pead, pvc)	140	140	140
hf (m)	14.59	17.44102	17.61084
Velocity check (m/s)	1.10	1.16	1.08
Power (kW)			
Gravity	9.81	9.81	9.81
hf (m)	14.59	17.44	17.61
Elevation	20.00	30.00	40.00
Height	28.00	38.00	48.00
Efficiency	0.80	0.80	0.80
Density	1000	1000	1000
Power (kW)	17.87	38.18	85.48
Power (KWh/m3)	0.1451	0.1888	0.2235

Continue Table A.12 Head loss and energy calculation for reuse strategies

	DW20	DW50	DW100
Zone A	1,800.00	1,800.00	1,800.00
Zone B	0.00	700.00	700.00
Zone C	0.00	0.00	3,000.00
Zone D	900.00	900.00	900.00
Zone E	0.00	2,500.00	2,500.00
Zone F	0.00	0.00	1,500.00
Total distance (m)	1,440.00	2,860.00	5,260.00
Reuse flow			
$m^{3/}y$	539,401.26	1,350,446.75	2,699,263.71
m^3/h	123.15	308.32	410.85
m^3/s	0.0342	0.0856	0.1141
Head Loss			
Flow (m^3/s)	0.034	0.086	0.114
Length (m)	1440.00	2860.00	5260.00
Diameter (m)	0.199	0.315	0.354
Diameter (in)	8	12	14
Coefficient	140	140	140
hf (m)	8.14	9.53	16.84060
Velocity check (m/s)	1.10	1.10	1.16
Power			
Gravity	9.81	9.81	9.81
hf (m)	8.14	9.53	16.84
Elevation	20.00	30.00	40.00
Height	28.00	38.00	48.00
Efficiency	0.80	0.80	0.80
Density	1000	1000	1000
Power (kW)	15.16	49.92	90.74
Power (KWh/m^3)	0.1231	0.1619	0.2209

Continue Table A.12 Head loss and energy calculation for reuse strategies

	DG20	DG50	DG100
Zone A	1,800.00	1,800.00	1,800.00
Zone B	0.00	700.00	700.00
Zone C	0.00	0.00	3,000.00
Zone D	900.00	900.00	900.00
Zone E	0.00	2,500.00	2,500.00
Zone F	0.00	0.00	1,500.00
Total distance (m)	1,440.00	2,860.00	5,260.00
Reuse flow			
m^3/y	369,143.51	924,722.86	1,848,994.87
m^3/h	84.28	211.12	281.43
m^3/s	0.0234	0.0586	0.0782
Head Loss			
Flow (m^3/s)	0.023	0.059	0.078
Length (m)	1440.00	2860.00	5260.00
Diameter (m)	0.160	0.248	0.315
Diameter (in)	6	10	12
Coefficient	140	140	140
hf (m)	11.87	15.10	14.80
Velocity check (m/s)	1.17	1.21	1.00
Power			
Gravity	9.81	9.81	9.81
hf (m)	11.87	15.10	14.80
Elevation	20.00	30.00	40.00
Height	28.00	38.00	48.00
Efficiency	0.80	0.80	0.80
Density	1000	1000	1000
Power (kW)	11.45	38.18	60.20
Power (KWh/m^3)	0.1358	0.1809	0.2139

Appendix.9 KPI

Table A.13 Summary of total KPI in all strategies and BAU

KPI's	BAU	C20	C50	C100	DW20	DW50	DW100	DG20	DG50	DG100
Potable water (m^3x10^6/y)	6.58	6.24	5.72	4.92	6.24	5.70	4.80	6.35	5.99	5.37
Energy (kWh/m^3)	1.15	1.12	1.08	1.02	1.15	1.15	1.16	1.15	1.15	1.15
GHG ($kgCO_2/m^3$)	0.84	0.83	0.81	0.78	0.82	0.80	0.75	0.83	0.83	0.80
Eu (gPO_4/m^3)	22.65	21.96	20.80	18.92	21.40	19.98	16.19	22.24	21.75	20.41
Ac (gSO_4/m^3)	10.38	10.39	10.37	10.33	9.84	9.24	7.58	10.25	10.10	9.65
BOD (Tonne/y)	275.25	266.29	251.83	229.62	257.73	237.54	186.49	270.88	265.20	250.64
COD (Tonne/y)	592.88	605.37	552.81	514.69	561.09	524.23	431.90	585.71	576.13	551.72
TN (Tonne/y)	121.10	116.43	108.64	95.96	114.36	106.67	86.03	120.68	119.88	117.56
TP (Tonne/y)	38.05	36.62	34.27	30.45	35.90	33.48	26.99	37.31	36.42	34.01
TSS (Tonne/y)	313.82	324.28	314.20	298.73	299.51	282.85	241.54	310.17	305.35	293.23

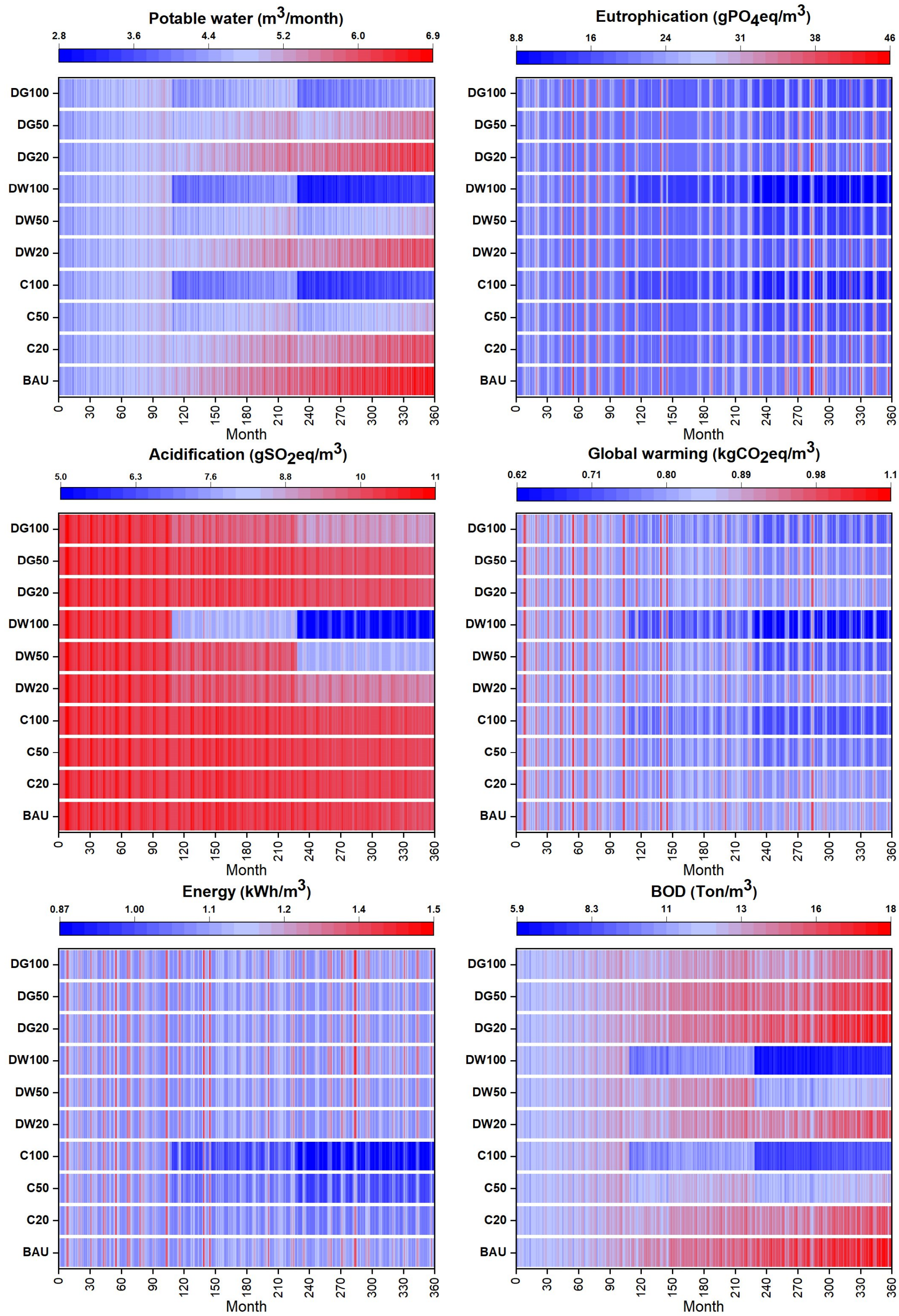

Figure A.1 Monthly results of six KPI's in all strategies and BAU. Blue colour indicates the lower values and red colour the maximum. X axis is the month over the planning horizon, Y axis is each of the strategies. From bottom to top is BAU, centralised, DW and DG from minor to major reuse adoption. Potable water is million of cubic meters.

Appendix.10 Results of water flows

Table A.14 Water flows in the UWS ($m^3 x 10^6/y$)

Source		BAU	C20	C50	C100	D20	D50	D100	DG20	DG2	DG3
WS	Water extraction	12.07	11.42	10.48	9.01	11.42	10.45	8.82	11.62	10.93	9.78
	Water demand	6.60	6.24	5.72	4.92	6.24	5.70	4.80	6.36	5.99	5.37
	Total Leakage	5.44	5.17	4.76	4.09	5.17	4.74	4.02	5.25	4.94	4.41
	Delivered potable	6.58	6.24	5.72	4.92	6.24	5.70	4.80	6.35	5.99	5.37
Sewer	Stormwater	3.44	3.44	3.44	3.44	3.44	3.44	3.44	3.44	3.44	3.44
	Sewage inflow	5.47	5.49	5.49	5.49	5.13	4.59	3.69	5.24	4.87	4.26
	Overflow	1.92	1.92	1.92	1.92	1.90	1.87	0.60	1.90	1.88	1.85
WWTW	Inflow	6.99	7.01	7.01	7.01	6.67	6.16	5.31	6.78	6.43	5.85
	Outflow	6.36	6.37	6.38	6.38	6.04	5.54	4.70	6.15	5.81	5.23
	Overflow	0.63	0.63	0.63	0.63	0.63	0.62	0.60	0.63	0.62	0.61
Receiving body	Treated	6.36	6.01	5.49	4.69	6.04	5.54	4.70	6.15	5.81	5.23
	Untreated	2.55	2.55	2.55	2.55	2.53	2.49	2.43	2.53	2.51	2.46
WR	Collected	0.00	0.36	0.89	1.69	0.36	0.90	1.80	0.25	0.62	1.23
	Delivered	0.00	0.36	0.89	1.69	0.36	0.90	1.80	0.25	0.62	1.23

WS: Water supply, WWTW: Wastewater treatment work; WR: Water reuse

Appendix.11 Contribution analysis of GWP

Table A.15 Contribution of inputs to global warming ($kgCO_2/m^3$)

Source		Emission	BAU		C20		C50		C100		DW20	
WS	Electricity	CO_2	0.34	(40.6%)	0.32	(38.9%)	0.30	(36.6%)	0.25	(32.6%)	0.32	(39.3%)
	Chem	CO_2	0.00	(0.27%)	0.00	(0.3%)	0.00	(0.2%)	0.00	(0.2%)	0.00	(0.3%)
WW	Treatment	CO_2	0.17	(19.8%)	0.17	(20.2%)	0.17	(20.6%)	0.17	(21.4%)	0.16	(19.3%)
	Fuel	CO_2	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
	Chem	CO_2	0.01	(1.0%)	0.01	(1.1%)	0.01	(1.1%)	0.01	(1.1%)	0.01	(1.0%)
	Biogas	CH_4	0.15	(18.4%)	0.15	(18.7%)	0.15	(19.1%)	0.15	(19.8%)	0.15	(17.9%)
	Renewable electricity	CO_2	- 0.01	-(1.6%)	- 0.01	-(1.6%)	- 0.01	-(1.6%)	- 0.01	-(1.7%)	- 0.01	-(1.5%)
	Sludge-landfill	CH_4	0.06	(7.0%)	0.06	(7.1%)	0.06	(7.3%)	0.06	(7.6%)	0.05	(6.7%)
	Sludge-landfill	N_2O	0.01	(1.1%)	0.01	(1.1%)	0.01	(1.1%)	0.01	(1.2%)	0.01	(1.0%)
	Sludge-fert	CH_4	0.08	(9.9%)	0.08	(10.0%)	0.08	(10.3%)	0.08	(10.7%)	0.08	(9.5%)
	Sludge-fert	N_2O	0.05	(5.5%)	0.05	(5.6%)	0.05	(5.8%)	0.05	(6.0%)	0.04	(5.3%)
	SSP	CO_2	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)
	Urea	CO_2	- 0.02	-(1.9%)	- 0.02	-(2.0%)	- 0.02	-(2.1%)	- 0.02	-(2.2%)	- 0.02	-(1.8%)
Rw	Electricity	CO_2	0.00	(0.0%)	0.01	(0.7%)	0.01	(1.5%)	0.03	(3.3%)	0.03	(3.2%)
UWS	Total	CO_2	0.84	(100%)	0.83	(100%)	0.81	(100%)	0.78	(100%)	0.82	(100%)

…continue Table A.15 Contribution of inputs to global warming (kgCO_2/m^3)

Source		Emission	DW50		DW100		DG20		DG50		DG100	
WS	Electricity	CO_2	0.30	(37.7%)	0.25	(33.2%)	0.33	(39.4%)	0.31	(38.0%)	0.28	(34.4%)
	Chem	CO_2	0.002	(0.3%)	0.002	(0.2%)	0.002	(0.3%)	0.002	(0.3%)	0.002	(0.2%)
WW	Treatment	CO_2	0.15	(18.5%)	0.12	(16.3%)	0.16	(19.3%)	0.15	(18.7%)	0.14	(17.1%)
	Fuel	CO_2	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
	Chem	CO_2	0.01	(1.0%)	0.01	(0.9%)	0.01	(1.0%)	01	(1.0%)	0.01	(0.9%)
	Biogas	CH_4	0.14	(17.1%)	0.11	(15.2%)	0.15	(17.9%)	0.14	(17.4%)	13	(15.8%)
	Ren electricity	CO_2	- 0.01	-(1.5%)	- 0.01	-(1.3%)	- 0.01	-(1.5%)	- 0.01	-(1.5%)	- 0.01	-(1.3%)
	Sludge-landfill	CH_4	0.05	(6.3%)	0.04	(5.3%)	0.06	(6.9%)	0.06	(6.9%)	0.05	(6.8%)
	Sludge-landfill	N_2O	0.01	(1.0%)	0.01	(0.8%)	0.01	(1.1%)	0.01	(1.1%)	0.01	(1.1%)
	Sludge-fert	CH_4	0.07	(9.0%)	0.06	(7.6%)	0.08	(9.8%)	0.08	(9.8%)	0.08	(9.6%)
	Sludge-fert	N_2O	0.04	(5.0%)	0.03	(4.2%)	0.05	(5.5%)	0.05	(5.5%)	0.04	(5.4%)
	SSP	CO_2	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)	- 0.00	-(0.1%)
	Urea	CO_2	- 0.01	-(1.8%)	- 0.01	-(1.5%)	- 0.02	-(1.9%)	- 0.02	-(1.9%)	- 0.02	-(1.9%)
Rw	Electricity	CO_2	0.06	(7.4%)	0.14	(19.1%)	0.02	(2.2%)	0.04	(4.9%)	0.10	(12.2%)
UWS	Total	CO_2	0.80	(100%)	0.75	(100%)	0.83	(100%)	0.83	(100%)	0.80	(100%)

Appendix.12 Contribution analysis of Eutrophication

Table A.16 Contribution of inputs to eutrophication (gPO_4/m^3)

Source		Emission	BAU		C20		C50		C100		DW20	
WS	Elec	PO_4	0.11	(0.5%)	0.11	(0.5%)	0.10	(0.5%)	0.08	(0.4%)	0.11	(0.5%)
	Chem	PO_4	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
SW	Overflow	TP	1.66	(7.3%)	1.68	(7.7%)	1.71	(8.2%)	1.74	(9.2%)	1.58	(7.4%)
		NO_3	0.02	(0.1%)	0.02	(0.1%)	0.02	(0.1%)	0.02	(0.1%)	0.02	(0.1%)
		COD	0.65	(2.9%)	0.65	(3.0%)	0.65	(3.1%)	0.65	(3.4%)	0.62	(2.9%)
WW	Elec	PO_4	1.08	(4.8%)	1.08	(4.9%)	1.08	(5.2%)	1.08	(5.7%)	1.03	(4.8%)
	Chem	PO_4	0.05	(0.2%)	0.05	(0.2%)	0.05	(0.3%)	0.05	(0.3%)	0.05	(0.2%)
	Discharge	TP	15.97	(70.5%)	15.29	(69.6%)	14.17	(68.2%)	12.36	(65.4%)	15.05	(70.3%)
	Discharge	NO_3	0.26	(1.1%)	0.24	(1.1%)	0.23	(1.1%)	0.20	(1.0%)	0.24	(1.1%)
	Discharge	COD	1.33	(5.9%)	1.28	(5.8%)	1.19	(5.7%)	1.06	(5.6%)	1.25	(5.8%)
	Ren electricity	PO_4	- 0.004	(0.0%)	- 0.004	(0.0%)	- 0.004	(0.0%)	- 0.004	(0.0%)	-.004	(0.0%)
	Sludge landfill	NH_3	1.60	(7.1%)	1.61	(7.3%)	1.62	(7.8%)	1.62	(8.6%)	1.50	(7.0%)
	SSP	PO_4	- 0.01	-(0.1%)	- 0.01	-(0.1%)	- 0.01	-(0.1%)	- 0.01	-(0.1%)	-0.01	-(0.1%)
	Urea	PO_4	-0.04	-(0.2%)	- 0.04	-(0.2%)	- 0.04	-(0.2%)	- 0.05	-(0.2%)	-0.04	-(0.2%)
Rw	Elec	PO_4	0	(0.0%)	0.002	(0.0%)	0.004	(0.0%)	0.009	(0.0%)	0.009	(0.0%)
UWS	Total	All	22.65	100%	21.96	100%	20.80	100%	18.92	100%	21.40	(100%)

…continue Table A.16 Contribution of inputs to eutrophication (gPO_4/m^3)

Source		Emission	DW50		DW100		DG20		DG50		DG100	
WS	Elec	PO_4	0.1	(0.5%)	0.08	(0.5%)	0.11	(0.5%)	0.10	(0.5%)	0.09	(0.4%)
	Chem	PO_4	0.0	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)	0.00	(0.0%)
SW	Overflow	TP	1.5	(7.4%)	1.27	(7.8%)	1.63	(7.3%)	1.59	(7.3%)	1.49	(7.3%)
		NO_3	0.0	(0.1%)	0.01	(0.1%)	0.02	(0.1%)	0.02	(0.1%)	0.02	(0.1%)
		COD	0.6	(2.9%)	0.51	(3.1%)	0.64	(2.9%)	0.63	(2.9%)	0.60	(2.9%)
WW	Elec	PO_4	1.0	(4.8%)	0.80	(4.9%)	1.04	(4.7%)	1.00	(4.6%)	0.89	(4.4%)
	Chem	PO_4	0.0	(0.2%)	0.04	(0.2%)	0.05	(0.2%)	0.05	(0.2%)	0.04	(0.2%)
	Discharge	TP	14.1	(70.5%)	11.28	(69.7%)	15.66	(70.4%)	15.29	(70.3%)	14.27	(69.9%)
	Discharge	NO_3	0.2	(1.1%)	0.18	(1.1%)	0.25	(1.1%)	0.25	(1.2%)	0.25	(1.2%)
	Discharge	COD	1.2	(5.8%)	0.93	(5.7%)	1.31	(5.9%)	1.29	(5.9%)	1.24	(6.1%)
	Ren electricity	PO_4	-0.004	(0.0%)	-0.003	(0.0%)	-0.004	(0.0%)	-0.004	(0.0%)	-0.004	(0.0%)
	Sludge landfill	NH_3	1.4	(7.0%)	1.10	(6.8%)	1.58	(7.1%)	1.56	(7.2%)	1.49	(7.3%)
	SSP	PO_4	-0.01	-(0.1%)	-0.01	-(0.1%)	-0.01	-(0.1%)	-0.01	-(0.1%)	-0.01	-(0.1%)
	Urea	PO_4	-0.04	-(0.2%)	-0.03	-(0.2%)	-0.04	-(0.2%)	-0.04	-(0.2%)	-0.04	-(0.2%)
Rw	Elec	PO_4	0.02	(0.1%)	0.05	(0.3%)	0.006	(0.0%)	0.013	(0.1%)	0.03	(0.2%)
UWS	Total	All	19.98	(100%)	16.19	(100%)	22.24	(100%)	21.75	(100%)	20.41	(100%)

Appendix.13 Contribution analysis of Acidification

Table A.17 Contribution of inputs to acidiphication (gSO_2/m^3)

Source		Emission	BAU		C20		C50		C100		DW20	
WS	Elec	SO_2	0.89	(8.53%)	0.84	(8.07%)	0.77	(7.42%)	0.66	(6.40%)	0.84	(8.52%)
	Chem	SO_2	0.02	(0.15%)	0.01	(0.14%)	0.01	(0.13%)	0.01	(0.11%)	0.01	(0.15%)
WW	Elec	SO_2	0.43	(4.17%)	0.43	(4.18%)	0.43	(4.19%)	0.43	(4.20%)	0.41	(4.18%)
	Chem	SO_2	0.06	(0.59%)	0.06	(0.59%)	0.06	(0.59%)	0.06	(0.59%)	0.06	(0.59%)
	Biogas	H_2S	0.69	(6.67%)	0.69	(6.68%)	0.69	(6.69%)	0.69	(6.72%)	0.66	(6.68%)
	Ren EE	SO_2	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.33%)
	Sludge	NH_3	8.59	(82.79%)	8.64	(83.14%)	8.68	(83.68%)	8.73	(84.44%)	8.08	(82.09%)
	SSP	SO_2	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)
	Urea	SO_2	-0.26	-(2.54%)	-0.27	-(2.58%)	-0.27	-(2.65%)	-0.28	-(2.75%)	-0.25	-(2.53%)
Rw	Elec	SO_2	0.00	(0.00%)	0.01	(0.14%)	0.03	(0.31%)	0.07	(0.66%)	0.07	(0.69%)
UWS	Total	SO_2	10.38	(100%)	10.39	(100%)	10.37	(100%)	10.33	(100%)	9.84	(100%)

…continue Table A.17 Contribution analysis to Acidification (gSO_2/m^3)

Source		Emission	DW50		DW100		DG20		DG50		DG100	
WS	Elec	SO_2	0.79	(8.51%)	0.65	(8.56%)	0.85	(8.31%)	0.82	(8.07%)	0.82	(8.45%)
	Chem	SO_2	0.01	(0.15%)	0.01	(0.15%)	0.02	(0.15%)	0.01	(0.15%)	0.01	(0.13%)
WW	Elec	SO_2	0.39	(4.18%)	0.32	(4.21%)	0.42	(4.08%)	0.40	(3.98%)	0.36	(3.68%)
	Chem	SO_2	0.05	(0.59%)	0.04	(0.59%)	0.06	(0.57%)	0.06	(0.56%)	0.05	(0.52%)
	Biogas	H_2S	0.62	(6.69%)	0.51	(6.73%)	0.67	(6.52%)	0.64	(6.36%)	0.57	(5.89%)
	Ren EE	SO_2	-0.03	-(0.33%)	-0.03	-(0.33%)	-0.03	-(0.32%)	-0.03	-(0.31%)	-0.03	-(0.29%)
	Sludge	NH_3	7.50	(81.08%)	5.89	(77.69%)	8.49	(82.82%)	8.36	(82.78%)	7.98	(82.71%)
	SSP	SO_2	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)	0.00	-(0.03%)
	Urea	SO_2	-0.23	-(2.52%)	-0.19	-(2.48%)	-0.26	-(2.56%)	-0.26	-(2.59%)	-0.26	-(2.66%)
Rw	Elec	SO_2	0.15	(1.67%)	0.37	(4.91%)	0.05	(0.46%)	0.11	(1.04%)	0.25	(2.63%)
UWS	Total	SO_2	9.24	(100%)	7.58	(100%)	10.25	(100%)	10.10	(100%)	9.65	(100%)

Appendix.14 Sensitivity analysis results

Table A.18 Summary of sensitivity indexes

Parameter	GWP	EuP	AcP
TSS_{HB}	0.006	0.019	0.046
TSS_{Ki}	0.026	0.077	0.179
TSS_{WM}	0.036	0.102	0.222
TSS_{Sh}	0.014	0.041	0.098
TSS_{TO}	0.151	0.393	0.737
TSS_{Ind}	0.035	0.101	0.228
TP_{HB}	0.000	0.023	0.000
TP_{Ki}	0.000	0.102	0.000
TP_{WM}	0.000	0.223	0.000
TP_{Sh}	0.000	0.094	0.000
TP_{TO}	0.000	0.259	0.000
TP_{Ind}	0.000	0.218	0.000
TN_{HB}	0.001	0.000	0.000
TN_{Ki}	0.001	0.000	0.000
TN_{WM}	0.000	0.000	0.000
TN_{Sh}	0.001	0.000	0.000
TN_{TO}	0.013	0.007	0.004
TN_{Ind}	0.002	0.001	0.001
COD_{HB}	0.000	0.003	0.000
COD_{Ki}	0.000	0.007	0.000
COD_{WM}	0.000	0.010	0.000
COD_{Sh}	0.000	0.003	0.000
COD_{To}	0.000	0.037	0.000
COD_{Ind}	0.000	0.015	0.000
BOD_{HB}	0.000	0.000	0.000
BOD_{Ki}	0.000	0.000	0.000
BOD_{WM}	0.000	0.000	0.000
BOD_{Sh}	0.000	0.003	0.000
BOD_{To}	0.000	0.037	0.000
BOD_{Ind}	0.000	0.015	0.000

Table A.19 Summary of statistics, sensitivity indexes and KPI studied related to total suspended solids

	Mean	S.D.	C.V.	SI
TSS_{HB}	296.28	161.92	54.65%	-
GWP	9,683.89	34.34	0.36%	0.006
EuP	239.01	2.52	1.06%	0.019
AcP	543.54	13.55	2.49%	0.046
TSS_{Ki}	717.48	339.66	47.34%	-
GWP	9,719.84	120.05	1.24%	0.026
EuP	241.65	8.82	3.65%	0.077
AcP	557.73	47.39	8.50%	0.179
TSS_{WM}	943.03	531.62	56.37%	-
GWP	9,905.43	200.43	2.02%	0.036
EuP	255.29	14.73	5.77%	0.102
AcP	630.98	79.11	12.54%	0.222
TSS_{Sh}	255.78	144.81	56.62%	-
GWP	9,654.21	75.07	0.78%	0.014
EuP	236.83	5.52	2.33%	0.041
AcP	531.83	29.63	5.57%	0.098
TSS_{TO}	2,093.65	959.89	45.85%	-
GWP	10,449.93	723.78	6.93%	0.151
EuP	295.30	53.18	18.01%	0.393
AcP	845.90	285.68	33.77%	0.737
TSS_{Ind}	599.86	232.60	38.78%	-
GWP	9,795.01	131.65	1.34%	0.035
EuP	247.17	9.67	3.91%	0.101
AcP	587.40	51.96	8.85%	0.228

S.D. Standard Deviation; C.V. Coefficient of variation; S.I. Sensitivity index. GWP is in Thousands of Tons CO_2eq/y; EuP is in TonPO_4eq/y and AcP is in Ton SO_2eq/y.

Table A.20 Summary of statistics, sensitivity indexes and KPI studied related to total phosphorus

	Mean	S.D.	C.V.	SI
TP_{HB}	13.65	7.22	0.53	-
GWP	9,625.13	0.12	0.00	0.0000
EuP	238.86	2.90	0.01	0.0229
AcP	520.41	0.00	0.00	0.0000
TP_{Ki}	38.33	20.85	0.54	-
GWP	9,624.67	0.56	0.00	0.0001
EuP	250.31	13.94	0.06	0.1024
AcP	520.41	0.00	0.00	0.0000
TP_{WM}	85.32	49.68	0.58	-
GWP	9,623.73	1.43	0.00	0.0003
EuP	273.46	35.45	0.13	0.2226
AcP	520.41	0.01	0.00	0.0000
TP_{Sh}	24.36	14.25	0.59	-
GWP	9,624.53	0.56	0.00	0.0001
EuP	253.69	13.98	0.06	0.0942
AcP	520.41	0.00	0.00	0.0000
TP_{TO}	50.53	20.65	0.41	-
GWP	9,623.54	1.19	0.00	0.0003
EuP	278.26	29.47	0.11	0.2591
AcP	520.41	0.01	0.00	0.0000
TP_{Ind}	50.05	17.53	0.35	-
GWP	9,624.86	0.76	0.00	0.0002
EuP	245.47	18.78	0.08	0.2185
AcP	520.41	0.00	0.00	0.0000

S.D. Standard Deviation; C.V. Coefficient of variation; S.I. Sensitivity index. GWP is in Thousands of Tons CO2eq/y; EuP is in TonPO4eq/y and AcP is in Ton SO2eq/y.

Table A.21 Summary of statistics, sensitivity indexes and KPI studied related to total nitrogen

	Mean	S.D.	C.V.	SI
TN_{HB}	53.88	29.95	55.59%	-
GWP	9,619.62	3.49	0.04%	0.0007
EuP	234.78	0.05	0.02%	0.0004
AcP	520.32	0.06	0.01%	0.0002
TN_{Ki}	39.79	19.95	50.14%	-
GWP	9,629.22	3.87	0.04%	0.0008
EuP	234.65	0.06	0.02%	0.0005
AcP	520.48	0.06	0.01%	0.0002
TN_{WM}	20.71	11.41	55.08%	-
GWP	9,624.11	2.36	0.02%	0.0004
EuP	234.72	0.03	0.01%	0.0003
AcP	520.39	0.04	0.01%	0.0001
TN_{Sh}	26.86	13.59	50.61%	-
GWP	9,621.92	3.87	0.04%	0.0008
EuP	234.75	0.06	0.02%	0.0005
AcP	520.36	0.06	0.01%	0.0002
TN_{TO}	296.42	114.27	38.55%	-
GWP	9,573.00	47.27	0.49%	0.0128
EuP	235.45	0.68	0.29%	0.0074
AcP	519.55	0.78	0.15%	0.0039
TN_{Ind}	62.50	21.84	34.94%	-
GWP	9,621.41	6.78	0.07%	0.0020
EuP	234.76	0.10	0.04%	0.0012
AcP	520.35	0.11	0.02%	0.0006

S.D. Standard Deviation; C.V. Coefficient of variation; S.I. Sensitivity index. GWP is in Thousands of Tons CO2eq/y; EuP is in TonPO4eq/y and AcP is in Ton SO2eq/y.

Table A.22 Summary of statistics, sensitivity indexes and KPI studied related to chemical oxygen demand

	Mean	S.D.	C.V.	SI
COD_{HB}	848.35	374.77	44.18%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	235.23	0.31	0.13%	0.0030
AcP	520.41	0.00	0.00%	0.0000
COD_{Ki}	1,135.43	647.05	56.99%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	235.03	0.89	0.38%	0.0067
AcP	520.41	0.00	0.00%	0.0000
COD_{WM}	1,588.50	794.60	50.02%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	236.45	1.17	0.49%	0.0099
AcP	520.41	0.00	0.00%	0.0000
COD_{Sh}	375.99	161.85	43.05%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	234.86	0.33	0.14%	0.0032
AcP	520.41	0.00	0.00%	0.0000
COD_{To}	3,030.23	1240.26	40.93%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	237.73	3.64	1.53%	0.0374
AcP	520.41	0.00	0.00%	0.0000
COD_{Ind}	1,651.82	787.63	47.68%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	236.69	1.74	0.73%	0.0154
AcP	520.41	0.00	0.00%	0.0000

S.D. Standard Deviation; C.V. Coefficient of variation; S.I. Sensitivity index. GWP is in Thousands of Tons CO2eq/y; EuP is in TonPO4eq/y and AcP is in Ton SO2eq/y.

Table A.23 Summary of statistics, sensitivity indexes and KPI studied related to biochemical oxygen demand

	Mean	S.D.	C.V.	SI
BOD_{HB}	319.81	161.78	50.59%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	234.70	0.00	0.00%	0.0000
AcP	520.41	0.00	0.00%	0.0000
BOD_{Ki}	875.26	340.74	38.93%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	234.70	0.00	0.00%	0.0000
AcP	520.41	0.00	0.00%	0.0000
BOD_{WM}	703.98	381.49	54.19%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	234.70	0.00	0.00%	0.0000
AcP	520.41	0.00	0.00%	0.0000
BOD_{Sh}	376.21	162.34	43.15%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	234.86	0.33	0.14%	0.0032
AcP	520.41	0.00	0.00%	0.0000
BOD_{To}	3,033.92	1241.05	40.91%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	237.74	3.64	1.53%	0.0375
AcP	520.41	0.00	0.00%	0.0000
BOD_{Ind}	1,651.23	787.56	47.70%	-
GWP	9,625.30	0.00	0.00%	0.0000
EuP	236.69	1.74	0.73%	0.0154
AcP	520.41	0.00	0.00%	0.0000

S.D. Standard Deviation; C.V. Coefficient of variation; S.I. Sensitivity index. GWP is in Thousands of Tons CO2eq/y; EuP is in TonPO4eq/y and AcP is in Ton SO2eq/y.