Assessing the environmental impacts of a complex urban water system based on the life cycle assessment framework: development of a versatile model and advanced water deprivation indicators

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Merci, merci, merci à vous tous, vous me rendez aphone

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Acronyms and abbreviations

AC: Acidification

BOD: Biological oxygen demand

C: Consumption (also noted WC – water consumption – in Chapter 3)

CC: Climate change

CF: Characterization factor

CED: Cumulative energy demand

COD: Chemical oxygen demand

CTA: Consumption-to-availability

D: Discharge

DS: Damage score

DWP: Drinking water production

DWD: Drinking water distribution

EE: Eco-efficiency

EQ: Ecosystem quality

ET: Evapotranspiration

EWR: Environmental water requirements

FET: Freshwater ecotoxicity

FEu: Freshwater eutrophication

FU: Functional unit

HH: Human health

HT: Human toxicity

I: Impact

IS: Impact/service

IR: Ionizing radiation

IUWM: Integrated urban water management

IWRM: Integrated water resource management

LCA: Life cycle assessment

LCI: Life cycle inventory

LCIA: Life cycle impact assessment

MEu: Marine eutrophication

OOP: Object-oriented programming

P: Precipitation

PAF: Potentially affected fraction

PDF: Potentially disappeared fraction

PNOF: Potentially not occurring fraction

R: Release (also noted WR – water release – in Chapter 3)

RO: Runoff

S: Services

SD: Species density

SEDIF: Syndicat des Eaux d'Île-de-France

SEOL: Sludge end of life

SIAAP: Syndicat Interdépartemental pour l'Assainissement de l'Agglomération Parisienne

SRB: Sub-river basin

SWC: Stormwater collection

TEu: Terrestrial eutrophication

U: User

UWS: Urban water system

V: Water volume

W: Withdrawal (also noted WW – water withdrawal – in Chapter 3)

WA: Water availability

WD: Water deprivation

WFD: Water framework directive

WIIX: Water impact index

WTA: Water-to-availability

WWC: Wastewater collection

WWT: Wastewater treatment

Preface

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The thesis is essentially based on the following papers, which have either been published, or submitted in international peer-reviewed journals:

- Loubet, P., Roux, P., Núñez, M., Belaud, G., & Bellon-Maurel, V. (2013). Assessing Water Deprivation at the Sub-river Basin Scale in LCA Integrating Downstream Cascade Effects. *Environmental Science & Technology*, *47*(24), 14242–9. doi:10.1021/es403056x
- Loubet, P., Roux, P., Loiseau, E., & Bellon-Maurel, V. (2014). Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature. *Water Research*, *67*(0), 187–202. doi:10.1016/j.watres.2014.08.048
- Loubet, P., Roux, P. & Bellon-Maurel, V. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 1 – formalism & framework for a modular approach. *Submitted to Water Research*
- Loubet, P., Roux, P., Guerin-Schneider L. & Bellon-Maurel, V. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 2 – Learning points from the assessment of water management scenarios in Paris suburban area. *Submitted to Water Research*

The work included in the thesis was presented in oral communications and posters in international conferences:

- Loubet, P., Bayart, J., & Danel, P. (2011). Measuring the Water Impact Index of water services. In *Ecotech & Tools*. Montpellier, France.

- Loubet, P., Roux, P., Nunez, M., & Bellon-Maurel, V. (2013). Assessing water deprivation at sub-river basin scale in LCA integrating downstream cascade effects. In *SETAC Europe 23rd Annual Meeting*. Glasgow, UK.
- Loubet, P., Roux, P., & Bellon-Maurel, V. (2014). Modelling technique for territorial LCA applied to urban water systems : evaluation of prospective scenarios in mega cities. In *SETAC Europe 24th Annual Meeting*. Basel, Switzerland.
- Loubet, P., Roux, P., Nunez, M., & Bellon-Maurel, V. (2014). Sub-river basin scale water deprivation at midpoint and endpoint levels in LCIA. In *SETAC Europe 24th Annual Meeting*. Basel, Switzerland.

Chapter 1. General introduction

« 361 degrés de rotation, du rien au tout, et puis du tout au rien.

Juste que nous ne sommes rien du tout, en fait on sait rien, c'est tout »

Akhenaton – Mon texte, le savon

Content of Chapter 1

1.1. Towards sustainable cities: the challenge of urban water systems (UWS)

Since the yearly 1970's, the mankind has raised awareness about the natural environment vulnerability. In its famous report, the club of Rome warned about the finite natural resources and discussed the limits to growth (Meadows et al., 1972). Indeed, in a biophysical system with finite resources, it is impossible for an economy based on these resources to grow infinitely. From these alarming signals, new concepts have risen. Among them, the sustainable development posits a desirable future state for human societies in which living conditions and resource-use meet human needs without undermining the sustainability of natural systems and the environment, so that future generations may also have their needs met (Brundtland, 1987). More radical concepts, such as the "degrowth", question the idea of development and propose a window of opportunity for political changes that will make the inevitable economic recession socially and environmentally sustainable (Kallis, 2011).

In this context of transition, the key role of cities was emphasized (Beck, 2011). After a twentieth century marked by considerable rural flight, the world has never been that urbanized. The world's population has reached 7 billion, and more people live in cities than in rural areas (United Nations, 2012). Megacities, defined as a metropolitan area with a total population in excess of 10 million people are becoming more and more common. As of today, there are 30 megacities in existence (Population Reference Bureau, 2013) and some of them are or will be facing acute problems, particularly related to water (Abderrahman, 2000). The urban sprawl poses challenges for urban planners, as it causes congestion, environmental degradation and increases the cost of service delivery (UN-Habitat, 2009). There is a need to rethink and modify the standards and principles for urban planning.

To meet the water challenges at the city scale, the integrated urban water management framework (IUWM) has been developed (Global Water Partnership Technical Committee, 2012). It aims at improving water management for different purposes within the urban area both in terms of quality and quantity. Nested within the broader framework of integrated water resources management (IWRM) (Global Water Partnership Technical Advisory Committee, 2000), it can contribute to meet water challenges at a river basin scale. By doing so, the IUWM framework enable stakeholders to look at the system holistically and facilitate the development of innovative solutions for urban water management. However, there is still room in the framework for tools and methods that can help managers to evaluate urban water system (UWS) sustainability and prospective scenarios. Measuring all environmental impacts associated with human activities is a necessary condition to reduce their footprint. Amongst the available tools for assessing environmental impacts of such systems, life cycle assessment (LCA) has already proven its worth. The LCA method is explained in the following section by underlining briefly its main forces and its limitations for the assessment of urban water systems.

1.2. To measure is to know: introduction to life cycle assessment (LCA)

LCA is a standardized approach for environmental evaluation (ISO, 2006a) and is widely recognized at world wide scale. This tool quantifies impacts of a product or a service within all its life cycle stages, i.e., from cradle-to-grave. It includes raw material extraction through materials processing, manufacture, distribution, use, repair and maintenance, and disposal or recycling. LCA is a multi-criteria approach that takes into account a wide range of impacts to the environment (e.g., climate change, eutrophication, resources depletion, etc.) and differs in this way from other tools such as carbon footprint, energy balance, as shown in Figure 1-1. The holistic nature of LCA allows identifying pollution shifting between impact categories, between life cycle stages or between different locations (Finnveden et al., 2009).

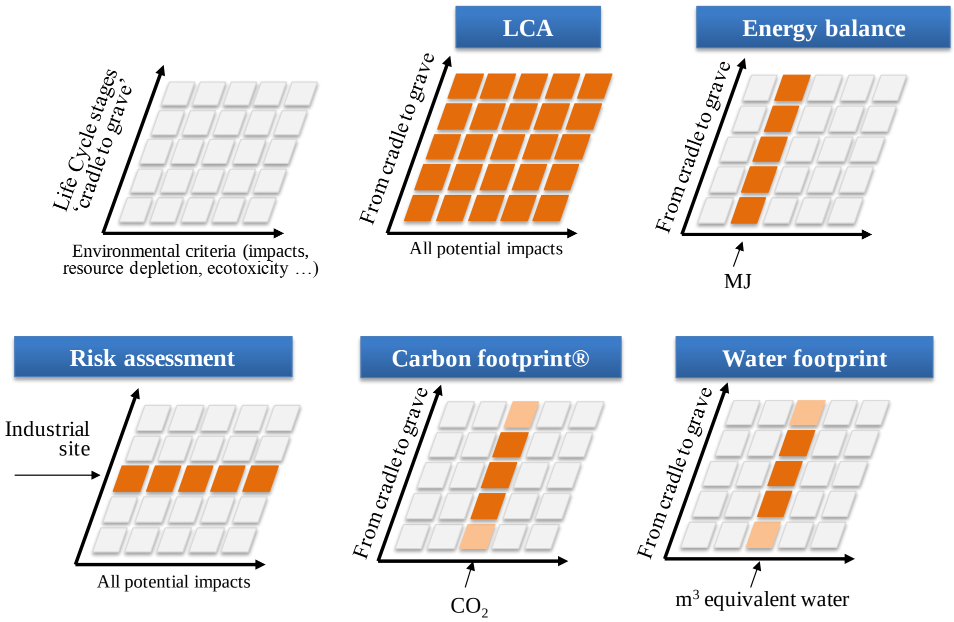


Figure 1-1. General criteria and life cycle stages from different environmental evaluation methodologies. Adapted from Risch et al. (2012)

LCA provides two main types of indicators. Midpoint indicators assess a change in the environment (an environmental mechanism) that links a human intervention (e.g., emission of CO_2) to a problem (e.g., global warming potential). Generally, endpoint indicators quantify the damages on areas of protection, generally human health, ecosystem quality and resources, due to these problems. These relations are described by cause-effect pathways. A representation of major pathways is shown in Figure 1-2.

Another LCA key feature is that it is based on a functional approach: potential impacts of a product or a service are quantified per unit of provided service, namely the functional unit. For a given service (e.g., "to participate to a meeting"), it allows to compare contrasted systems (e.g., train, car and videoconferencing).

LCA was initially developed according to a product-oriented approach, with the aim to bring information on goods and services to the public (eco-labeling), to decision makers or to industries for eco-design purpose. Recent proposals have been made to adapt the LCA framework in order to broaden its scope towards larger scale systems such as cities (Loiseau et al., 2013). This is a relevant scale to assess environmental impacts of urban water systems. However, LCA studies can be time consuming and their application to large systems such as megacity UWS requires a huge amount of data. In addition to diagnosis purposes, the evaluation of forecasting scenarios would also require important modeling efforts. Therefore, in line to the analysis of (Schulz et al., 2012), there is a great need for developing simplified procedures to easily provide stakeholders indicators about the environmental performance of UWS and their forecasting scenarios.. This means creating new procedures for modeling UWS, in order to easily feed LCA analysis.

In addition to methodological needs in terms of UWS modeling for matching LCA models, another challenge in LCA applied to UWS is the assessment of water use impacts. Water is both a resource and an environmental compartment, and its consideration within environmental evaluation raises some challenges, today unresolved, as pointed out hereafter.

1.3. Water in environmental evaluations

1.3.1. Water, a unique resource and a sensitive environmental habitat

Water has this specific property to be both a resource for humans and an environmental habitat, explaining the many concerns we place on this "blue gold". Of course, water is not as scarce as gold. On the contrary, it is a renewable resource and water moves continually on earth through a cycle. There are approximately 1,400,000,000 km^3 of water on earth but only 3% is freshwater; of which 69% is locked up in glaciers and snow (Oki and Kanae, 2006). The remaining water is usable for human but it is poorly distributed within the world. More than 2.5 billion people face water scarcity during at least one month of the year (Hoekstra et al., 2012), meaning that sufficient available water resources are lacking for meeting demands of water usages. Human interventions exacerbate the situation. This is principally due to agriculture that is responsible of 70% of water withdrawals, whereas domestic users are 12% and industrial users 18% (FAO, 2012). The future is not bright, as climate change and population growth tend to increase this threat (Vorosmarty, 2000). Besides the issue of quantity, the limited access to water is also linked to water quality. Degradation of water quality leads to unavailable water resources for certain usages (Peters and Meybeck, 2000).

Freshwater is also an environmental habitat that can be affected by water scarcity and pollutions. In terms of biological value, rivers contain a rich and varied biota, i.e., at least 100,000 species, almost 6% of all described species (Dudgeon et al., 2006). Ecosystem destruction due to water abstraction, habitat alteration incurred by damming or water transferring, changes in water chemistry because of pollutions, and species removal and additions are the main disturbances from anthropogenic activities (Malmqvist and Rundle, 2002).

All these concerns show the importance of assessing impacts of water use and pollution on species, i.e., on ecosystems and human health. Such methods have been increasingly developed as shown hereafter.

1.3.2. Water footprint and water in LCA

In the beginning of the 2000's, the concepts of "virtual water" and "water footprint" have been developed in order to account for these water issues in supply chains (Allan, 1998; Hoekstra and Hung, 2002). It accounts for the withdrawal of surface and ground water (named "blue water"), the evapotranspiration of rainwater (named "green water") and the pollution of freshwater (named "grey water"). It results in amounts of equivalent cubic meters needed to produce the targeted goods or services: for example one kilogram of beef represents 15 400 L of water (Mekonnen and Hoekstra, 2010) or one kilogram of coffee almost 19 000 L (Mekonnen and Hoekstra, 2011). These considerable amounts raise awareness in the public. However, the interpretation of this volumetric approach is questionable. For example, one cubic meter of water transpired in a wet area (e.g. Scotland) is not equivalent to one cubic meter of water consumed in a dry one (e.g. in the Colorado). In addition, the quantification of pollution in “grey-water” is based on a dilution volume approach, which does not consider substance fate, contrarily to what life cycle impact assessment (LCIA) models do.

Alternatively to the virtual water concept, LCA characterizes the inventory data in order to quantify potential impacts and damages to the environment. Originally, LCA assesses only impacts and damages on aquatic ecosystems through the categories freshwater eutrophication, and ecotoxicity. The assessment of water use is at an early development stage but new methods are currently developed and certain ones are operational (Kounina et al., 2012). In this context, the recently developed water footprint standard (ISO, 2013) states that a water footprint profile should be presented as a compilation of LCIA results related to water: water use, eutrophication, freshwater ecotoxicity, etc. (Figure 1-2).

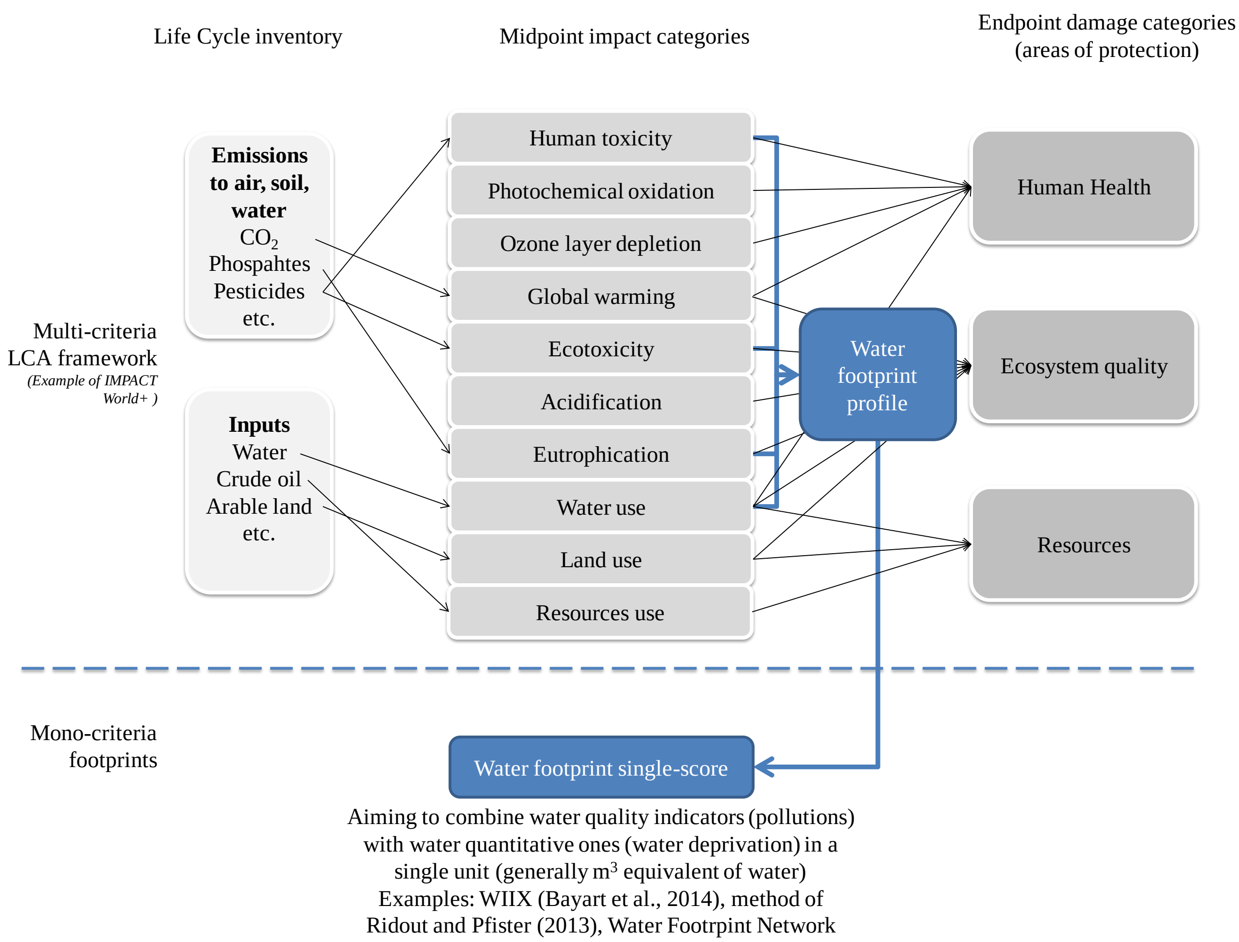


Figure 1-2. Main impact pathways in LCA and presentation of the water footprint profile and single-score. Adapted from Impact World+ (http://www.impactworldplus.org/en/index.php) and Boulay et al. (2014)

The application of these approaches to UWS is highly needed as urban systems play a key role in the water management at the scale of river basin, and there is a need to assess properly their impacts related to water for mitigation purposes. In terms of quality, UWS have a significant role to ensure good quality of the rivers (Niemczynowicz, 1999). In terms of quantity, even if the urban systems is often not the main water consumer compared to irrigated agriculture, it may have an influence on water deprivation; the variety of water sources that the UWS can use in a river basin, as well as the distance between withdrawals and releases points lead to different water deprivation levels (Jønch-Clausen and Fugl, 2001). However, several challenges remain in the evaluation of water deprivation impacts. The main one is that the current scale for assessing water deprivation is the river basin (Pfister et al., 2009), which is not appropriate to the evaluation of urban water systems that can use many water sources within a same river basin and that typically release water far from the withdrawal points.

1.4. Objectives of the thesis

With the aim to address the urgent need for tools to easily supply stakeholders with indicators about the environmental performance of UWS and forecasting scenarios, the research question of this thesis is:

"Is it possible to model a urban water system in order to assess the environmental impacts it induces in regards with services provided to the users, using the conceptual framework of LCA ?"

This global question is approached through two axes, each one related to a crucial phase of LCA. In the goal & scope and life cycle inventory (LCI) phases, the question is: "how to model the UWS of big cities, in order to be at the same time, simple to implement, representative of a given UWS scenario, and compliant to LCA specifications?" In the LCIA phase, the question is: "regarding the fact that UWS will have major qualitative and quantitative effects on the water compartment, how to better take this effects into account?" Following these two axes, five sub-objectives are defined:

1. **Identifying the main methodological challenges** related to LCA applied to urban water systems and demonstrate the need for a standardized approach.
2. **Refining the impact category related to water deprivation, at an appropriate scale,** in order to make it applicable and relevant for urban water systems.
3. **Accounting for quality of urban water flows** taking into account existing LCIA and water footprint methods

4. **Developing a model** and an associated formalism that reduces the complexity of the system and that is versatile enough to implement forecasting scenarios, while being still relevant for life cycle assessment.
5. **Demonstrating the capacity of the model to address stakeholder's expectations** when evaluating forecasting scenarios.

Each sub-objective is addressed by a chapter of the thesis referring to a scientific publication (either published or submitted), as described hereafter and summarized in Figure 1-3.

After the introduction (**Chapter 1**)**, Chapter 2** is a review which aims at comparing papers dealing with LCA of the entire UWS (including drinking water production and distribution, as well as wastewater collection and treatment): 18 different case studies have been found. It is based on a compilation and analysis of LCA results for urban water systems, and it ends up by the identification of several guidelines for streamlining LCA of UWS and of methodological challenges for the future.

From the guidelines and challenges pointed out within the review chapter, **Chapter 3** and **Chapter 4** propose original approaches in order to better take into account water-related impacts in urban water system (respectively the quantitative and qualitative aspects).

More specifically, **Chapter 3** presents the development of a methodology to assess water deprivation issues at the sub-river basin scale in LCA integrating "downstream cascade effects", i.e. effects of withdrawals on downstream users and ecosystems. Following the present framework used to assess impacts of water deprivation, this method differentiates the withdrawal and release points within a river basin. It is based on a two-steps approach that first defines the "local water scarcity" at the sub-river basin scale and, second, computes water deprivation for downstream users. The methodology is then validated on two different river basins. Whereas Chapter 3 focuses on the quantitative impact of water use, **Chapter 4** reviews current approaches to assess the qualitative impacts of water use. It aims at assessing the damage scores of the different water flows found within the UWS, and to classify these flows.

The development of these methods is a prerequisite for the development of the UWS model which is presented in **Chapter 5**, i.e., the core of the thesis. This model, named WaLA (for Water system Lifecycle Assessment), is elaborated to tackle the methodological issues of LCA applied to UWS, which have been pointed out in chapter 2. It integrates the developments described in chapters 3 and 4. The model is based on a formalism which defines a generic component that characterize both water users and water technologies. These components can be interconnected and interoperated, and are linked to water resources. This enables to build a representation of a UWS scenarios through a modular approach. The model is implemented within a Matlab/Simulink user-friendly interface. It computes environmental impacts induced by the system, as well as services provided to the users. It is tested on a theoretical case study.

Chapter 6 is the application of the model to forecasting scenarios. It aims at verifying the capacity of the versatile model to assess scenarios and address stakeholders' questions. The chosen case study is Paris suburban area. Several scenarios related to changes of water users, water resources and water technologies are studied.

Finally, a discussion about the two main outcomes of the thesis, i.e. (i) the LCIA model for assessing water deprivation at the sub-river basin scale, and (ii) the WaLA model for the LCA of UWS and its perspectives, is provided in **Chapter 7**.

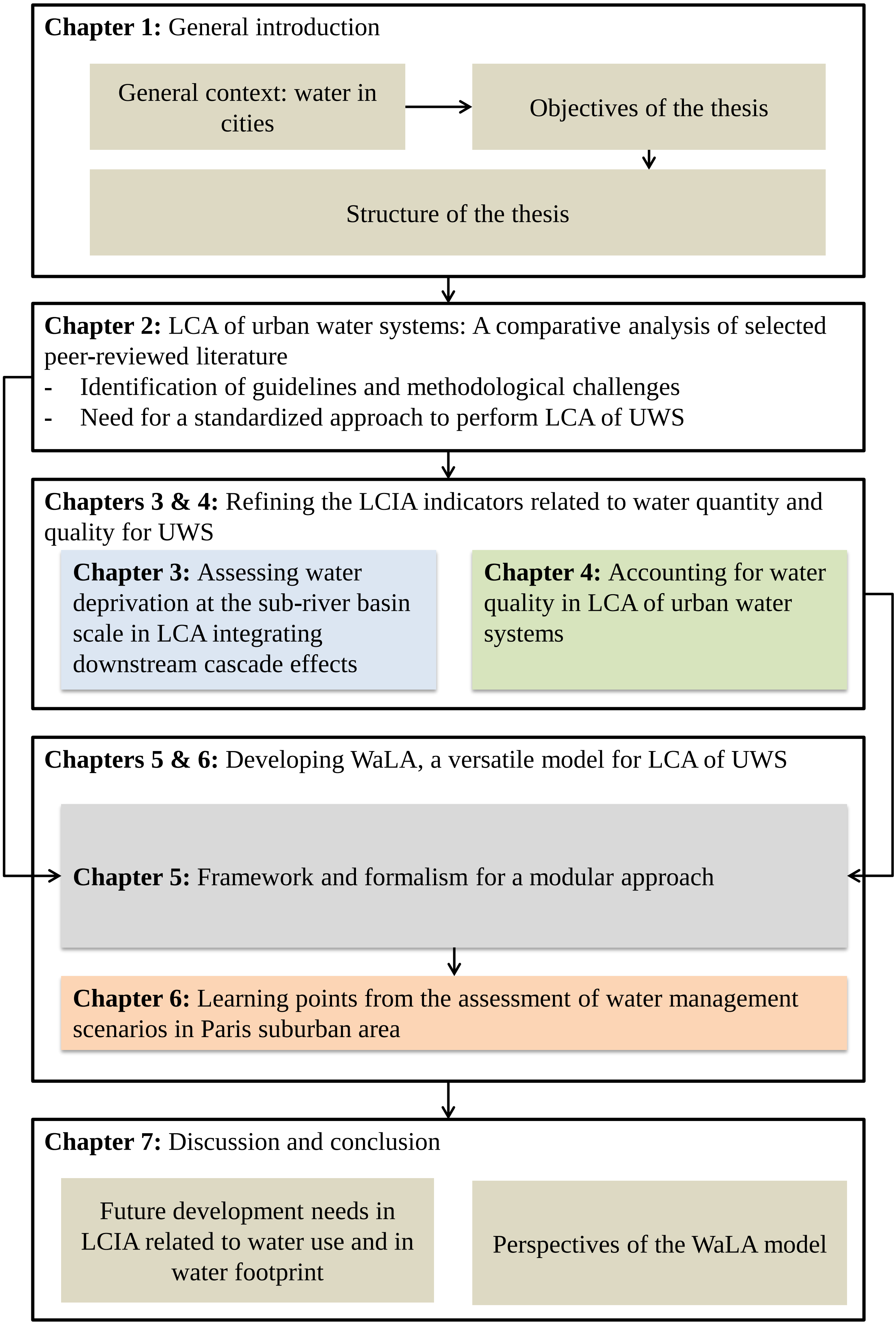


Figure 1-3. Structure of the thesis

Chapter 2. Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature

« Les égouts ont refoulé, la bécosse a débordé

Y'avait des coliformes fécaux qui flottaient su'l terrazzo »

Les Cowboys Fringants – Le plombier

This chapter aims at reviewing papers dealing with LCA applied to water technologies in order to identify the main methodological challenges in that field. It compiles all LCA papers related to water technologies, out of which 18 LCA studies deals with whole urban water systems (UWS). A focus is carried out on these 18 case studies which are analyzed according to criteria derived from the four phases of LCA international standards. The results show that whereas the case studies share a common goal, i.e., providing quantitative information to policy makers on the environmental impacts of UWS and their forecasting scenarios, they are based on different scopes, resulting in the selection of different functional units and system boundaries. A quantitative comparison of life cycle inventory (LCI) and life cycle impact assessment (LCIA) data is provided, and the results are discussed. It shows the superiority of information offered by multi-criteria approaches for decision making compared to that derived from mono-criterion. From this review, recommendations on the way to conduct the environmental assessment of UWS are given, e.g., the need to provide consistent mass balances in terms of emissions and water flows. Remaining challenges for urban water system LCAs are identified, such as a better consideration of water users and resources and the inclusion of recent LCA developments (territorial approaches and water-related impacts). This chapter refers to the following published paper: “Loubet, P., Roux, P., Loiseau, E., & Bellon-Maurel, V. (2014). Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature. Water Research, 67(0), 187–202. doi:10.1016/j.watres.2014.08.048”

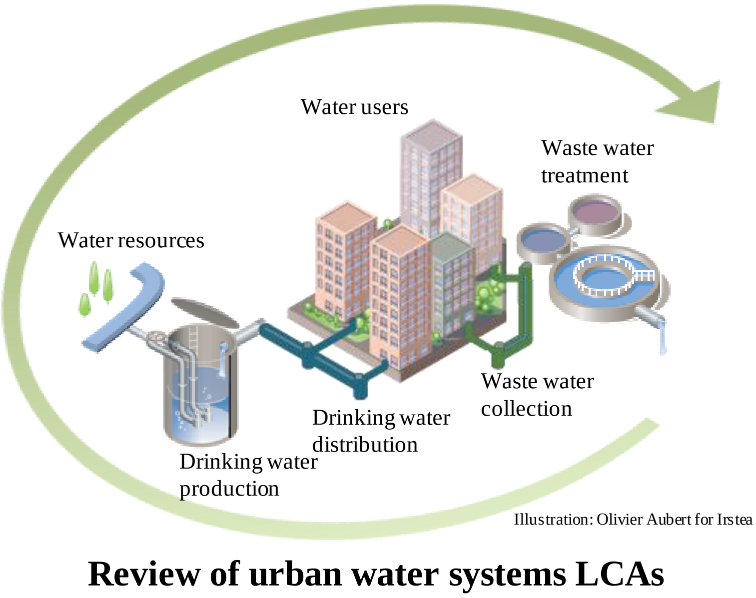


Figure 2-1. Graphical abstract of Chapter 2

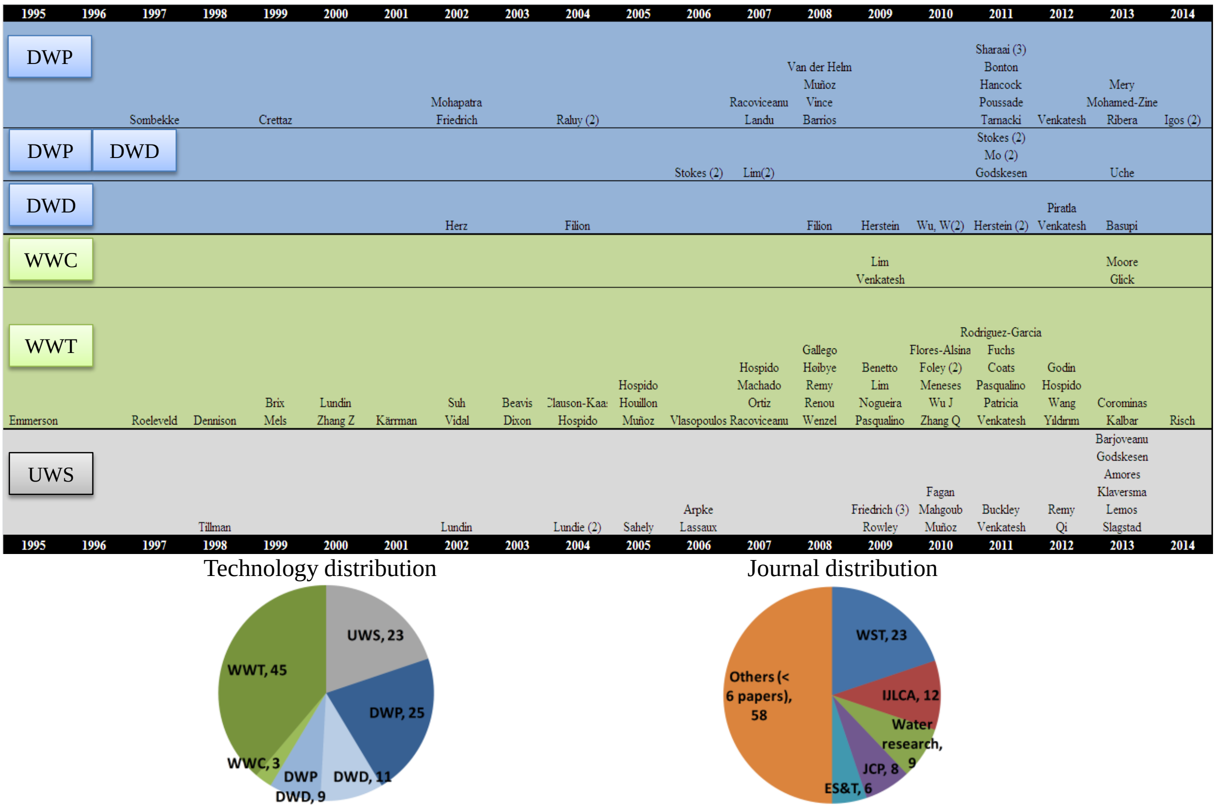
Content of Chapter 2

2.1. Introduction

In 2012, about half of the world's population lived in urban areas. This figure is expected to swell to 60% by 2030 (United Nations, 2012). Domestic, commercial and industrial water demand is consequently growing in cities. In the meantime, water scarcity is increasing, leading to water competition between users (World Water Assessment Programme UN, 2009). The degradation of water quality due to various forms of pollution has led to higher costs (both financial and environmental) in water treatment. Hence, water management is a significant challenge in the administration of growing cities. Urban water systems (UWS) are complex, as they are composed of many components that are often managed separately (raw water abstraction, drinking water production and distribution, water usage, wastewater collection and treatment, etc.). Integrated urban water management (IUWM) is a holistic approach that integrates water sources, water-use sectors, water services and water management scales (Global Water Partnership Technical Committee, 2012). The development of IUWM requires quantitative tools to assess the environmental impacts of UWS, in order to manage them in a sustainable way.

In the last 20 years, life cycle assessment (LCA) has proven its worth in the evaluation of the environmental sustainability of water systems. LCA is a standardized method (ISO, 2006b) used to assess the environmental performance of a product, service or activity from a life cycle perspective. LCA makes it possible to identify environmental hotspots within systems for eco-design purposes and helps at avoiding pollution shifts between impact categories (e.g., toxicity and eutrophication versus climate change) or between life cycle stages (e.g., treatment and discharge versus sludge end-of-life) (Finnveden et al., 2009).

LCA has been applied to water technology assessment since the late 1990s (Figure 2-2). Early LCAs focused on parts of the urban water system, mainly wastewater treatment (WWT) (Emmerson et al., 1995) and drinking water production (DWP) (Sombekke et al., 1997). Since 2005, the number of LCA studies has sharply increased. While some papers deal specifically with drinking water distribution (DWD), few focus on wastewater collection (WWC). Concerning the geographical distribution, more than half of the case studies are located in Europe, while the others are distributed in North America, Australia, South Africa, China and Southeast Asia (Figure 2-3).



Each paper is named by the name of first author ("et al." has been removed for clarity). Numbers within brackets show the numbers of papers published corresponding to each case study. Abbreviations of journals: WST: Water Science and Technology, IJLCA: International Journal of Life Cycle Assessment, JCP: Journal of Cleaner Production, ES&T: Environmental Science & Technology.

Figure 2-2. Timeline and journal distribution of water technology LCA papers.

Lundin and Morrison (2002) proposed the first framework based on LCA to assess the environmental impacts of UWS. Kenway et al. (2011) and Nair et al. (2014) reviewed the water-energy nexus in UWS, focusing on energy use and climate change. A review of LCA water treatment studies has been published by Buckley et al. (2011), focusing on South Africa. Recently, Corominas et al. (2013) published a complete review of wastewater treatment plant LCAs with the inclusion of some urban water system LCAs. More particularly, Yoshida et al. (2013) reviewed LCAs of sewage sludge management.

However, none of these studies provide a review of LCAs related to the whole UWS. Therefore, this paper aims to provide a comprehensive review of urban water system LCAs. Case studies are selected from a compilation of all LCA papers related to water technologies. They are then analyzed using criteria from the 4 phases described in LCA international standards, goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation. The comparison allows pointing out the main methodological guidelines in the assessment of urban water system regarding critical points such as the system multi-functionality, the LCI and the LCIA related to water, both in terms of quantity and quality. Future research needs in order to perform a comprehensive environmental assessment in regards with the IUWM requirements to integrate each parts of the system (i.e., water resources, users and technologies) are also discussed.

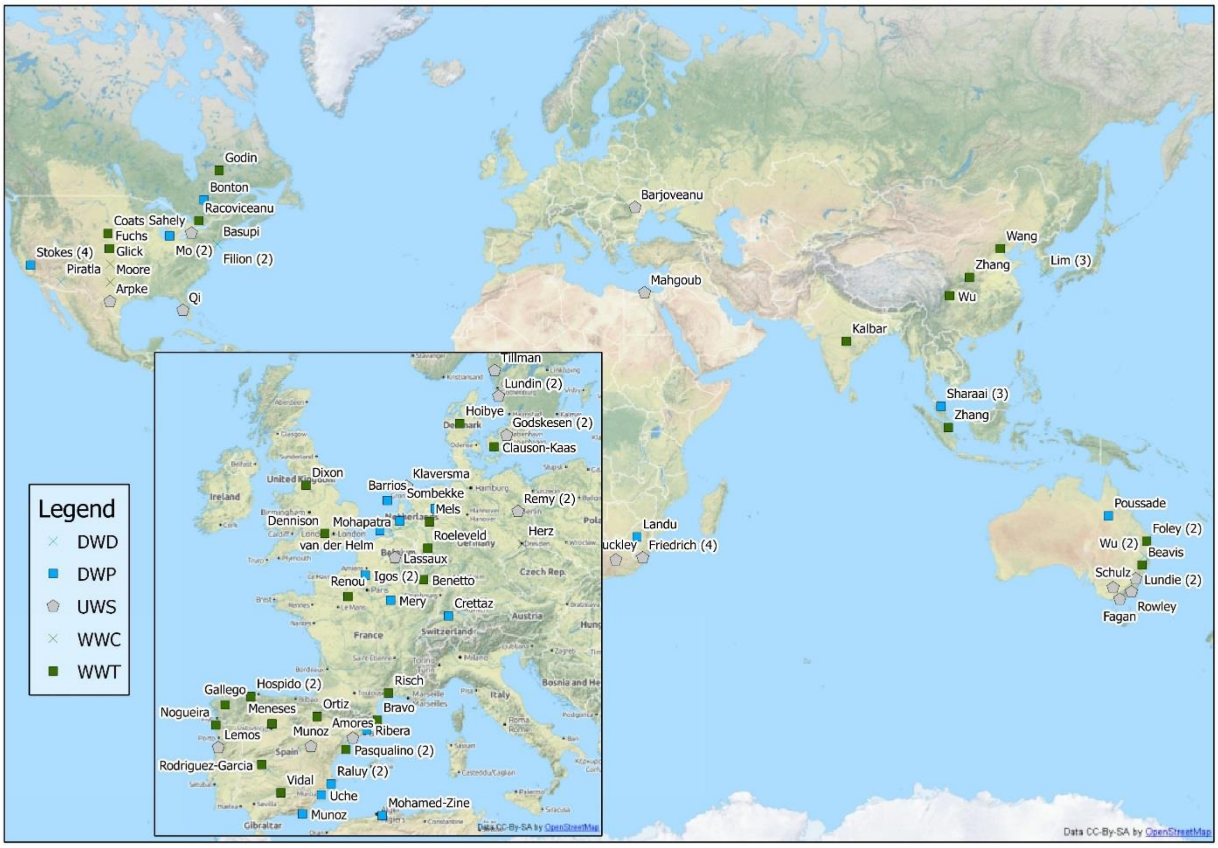


Figure 2-3. Map of LCA papers focusing on water technology, when location of the case study is available. Names refer to first authors of the papers. Numbers in brackets refer to the number of papers related to this author. When the city is unknown, the location is placed randomly within the country.

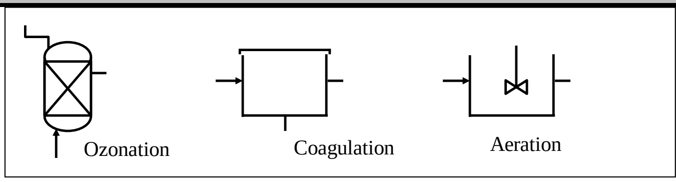
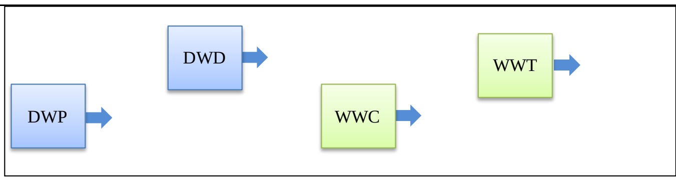
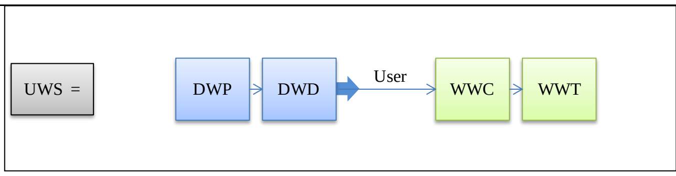
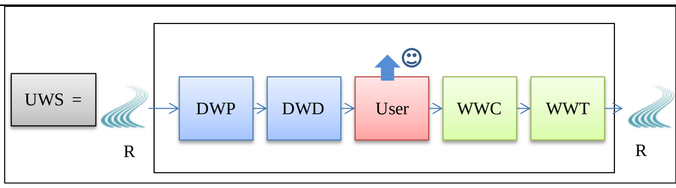
2.2. Material and methods

2.2.1. Selection of LCA papers dealing with UWS

Water technologies LCA papers can be separated according to three different nested scales: (i) "urban water systems (UWS)" which comprise (ii), "water technologies" (plants or networks) which in turn comprise (iii), "unit processes", as shown in Table 2-1.

Water technologies are classified using 4 categories: drinking water production (DWP) plant, drinking water distribution (DWD) network, wastewater collection (WWC) network and wastewater treatment (WWT) plant. The function of DWP and WWT plants is to improve water quality, while the function of DWD and WWC networks is to transfer water. The present review does not aim at compiling papers related to the unit process scale; therefore we only compiled papers at water technologies and UWS scales. Urban water system case studies are then selected according to the two following criteria, i.e., (i) they should include several water technologies (i.e., comprising at least DWP and WWT) and (ii) they should be partial or full LCA as long as they include one impact category or a multi-criteria impact assessment.

Table 2-1. Classification of papers dealing with LCA of water technologies.

Scale	**Built from**	**Number of papers**	**Scheme** *Legend:* = Functional Unit; = Water flow; = provided service to users
Unit process	Physical models	*	
Plant: DWP, WWT; or Network: DWD, WWC	Unit processes	100+	
Technological urban water system (combination of technologies)	Plants and networks	24	
Urban water system as a combination of technologies, users and resources	Plants, networks, users and water resources	0	

* Includes several papers not compiled in the present review, but two PhD dissertation have compiled most of the models used for drinking water production (Mery, 2012; Vince, 2007). R = Resources, DWP = Drinking water production, DWD = Drinking water distribution, WWC = Wastewater collection, WWT = Wastewater treatment

2.2.2. Analysis grid of LCA papers focusing on whole UWS

The case studies analysis follows the four steps of LCA according to ISO (2006): (phase 1) definition of goal and scope, (phase 2) life cycle inventory (LCI), (phase 3) life cycle impact assessment (LCIA) and (phase 4) interpretation of the results. For each phase, a set of criteria has been selected from the ISO and ILCD guidelines (EC - JRC - IES, 2010a). The set of criteria is detailed below and a summary is provided in Table 2-2.

Table 2-2. Description of criteria taken into account within the review

LCA Phase	Qualitative criteria	Quantitative criteria
Phase 1 – Goal and scope definition	- Goal - System boundaries - Life cycle steps considered	- Functional Unit - Geographic location - Number of inhabitants
Phase 2 – LCI	- Source of foreground data (ad-hoc measurements, literature, etc.) - Source of background data (databases)	- Electricity consumption data - Water flows data - Water consumption data
Phase 3 – LCIA	- LCIA method selected - Impacts and damages taken into account - Normalization (yes/ no?) - Weighting (yes/ no?)	- Climate change impacts data - Eutrophication impacts data - Water consumption impacts estimation - Single score data
Phase 4 – Interpretation	- Mono or multi criteria - Sensitivity check	- Contribution analysis from technologies and group of processes

2.2.2.1. Criteria for LCA phase 1 – goal and scope

The studies' goals are compared according to their intended applications and the reasons for carrying out the studies. A focus is placed on whether or not the studies intend to evaluate prospective scenarios, and if this is the case, whether or not a classification of scenarios is conducted. The analysis of the scope definition includes (i) the choice of functional unit (FU); (ii) key information about the system (geographic location, number of inhabitants); (iii) the definition of system boundaries; (iv) the life cycle steps considered; and (v) allocation procedures. Concerning the boundaries, the analysis investigates whether or not the case studies include foreground technologies (DWP, DWD, WWC, WWT or others), sludge end-of-life (within DWP and WWT), transportation of sludge, chemicals, consumables and fuels. Concerning the life cycle step, the inclusion of construction (both infrastructure components and associated civil works), operation, and deconstruction is reviewed.

2.2.2.2. Criteria for LCA phase 2 – life cycle inventory

The analysis of the LCI phase deals with the procedures used to collect foreground and background data (i.e., source of data) and the completeness of the inventories. It also aims at collecting data and providing a quantitative analysis of electricity consumption and water flow inventories.

Electricity consumption is represented according to the contributions of the different technologies (DWP, DWD, WWC, and WWT). Data related to water abstraction by pumping are included within DWD since it is a “water transfer” technology and not a form of water treatment. Results found in the case studies are compared according to three different approaches, having different metrics: (i) process approach, in kWh per m^3 of water processed by the technology, (ii) technological system approach, in kWh per m^3 of water delivered to the end users and (iii) territorial system approach, in kWh per capita per year. This classification follows the definition of process and system approaches from Friedrich et al. (2009a). Calculations are performed using data found in the papers when available and eq (1) and (2). These LCI data are only collected and computed for the baseline scenario of the case studies.

$$E_{/m^3 user} = E_{/m^3 process} \cdot \frac{V_{process}}{V_{user}} \qquad (1)$$

$$E_{/capita/year} = E_{/m^3 user} \cdot Vdem_{/capita/year} \qquad (2)$$

Where $E_{/m3process}$ is the technology electricity consumption for 1 m^3 at the input of the technology (kWh/m^3 at the process), $E_{/m3user}$ is the technology electricity consumption for 1 m^3 provided to the user (kWh/m^3 at the user) and $E_{/capita/year}$ is the technology electricity consumption per capita during one year (kWh/capita/year), $V_{process}$ is the water flow rate at the input of the technology (m^3/year), V_{user} is the water flow rate delivered to the users (m^3/year), and $Vdem_{/capita/year}$ is the specific water demand per capita (m^3/year/capita).

Beyond the energy consumption, water flow data are collected from the case studies and equilibrated water balances are then checked. When available, water consumption data, defined as the water evaporated or transpired through the system (Bayart et al., 2010), is collected. If these data are not available, we estimated them by considering a simplified assumption that 50% of the water losses within the system are evaporated or transpired and are considered as water consumption. The remaining 50% is considered as water returned to the environment. This first estimation of water consumption does not take into account the specific climatic conditions of each case study, as done by Risch et al. (2014). Also, water that is released to the sea is considered as lost for the local environment and is considered as water consumption.

A qualitative analysis of direct emissions (to air, soil and water) is performed, including emissions to water from each technology, sludge emissions to the soil from DWP and WWT and emissions to air from WWT.

2.2.2.3. Criteria for LCA phases 3 and 4 – life cycle impact assessment and interpretation

The criteria used for analyzing the LCIA phase of the various case studies include the chosen LCIA methodology, the list of selected impact categories at both the midpoint and endpoint levels and the presence of normalization and weighting, which are optional elements. The weighting steps and associated single scores are based on value choices and are not scientifically based (ISO, 2006b). Specific LCIA results are collected and compared among the studies for relevant and available impact categories, i.e., climate change, eutrophication, and single score. These data are only collected for the basis scenario of the case studies.

Most of the examined studies were performed before the recent advances in the inclusion of water use impacts in LCIA. These new methods provide indicators at the midpoint and endpoint level that are geographically differentiated at the country and river basin scales and that take into account water availability heterogeneity around the world (Kounina et al., 2012).We aim at evaluating water use impacts on the same basis, when possible. For this purpose, the process is the following: inventory data of water consumption obtained from the LCI (section 2.2.2) are converted into Eco-indicator 99 and ReCiPe damages (ecosystem, human health, resources) according to the method of Pfister et al. (2011, 2009). Damage scores are converted to a single score and compared to the original single scores (only those obtained from Eco-indicator 99 or ReCiPe) found in the papers that do not take into account water use damages. The Eco-indicator 99 single score is calculated using default normalization and the Hierarchist perspective (Goedkoop and Spriensma, 2001). The ReCiPe single score is calculated using European normalization, the Hierarchist perspective and average weighting factors. Even though research on water use impacts is still ongoing, we decided to apply the Pfister et al. approach because it is operational and compatible with both Eco-indicator 99 and ReCiPe units, and because characterization factors (CFs) at the endpoint level are available on a global scale. We decided to compute single score in order to be able to compare our computations with results found in the paper on a same basis, even if weighting step is questionable (ISO, 2006b).

The analysis of the interpretation phase includes the identification of hot spots based on the relative contributions from technologies and from types of contributors (electricity, chemicals, direct emissions, infrastructures). Finally, we determine whether a sensitivity check had been performed (i.e., sensitivity analysis and uncertainty analysis).

2.3. Results

Twenty-four papers dealing with LCAs of urban water system were found, as shown in Figure 2-2. However, two papers compiled several LCAs of technologies without studying the whole system (Godskesen et al., 2011; Klaversma et al., 2013) and were not considered in our review. Also two case studies were covered by several papers: Friedrich et al. (2009) was also covered by 3 other references (Buckley et al., 2011; Friedrich and Pillay, 2007; Friedrich et al., 2009b) that studied Durban UWS, and Lundie et al. (2004) was also covered by Rowley et al. (2009) that studied Sydney UWS. Therefore, six papers were disregarded and the review focused on eighteen case studies.

Table 2-3 presents the key points of the analysis grid. The papers studied medium towns to big cities and whole regions, ranging from 8 500 houses to 20 million inhabitants, with 39% of the papers dealing with case studies that have more than 1 million inhabitants.

2.3.1. LCA phase 1 - goal and scope

2.3.1.1. Goal of the studies

All of the studies aimed to provide quantitative information to policy makers on the environmental profiles and hot spots of UWS. Among the studies, 78% also evaluated prospective scenarios that could improve the environmental performance of the systems. Fagan et al. (2010) and Schulz et al. (2012) studied nonexistent or developing urban areas in Australia and thus also aimed at eco-designing UWS.

Three main types of scenarios that can be combined have been identified in the concerned papers: (i) change or improvement of a technology (e.g., the construction of a new treatment plant or an increase in the connection rate of a wastewater collection system), (ii) change of water resources, (e.g., abstracting water from another river, releasing wastewater into the sea) and (iii) change of users (e.g., increase of the population, change of users' behavior). According to our review, all of the scenarios found in the literature can be categorized into one or more of these three types.

Table 2-3. Key points of the analysis of the reviewed papers

Reference	Country	City /region	Population	Scenarios number	FU	Boundaries	Life cycle steps	LCIA method
(Amores et al., 2013)	Spain	Tarragona	145 000	3	1 m^3	DWP, DWD, WWC, WWT	Op, Cons (pipes)	CML-IA
(Lemos et al., 2013)	Portugal	Aveiro	78 450	5	1 m^3	DWP, DWD, WWC, WWT, Adm	Op, Cons (pipes)	ReCiPe
(Slagstad and Brattebø, 2014)	Norway	Trondheim	171 000	0	1 city/year	DWP, DWD, WWC, WWT	Op, Cons	ReCiPe
(Godskesen et al., 2013)	Denmark	Copenhagen	520 000	4	1 m^3	DWP, DWD, WWC, WWT, Users	Op, Cons	EDIP 1997
(Barjoveanu et al., 2013)	Romania	Iasi City	261 384	4	1 m^3	DWP, WWC, WWT	Op, Cons (pipes)	CML-IA a ecoscarcity 2006
(Schulz et al., 2012)	Australia	Kalkallo	86 000	3	1 city/year	DWP, DWD, WWC, WWT	Op, Cons	none
(Qi and Chang, 2012)	United States of America	Manatee County	323 833	20	1 m^3	DWP, DWD, WWC, WWT	Op, Cons	None (only CC)
(Remy and Jekel, 2012)	Germany	Berlin (part)	_	3	1 capita/year	DWP, WWT	Op, Cons, Decons	None (only CED)
(G Venkatesh and Brattebø, 2011)	Norway	Oslo	529 800	0	1 capita/year	DWP, DWD, WWC, WWT	Op	CML-IA
(Fagan et al., 2010)	Australia	Aurora	8500 houses	3	None	DWP, DWD, WWC, WWT, Users	Op, Cons	Eco-indicator 95
(Mahgoub et al., 2010)	Egypt	Alexandria	3 700 000	6	1 m^3	DWP, DWD, WWC, WWT	Op	Eco-indicator 99
(Muñoz et al., 2010)	Spain	Mediterranean region	20 000 000	2	1 m^3	DWP, DWD, WWC, WWT	Op, Cons	CML-IA and CED
(Friedrich et al., 2009a)	South Africa	Durban	3 100 000	4	1 m^3	DWP, DWD, WWC, WWT	Op, Cons	CML-IA
(Lassaux et al., 2006)	Belgium	Walloon region	3 500 000	5	1 m^3	DWP, DWD, WWC, WWT	Op, Cons	Eco-indicator 99 and CML-IA
(Arpke and Hutzler, 2006)	United States of America	_	_	0	None	DWP, DWD, WWT, Users	Op	BEES
(Sahely et al., 2005)	Canada	Toronto	2 600 000	0	None	DWP, DWD, WWC, WWT	Op	None (only CC and CED)
(Lundie et al., 2004)	Australia	Sydney	4 500 000	8	1 city/year	DWP, DWD, WWC, WWT, Adm	Op, Cons	CML-IA
(Tillman et al., 1998)	Sweden	Bergsjon Hamburgsund	14 300	2	1 capita/year	DWP, DWD, WWC, WWT	Op, Cons	None (only CED)

DWP = Drinking water production, DWD = Drinking water distribution, WWC = Wastewater collection, WWT = Wastewater treatment, Adm = Water administration, Op = Operation, Cons = Construction, Decons = Deconstruction, CC = Climate Change, CED = Cumulative Energy Demand. “_” = No data available.

2.3.1.2. Scope: functional unit

A total of 50% of the studies defined the FU as the “provision and treatment of 1 m^3 of water at the user” or the equivalent, which can be summarized as “1 m^3” whereas a total of 17% of the studies defined the FU as the “provision and treatment of water per capita for one year” or the equivalent, which can be summarized as “1 capita/year”. A total of 17% of the studies defined the FU as the “provision and treatment of water for the city and one year”, which can be summarized as “1 city/year” Three papers did not define any FU, but implicitly consider “1 m^3” (Arpke and Hutzler, 2006; Sahely et al., 2005) or ”1 city/year” (Fagan et al., 2010).

2.3.1.3. Scope: boundaries, life cycle steps, allocation procedures

All of the studies considered at least DWP and WWT in the boundaries of the systems, which is straightforward since it is the criterion of selection of the papers. Fifteen (83%) studies include all the main water technologies (DWP, DWD, WWC and WWT).

Only three papers, i.e., Fagan et al. (2010), Arpke and Hutzler (2006) and Godskesen et al. (2013), considered water users (domestic and industrial) as a part of the system. This acknowledges that users can have an impact on the environment, for instance when using technologies such as water heaters or in relation to direct water release at the user’s location. Lemos et al. (2013) and Lundie et al. (2004) included water management administration (office buildings, vehicle fleets, etc.).

WWT sludge end-of-life was taken into account in twelve (61%) studies (combinations of agricultural application, landfill, incineration and composting). Amores et al. (2013) also took into account DWP sludge end-of-life (recycled in a cement plant), but they do not provide information on its contribution to the DWP process. Among the studies that took into account WWT sludge end-of-life, six used substitution by chemical fertilizers and one used system expansion integrating fertilization and energy production in the system functions in order to take into account the environmental benefits of sludge end-of-life (Remy and Jekel, 2012). Four studies did not consider environmental benefits for sludge.

Concerning the life cycle steps, all the studies included the operational phase. Three studies took into account the pipe infrastructure (DWD and WWC), and ten studies include the infrastructure of the whole system. However, only the needed components and materials were taken into account for the infrastructure, and none of these studies accounted for the necessary civil works (e.g., excavation) associated with construction.

2.3.2. LCA phase 2 - life cycle inventory

Inventories of foreground flows (use of energy, chemicals, quantity and quality of water, etc.) were mostly collected from site specific data gathered in internal reports or databases. Other foreground flows (such as infrastructure) were collected from estimations and data in the literature. Eleven (61%) studies provided the reference or the source of these data. Foreground data were often assumed to be of fair quality but only Lemos et al., (2013) provided indications on the data quality, classifying data from low quality to high quality whereas Friedrich et al. (2009a) and Qi and Chang (2012) commented data quality. Concerning background data, twelve studies used ecoinvent (Frischknecht et al., 2007) as a database for background processes, two used the GaBi database (PE International), and three used other sources. Twelve (67%) studies provided LCI data; a comparison of LCI results regarding energy and water flows is presented below.

2.3.2.1. Operation (energy)

The energy for water technologies (pumps, stirring reactors, retro washing, etc.) is electricity. Electricity consumptions of eleven case studies are presented in Table 2-4, following the three metrics introduced in section 2.2.2.2.

Table 2-4. Electricity consumption of the technologies composing UWS in 11 studies.

	kWh/m^3 process				kWh/m^3 user					kWh/capita/year				
Reference	DWP	DWD	WWC	WWT	DWP	DWD	WWC	WWT	total	DWP	DWD	WWC	WWT	total
(Amores et al., 2013)	0.37	0.48	0.00	1.09	0.44	0.58	0.00	1.09	2.11	34	45	0	85	165
(Godskesen et al., 2013)	_	_	_	_	0.18	0.10	0.08	0.68	1.03	10	6	4	39	59
(Lemos et al., 2013)	0.64	0.15	0.21	0.87	0.88	0.21	0.21	0.73	2.04	49	12	0	41	101
(Barjoveanu et al., 2013)	0.04	0.27	0.04	0.17	0.07	0.45	0.04	0.14	0.69	10	63	5	19	97
(Slagstad and Brattebø, 2014)	-	0.17	0.00	0.14	-	0.25	0.00	0.32	0.58	-	20	0	26	47
(Venkatesh and Brattebø, 2012)	0.23	0.18	0.06	0.75	0.29	0.22	0.06	0.88	1.44	51	39	10	156	256
(Muñoz et al., 2010) EWRT avg*	0.55	0.50	_	0.30	0.67	0.61	_	0.30	1.58	34	31	_	15	79
(Friedrich et al., 2009a)	0.09	0.10	0.14	0.44	0.12	0.14	0.14	0.26	0.67	_	_	_	_	_
(Lassaux et al., 2006)	0.21	0.18	0.00	0.31	0.30	0.25	0.00	0.24	0.79	18	15	0	14	47
(Arpke and Hutzler, 2006) low	0.34	0.11	_	0.21	_	_	_	_	_	_	_	_	_	_
(Arpke and Hutzler, 2006) high	0.37	0.44	_	0.77	_	_	_	_	_	_	_	_	_	_
(Sahely et al., 2005)	_	0.60	_	0.47	_	_	_	_	_	_	_	_	_	_
(Lundie et al., 2004)	0.08	0.24	0.06	0.41	0.08	0.25	0.06	0.33	0.73	11	34	8	45	98
Median	**0.23**	**0.21**	**0.05**	**0.43**	**0.23**	**0.25**	**0.06**	**0.33**	**0.91**	**18**	**31**	**2**	**39**	**97**
Average	**0.26**	**0.28**	**0.06**	**0.49**	**0.30**	**0.31**	**0.06**	**0.50**	**1.17**	**24**	**29**	**3**	**49**	**105**
Standard deviation	0.21	0.17	0.08	0.31	0.28	0.18	0.07	0.32	0.58	18	18	4	46	67

DWP = Drinking water production, DWD = Drinking water distribution, WWC = Wastewater collection, WWT = Wastewater treatment. "_" = No data available. *avg means the average value between optimistic and pessimistic EWRT scenario.

In most studies, the highest share of electricity consumption was due to WWT, closely followed by DWD and DWP. It should be noted that DWP electricity consumption might be overestimated in some studies since part of that energy might be used for pumping at the exit of the DWP plant and should thus be allocated to DWD network. WWC was negligible since this water transfer is mostly driven by gravity. According to the "technological system" approach, UWS require 0.58 to 2.11 kWh per m^3 delivered to the user. The "territorial system" approach yields a different classification of the case studies, ranging from 47 to 256 kWh per capita per year. Compared with average European electricity consumption, UWS contribute for 1 – 2% of the total consumption which is approximately 5 700 kWh/capita (European Environment Agency, 2008).

These results emphasize the importance of functional unit choice and the consideration of users' behavior. It should be noted that no DWP desalination data is included in Table 2-4 since no case study included it in basis scenarios. Muñoz et al. (2010) gave values ranging from 1 kWh/m^3 of water produced (optimistic value for brackish water desalination) to 4 kWh/m^3 of water produced (pessimistic value for seawater desalination) in their prospective scenarios. Arpke and Hutzler (2006) also considered electricity consumption for water heaters and found a consumption of 63 kWh/m^3 to heat water in the United States. In this study, the proportion of hot water used is 10% in office buildings and 46% in apartments (domestic use). This results in overall user water heating electricity consumption ranging from 6.3 kWh/m^3 for office user to 29 kWh/m^3 for domestic user. This energy amount is 5 to 23 times greater than the average electricity consumption in all other technologies of the urban water system.

2.3.2.2. Direct water flows

Half of the studies indicated the volumes of water flows within the system. Table 2-5 shows water flows at the input of the technologies, after data normalization for 1 m^3 at the user.

Two studies considered the water losses of DWP (Friedrich et al., 2009a; Slagstad and Brattebø, 2014), finding values of 4 % and 8 % (respectively). The average DWD losses were 25%. Wastewater flows were greatly variable because systems may have combined sewer systems, separated sewer systems, or both. Studies did not provide a comprehensive water balance according to the framework of Bayart et al. (2010), i.e., the total amount of water withdrawn and released within the local environment, as well as the water evaporated to the global environment or released to the sea (consumptive use). Our rough estimation of water consumption through the systems shows a range from 0.13 to 1.11 m^3 of water consumption for 1 m^3 of water at the user.

Table 2-5. Water flows through the different components of the UWS and associated impacts from 8 studies.

	Water flow inventory (/m^3 at the user)								Water consumption damages*				
Reference	DWP	DWD	User	WWC	WWT	Unww	Released water to sea?	Estimated WC	ReCiPe damage/m^3			ReCiPe single score/m^3	EI99 single score/m^3
Unit (/m^3 user)	m^3	m^3	m^3	m^3	m^3	m^3		m^3	HH (E-06 DALY.yr)	EQ (E-09 species.yr)	Res ($)	Pt	Pt
(Amores et al., 2013)	1.20	1.20	1.00	1.00	1.00	0	Y	1.10	0	7.005	1.825	1.200	0.135
(Barjoveanu et al., 2013)	1.70	1.70	1.00	0.81	2.09	0.193	N	0.44	0.002	1.240	0	0.003	0.014
(Lemos et al., 2013)	1.38	1.38	1.00	0.84	0.84	0	Y	1.11	0	2.099	0	0.005	0.009
(Slagstad et al. 2013)	1.60	1.47	1.00		2.28	0.115	N	0.30	0	0.338	0	0.001	0.005
(Venkatesh et al., 2012)	1.25	1.25	1.00	1.17	1.17	0	N	0.13	0	0.141	0	0.000	0.006
(Friedrich et al., 2007)	1.47	1.42	1.00	0.60	0.60	0	Y	1.04	0.365	4.981	0	0.018	0.033
(Lassaux et al., 2006)	1.42	1.42	1.00	0.78	0.78	0	N	0.32	0	0.796	0	0.002	0.012
(Lundie et al., 2004)	1.05	1.05	1.00	0.81	0.78	0	Y	0.92	0	3.545	0	0.008	0.019
Average	1.38	1.36	1.00	0.86	1.19			0.67				0.155	0.029

DWP = Drinking water production, DWD = Drinking water distribution, WWC = Wastewater collection, WWT = Wastewater treatment, Unww = Untreated wastewater, WC = Water consumption, * Damages are computed with regard to water consumption using Pfister et al. (2009).

2.3.2.3. Direct emissions (water, air and soil)

DWP direct emissions to water were not considered and only Amores et al. (2013) studied DWP sludge emissions. Furthermore, none of the studies addressed emissions from the sewage network (WWC).

A total of 61% of the studies inventoried direct emissions to water from WWT effluent release, and 44% of the studies accounted for emissions to air from WWT. This lack of consideration is mainly because several studies only focused their environmental assessments on the energy use and/or the infrastructures of the systems (Arpke and Hutzler, 2006; Godskesen et al., 2013; Sahely et al., 2005; G. Venkatesh and Brattebø, 2011).

Concerning the pollutants taken into account in WWT, emissions to water always included nitrogen (total nitrogen or nitrates, nitrites and ammonia) and phosphorus (total phosphorus, phosphates). Six studies included COD and/or BOD. Heavy metals emissions to water were only considered in one study (Fagan et al., 2010). Air emissions mostly included nitrous oxide (N_2O) (five studies), CO_2 (four studies), CH_4 (three studies) and occasionally other pollutants (particulates, volatile compounds, CO, SO_2). Emissions to soil (from sludge spreading) included heavy metals in three studies. Equilibrated mass balances of pollutants were not provided in the reviewed studies.

2.3.3. LCA phases 3 and 4 – life cycle impact assessment and interpretation

2.3.3.1. Impacts taken into account

Four (22%) studies performed a mono-criterion assessment, evaluating the impacts on climate change and/or the cumulative energy demand of the urban water system (Qi and Chang, 2012; Remy and Jekel, 2012; Sahely et al., 2005; Tillman et al., 1998). Seven studies applied CML-IA (Guinée et al., 2002), three applied Eco-indicator 99 (Goedkoop and Spriensma, 2001), two applied ReCiPe (Goedkoop et al., 2009) and one applied EDIP (Potting and Hauschild, 2004). Lassaux et al. (2006) used two methods (Eco-indicator 99 and CML-IA) and found similar results. None of the studies showed endpoint indicator results according to the three areas of protection (human health, ecosystem quality and resources).

When only considering multi-criteria studies (fourteen studies), 100% of the papers included climate change and eutrophication, 89% included acidification, 44% included ecotoxicity (marine, aquatic or terrestrial), and only 22% included water use impacts. In four studies, raw midpoint results were not displayed, and results were only shown in the single score, thus omitting useful information. Hence, we focused on climate change impacts, water use impacts and water pollution impacts.

2.3.3.2. Climate change impacts

A total of sixteen studies calculated the impact on climate change, and results were available in six studies. The impacts ranged from 0.51 to 1.57 kg CO_2 eq/m^3 at the user (Figure 2-4) and are highly dependent on electricity consumption and the electricity mix used in each country. Lundie et al. (2004) (Sydney) and Friedrich et al. (2009a) (Durban) indicated relatively high impacts on climate change, whereas their electricity consumption was relatively low in comparison with other studies. This is because the electricity mixes used in their countries generate twice the amount of GHG emissions (respectively, 1.03 kg CO_2 eq/kWh for Australia and 0.97 kg CO_2 eq/kWh for South Africa) than in other case studies (e.g., 0.45 kg CO_2 eq/kWh in Spain) (Itten et al., 2013).

One study took into account the contribution of user-related water technologies, and found out that 93% of the impacts on climate change were related to electric water heating systems (Fagan et al., 2010).

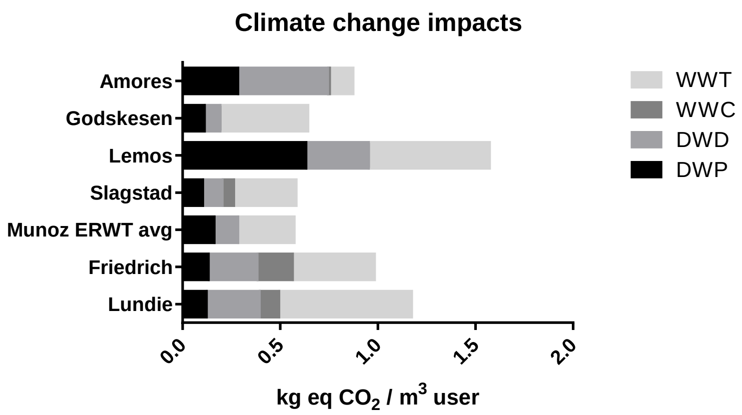


Figure 2-4. Climate change impacts of the technologies composing the UWS of 6 studies.

2.3.3.3. Water use impacts

Three studies have taken into account the water use impacts. Amores et al. (2013) and Muñoz et al. (2010) use freshwater ecosystem impact (FEI), which is calculated from the withdrawal-to-availability (WTA) ratio of the river basins where water is withdrawn or released (Milà i Canals et al., 2008). The case study published by Muñoz et al. (2010) is relevant because water is withdrawn and released in different basins and it justifies the use of indicators differentiated at the river basin scale. Godskesen et al. (2013) used CF values determined using the methodology of Lévová and Hauschild (2011), which is also based on WTA. It should be noted that they considered all of the withdrawn water as consumed water because wastewater is returned to the sea and thus lost to the local freshwater environment.

We have computed water use impacts from the water consumption estimations described in section 2.3.2.2 and from the ReCiPe and Eco-indicator 99 endpoint single score CFs (Table 2-5). The results range from 0.002 to 0.149 EI99-Point/m^3 at the user and from 0.0011 to 1.200 ReCiPe-Point/m^3 at the user. These huge variations are caused by the regional water scarcity context. Ecosystem damages vary from one order of magnitude. Human health damage is pointed out in two studies only, located in South Africa (Friedrich et al., 2009a) and Romania (Barjoveanu et al., 2013), where the Human Development Index (HDI) is below 0.88. Resources damages are also identified in one study only, located in Spain (Amores et al., 2013), where the water stress defined by the WTA is higher than 1 (Pfister et al., 2009).

Single score results related to water use can be compared to single score of the whole urban water system (see section 2.3.3.5). EI99 and ReCiPe single score results for the whole urban water system were collected from two studies. Lassaux et al. (2006) found 0.4 EI99-Pt/m^3 at the user and Lemos et al. (2013) found 0.151 ReCiPe-Pt/m^3 at the user, respectively 100 and

30 times higher than the water use single score. However, these 2 studies were located in areas with low water scarcity (Belgium and Portugal). Other locations, in areas of scarce water might find a high contribution of water use damage to the total score, such as Amores et al. (2013).

2.3.3.4. Water pollution impacts

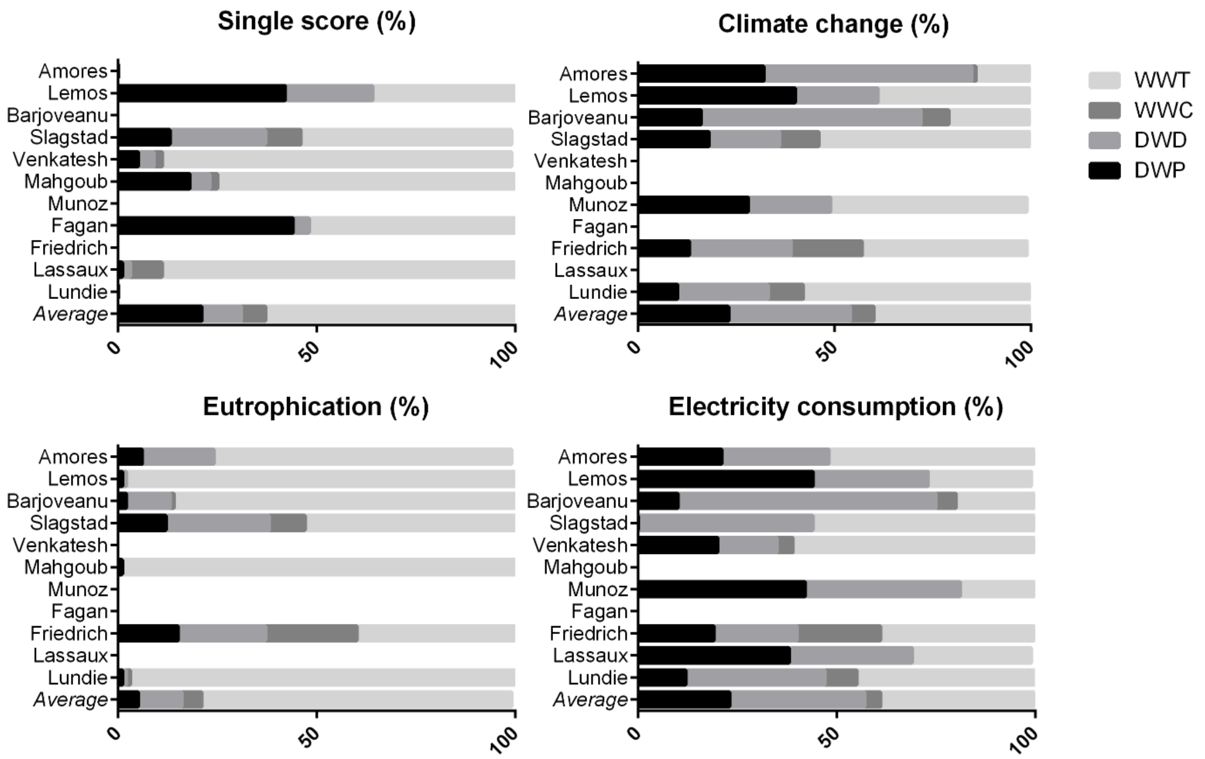
Eutrophication, ecotoxicity and acidification are major direct impacts generated by UWS. Eutrophication figures were available in and gathered from eight studies. Since the units are different, only the relative contributions of the technologies are compared. WWT contributes to the highest share of impacts due to the release of treated water containing residual amounts of eutrophicating substances (Figure 2-5). This direct contribution accounted for more than 50% of the total eutrophication impacts. Marine eutrophication was assessed in one study with ReCiPe (Lemos et al., 2013), whereas the other studies only regarded freshwater eutrophication.

Concerning ecotoxicity, none of the studies examined used the consensual method Usetox (Rosenbaum et al., 2008), because it was not included in the selected LCIA methods. Muñoz et al. (2010) chose not to include toxicity-related impacts because of the lack of information on the toxicity effects of emerging pollutants. However, some studies provided a full inventory of toxic substances. Hence, direct ecotoxicity impacts were mostly caused by background processes.

2.3.3.5. Normalization, weighting

Eight studies used normalization and four of these displayed normalization results at the midpoint level (all with European values). From these studies, the impacts with the greatest contribution were all related to water pollution, i.e., eutrophication (2 studies), marine ecotoxicity (1 study) and acidification (1 study).

Seven studies provided a single score after weighting. Weighting factors depend on the selected methods, including the hierarchist perspective with average weighting in ReCiPe or Eco-indicator 99, Eco-indicator 95 weighting adapted to Australian data, and weighting provided by CML-IA or the USEPA scheme. Because of the discrepancy, single score results from these different studies cannot be compared.



Only DWP, DWD, WWC and WWT are taken into account. Spaces left blank mean that no data were available. The results of Fagan et al. (2010) do not include user contribution.

Figure 2-5. Technology contribution analysis of LCA single score, climate change & eutrophication impacts and electricity consumption inventory.

2.3.3.6. Contribution analysis

A total of fifteen studies provided a contribution analysis of the technologies (i.e., DWP, DWD, WWC and WWT) used in the systems. They are presented in Figure 2-5. Regardless the impact category analyzed, the highest contributions came from WWT (average contribution of 66% to single score, 44% to climate change, 78% to eutrophication and 39% to electricity consumption). Following, DWP and DWD had equivalent contributions. WWC had a low contribution in all criteria. Water administration, which has been studied in two papers, did not contribute to a large share of the impacts. However, water users, which were also included in two studies, contributed to a large share of the impacts: Fagan et al. (2010) found a contribution of 50% on the single score result, mainly because of water heating.

Eight studies provided a contribution analysis according to types of contributors (such as energy, chemicals, infrastructures, direct emissions, etc.). In all cases, electricity contributes to the largest share of impacts. A contribution of infrastructures was considered in seven studies. Three of these studies found high contributions, i.e., more than 20% (Fagan et al., 2010; Lassaux et al., 2006; Slagstad and Brattebø, 2014), whereas the other four studies found lower contributions, i.e., less than 10% (Lemos et al., 2013; Lundie et al., 2004; Remy and

Jekel, 2012; Schulz et al., 2012). Infrastructure can differ depending on the density and topography of cities and thus can lead to different shares of the impacts. However, the results showed that infrastructure should be considered and is most likely under-estimated since civil work is not taken into account, as noted by Roux et al. (2010).

2.3.3.7. Sensitivity check

Sensitivity analysis has been performed in 50% of the studies. The evaluation of scenarios (done in 78% of the cases) can also be considered as sensitivity analysis. This is done was done by comparing with basis scenario values and by showing the increase or decrease for each category of impact. This review does not collect LCI and LCIA results from the prospective scenarios. Finally, a proper uncertainty analysis with Monte Carlo simulation has been provided in only one study (Muñoz et al., 2010).

2.4. Discussion and perspectives

In addition to providing a comprehensive analysis of data and figures, results section point out several questions associated with UWS LCAs. This section discusses the most relevant issues, following all LCA phases as well as additional focuses on uncertainty and decision makers' issues. Based on that, recommendations are proposed and remaining methodological challenges are identified.

2.4.1. Goal and scope

2.4.1.1. Functional unit

FUs defined in the reviewed studies ("1m3", "1 capita/year" or "1 city/year") are linked to the goal and scope and are related to the functionality of the systems. The "1 m3" FU represents water as a product processed and distributed by a technological system and is linked to the efficiency of the system. In this case the functionality is to produce, to deliver or to treat water and to deliver it at the users' location. On the other hand, the "1 capita/year" FU depicts water as a provided service to a user within an integrated urban water system. The functionality is to provide enough water (both in terms of quantity and quality) for users. Therefore, this FU includes the behavior of the user. In the case studies, the volume used per capita ranges from 50 to 177 m3 per year. Hence, depending on the FU (based on volume or capita), the results can radically change. If a policy for the integrated urban water management reduces the water use per capita, the impacts per m3 will slightly not change, whereas the impacts per capita will likely be reduced, assuming that WWC and WWT can face marginal variations in flows.

This improvement is not due to a technological change within the urban water system, but to a change within the whole system, which includes the user. The amount of water use per capita is dependent on the climate, the socioeconomic level of the country, the awareness of the users, etc. The 1 city/year" FU "is relevant when comparing the overall impacts of different UWS management scenario. It is also an interesting approach in order to solve the issue faced with the FU "1 capita/year" that only defines one kind of user (domestic), whereas other users such as industries, services, etc. should be taken into account.

2.4.1.2. Boundaries of the system

Since the majority of energy consumption stem from water heating which is mainly done at the user's place, the inclusion of the water users' technologies should be questioned. If the goal and scope of the LCA is to only assess different technologies (of DWP, DWD, WWT, etc.), users and their water heating system can be excluded; but if the goal and scope claims to study the entire urban water system, it cannot. In the latter case, even if the main energy consumption is due to water heating, the other contributors (direct emissions in air, water, soil, chemicals and infrastructures, etc.) should not be neglected. While energy consumption is the greater contributor of several specific impacts categories (ionizing radiation, abiotic depletion, etc.) the other contributors predominantly affect other impact categories such as eutrophication, toxicity, water deprivation.

Also, the status of sludge is still controversial: it can be considered either as a by-product when it has an economic value (due to its mineral, organic or energetic content) or as a waste when the value is equal or less than zero (Frischknecht, 1998). The review shows that both considerations have been chosen. However, these statuses are dependent on the today's economy and the local context of the studies. When evaluating sludge as a by-product, several options can be adopted in order to take into account its environmental benefit: substitution with a fertilizer or another energy source, expansion of the system including supplementary functions or allocation (EC - JRC - IES, 2010b). Allocation, which should be avoided according to ISO 14044, has never been used and is clearly not adapted to assess sludge. ISO rather recommends to use expansion of the system but do not mention substitution, even if we can consider that both alternatives are equivalent (Heijungs, 2013).

2.4.1.3. Towards a territorial/city LCA approach

As an urban water system is part of a given territorial system, its environmental evaluation could benefit from recent research on the adaptation of the LCA framework to territorial assessment (Loiseau et al., 2013). This approach proposes to define the reference flow (i.e. the LCA input) as the association of a given territory and a specific land-planning scenario. This adaptation allows considering all the services provided by the so-called reference flow and is thus suitable to UWS which are multi-functional (provision of water for domestic, industrial, recreational users) and which are associated to planning scenarios (choices of resources abstraction and technologies, city growth, etc.). In such a multi-functional system, the functional unit is no longer a unity but becomes a vector of services which can be assessed in a qualitative (based on stakeholder involvements) or quantitative (based on statistic and economic data and models) way. This would enable the evaluation of different FUs ("1m3", "1 capita/year", "1 city/year") in the same time to calculate several eco-efficiency ratios and compare them (Seppäläa and Melanen, 2005). This adaptation requires first to clearly identify the different kind of water users (Bayart et al., 2010; Boulay et al., 2011) and which services are provided by the UWS to them.

2.4.2. Life cycle inventory

2.4.2.1. Mass balances

In the inventory phase, a major challenge is the provision of equilibrated mass balances of water and of pollutants at each stage of the UWS. There is a particular need to formalize the water balance within UWS for LCA purposes and to evaluate the different water flows. A water technology can exchange water with three different compartments: the technosphere (i.e., other technologies and users), the local environment (i.e., the (sub) river basin where the technology withdraws and releases water) and the global environment (i.e., the atmosphere, where water is evapotranspirated and ultimately consumed from the local water cycle, or the sea) (Loubet et al., 2013). The Quantis Water database (Quantis, 2011), which is implemented within ecoinvent 3.01, already provides a comprehensive water inventory for industrial processes. Research is still required to compute water balances in other water processes, especially the networks (the share of leaks that are evaporated or returned to surface and ground water), and at the users' place differentiating domestic, and industries. WWT might also be an important water consumer, particularly in the case of reed bed filters or lagoon treatments (Risch et al., 2014).

Mass balances of pollutants should be performed, first at the WWT scale (Risch et al., 2011), and also in the other components of the urban water system scale, because the fluxes of pollutants emitted to the environment from the other technologies are often disregarded. Studies focusing on DWP have shown that the impacts due to emissions of metals from chemicals (e.g., aluminum) in water and soil should not be neglected (Igos et al., 2014). Concerning WWC, emissions of methane, nitrous oxide and hydrogen sulfide should be quantified (Guisasola et al., 2008; Hjerpe, 2005; Short et al., 2014).

Additionally, pollutants that are released within the environment might come from the same local environment. For example, a drinking water plant withdraws water from the environment, removes pollutants from this water and finally releases them within the same local environment (as water release or sludge). Therefore, pollutants that are withdrawn from the local environment should not be accounted for when they are released again within the same environment.

2.4.2.2. Sources of data

Registers such as the European Transfer Pollutant Transfer Register (E-PRTR) can be used to provide accessible, standardized and up to-data direct data emission (to air, soil, water) from industries and thus water technologies (Yoshida et al., 2014). Nevertheless, such database does not provide data for electricity or chemical consumption whereas this review showed that these processes contribute for a large share of impact. Data gathering at the plant scale is still needed since they are mainly site-specific. Energy demand for water transfer technologies (DWD and WWC) highly depends on the density and the topography of the city and on the locations of raw water abstraction and wastewater release. Energy demand for water treatment technologies (DWP and WWT) also depends on the quality of input and output water.

Another challenge is the gathering of inventory data for future scenarios. New technologies should be assessed, such as alternative WWT plants (Foley et al., 2010), microtunnelling technologies for DWP (Piratla et al., 2012), etc. An effort should be made on the knowledge regarding infrastructures and civil works associated since important trade-offs can occur between operation and construction of new infrastructures (Roux et al. 2010). Future effects of climate change on urban water system should be taken into account, primarily regarding the choice of water resources, since it is a key issue for future scenarios (Short et al., 2012).

2.4.3. Life cycle impact assessment

The review showed that several impact categories are significant for UWS, in particular those in links with water quality and quantity. Thus, mono-criterion approaches such as carbon footprint and energy-balances should be avoided in the future. The role of UWS is central within water resource management. The evaluation of the direct impacts of these systems on water resource should thus be improved and relevant LCIA methods for UWS should be refined. Water footprint methodologies are often cited to meet this issue. They have been developed outside and inside the scope of LCA (Hoekstra et al., 2011; Kounina et al., 2012), to evaluate the impacts on water as a resource (quantitative issues) and water as a compartment receiving pollution (qualitative issues). The impact assessment of water use is recent and no consensus has been reached yet. New approaches are currently being developed in order to improve the geographical and temporal resolution of the characterization factors (Pfister and Bayer, 2014), as well as the link between midpoint and endpoint damages. Loubet et al. (2013) developed a method relevant for UWS studies, that differentiates impacts at the sub-river basin scale and takes into account downstream cascade effects of water withdrawal. It makes possible to compare scenarios in which different withdrawal and release locations are proposed within the same river basin. Otherwise, conventional methods use the same water stress indicator for the entire river basin and are therefore unable to discriminate such scenarios. As for the LCI phase, effects of climate change on water deprivation indicators at the global scale should be taken into account when computing forecasting scenarios as a first study did for Spain (Nunez et al., submitted).

Impact assessment of water pollution also needs improvements in the time and space resolution, especially for eutrophication. New methodologies within the LC-Impact project address regionalized freshwater and marine eutrophication, both at the midpoint and endpoint level (Azevedo et al., 2013; Cosme et al., 2013). These methodologies should be of great interest for urban water system LCAs. Concerning ecotoxicity, the relevancy of heavy metals characterization should be revisited (Muñoz et al., 2008) Furthermore, the assessment of pathogens on human health was not yet possible and sharply limited water system environmental assessments, but a recent work opens interesting perspective (Harder et al., 2014).

2.4.4. Uncertainty management

Three main sources of uncertainty can be addressed in LCA according to ILCD: stochastic uncertainty of LCI data and LCIA methods, uncertainty due to choices and lack of knowledge of the studied system. Stochastic uncertainties linked to foreground data should be definitely quantified in future studies, especially for significant flows such as water (quantity and quality) and energy. When uncertainties are not known, standard deviation can be estimated with the pedigree approach (Weidema and Wesnæs, 1996). This requires defining data quality indicators. Stochastic uncertainties linked to background processes and LCIA methods are inherent to LCA studies and are already provided within the database. Second, uncertainties due to choices are already treated by some of the reviewed papers with the provision of sensitivity analysis and in a lesser extent with the evaluation of different scenarios. Worst and base case scenario should also be computed, as done by Muñoz et al. (2010). Finally, paucity of data in developing countries is a real challenge (Sonnemann et al., 2013) and can be barrier to conduct LCAs: at present, UWS LCAs are conducted in developed countries, as shown in Figure 2-3.

2.4.5. Towards integrating LCA results for UWS decision-makers

Decision-making process is dependent on the stakeholders that have different goal and scope regarding urban water system management, and two of them are discussed here after.

(i) For decision about future investments done by regional and local authorities at the scale of a river basin or a city, forecasting scenarios should be evaluated in order to inform on their potential environmental impacts. There is a need for a common formalism and associated tools that can model water users, water technologies and water resources in an integrated way in order to facilitate scenario building and their analysis by decision-makers. Simplified or streamlined tools which have the capacity to provide results with less time and data requirements are needed, as stated by Schulz et al. (2012). This is also relevant when decisions with potential large environmental consequences have to be made in short time. These models should tackle the methodological challenges pointed out in this review.

(ii) For day-to-day management of water services done by operators, LCA could be used to select the most interesting solution on an environmental point of view. For instance, it is the case when managing the water production from different DWP which withdraw water at different locations. However, data gathering, temporal and spatial scales as well as uncertainties of current LCA models are a barrier and such an application would require large developments and can't be expected at short term. Particularly, traditional LCA models associated with annual time step are not suited for this goal and dynamic tools running at a hourly or daily time step would be needed, as the one developed by Fagan et al. (2010).

More generally, efforts on communicating and teaching stakeholders with LCA methodology should be made (Corominas et al., 2013). These wider questions related to decision-making are generic in LCA.

2.5. Conclusions

This chapter reviews urban water system LCAs and provides a synthesis and analysis of the main LCI and LCIA results available. It shows that LCA offers an interesting holistic approach for evaluating UWS. This review highlights several recommendations and challenges on the way to conduct the LCA of UWS. These guidelines are summarized below:

- When assessing an integrated UWS as a whole, the definition of the functional unit should include the water user since the function of the system is to comply with users' water demand (both in terms of quality and quantity).
- The multi-functional urban water system LCAs should take advantage of the adaptation of the LCA framework to territorial assessment (Loiseau et al., 2013).
- Forecasting scenarios definition should combine and differentiate changes of water technologies, water users and water resources.
- Boundaries of the system should include each step (construction, operation and deconstruction). A specific focus should be done on civil works associated with the networks.
- Appropriate inventory of all water flows should be provided: water flows within the technosphere, water withdrawn and released to the local environment and water evapotranspiration to the atmosphere (water consumption).
- Mass balance of pollutants (to air, water and soil), particularly nitrogen, phosphorus, Carbon, should be equilibrated along the whole system.
- LCIA developments now enable full and comprehensive multi-criteria assessment of urban water system. Thus mono-criterion approaches such as carbon footprint should be avoided in order to prevent pollution shifting, especially on water related impacts such as eutrophication, ecotoxicity and water deprivation.

- Recent advances in impact assessment models related to water use and water quality (eutrophication, ecotoxicity) should be implemented. Spatial and temporal differentiation at an appropriate scale enables site specific assessments that are useful to assess UWS.
- Efforts should be made to include uncertainty analysis, going beyond the sensitivity analysis.

This review also paves the way for further research, with the aim of developing a standardized approach for assessing the environmental performance of UWS, a current burning issue. This is the aim of the following chapters.

Chapter 3. Assessing water deprivation at the sub-river basin scale in life cycle assessment integrating downstream cascade effects

« *Introspections comme remèdes, une attitude neuve*
Du torrent à la rivière, de la rivière au fleuve
Dans la mer, molécules éparpillées, noyées, évaporées
Retour métempsychique dans les cieux sous forme de nuages chargés
Pluies cycliques, éveil du disque
L'âme purifiée revient sur terre telle un phénix »
Akhenaton – Entrer dans la légende

Chapter 2 identified several methodological challenges for the LCA of UWS. One of the most burning issues is the impact assessment of water deprivation at an appropriate scale. Indeed, physical water deprivation at the midpoint level is assessed in water-related LCIA methods using water scarcity indicators at the river basin scale. Although these indicators represent a great step forward in the assessment of water-use-related impacts in LCA, significant challenges still remain in improving their accuracy and relevance. This chapter presents a methodology that can be used to derive midpoint characterization factors for water deprivation taking into account downstream cascade effects within a single river basin. This effect is considered at a finer scale than the one of a river basin, because water can be withdrawn in one location of the water basin and released in one other, far away; therefore the river basin must be split into different sub-units. The proposed framework is based on a two-step approach. First, water scarcity is defined at the sub-river basin scale with the consumption-to-availability (CTA) ratio, and second, characterization factors for water deprivation (CF_{WD}) are calculated, integrating the effects on downstream sub-river basins. The sub-river basin CTA and CF_{WD} were computed for two different river basins based on runoff data, water consumption data and a water balance. The results show significant differences between the CF_{WD} in a given river basin, depending on the upstream or downstream position. Finally, an illustrative example is presented, in which different land planning scenarios, taking into account additional water consumption in a city, are assessed. This work demonstrates how crucial it is to localize the withdrawal and release positions within a river basin. This chapter refers to the following published paper: "Loubet, P., Roux, P., Núñez, M., Belaud, G., & Bellon-Maurel, V. (2013). Assessing Water Deprivation at the Sub-river Basin Scale in LCA Integrating Downstream Cascade Effects. *Environmental Science & Technology*, *47*(24), 14242–9. doi:10.1021/es403056x"

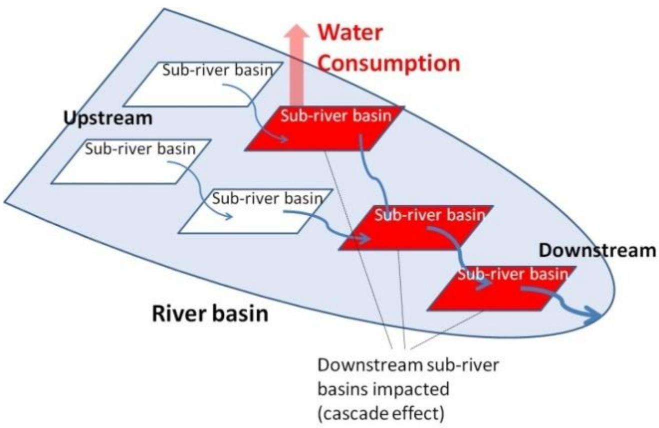


Figure 3-1. Graphical abstract of Chapter 3

Content of Chapter 3

3.1. Introduction

Water scarcity affects a significant share of the world's population and many sensitive ecosystems. It is an increasing threat because of the combination of population growth, economic development and potential regional impacts of climate change on water availability. To address this major environmental issue, methods aimed at assessing the environmental impacts of human activities on water resources have been developed in recent decades. Life cycle assessment (LCA) is a multi-indicator method that estimates the environmental burdens of a product system along its entire life cycle (ISO, 2006a). LCA now complements existing indicators related to water quality issues (pollution) by providing indicators related to the quantitative effects associated with water consumption.

Several life cycle impact assessment (LCIA) approaches for water scarcity have been proposed and compared (Kounina et al., 2012). Recent midpoint approaches (Frischknecht et al., 2006; Lévová and Hauschild, 2011; Milà i Canals et al., 2008; Pfister et al., 2009) evaluate water deprivation using water scarcity indicators that quantify the relationship between water withdrawal or consumption and the amount of water availability at a given location. Midpoint methods are geographically differentiated at the country and river basin scales and take into account water availability heterogeneity around the world. They are all based on global water models, such as Watergap (Alcamo et al., 2003), which evaluate water availability and water consumption. Hoff et al. (2010) reviewed and compared some of these global water models, which also provide water scarcity indicators unrelated to LCA (Hoekstra et al., 2012; Smakhtin et al., 2004; Wada et al., 2011).

While current water deprivation indicators are currently in use to assess water consumption related impacts in LCA, significant challenges still remain in improving their relevance and accuracy. First, they are based on water scarcity indices that compare water demand to the available water in an area. These water scarcity indices are related to the state of the river basin in which the water is consumed. Nevertheless, within the LCA framework, the definition of water deprivation should be related to the downstream effects of a specific human activity: consumption at the source of a river would deprive more users and ecosystems than consumption at the mouth of the same river. According to Vörösmarty et al. (2005), this upstream-downstream perspective is important when considering the needs of competing users. Also, Falkenmark (2000) states that in an integrated basin approach, side effects of water-impacting land use conversions upstream on water-dependent activities and on ecosystem health downstream have to be considered. Thus, differentiation between upstream and downstream water consumption within a river basin should be taken into account. Second, current aggregation scales do not always satisfy the required level of detail (Jeswani and Azapagic, 2011; Milà i Canals et al., 2008). This is especially true for large countries or large river basins with heterogeneous internal water availability and for systems where water is expected to become one of the main environmental issues (Jeswani and Azapagic, 2011). Third, current LCA indicators do not consider the fact that additional water consumption alters the sensitivity of a river basin to water scarcity, which would be encountered in the LCA of a large expansion of an irrigated area or of a growing megapolis (Berger and Finkbeiner, 2013). This means that current midpoint characterization factors are only suitable for the study of systems that are characterized by marginal water consumption. Furthermore, temporal specification beyond the year scale has been mostly disregarded in water scarcity indicators. Although temporally resolved methods take into account the intra-annual variations in water flows, they only provide a single indicator for the whole year (Pfister et al., 2009; Smakhtin et al., 2004). This resolution is adequate for cases in which water consumption is constant throughout the year, but this is not usually the case for the heaviest water users. Only Hoekstra et al. (2012), Wada et al. (2011) and Pfister and Baumann (2012) have calculated monthly water scarcity indicators, thus offering more temporal precision for impact evaluation.

The objective of this chapter is to provide a new reproducible methodology for assessing water deprivation at the sub-river basin scale to better capture the environmental impacts of water consumption at the midpoint level in LCA. This two-step framework aims to define, at the sub-river basin scale, (i) the consumption-to-availability (CTA) ratio and (ii) the water deprivation characterization factor (CF_{WD}). While CTA shows the current water scarcity state of a sub-river basin, the CF_{WD} assesses the cascade effect of water deprivation in a sub-river basin on the downstream impacted sub-river basins. Finally, the methodology is applied to two river basins according to a chosen scale, and an illustrative example is provided through a case study.

3.2. Methods

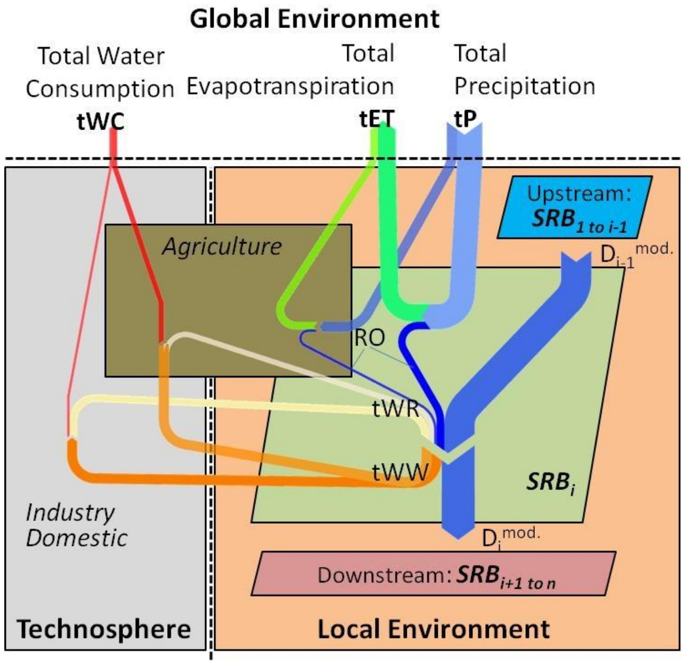
The proposed methodology can be applied at any scale but requires that the assessed river basin be split into different sub-river basins (SRB). The following hypothesis, assumptions and data are chosen to make the methodology applicable on a global scale for the future provision of global indicators.

3.2.1. Water scarcity: consumption-to-availability ratio

The framework is based on a water balance at the sub-river basin scale (Figure 3-2) that aims to define water consumption and water availability.

3.2.1.1. Water balance

Three compartments can be distinguished: (i) the global environment, i.e., the global water cycle, (ii) the local environment, which is the local water cycle within the considered river basin, and (iii) the technosphere, which represents the human activities within the river basin. A river basin is defined as the total land area that drains water to a sea or an ocean. The LCA literature also uses the term watershed instead of river basin (Pfister et al., 2009). The river basin is considered to be linear and is divided into n sub-river basins, from SRB_1 (i.e., the most upstream position) to SRB_n (i.e., the most downstream position). Three sub-compartments are defined based on their relative location within the river basin, i.e., the assessed sub-river basin, denoted SRB_i, its upstream sub-river basins, denoted $SRB_{1\ \text{to}\ i-1}$, and its downstream sub-river basins, denoted $SRB_{i+1\ \text{to}\ n}$.



a=agriculture, id=industrial and domestic, t=total, P=Precipitation (m^3), ET=Evapotranspiration (m^3), RO=generated runoff in SRB_i (m^3), D_{i-1}=discharge from upstream sub-river basin (m^3), D_i=discharge to downstream sub-river basin (m^3), WW=water withdrawal (m^3), WR=water release (m^3), WC=water consumption (m^3).

Figure 3-2. Water balance at the sub-river basin scale.

Three types of flow enter and leave the assessed sub-compartment SRB_i. (i) “Global environment” water flows consist of the input precipitation (P) and the output evapotranspiration (ET). The difference between P and ET generates local runoff (RO_i) on

SRB_i. (ii) "Local environment" water flows consist of the input discharge D_{i-1} from upstream SRB_{i-1} and the output discharge D_i to downstream SRB_{i+1}. (iii) "Technosphere" water flows represent the input water release (tWR) and water withdrawal (tWW) for human activities. It should be stated that the quality of the water that is released to the environment might be different than that of the withdrawn water. This is an important issue currently assessed in LCA only by water quality indicators at the midpoint level, such as eutrophication, ecotoxicity and acidification. Pathways linking quality changes at the midpoint level and potential effects at the endpoint level (i.e., the effect of water deprivation due to quality losses on humans) are addressed by Boulay et al. (2011) and still need to be improved (Kounina et al., 2012). These issues are not within the scope of the present chapter, which only addresses quantitative issues.

3.2.1.2. Water consumption definition

The difference between withdrawal and release is the total water consumption output (tWC). tWC takes into account evaporation, transpiration and water incorporated in products within the technosphere. tWC crosses the boundary between the technosphere and the global environment, and the water represented by tWC is no longer available to the local environment. In Figure 3-2, two types of human activities are distinguished, i.e., agriculture and industry & domestic use.

Agriculture is a specific human activity in terms of water consumption because system boundary between the technosphere and the local environment is not easy to define and not consensual. Evapotranspiration on agricultural fields that comes from precipitation (often called green water in the literature)(Hoekstra et al., 2011) could be considered as human-activity related water consumption. Nevertheless, we choose to assign irrigation-fed fields to the technosphere, and natural precipitation-fed fields to local environment. Only water evapotranspiration occurring in the technosphere is considered as water consumption. The LCA community is currently discussing the indirect impacts on downstream water availability linked to evapotranspiration changes under human land occupation compared to a reference situation (Núñez et al., 2013a). This issue is closely linked to the current efforts to define a natural terrestrial land reference state. The discussion of these wide-ranging issues is beyond the scope of this chapter.

According to Wada et al. (2011), if groundwater is drawn at a renewable rate, i.e., the extraction does not outstrip recharge, it can be considered as surface water because both flows are interconnected and groundwater withdrawal would only decrease river base flow. If groundwater abstraction exceeds the natural recharge, overexploitation occurs, and there is groundwater depletion. We consider non-renewable as well as fossil groundwater depletion to be a different environmental issue to that of renewable water deprivation because it affects abiotic water resources (Milà i Canals et al., 2008), as shown in Figure 3-3. Global water depletion has been quantified by Wada et al. (2010) We subtracted this amount of groundwater depletion from total water consumption to get the total blue water consumption.

Based on the above assumptions, the chapter focuses now on the impacts caused by total surface water (from river and renewable groundwater) consumption. tWC data at a resolution of 5 arc minutes are taken from Hoekstra et al. (2012).

3.2.1.3. Water availability definition

Van Beek et al. (2011) defined three hydrologic regimes, reflecting different human interference levels, to evaluate water availability. Natural discharge ($D^{nat.}$) is the discharge that would occur without any human interference, regulated discharge ($D^{reg.}$) is that in which natural discharge is altered by reservoir operations, and modified discharge ($D^{mod.}$) is the regulated discharge minus the total water consumption resulting from human activities. Here, we assume that the total water availability (tWA) in a sub-river basin is the regulated discharge. This assumption has been established by Smakhtin et al. (2004) and Hoekstra et al. (2012) Natural discharge could be used as a reference when assessing the impact of anthropogenic flow regulation systems, such as large reservoirs.

Discharge data at a 30-arc minute resolution were obtained from the Composite Runoff v1.0 database (Fekete, 2002). This database provides modified runoff data ($RO^{mod.}$) that take into account human activities (reservoir operations and water consumption) by combining a simulated runoff model and actual river discharge measurements from gauging stations managed by the Global Runoff Data Centre.

From this database, the modified discharge $D^{mod.}$ for each SRB_i is computed as the sum of upstream modified runoff (eq (3)):

$$D_i^{mod.} = \sum_{j=1}^{i} (RO_j^{mod.}) \tag{3}$$

Because the modified discharge is the regulated discharge minus total water consumption, upstream tWC is added to $D^{mod.}$ to get $D^{reg.}$ (eq (4)):

$$D_i^{reg.} = D_i^{mod.} + \sum_{j=1}^{i} tWC_j \tag{4}$$

As mentioned above, tWA is equal to $D^{reg.}$. Environmental Water Requirements (EWR), i.e., the flows needed to maintain ecological functions, are accounted for in different ways in global water scarcity indicators. Smakhtin et al. (2004) and Hoekstra et al. (2012) subtracted EWR from the total available water. Pfister et al. (2009) took into account EWR by assuming that water scarcity does not vary linearly with water availability; they chose to modify their water stress index by calculating nonlinear values set between 0.01 and 1. We used the first option, i.e., we assumed that the real available water WA, is the difference between tWA and EWR. EWR is not defined for each river basin or sub-river basin because its evaluation must take into account the hydrological properties of the river. Richter et al. (2012) have proposed a presumptive standard for environmental flow protection to be used in cases where river basin-specific studies have not yet been performed. It is stated that a moderate level of protection is provided when flows are altered by 11-20%. In this case, there will be minimal changes in ecosystem functions. In keeping with Hoekstra et al. (2012), EWR was set at 80% of tWA. Therefore, the real water availability in SRB_i, denoted by WA_i, is:

$$\mathrm{WA_i} = (1 - \%\mathrm{EWR}) \cdot \mathrm{D_i^{reg.}} \tag{5}$$

Where %EWR is the percentage of tWA that can be consumed without causing any change to the ecosystems (unitless).

3.2.1.4. Consumption-to-availability ratio

In current LCA indicators (Frischknecht et al., 2006; Lévová and Hauschild, 2011; Milà i Canals et al., 2008; Pfister et al., 2009), the withdrawal-to-availability (WTA) ratio is routinely chosen to characterize water scarcity. The difference between “water withdrawal” and “water consumption” is that water consumption takes into account water that is returned to the flow (i.e., withdrawal minus release). It appears that water consumption is more relevant when water scarcity issues are being addressed in LCA because released water is made available again in the ecosystem for new users (Bayart et al., 2010). In these conditions, we apply the CTA ratio as previously done or suggested (Berger and Finkbeiner, 2013; Boulay et al., 2011b; Hoekstra et al., 2012; Wada et al., 2011).

$$\mathrm{CTA} = \frac{\mathrm{tWC}}{\mathrm{WA}} \tag{6}$$

Where tWC is the total water consumption (m^3) and WA is the available water (m^3) in the river basin. When tWC is above 20% of tWA, CTA is above 1 and there is a moderate to major change in natural structure and ecosystem function.

This definition is modified at the sub-river basin scale (illustrated by the Figure B-1 in Annex B.1). In an SRB_i, the total local water consumption (tWC_i) plus the upstream water consumption ($tWC_{1 \text{ to } i-1}$) lowers the local water availability (WA_i), quantified by:

$$CTA_i = \frac{\sum_{k=1}^{i} tWC_k}{WA_i} \tag{7}$$

This ratio shows the local water scarcity, which is a characteristic of SRB_i, but does not depict how specific water consumption in this SRB_i would lower downstream water availability.

3.2.2. Characterization factors for water deprivation

The LCA literature always considers a river basin as a whole and does not provide values for specific locations within the river basin. It is assumed that the total water consumption (or withdrawal) within a river basin affects the water availability of the river basin, as depicted by eq (6).

In reality, the water consumed at a specific location only affects SRBs downstream from this location: specific water consumption in SRB_i will affect SRB_i to SRB_n. This causes a cascade effect on potential downstream usages and ecosystems, something that is not captured by water scarcity indicators. This effect can be measured by the sum of downstream CTA ratios. Nevertheless, downstream SRBs are affected differently because they vary in terms of area, water volume, density of population, etc. This means that each downstream impact should be weighted by a chosen parameter p (area, water discharge, etc.). Consequently, the characterization factor for water deprivation in SRB_i is the weighted sum of all downstream CTA ratios (including itself), as described in eq (8):

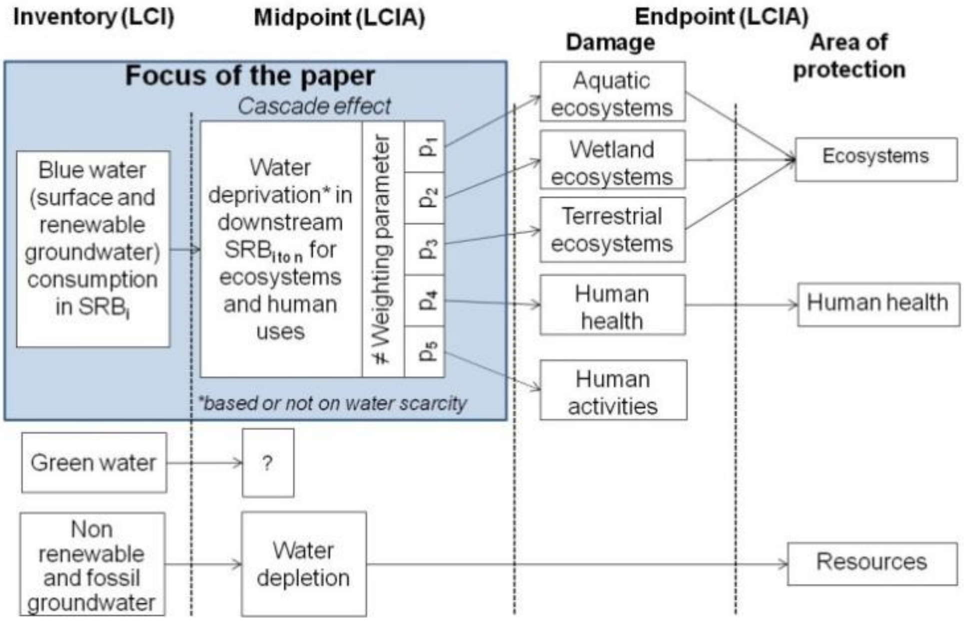
$$CF_{WD,i} = \frac{1}{\overline{p} \cdot \overline{N}_{down}} \sum_{j=i}^{n} \left(CTA_j \cdot p_j \right) \tag{8}$$

Where p_j is the chosen weighting parameter of downstream SRB_j, $\overline{p}$ is the average value of the weighting parameters among all the SRBs within the river basin and $\overline{N}_{down}$ is the average number of SRBs downstream from each SRB within the river basin. The CF_{WD} fulfills two requirements: (i) For each SRB_i, the product of CTA_j and p_j must be constant because regardless of the location of upstream water consumption, the impact of this upstream consumption on SRB_i will be the same; (ii) the average CF_{WD} within a river basin should be in the range of the river basin CTA value in order to be able to compare CF_{WD} of SRBs with CTA ratios at a higher scale (river basin or country).

3.2.3. Midpoint assessment: choice of the weighting parameter

The CF_{WD} aims to assess downstream water deprivation. It is based on water scarcity and thus can be used as a midpoint indicator as done by previous authors (Boulay et al., 2011b; Frischknecht et al., 2006; Milà i Canals et al., 2008; Pfister et al., 2009) and suggested within the framework of Bayart et al. (2010) and the review of Kounina et al. (2012). The choice of the weighting parameter can differently reflect the downstream potentially affected entities (human population, terrestrial areas and freshwater volumes). It is important to note that there is not yet empirical evidence of a link between midpoint indicators based on water scarcity and damages on ecosystems (Kounina et al., 2012). This midpoint-endpoint link is further discussed in the "Environmental relevance" section (Discussion chapter) where it is demonstrated how the downstream cascade effect could be adapted to endpoint indicators that are not based on water scarcity.

We consider the following hypotheses, which assume a homogeneous climate and ecosystem river basin: the terrestrial species potentially affected within an SRB are a function of the SRB surface, the aquatic species potentially affected are a function of the river water volume, the wetland-dependent species potentially affected are a function of the wetland surface area, and human health is related to the amount of users that are deprived of water. The proposed weighting parameters area, river volume and number of inhabitants are applied in the calculation of three different CF_{WD} values. The SRB areas are taken from the HYDRO1k database (U.S. Geological Survey Center for Earth Resources Observation and Science, 2004). The river volumes are calculated as done by Hanafiah et al. (2011) (see Annex B.6). The numbers of inhabitants are taken from the GPWV3 database (Center for International Earth Science Information Network (CIESIN) and Centro Internacional de Agricultura Tropical (CIAT), 2005).



p1=volume within the rivers, p2=wetland area, p3=sub-river basin area, p4=population.

Figure 3-3. Summary of cause-effect chains leading from water consumption inventory to different areas of protection, adapted from Kounina et al. (2012)

3.2.4. Water deprivation midpoint impacts

Potential midpoint impacts on the water deprivation of a studied system are calculated based on the difference between water withdrawal and water release, characterized by their respective CF_{WD} values, as previously done by Boulay et al. (2011):

$$I_{WD} = WW \cdot CF_{WD,A} - WR \cdot CF_{WD,B} \tag{9}$$

where I_{WD} is the midpoint impact of water deprivation (m^3 equivalent or m^3 eq.), WW is the water withdrawal volume of the studied system that occurs at location A (m^3), WR is the water release volume of the studied system that occurs at location B (m^3), and $CF_{WD,A}$ and $CF_{WD,B}$ characterize locations A and B, respectively. If WW and WR occur at the same location A, then I_{WD} can be simplified as the product of WC and $CF_{WD,A}$.

3.2.5. Identifying upstream and downstream SRBs to streamline CTA and CF_{WD}

Figure 3-2 shows a linear river basin scheme. In reality, river basin topology is much more complex and is generally composed of many tributaries.

Tributaries of a specific sub-river basin were extracted from the HYDRO1k drainage basin database (U.S. Geological Survey Center for Earth Resources Observation and Science, 2004). This database covers a global scale while offering a 0.5 arc minute resolution. Runoff and water consumption data previously defined are available at the grid cell scale. They are obtained for the different sub-river basins as the average values of the grid cells contained within the SRB boundaries. The upstream and downstream sub-river basins for each SRB have been identified from the Pfafstetter sub-river basin coding system (Pfafstetter, 1989) provided by the HYDRO1k database. From this identification and RO_i and tWC_i data, CTA_i and $CF_{WD,i}$ are respectively computed with eq (7) and (8). The reproducible procedure is available in the Annex B.3.

3.2.6. Illustrative case study

3.2.6.1. Characterizing river basins

The proposed framework was applied to calculate the CF_{WD} on two river basins: the Seine, in France, and the Guadalquivir, in Spain. These river basins were chosen due to their diverse climatic conditions and because they face high human pressure (Hoekstra et al., 2012).

3.2.6.2. Assessing land planning scenarios

To apply sub-river basin scale CF_{WD} values to a case study, we assessed the impacts of a hypothetical urbanization development on different water bodies in the greater Paris area. This development would attract 200 000 inhabitants, as well as industries that would withdraw an additional 8 million cubic meters of water every year. We assumed that 90% of the water withdrawal would be released to the environment, i.e., 7.2 million m^3. Figure 3-4 shows the locations of the different withdrawal and release location options: WW at point A (SRB id41) at a current drinking water plant on the Oise river, WW at point B (SRB id20) at the source of the Eure river where an aqueduct conveys water to Paris, and WR at point C (SRB id30), located at the current wastewater treatment plant of Achères. Two scenarios were analyzed, each one combining one withdrawal and one release location option: (S1) WW at point A, WR at point C and (S2) WW at point B, WR at point C. In the different scenarios, withdrawal and release do not occur in the same SRB. The midpoint impact of water deprivation is calculated from eq (9). The scenarios are then compared to the case where geographic location within the river basin is not taken into account: CTA is considered as the river basin CF_{WD}, as is typically the case in LCA.

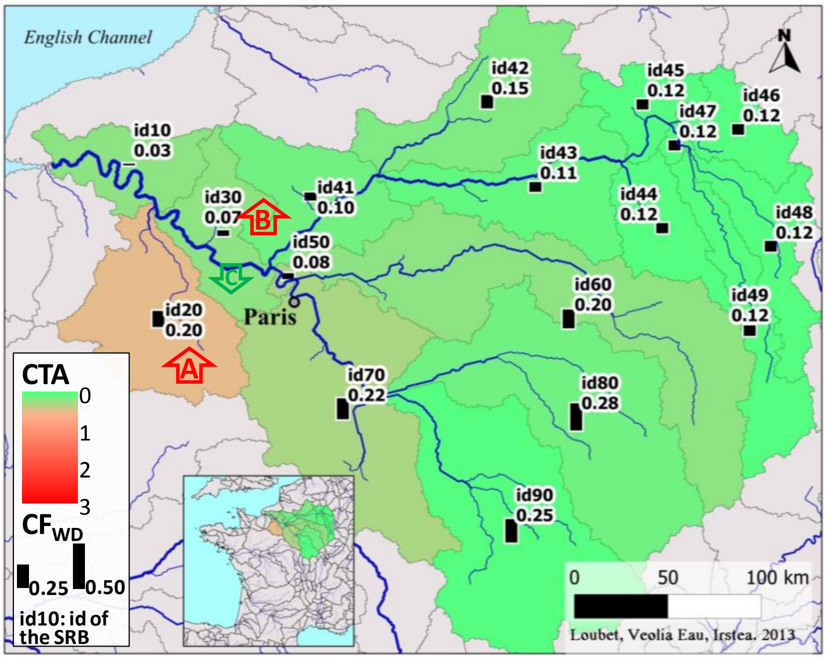
3.3. Results

3.3.1. CTA and CF_{WD} for selected sub-river basins

Figure 3-4 and Figure 3-5 show CTA and the CF_{WD} (p = area) of the SRB constituting the Seine river basin and the Guadalquivir river basin (full results and raw data are presented in Annex B.4: Table B-3 and Table B-4). Obviously, SRBs that have the highest CF_{WD} are located at the source of the rivers because water consumption in these locations affects a greater downstream area. SRBs located at the mouth of the river basin (i.e., the most downstream position) have the lowest CF_{WD} because no other downstream SRBs are affected by their water consumption. This is consistent with the fact that seawater is an unlimited resource and does not contribute to any related water consumption impacts (Milà i Canals et al., 2008). Low CF_{WD} values in downstream locations guarantee continuity with the CF_{WD} of seawater resources, which are equal to 0.

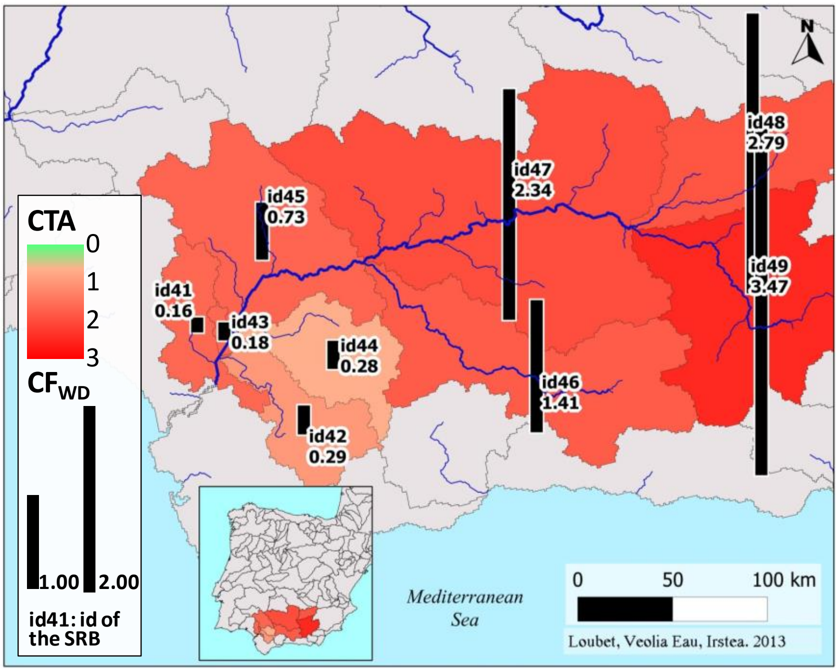
As shown in Figure 3-6, the CF_{WD} inevitably decreases in downstream SRBs regardless of the weighting parameter p (area, volume, inhabitants). Nevertheless, there are differences depending on the choice of p. When p is the volume, the CF_{WD} values are generally much higher in the selected lines because each SRB affects the last downstream SRB, which has a high volume. Thus, the effect of water deprivation on aquatic ecosystems is high in the most downstream SRB, and all CF_{WD} values increase. Area- and population-weighted CF_{WD} results follow the same trends in the first selected line of the Seine and the selected line of the Guadalquivir because the population density is well-distributed within these SRBs. However, in Figure 3-6.b, the CF_{WD} increases noticeably in the upstream position because most of the greater Paris area is in SRB id70. Consequently, water consumption in SRBs id70 and 90 deprives this large share of the population. In this case, it should be noted that the human health of the population will not be damaged because the region is developed and compensation with backup technology can occur (Boulay et al., 2011b). Nevertheless, damages are not accounted for in the present framework, only the water deprivation is quantified.

CTA does not follow any specific trends. Depending on local conditions, CTA can alternatively increase in downstream SRBs (Figure 3-6.a) or first decrease and then increase (Figure 3-6.c). In addition, CTA_n, which characterizes the most downstream SRB (at the mouth of the river), is equal to the river basin CTA. In fact WA_n is equal to $WA_{river\ basin}$ because they are both the total sum of runoff occurring in the river basin, and $tWC_{1\ to\ n}$ is equal to $tWC_{river\ basin}$ because they are both the total sum of water consumption occurring in the river basin. Table B-5 in Annex provides a comparison between the different methods used to assess water scarcity at the river basin scale.



Numbers give the simplified SRB coding (idxx) from the Pfafstetter system (two last digits) and the CF_{WD}. Red and green arrows show the pumping and release locations of the illustrative example.

Figure 3-4. Sub-river basin CF_{WD} (p=area) and CTA of the Seine river basin (France)



Numbers give the simplified SRB coding (idxx) from the Pfafstetter system (two last digits) and the CF_{WD}.

Figure 3-5. Sub-river basins CF_{WD} (p=area) and CTA of the Guadalquivir river basin (Spain)

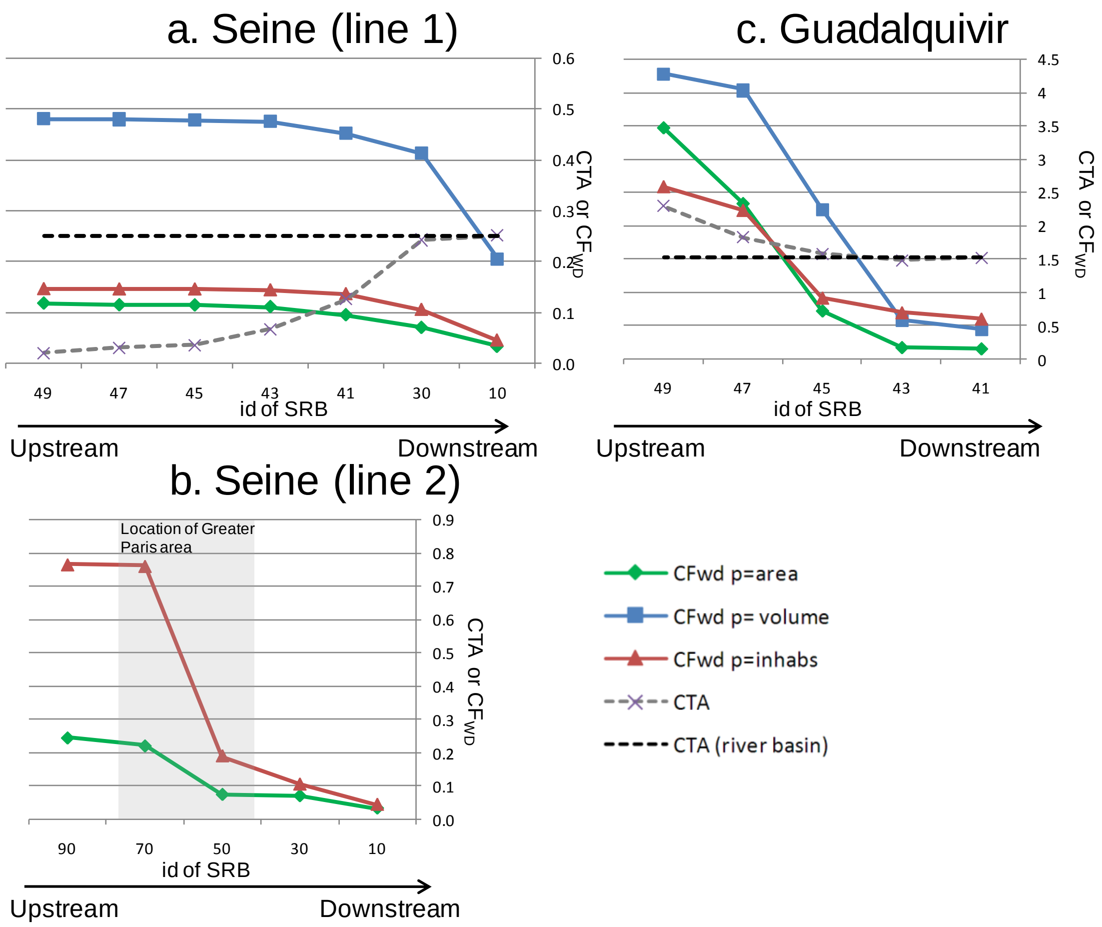


Figure 3-6. CF_{WD} and CTA evolution from upstream to downstream locations in three selected lines.

3.3.2. Results of land planning scenarios

Water deprivation impacts (I_{WD}) are calculated with eq (9) and p=area. The I_{WD} for scenario S1 (WW in point A and WR in point C) and scenario S2 (WW in point B and WR in point C) are 0.88 Mm^3 eq. and 0.26 Mm^3 eq., respectively. These results show how diverse the impact on water deprivation can be depending on the withdrawal and release locations. If the water deprivation impact were calculated using the river basin scale CTA ratio (S3), the result would be 0.20 Mm^3 eq. for both, which is lower than the values obtained in the two scenarios because scenarios S1 and S2 describe two situations where the water is released downstream from the withdrawal position.

3.4. Discussion

The method was evaluated against the specific criteria for water use impacts defined in the ILCD handbook (EC - JRC - IES, 2010c). Following the guidelines provided in this handbook facilitated the comparison between our method and other approaches used to assess water use in LCA.

3.4.1. Completeness of scope

The framework provides a methodology to account for the environmental impacts related to water deprivation in LCA. The method focuses on the impacts on downstream ecosystems and users linked to water consumption from rivers and renewable aquifers. It takes into account downstream cascade effects. Degradative use (water quality alteration) is not considered. We propose to use the concept of "water deprivation" in LCA, which is assessed with the CF_{WD}, instead of the concept of "water scarcity", which is assessed with CTA. Because the indicator is regionalized at the SRB scale, it captures the variability of water availability within the same river basin, thus providing a geographically detailed CF. The SRB spatial differentiation is relevant for foreground systems where water consumption is of high importance for the river basin management: urban water system, large irrigated areas, etc. Concerning conventional products LCAs, focusing on local impacts from the whole product system is not necessary (Hauschild, 2006) and a river basin or country scale approach remains sufficient (i.e. for background systems and foreground systems without identified water issues).

3.4.2. Environmental relevance

This framework aims to assess the effects of water use on downstream deprivation at midpoint level but not the potential damages associated to deprivation (endpoint level). By weighting the CF_{WD} by either the area, the water volume or the population of downstream SRBs, different midpoint indicators are proposed, reflecting potential water deprivation on downstream users and ecosystems. Other weighting parameter can be applied, such as water withdrawals of specific activities and thus can depict the deprivation on human usages which are not directly linked to population, e.g. agriculture or specific industries.

The cascade effect methodology could also be used to calculate current endpoint indicators at the SRB scale. As methods covering human health mostly use water scarcity indicators (Boulay et al., 2011b; Pfister et al., 2009), our proposed indicator could be adapted to their calculation at SRB scale. Most of the existing endpoint approaches considering ecosystem quality do not include water scarcity as an element of the equations (Kounina et al., 2012) (e.g., methods assessing impacts on terrestrial (Pfister et al., 2009), aquatic (Hanafiah et al., 2011), or wetland ecosystems (Verones et al., 2012)). In these cases, the present methodology for assessing downstream cascade effects could also be applied by replacing CTA with the appropriate effect factor. As an illustration of the applicability, we have adapted the CFs developed by Hanafiah et al. (2011) to consider downstream cascade effects on freshwater fish species at the endpoint level (adaptation and results are available in Annex B.6). As soon as a consensus will be reached for a set of consistent midpoint and endpoint indicators, the cascade effect methodology could be applied on water scarcity and non-water scarcity based indicators (including different weighting). Resource depletion falls outside the scope of the present study because this area of protection is only damaged by water withdrawals made from non-renewable and fossil groundwater, as shown in Figure 3-3.

3.4.3. Scientific robustness and certainty

Uncertainties, as well as the geographic and temporal resolution of the models, are discussed below, and the main needs for further developments are identified.

(i) The HYDRO1K database, which provides SRB boundaries, is based on the USGS 30 arc-second digital elevation model (DEM) of the world (GTOPO30). GTOPO30 is a digital elevation model and can be inconsistent when dealing with flat areas as it can lead to the generation of incorrect sub-river basin boundaries. This was the case with the Seine river basin, where two SRBs located in the eastern part actually belong to the Meuse river basin. This problem is recurrent in DEM (Holmes et al., 2000) and can be tackled by using more accurate databases (e.g., BD TOPO® for France). However, such databases are not available on a world-wide scale. Here, the choice of geographical scale is practical because the database used is available globally and provides physical SRBs and a river ordering scheme. We suggest using this database for global application.

(ii) In global water models, tWC is estimated from GIS data related to population density, country statistics and land cover data. Moreover, water withdrawal and release do not automatically take place in the sub-river basin or river basin where water consumption actually occurs (for example, a city that draws water from canals or long-distance pipes). It would be more relevant to use water withdrawal and release location information, i.e., a bottom-up approach instead of a top-down one. This type of data is much more difficult to obtain than GIS–based water consumption estimates. In addition, water can be withdrawn from desalinized sea and ocean water, which are not accounted for in this framework. Thus, these inputs should not be accounted for in blue water deprivation. Recently, Wada et al. (2011) have taken into account such inputs and have subtracted them from water demand. Further developments should take into account these advances. Regarding temporal resolution, seasonality should be applied to domestic and industrial water consumption, as has been recently done with monthly data (Hoekstra et al., 2012; Wada et al., 2011). Additionally, in the case of non-marginal water consumption of a studied system, CF_{WD} recalculations would be necessary to take into account additional WC within tWC.

(iii) Water availability is calculated from runoff and water consumption databases and needs the same refinement as specified above. Moreover, the environmental water requirement definition, although taken from the literature, is somewhat arbitrary and merits further study. Lastly, monthly values for water availability should be computed.

3.4.4. Documentation, transparency and reproducibility

Sub-river basin topology, WA and WC data are available and accessible online on a world-wide scale. However, as detailed above, more recent and accurate databases (van Beek et al., 2011; Wada et al., 2011) can be used to recalculate the CF_{WD} using our reproducible method.

3.4.5. Applicability

This methodology is mainly targeted to LCA practitioners who study foreground systems where water is a main issue. This requires having an access to inventory data at the local scale in particular withdrawals and discharges locations which are generally available from the stakeholders who intend to study systems with a focus on water.

In cases where several alternatives of water withdrawal locations are available within a given river basin (e.g., for irrigated land area or water provision in big cities), land planners need tools to assess the relevancy of the various water resources options. This is also the case when the withdrawal location is far away from release location (e.g., water transfers between river basins). The illustrative example confirmed that the localization of water withdrawal and release within a river basin is important because it can lead to different impacts and demonstrated the applicability of the methodology.

The localized assessment of water consumption impacts can also be useful for the emerging territorial LCAs which assess land planning options within a territory (Loiseau et al., 2013). Beyond the scope of LCA, water managers could use this indicator as a stand-alone one for comparing different resources in an upstream/downstream perspective. UN Water notes that imbalances between availability and demand, intersectoral competition and interregional and international conflicts all bring water issues to the fore (UN-Water, 2006). The proposed framework for assessing water deprivation provides an efficient tool for coping with these challenges at a proper scale, i.e., the sub-basin.

3.4.6. Outlook

It is intended to develop the CF_{WD} at a world-wide scale, as done by current LCA indicators. To be able to generalize the methodology at this scale, the main simplifications made in this study would have to remain. Average river basin and country scale CFs could also be calculated. Finally, following the recommendations of Mutel et al. (2012) regarding the spatial scale of impact assessment, minimization of global spatial autocorrelation should be applied to aggregate small spatial units and build typologies of sub-river basins.

Chapter 4. Accounting for quality of urban water flows taking into account existing LCIA and water footprint methods

« *Don't go near the water*

Don't you think it's sad

What's happened to the water

Our water's going bad »

Beach boys – Don’t go near the water

In addition to water quantity issues presented in Chapter 3, impact assessment associated with water quality should be taken into account in LCA of UWS. Chapter 4 proposes to review different LCIA method to assess water quality of urban water flows from their associated nutrient and chemical composition. Damage scores of urban water flows (e.g., water resources, wastewater, etc.) are computed with Impact 2002+, ILCD and ReCiPe, and compared to damage scores of states of water from the water framework directive (WFD). These damage scores are also used to build up an advanced water quality indicators for the Water Impact Index (WIIX), a water footprint single score. From the results, a classification of urban water flows according to their associated damage scores is built. It classifies urban water flows into five main types, in order to implement it in the model presented in the Chapter 5.

Content of Chapter 4

4.1. Introduction

Water can be polluted from many chemical substances emitted by human activities. This is a threat to water users including human and ecosystems. Ensuring good quality within the environment is a growing challenge in order to cope with all water usages. In this context the evaluation of water quality is typically done through the analysis of the composition of water. However, the analysis of raw composition of water including lots of parameters (chemical, biological, physical) can be difficult to communicate. The use of water quality indices simplifies this large amount of data provided by water analysis, by aggregating the information into a single indicator. Such indices have been widely developed in the past, for example to strengthen communication for the public , to better inform decision makers (Carvalho et al., 2010; Dadolahi-Sohrab et al., 2012) or to develop high scale policies such as the European water framework directive (WFD) (Official Journal of the European Communities, 2000).

In addition, there is an increasing demand from industries for developing single scores for water footprint, which would include water quality assessment. Several single score indicators which take into account water quality have been developed such as the Water Impact Index (WIIX) (Bayart et al., 2014), the single-score stand-alone water footprint index of Ridoutt and Pfister (2012) or the method from the Water Footprint Network (Hoekstra et al., 2011). According to the recent international standards (ISO, 2013), water footprint should refer to the potential impact occurring because of water use and pollutions. This is done with characterization factors (CF) in life cycle assessment (LCA). CFs quantifies the extent to which each emission (to air, soil or water) contribute to different environmental impacts and damages. They therefore enable to aggregate amounts of chemical compounds diffused in the media, into impact or damage scores. Life cycle impact assessment (LCIA) methods, which carry out this transformation, take into account fate, exposure and effect of the pollutants, thus strengthen the evaluation of their potential impacts and damages. Ridoutt and Pfister (2012) use LCIA methods in order to build their water footprint single score. However, no study has fully explored and discussed the use of LCA impact and damage score computation methods to build water quality indices and classify water flows of urban water systems (UWS).

The objectives of this chapter are: (i) to compare damage scores of natural water resources with classification of water from the water framework directive, (ii) to analyze the water quality of urban water flows according to their associated damage scores and different LCIA methods, (iii) to apply the damage scores of urban water flows to a simplified water footprint methodology, and (iv) finally, it is to build a typology of urban water flows based on these results, in order to implement it in the it in the model we will develop in Chapter 5.

4.2. Material and methods

The proposed methodology follows three steps: (step 1) identification of typical water flows found in UWS and definition of their chemical composition, (step 2) characterization and aggregation of the water flow compounds in damage scores and (step 3) implementation of the proposed damage score to an existing water footprint method (advanced water impact index - WIIX). These three steps are presented in the following sections.

4.2.1. Identification of urban water flows and their associated composition

Different types of water associated with diverse quality can be identified in UWS:

- Natural water resources: surface water, ground water, sea water, rainwater
- Produced water: drinking water, industrial water
- Raw wastewater generated by users: domestic wastewater, industrial wastewater
- Water effluents from drinking water plants (DWP effluent)
- Water effluents from wastewater treatment plants (WWT effluent)

Twelve water flows are selected from ecoinvent process data dealing with urban water (i.e., input and output of WWT) and French context data, as detailed in Table 4-1. In addition, a set of 2534 analyses of water corresponding to different measurement stations within the French basins of Garonne, Loire, and Seine have also been selected to make a focus on natural water resources.

Table 4-1. Composition of selected water flows for nutrients and metals (non-exhaustive list). Concentrations highlighted in grey are not known and taken equal to the ones associated to a very good state

		Natural water resources					Produced water	Raw wastewater			Drinking water plant effluent	Wastewater plant effluent	
	Water	WFD : very good state	WFD : good state	WFD : moderate state	WFD : poor state	Seine river	Drinking water	French context wastewater	ecoinvent wastewater	Highly polluted wastewater	DWP effluent France	WWT effluent France	WWT effluent ecoinvent
	Source →	(Ministère de l'écologie et du développement durable, 2005)				(AESN, 2014)	(SEDIF, 2012)	Irstea	(Doka, 2009)	(Henze and Comeau, 2008)	(SEDIF, 2012)	(SIAAP, 2012)	(Doka, 2009)
Pollutants	CAS												
COD	-	2.00E+01	3.00E+01	3.00E+01	3.00E+01	1.77E+01	3.50E-01	6.46E+02	1.55E+02	1.20E+03	1.17E+01	5.50E+01	2.75E+01
BOD	-	3.00E+00	6.00E+00	6.00E+00	6.00E+00	2.09E+00	3.00E+00	2.65E+02	1.04E+02	5.60E+02	3.00E+00	1.30E+01	8.15E+00
Phosphore total (Pt)	7723140	5.00E-02	2.00E-01	5.00E-01	1.00E+00	1.57E-01	1.00E-02	9.40E+00	3.07E+00	2.50E+01	5.00E-01	9.00E-01	8.49E-01
Ion ammonium (NH4+)	14798039	1.00E-01	5.00E-01	2.00E+00	5.00E+00	8.75E-01	3.00E-02	5.49E+01	1.92E+01	9.64E+01	1.00E-01	9.51E+00	1.10E+01
Nitrate (NO3-)	14797650	1.00E+01	5.00E+01	5.00E+01	5.00E+01	2.45E+01	1.81E+01	2.50E+00	4.65E+00	1.11E+00	3.18E+00	4.25E+01	4.83E+01
Nitrite (NO2-)	14797650	1.00E-01	3.00E-01	5.00E-01	1.00E+00	5.16E-01	1.00E-02	4.00E-01	1.31E+00	8.21E-01	6.00E-02	1.00E-01	6.44E-01
Cadmium (Cd)	7440439	7.50E-05	7.50E-05	1.50E-04	1.50E-04	7.50E-05	1.00E-08	2.54E-04	2.81E-04	4.00E-03	7.50E-05	2.81E-04	2.81E-04
Mercury (Hg)	7439976	2.50E-05	2.50E-05	5.00E-05	5.00E-05	1.63E-05	1.00E-08	5.36E-04	2.00E-04	3.00E-03	1.50E-04	2.00E-04	2.00E-04
Arsenic (As)	7440382	2.10E-03	2.10E-03	4.20E-03	4.20E-03	1.01E-03	1.00E-08	1.49E-03	9.00E-04	2.10E-03	2.10E-03	4.20E-03	4.20E-03
Aluminum (Al)	7429905	1.00E-01	1.00E-01	2.00E-01	2.00E-01	1.00E-01	1.00E-08	1.20E+00	1.04E+00	1.00E+00	1.36E+00	1.04E+00	1.04E+00
Iron (Fe)	7439896	5.00E-02	5.00E-02	5.00E-02	5.00E-02	5.00E-02	1.00E-08	1.60E+00	7.09E+00	5.00E-02	5.00E-02	7.09E+00	7.09E+00
Chromium (Cr)	7440473	1.70E-03	1.70E-03	3.40E-03	3.40E-03	7.49E-04	1.00E-08	1.35E-02	1.22E-02	4.00E-02	1.70E-03	1.22E-02	1.22E-02
Copper (Cu)	7440508	7.00E-04	7.00E-04	1.40E-03	1.40E-03	2.25E-03	1.00E-08	8.49E-02	3.74E-02	1.00E-01	3.82E-03	3.74E-02	3.74E-02
Lead (Pb)	7439921	3.60E-03	3.60E-03	7.20E-03	7.20E-03	3.60E-03	1.00E-08	2.31E-02	8.63E-03	8.00E-02	8.10E-04	8.63E-03	8.63E-03
Zinc (Zn)	7440666	3.90E-03	3.90E-03	7.80E-03	7.80E-03	8.01E-03	1.00E-08	1.88E-01	1.09E-01	3.00E-01	9.80E-03	3.24E-02	3.24E-02

A defined set of pollutants was chosen from the European water framework directive (WFD) (Official Journal of the European Communities, 2000) and from its application in France, namely the “directive cadre sur l’eau” (Ministère de l’écologie et du développement durable, 2005). WFD classifies water quality according to its biological, hydromorphological and physico-chemical (both of three representing the ecological state) as well as its chemical state. Biological parameters (such as species richness) and hydromorphological parameters (such as hydrological regime) have been disregarded here since they do not correspond to typical inventory data for LCA. Therefore, only parameters defining the physico-chemical and the chemical states have been kept. These parameters were chosen since it is more likely that they are measured, especially for natural water resources, and also because the biological parameters are generally a consequence of the chemical and physico-chemical conditions.

The physico-chemical state is defined according to five states (bad, poor, moderate, good, very good). Each pollutant of the classification is compared to threshold values, as shown in Table 4-2. The water class depends on the worst status found for the pollutants describing the physico-chemical state of water (chemical oxygen demand, nitrogen and phosphorus compounds). Chemical state can only be described by two states, i.e. good or bad, depending on the concentration of fifty chemical compounds classified as “priority compounds” (metals, pesticides, etc.). If one pollutant exceeds the threshold between good and bad quality, the water is automatically classified as “bad chemical state”.

Compositions of selected water flows are inventoried from various sources. The missing data concerning pollutant concentrations for each water flow has been managed following the rule of thumb, hereafter: when a pollutant concentration was unknown for a water flow, it was set as equal to the threshold of the very good state of water for nutrients and half the threshold between good and bad state for chemical compounds. Nutrient and metal concentrations are presented in Table 4-1.

Table 4-2. Threshold values for the definition of physico-chemical state from the water framework directive applied in France

Parameters	Thresholds values defining the state				
	Very good	Good	Moderate	Poor	Bad
Oxygen					
Dissolved oxygen (mg O_2/L)	8	6	4	3	<3
BOD_5 (mg O_2/L)	3	6	10	25	>25
Dissolved organic carbon (mg C/L)	5	7	10	15	>15
Temperature					
Salmonid waters (°C)	20	21.5	25	28	>28
Cyprinid waters (°C)	24	25.5	27	28	>28
Nutrients					
Phosphates PO_4^{3-} (mg/L)	0.1	0.5	1	2	>2
Total phosphorus (mg/L)	0.005	0.2	0.5	1	>1
Ammonium NH_4^+	0.1	0.5	2	5	>5
Nitrites NO_2^- (mg/L)	0.1	0.3	0.5	1	>1
Nitrates NO_3^- (mg/L)	10	50	50	50	>50
Acidification					
Minimum pH	6.5	6	5.5	4.5	<4.5
Maximum pH	8.2	9	9.5	10	>10

4.2.2. Characterization of urban water flows

4.2.2.1. Identification and selection of LCIA categories

Each selected water flow is characterized according to the potential impact it would have if released to the environment. This is done in order to aggregate the different pollutants in impact or damage categories found in LCIA methods. Emissions to water compartment may affect aquatic ecosystems (because of freshwater/marine eutrophication and ecotoxicity), but also terrestrial ecosystem, (terrestrial ecotoxicity, terrestrial acidification) and human health (toxicity). These induced impacts are first due to the fact that emissions of pollutant to water have a fate and can be re-emitted to air and soil, and second water can be a pathway for human exposure (Rosenbaum et al., 2008).

There are different LCIA methodologies (e.g., Impact 2002+, ReCiPe, ILCD, etc.) which deliver different impact categories, or, for the same impact category, which may have different characterization factors. Therefore, depending on the LCIA methodologies, midpoint impact and damage categories affected by emissions to water differ. We listed below the considered impacts categories for three different LCIA methods and which area of protection is ultimately affected from emissions to water, i.e., either the ecosystem quality or human health.

Table 4-3. List of impact categories affected by emissions to water for three LCIA methods.

Area of protection affected	Impact type	Impact categories			
		Impact 2002+ (Jolliet et al., 2003)	ReciPe (Goedkoop et al., 2009)	ILCD (EC - JRC - IES, 2010a)	Abbre-viation
Ecosystem quality (E)	Eutrophication	Aquatic eutrophication	Freshwater eutrophication	Freshwater eutrophication (ReCiPe)	FEu
		-	Marine Eutrophication (only midpoint)	Marine Eutrophication (only midpoint, ReCiPe)	MEu
	Ecotoxicity	Aquatic ecotoxicity	Freshwater ecotoxicity	Freshwater ecotoxicity (UseTOX)	FET
		-	Marine ecotoxicity	-	MET
		Terrestrial ecotoxicity	Terrestrial ecotoxicity	-	TET
	Acidification	Aquatic acidification	-	-	AC
	Ionizing radiation	-	-	Ionizing radiation E	IR
Human Health (HH)		Ionizing radiation HH	Ionizing radiation HH	Ionizing radiation HH	IR
	Toxicity	Carcinogens	Human toxicity	Cancer (UseTOX)	HTC
		Non carcinogens		Non cancer (UseTOX)	HTNC

We considered that no ionizing compound is emitted within urban water flows and thus ionizing radiation will not be considered. Each water flow and its associated composition is characterized according to each LCIA method and its associated characterization factors for midpoint impacts and endpoint damages. Endpoint damage scores are aggregated for ecosystem and human health since they have the same unit:

$$DS_E^i = \sum_p \left[(CF_{FEu,p} + CF_{MEu,p} + \cdot CF_{FET,p} + CF_{TET,p} + CF_{MTET,p} + CF_{AC,p}) \cdot C_p^i \right] \quad (10)$$

$$DS_{HH}^i = \sum_p \left[(CF_{HTC,p} + CF_{HTNC,p}) \cdot C_p^i \right] \quad (11)$$

Where DS_E^i is the damage score to ecosystem associated with water flow i (e.g., species.yr/L for ReCiPe method), DS_{HH}^i, the damage score to human health associated with water flow i (DALY/L), $CF_{x,p}$ are the different characterization factors for pollutant p on impact categories x in endpoint units (e.g., species.yr/kg for ReCiPe) , C_p^i is the concentration of pollutant p in water flow i (kg/L). Categories of impact x correspond to categories detailed above.

It should be noted that for Impact 2002+ method for eutrophication, it has been considered an undefined river basin, meaning that both nitrogen and phosphorus emissions have an impact on aquatic eutrophication. Also, CF related to toxicity of phosphorous in ReCiPe has been disregarded since it concerns white phosphorus, which is an allotrope compound and not the form of phosphorus which is found in water flows.

4.2.2.2. Setting conversion factors to compare damage scores from different LCIA method

Since one of the proposed step is the aggregation of the water flows compounds in damage scores, it sounds interesting to assess the sensitivity to LCIA methods. For that purpose, damage scores are compared for Impact 2002+, ReCiPe, and ILCD recommended endpoint pathways. This comparison requires setting conversion factors between the various units used by each method. All the methods use the same unit for characterizing human health damages, i.e., disability-adjusted life year (DALY) but different ones for ecosystem quality: Impact 2002+ uses PDF.m^2.yr – with PDF standing for "potentially disappeared fraction of species", ReCiPe uses species.yr, and ILCD uses species.yr for eutrophication damages (based on ReCiPe) and PAF.m^3.d – with PAF standing for "potentially affected fraction of species" - for ecotoxicity (based on UseTOX).

Therefore, all damages on ecosystem scores are translated into species.yr in order to compare the different methods. Conversion factors between units are necessary. (i) Dong et al. (2013) considered that 1 PAF (potentially affected fraction of species) = 1 PDF (potentially disappeared fraction of species) = 1 PNOF (potentially not occurring fraction of species), whereas Humbert et al. (2012) considered alternatively 2 PAF per PDF. This equivalence is therefore questionable and highly uncertain. We chose 1 PAF per PDF in order to be compliant with Dong et al. (2013). (ii) Each individual endpoint score expressed in PAF, PDF or PNOF shall be converted into "loss of species" so that it captures the species distribution within each type of ecosystems (freshwater, marine water, and terrestrial). It enables to weight the damages on the basis of the total number of species on land and in water bodies. To make such a conversion, species densities (SD) are found from ReCiPe (Goedkoop et al., 2009): $SD_{freshwater}$ = 7.89E-10 m^{-3}, SD_{marine} = 3.46E-12 m^{-3}, $SD_{terrestrial}$ = 1.48E-08 m^{-2}. (iii) In order to convert freshwater eutrophication and ecotoxicity damages of Impact 2002+, which unit is PDF.m^2.yr, to PDF.m^3.yr, the amount of m^3 of water per m^2 of river, i.e., the river height has to be defined. Whereas Humbert et al. (2012) consider a value of 17.8 m^3/m^2 which seems overestimated, we have chosen a value of 3m^3/m^2 as suggested by Dong et al. (2013). (iv) It has been considered 365 days in 1 year.

It results in the conversion factors presented in Table 4-4.

Table 4-4. Conversion factor for endpoint ecosystem damages between LCIA categories

Method	Usetox	Impact 2002+	ReCiPe
Unit	PAF.m^3.d	PDF.m^2.yr	species.yr
Freshwater ecosystems	1	$1\,\text{PAF.m}^3.\text{d}\cdot\frac{1\,\text{PDF}}{1\,\text{PAF}}\frac{1\,\text{m}^2}{3\,\text{m}^3}\cdot\frac{1\,\text{yr}}{365\,\text{d}}$ $=9.12\cdot10^{-4}$	$1\,\text{PAF.m}^3.\text{d}\cdot\frac{1\,\text{yr}}{365\,\text{d}}\cdot7.89\cdot10^{-10}\,\text{m}^{-3}$ $=2.16\cdot10^{-12}$
Marine ecosystems	1	-	$1\,\text{PAF.m}^3.\text{d}\cdot\frac{1\,\text{yr}}{365\,\text{d}}\cdot3.46\cdot10^{-12}\,\text{m}^{-3}$ $=9.48\cdot10^{-13}$
	-	PDF.m^2.yr	species.yr
Terrestrial ecosystems	-	1	$1\,\text{PDF.m}^2.\text{yr}\cdot1.48\cdot10^{-8}\,\text{m}^{-2}$ $=1.48\cdot10^{-8}$

4.2.3. Implementation of the proposed damage score to a water footprint method (advanced water impact index - WIIX)

The water impact index (WIIX) is a simplified single indicator approach for water footprinting (Bayart et al., 2014) that includes issues related to water scarcity and water quality:

$$WIIX = \sum_i \left(V\_W_i \cdot Q_i^{WIIX} \cdot WSI_i\right) - \sum_i \left(V\_W_i \cdot Q_j^{WIIX} \cdot WSI_j\right) \tag{12}$$

Where Wi and Rj are quantities of water withdrawn from water body "i" and returned to water body "j", respectively (in volume unit), Q_i^{WIIX} and Q_j^{WIIX} are quality indices of water withdrawn from water body "i" and returned to water body "j", respectively (unitless), WSI_i and WSI_j are water scarcity indices for water bodies "i" and "j", respectively (unitless).

In this approach, water quality index Q^{WIIX} is based on the minimum ratio between pollutant concentrations in ambient quality standard water flows (for example WFD in Europe) and in the assessed water flow as shown in Eq. (13). It is comprised between 0 and 1 where 0 represents a bad water quality whereas 1 represents a good water quality.

$$Q_i^{WIIX} = \min_p \left(1; \frac{C_p^{ref}}{C_p^i}\right) \tag{13}$$

Where Q_i^{WIIX} is the quality index of WIIX (unitless, bounded between 0 and 1), C_p^i is the concentration of pollutant p in water flow i (kg/L) and C_p^{ref} is the concentration of pollutant p in the chosen reference flow (kg/L).

This approach is therefore based on the most penalizing pollutant, as done within the WFD when defining water classes. This simplification has a masking effect on the variation of other pollutants. Therefore, Bayart et al. (2014) discussed the possibility to build a quality index based on several pollutants, through an aggregation. We here propose to calculate an advanced WIIX quality indicator based on LCIA, which has the advantage of taking into account several pollutants in a single indicator as shown in eq. (14). Damage score on ecosystem is chosen because it takes into account both eutrophication and ecotoxicity.

$$Q_i^{WIIX+} = \min\left(1; \frac{DS_E^{ref}}{DS_E^i}\right) \quad (14)$$

Where Q_i^{WIIX+} is the advanced quality index of WIIX (unitless, bounded by 0 and 1), DS_E^{ref} is the damage score on ecosystem of the reference flow, and DS_E^i is the damage score on ecosystem of the studied flow i, both based on eq (10). Both Q_i^{WIIX} and Q_i^{WIIX+} are computed for water flows selected in section 4.2.1. The chosen reference is the good physico-chemical state (column 2 of Table 4-2) and good chemical state from WFD.

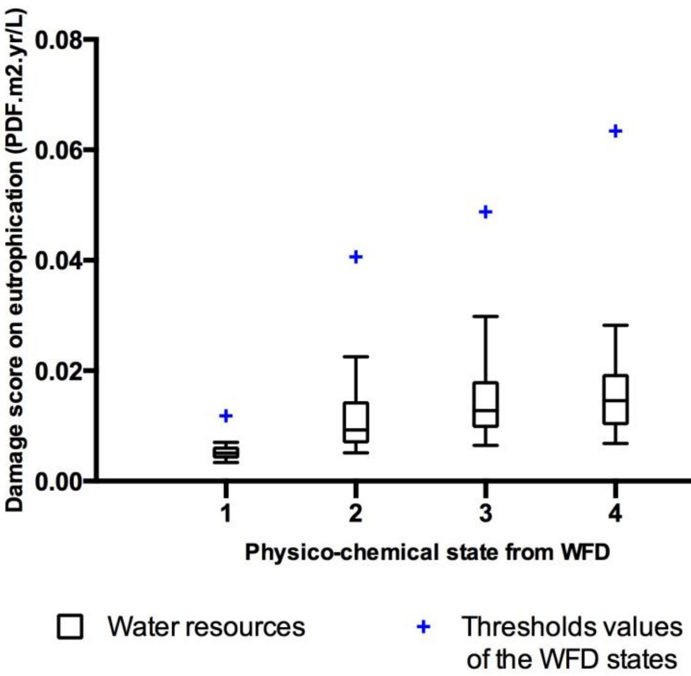
4.3. Results and discussion

Specific results found for damage scores of the natural water resource are first discussed, in comparison with the classifications of the WFD. For this purpose, only Impact 2002+ method is chosen to simplify the comparison. Then, damage scores results for all selected urban water flows are analyzed according to all LCIA methods. Finally damage scores are applied to compute the advanced quality index of WIIX and this index is compared to the original water quality index WIIX.

4.3.1. Damage scores analysis for natural water resources

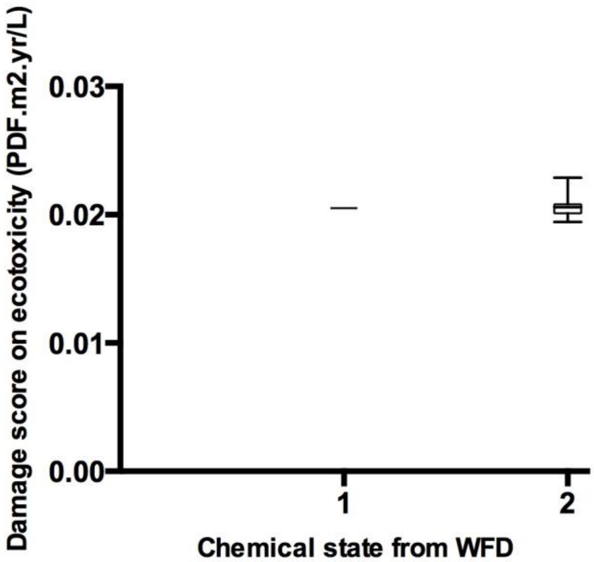
Figure 4-1 represents the damage scores of the stations on ecosystem (only including eutrophication) according to values measured for the physico-chemical state. There is a correlation between the state and the damage scores: obviously, better the physico-chemical state, lower the damage score. However, there is a high variability in damage scores for each state, except for the very good state. In addition, Figure 4-1 shows in blue the damage scores of the flows defined with threshold values for each physico-chemical parameters taken into account, i.e., the highest damage score that can be found for each class. For example, damage score of the flow defined with threshold value of water class 2 (i.e., good state class) have a similar damage score than the highest value found for class 5 (bad state class). It means that a water classified with good state could lead to more potential impact than a water classified with bad state. It demonstrates the limitations of the state definition depending on thresholds values. The aggregated damage score obtained from LCIA enables us to consider several pollutants and could be an interesting option to classify waters.

Damage scores to freshwater ecotoxicity and human toxicity can be compared to the chemical state of stations. Nevertheless, only two states are considered for chemicals: good or bad, which greatly limits the potential for comparison. State 1 (good) average, minimum and maximum values are similar. This is because when concentration of chemical compounds was not know, it was set to the concentration of the good state. Therefore, most water resources have the same damage score to ecotoxicity since there were considered to have the same concentration.



Box-and-whiskers figure details: box extends from the 25th to 75th percentiles, the line in the middle of the box is plotted at the median and whiskers refer to min and max.

Figure 4-1. Average damage score due to eutrophication of 2534 water resources versus physico-chemical state from the WFD, from 1 (very good state) to 5 (bad state); LCIA method is Impact 2002+.

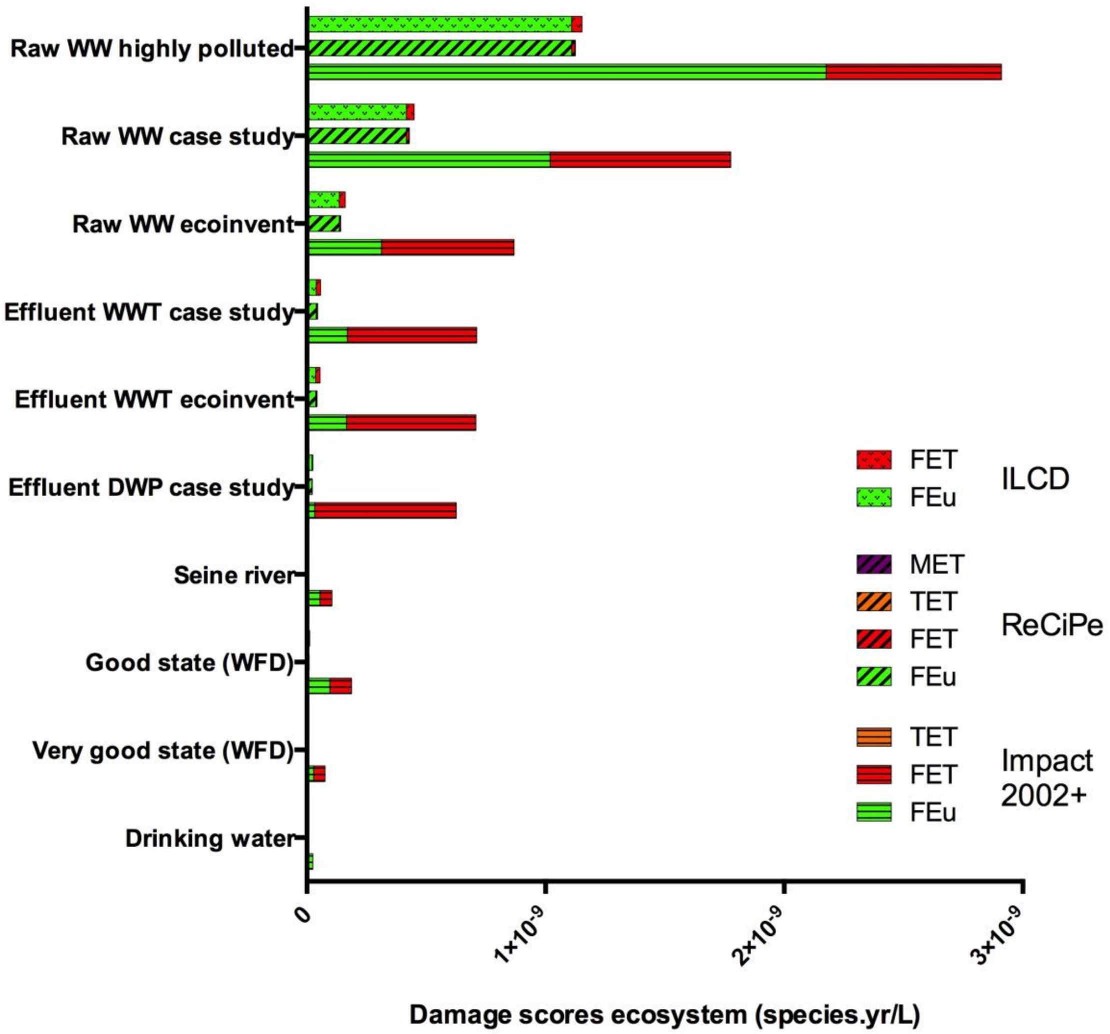


Box-and-whiskers figure details: box extends from the 25th to 75th percentiles, the line in the middle of the box is plotted at the median and whiskers refer to min and max.

Figure 4-2.: Average damage score due to ecotoxicity of 2534 water resources versus chemical state from WFD; LCIA method is Impact 2002+.

4.3.2. Analysis of damage scores of selected urban water flows

Figure 4-3 compare damage scores to ecosystems of the selected urban water flows, depending on the LCIA method. It also shows the contribution of eutrophication and ecotoxicity to total damage.



FEu = Freshwater eutrophication, FET = Freshwater ecotoxicity, TET : Terrestrial ecotoxicity, MET = Marine ecotoxicity

Figure 4-3. Damage scores on ecosystem (including eutrophication and ecotoxicity) of selected water flows assessed with different LCIA methods. All scores are converted in species.yr.

The three methods differentiate the different water flows with their damage scores, from the lower potential damage (drinking water) to the highest potential damage (raw highly polluted wastewater).When comparing all the methods with the same basis (species.yr/L), damages calculated with Impact 2002+ are much higher than the two others methods, which can be explained by several reasons.

First, regarding ecotoxicity, Impact 2002+ results show a higher contribution of ecotoxicity compared to others methods. This is because metals are overestimated in Impact 2002+, and some pollutants evaluated in Impact 2002+ (such as aluminum) are not evaluated in the two other methods. There is still an important challenge to assess metals in LCIA, as noted by Rosenbaum et al. (2008). For example, Usetox still considers characterization factors for metals as “interim” because the model does not account for speciation and other important specific processes for metals.

Second, concerning freshwater eutrophication, Impact 2002+ takes into account Nitrogen, phosphorus and COD whereas ReCiPe and ILCD only take phosphorus into account. However, the consideration of N emissions in Impact 2002+ for freshwater eutrophication is questionable. We chose to apply a damage CF for N emissions in Impact 2002+ by considering unknown river basin limiting nutrient, whereas most of river basins are phosphorus-limiting (and not affected by N emissions). This assumption is made to take into account N emissions at the endpoint level for freshwater eutrophication. Actually, nitrogen emissions affect marine eutrophication, which is still not assessed at the endpoint level in consensual methods. New models that develop fate and effect factors for marine eutrophication are currently being developed (Cosme et al., 2013) but the resulting characterization factors are still under research and it was not possible to include them in the current chapter (Dong et al., 2013).

Third, modeling choices to assess the damage are different for each method, and the conversion factors used are also uncertain, which also explain the different results.

In the context of urban water system, Impact 2002+ enables us to assess the potential damages of key pollutants, such as nitrogen, COD, aluminum, etc. However, the impact assessment models and the associated assumptions are subject to high uncertainties.

Regardless the chosen LCIA methods, contributions of the different kinds of ecotoxicity are similar: urban water flow pollutants contribute in majority to freshwater ecotoxicity, compared to marine and terrestrial ecotoxicity. Even if terrestrial ecotoxicity models are limited and are not always taken into account (e.g., UseTOX), the results clearly shows that urban water flows contribute significantly to eutrophication and freshwater ecotoxicity only.

However, sludge spreading, that was not considered here as it is not strictly a water flow, has an important contribution on terrestrial ecotoxicity.

For damages on human health (Figure 4-4), since only toxic compounds (and not nutrients) are taken into account, the differentiation between water qualities is limited: this is because concentrations of several toxic compounds are not known and are, therefore, set to “very good state” water quality in our “gap filling procedure”. Results are not satisfying, and further data on the composition of water flows is needed. However, the methods give similar results.

It also raises concerns on the present methodology with regards to human health. In order to assess damage scores, the original assumption is that the water flow is released to the aquatic environment. However, flows which stay within the technosphere can be exposed to human (such as drinking water) and thus would require modified characterization factors considering new exposure factors. This issue is important when assessing impacts of drinking water production but has never been explored so far.

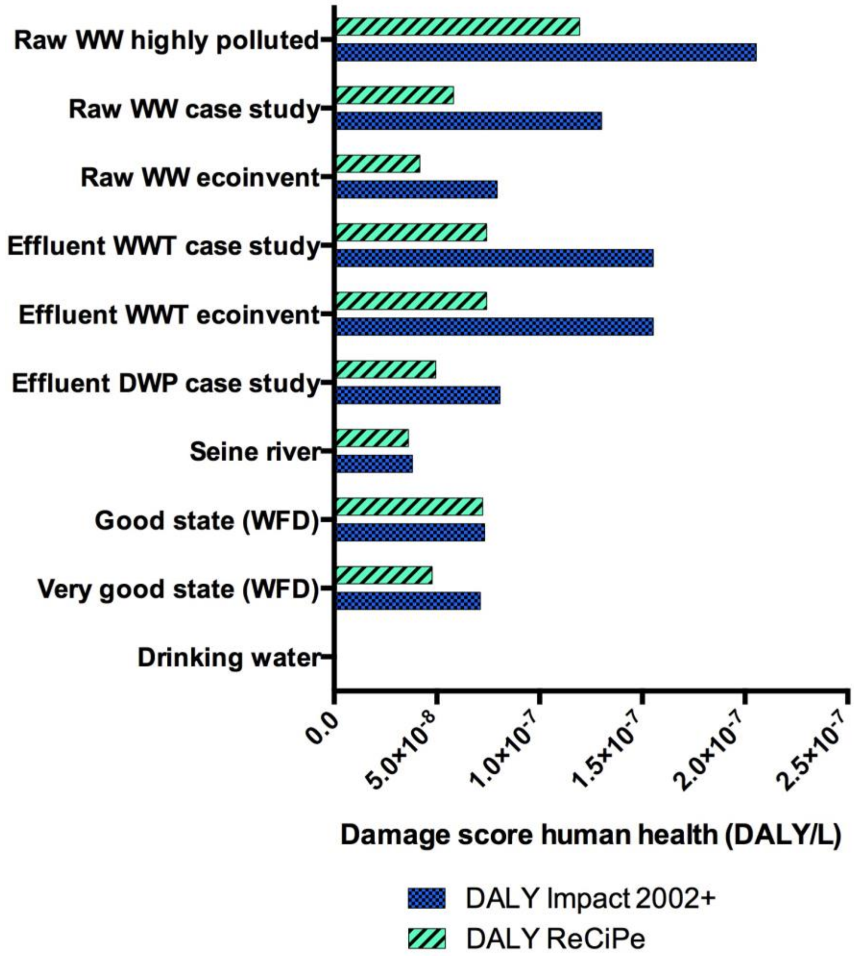


Figure 4-4. Damage scores on human health of selected water flows assessed with different LCIA methods.

4.3.3. Application to a water footprint method (Water Impact Index – WIIX)

Figure 4-5 represents the WIIX quality index score depending on the original approach (Q^{WIIX} eq. (13)) and the advanced approach (Q^{WIIX+} eq. (14)). The advanced Q^{WIIX+} allows a better differentiation of water. For example, different polluted water (i.e., effluents from WWT and DWP and raw wastewater) have a similar Q^{WIIX} close to 0, whereas Q^{WIIX+} clearly differentiate these types of flows, which have different water quality. This is because Q^{WIIX+} takes into account all pollutants and is less sensitive to the high concentration of one pollutant.

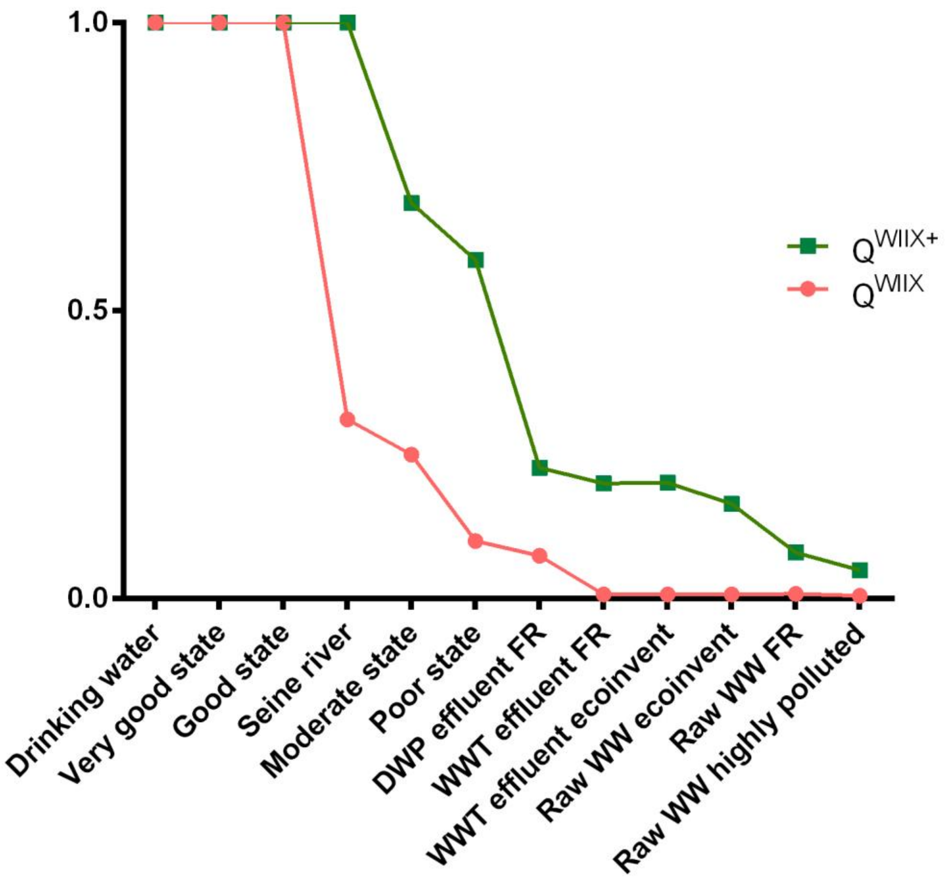


Figure 4-5. WIIX quality index related to the original approach and the advanced approach

4.4. Proposed classification of urban water flows

From the damage scores analysis, a preliminary classification of flows is set according to the ecosystem damage scores and the Q^{WIIX+} indicator. Damages to human health were disregarded since the differentiation of water types is not possible here, as shown in section 4.3.2. Five main types of water flows and their associated damage scores are defined to feed the UWS model: from A,best water quality, to D, worst quality (A - Produced water, B -

Natural water resources, C - Effluents from DWP and WWT, D - Raw wastewater), with a last type E, which represents “sludge”. These types follow specific ranges of damage scores and of Q^{WIIX+} as shown in Table 4-5. This classification also allows to use these typical values to evaluate an urban water flow of a certain type, without knowing exactly its composition. Within each type, several levels can be defined to be even more precise (e.g., A1, A2… Ai in “A - Produced water” type) and to represent all water flows of the case studies.

Table 4-5. Proposition of water types for urban water flows and corresponding damage scores to ecosystems

		Range of “damage score” for the proposed water quality type		Range of Q^{WIIX+}
Water flow types	Water flow indices as named in the WaLA model	Impact 2002+ ecosystem score (10^{-3} PDF.m^2.yr/L)	ReCiPe ecosystem score (10^{-12} species.yr/L)	
A - Produced water	A1, A2, …, Ai	0 – 11	0 – 1	1
B - Natural water resources	B1, B2, …, Bi	11 – 40	2.5 – 10	0.5 – 1
C - Effluents from DWP and WWT	C1, C2, …, Ci	15 – 100	10 – 50	0.2 – 0.5
D - Raw wastewater	D1, D2, …, Di	100 – 1000	100 – 1200	0 – 0.2
E – Sludge	E1, E2, …, Ei	-	-	-

4.5. Conclusions and outlook

Water-related LCIA methods have been applied to aggregate water composition of urban water flows into two quality scores: a damage score for ecosystem quality and a damage score for human health. Damage scores of natural water resources show a correlation with their physico-chemical and chemical state as described by WFD. However, it also points out the limits of the definition of water quality states built by WFD, which are based on threshold values. It opens an interesting discussion about using aggregated water quality scores based on LCA as new indicators to classify natural water resources. Within the LCA framework, Boulay et al. (2011) also developed categories of natural water resources depending on their functionalities towards different kind of users. Their approach is also based on threshold values, as the WFD. It would be interesting to evaluate the average damage scores for several flows within each of their categories. A further step would be to compare damage scores with ecological state, which is also based on biological and hydromorphological elements.

Studied urban water flows show a high sensitivity of the damage scores on ecosystems, allowing the flows to be differenciated. The Impact 2002+ method enables us to take into account a larger set of pollutants of importance, but relies on more uncertain models than ReCiPe or ILCD. Damage scores on human health do not permit such a differentiation because of lacking inventory data on toxic compounds. The implementation of the damage scores on ecosystems to the quality indicator of water footprint methodology (WIIX) has led to a new indicator, named Q^{WIIX+} which has proven its worth compared to the original quality indicator (Q^{WIIX}) based on the most penalizing pollutants. This kind of simplification, which consists in focusing only on water issues, clearly helps for the interpretation of the results: a water footprint is generally easier to interpret than results from a full multi-criteria approach. There is also an increasing demand from industries for developing this kind of metrics. Nevertheless, it should be stated that such a simplification doesn't allow the identification of pollution shifting to other impact categories, which would not be related to water.

Chapter 2 has shown that LCA has already proven its worth in assessing the environmental impacts of UWS but it also pointed out methodological challenges related to LCA of UWS, and the need for a standardized approach. Following this review, methodological developments related to the assessment of water deprivation and water quality are presented Chapter 3 and 4. The following chapter, which is the core of the thesis, aims to develop a framework and an associated model for the LCA of UWS, following the identified specifications and implementing the methodological advances.

Chapter 5. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 1 – formalism and framework for a modular approach

« On my block, it ain't no different than the next block »

Scarface – On my block

In this chapter, which is the heart of the thesis, we propose a versatile model, termed WaLA model (Water system Lifecycle Assessment), which reduces the complexity of the system while ensuring a good representation with regards to water issues and LCA requirements. Indeed, LCAs require building UWS models, which can be tedious if several scenarios are to be compared. The WaLA model is based on a framework that uses a “generic component” representing alternately water technologies and users, with their associated water flows, and the associated impacts due to emissions, operation and infrastructure. UWS scenarios can be built by inter-operating and connecting the technologies and users components in a modular and integrated way. The model calculates monthly outputs of life cycle impacts for a set of services provided to users, as defined by the scenario. It leads to the impact/service ratio (e.g., impact/capita) and useful pieces of information for UWS diagnosis or comparison of different scenarios. The model is implemented in a Matlab/Simulink interface thanks to object-oriented programming. The applicability of the model is demonstrated using a virtual case study based on ecoinvent processes. This chapter refers to the following paper submitted to Water Research: “Loubet, P., Roux, P. & Bellon-Maurel, V. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 1 – formalism & framework for a modular approach.”

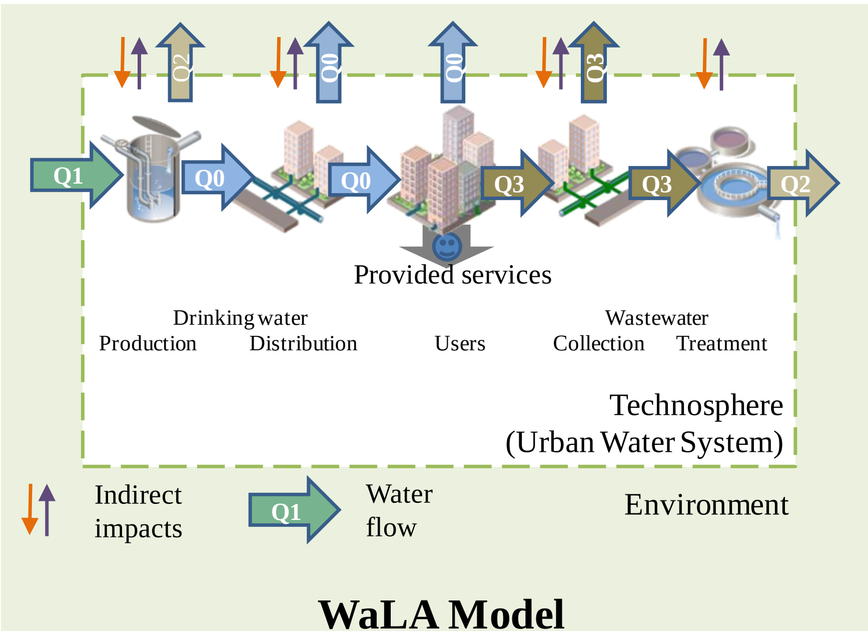


Figure 5-1. Graphical abstract of Chapter 5

Content of Chapter 5

Table 5-1. Specific glossary for the WaLA model (Chapters 5 and 6)

Technologies/Users designation		*Variables and parameters*	
DWP	Drinking water production	DEM	Total water demand (m^3/time)
DWD	Drinking water distribution	dem	Specific water demand (m^3/user/time)
SEOL	Sludge end of life	I	Impact matrix (e.g., kg CO2 eq)
SWC	Stormwater collection	i	Specific impact matrix (e.g., kg CO2 eq/m^3)
U	User	Q	Water quality index of a given flow
UWS	Urban water wystem	q	Water quality distribution vector
WH	Water heaters	S	Services provided (amount of users)
WWC	Wastewater collection	V	Volumetric water flow (m3/time)
WWT	Wastewater treatment	v	Volumetric water flow distribution vector
Flows designation		*Superscripts of I and i*	
C	Consumption	direct, air-soil	Direct impacts in link with air & soil emissions
P	Precipitation	direct, water	Direct impacts in link with water emissions
R	Release	indirect, support	Indirect impacts due to supporting activities
T_{in}	Technosphere in	indirect, chem	Indirect impacts due to chemicals
T_{out}	Technosphere out (liquid)	indirect, ener	Indirect impacts due to energy
T_{out2}	Technosphere out (sludge)	indirect, infra	Indirect impacts due to infrastructures
W	Withdrawal		

5.1. Introduction

Water management in cities faces many challenges, which are linked to water resources, water users and water technologies (Global Water Partnership Technical Committee, 2012). Decision makers require tools to assess the environmental impacts of urban water systems (UWS) and thereby compare technical solutions. Holistic approaches are required to evaluate all components of the system in an integrated way (Falkenmark, 1998).

A large amount of literature provides integrated UWS models. Mitchell et al. (2007) reviewed 65 studies, which predict water flows and water quality in cities. Some of these models go beyond calculating water quantity and water quality fluxes to include environmental aspects. For example, the SWITCH city water balance also aims to quantify energy consumption and simplified life cycle costs (Mackay and Last, 2010). Fagan et al. (2010) include environmental impact scores in their complex model for a specific UWS in Australia. However, none of these scoping tools include multi-criteria approaches, such as a full life cycle assessment (LCA)

A recent review shows that LCA is used more often to assess the environmental performance of UWS (Loubet et al., 2014). It highlights guidelines and the need for methodological frameworks in that field for all LCA phases. In the meantime, several scientific developments have occurred for LCAs to better assess impacts associated with water use (Kounina et al., 2012). These recent advances have been implemented in only a few UWS LCAs (Godskesen et al., 2013; Muñoz et al., 2010). However, LCA of UWS is still an open issue: in their review of the water-energy-greenhouse gas nexus of UWS, Nair et al. (2014) noted the interest of LCA but underlined the current static nature of the simulation tools.

The objective of this work is to model the complex UWS of megacities within the LCA framework with the aim of assessing its environmental impacts in relation to the services provided to water users. The model is termed “WaLA” as an acronym for “Water system Life-cycle Assessment”. It reduces the complexity of the system to easily implement forecasting scenarios while ensuring a good representation from the LCA perspective.

We first propose a framework and its associated modeling formalism based on a combination of generic components, representing either water technologies or water users. The technologies and users composing the UWS are then interoperated in an integrated way and connected to water resources to model the water flows and the associated impacts linked to water flows, operation and infrastructures. WaLA is run through a Matlab/Simulink (Mathworks Inc., 2007) graphical interface where the practitioner implements his own UWS scenarios in an interactive manner. Calculations of the model provide the LCIA results and the impact/service ratio. The model is finally tested on a first case study that represents a virtual UWS based on ecoinvent processes and assumptions.

5.2. Urban water system modeling

5.2.1. Specifications for an integrated UWS model

The objective of the WaLA model is to assess the environmental impacts of a UWS using the LCA framework. Therefore, the model should fulfill the requirements of the four phases defined in the international LCA standard (ISO, 2006a): goal and scope definition, LCI, LCIA and interpretation of the results. The proposed model should also be compliant with the specific methodological challenges associated with the LCA of UWS, as noted in Chapter 2 (Loubet et al., 2014), i.e., it should address issues/specifications related to the following points:

- (S1) Multi-functionality: UWS is a typical multifunctional activity (including domestic, industrial, agricultural, service users), whereas conventional LCAs were originally designed to assess a single service quantified by a functional unit. This issue can be solved by using the conceptual framework proposed by Loiseau et al. (2013) for LCA of regions, called “territorial LCA”.

- (S2) Modularity: There is an increasing demand for modeling forecasting scenarios in land and city management processes (Bach et al., 2014). A modular and interactive approach that simplifies the definition and modification of the UWS model is required for such forecasting.

- (S3) LCI and LCIA requirements linked to water quantity. As water is central in UWS , a precise accounting of water withdrawals, releases and consumption is therefore necessary; the model should follow the conceptual framework for assessing off-stream water use in LCI, as defined by Bayart et al. (2010), and the various LCIA methods that have been developed to assess the impacts related to water deprivation (Kounina et al., 2012).

- (S4) LCI and LCIA requirements linked to water quality: the mass balance of pollutants within the entire water systems must be satisfied for LCI (Risch et al., 2011). The model should also be able to include recent and future LCIA developments regarding water quality, particularly those related to eutrophication and toxicity.

- (S5) Accounting for all impacts associated with operation and infrastructures: In addition to impacts due to water deprivation and pollution, UWS generate impacts by operating the system and from infrastructures that should be considered to avoid burden shifting.

- (S6) Appropriate spatial scale: In a conventional LCA, the present trend is to conduct an impact assessment of water deprivation at the river basin scale (Pfister et al., 2009). Concerning UWS, the plurality of water resources within the basin is large and can lead to different impacts, depending on the location (upstream/downstream) of the water sources as shown in Chapter 2 (Loubet et al., 2013; Vörösmarty et al., 2005). Therefore, it is necessary to consider finer scales for UWS, i.e., the sub-river basin.

- (S7) Appropriate temporal scale: Whereas most UWS models include calculations at a daily scale (Mitchell et al., 2007), LCA is generally designed to assess impacts on a yearly basis. However, yearly timescales are not appropriate when water issues are being addressed because of high seasonal variations. Therefore, for a water-related impact assessment, a monthly scale appears appropriate to capture the climatic and hydrologic variations (Pfister and Bayer, 2014).

- (S8) Uncertainty management: Uncertainty has been disregarded in most previous LCAs applied to UWS (Loubet et al., 2014). The model should be able to compute uncertainties in the impact scores.

In the sections 5.2.3 and 5.2.4 each UWS modeling proposal is presented by following the first three LCA phases (goal & scope definition, LCI, LCIA), including a discussion for each relevant requirement (S1 to S8) defined above. The implementation of the model within a computer program is developed in section 5.2.6. The fourth LCA phase (interpretation) is addressed in detail within the application to a case study introduced in the section 5.2.7.

5.2.2. The general framework of the WaLA model

Figure 5-2 is a simplified representation of the general framework of the WaLA model. It is based on a combination of generic components, which have been instanced various parts of the UWS, i.e. water technology and user components. Both components are connected to water resources. The modularity requirement (S2) is achieved thanks to this modeling strategy: each the model is built; as a combination of components which are interoperable. Whereas Figure 5-2 is a representation of a basic UWS, a real UWS would be described as a combination of all the technologies used to run a UWS, linked to the various categories of users satisfied by this UWS.

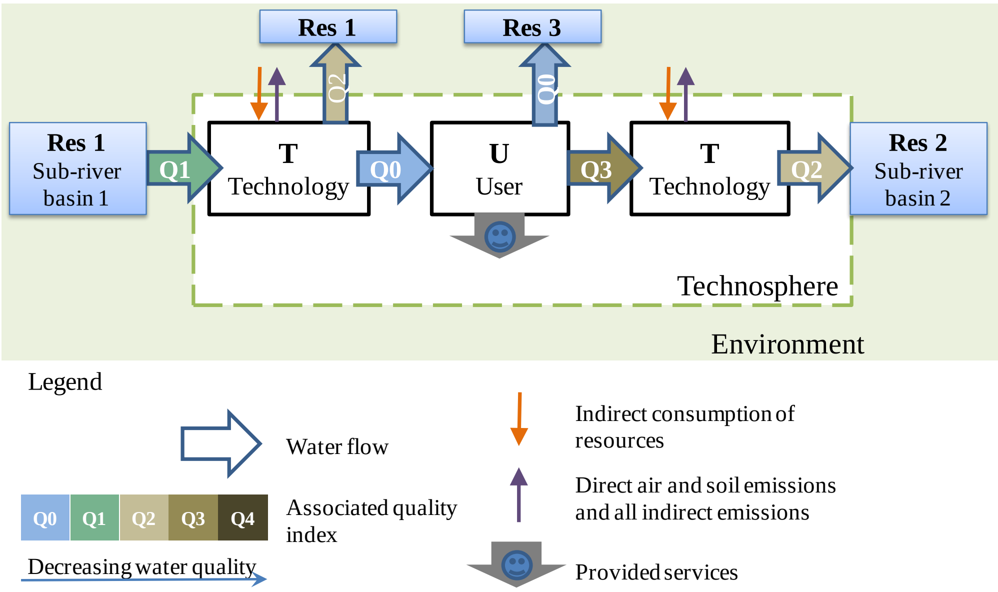


Figure 5-2. Simplified presentation of the modular formalism and boundaries of the urban water system.

5.2.3. Goal and scope definition

As stated in S1, the challenge related to the goal and scope of LCA applied to UWS is to cope with the multi-functionality of UWS at a regional scale (city, small region) (Loubet et al., 2014). Indeed, the main goal of UWS is to deliver water to customers and to manage associated wastewater, but the water demand and the quality of water may change according to the customer, e.g., domestic, industrial and agricultural users. Therefore, a single functional unit (e.g., 1 m^3 of water volume delivered) is far too restrictive to address all of the potential issues of UWS associated with all potential stakeholders. The application of the framework proposed by Loiseau et al. (2013), namely “territorial LCA”, has been proposed to solve this issue. According to this framework, the goal and scope is defined in three steps: first, a reference flow is chosen, i.e., the studied territory and associated scenario; second, the functions provided by the reference flow are identified; and finally, the most appropriate functions are selected and quantified. Here, we propose to define the reference flow as the UWS described above. Its associated functions are providing water to different kinds of users. The selection of the most relevant functions cannot be done arbitrarily and should be defined in accordance with stakeholder's issues. As a first approach, the functions are defined by a representative indicator of each type of user (e.g., the number of domestic users, industrial jobs), which are simply quantified through the description. Thus, we do not create a single functional unit but a set of functions represented by a vector of values termed **S**. An alternative function is defined as the volumetric amount of water supplied at the user's place, which can be useful to assess the efficiency of the "technological system". However, this function does not consider user behavior and may be less relevant when considering an integrated urban water system, including its social dimension (Loubet et al., 2014). Water users are characterized by their water demand (in terms of both quantity and quality), noted *dem* (m^3 of a given quality/user unit/unit of time). The total water demand of a user, termed *DEM*, is the product of the number of users *S* and the specific water demand *dem*.

5.2.4. LCI/LCIA associated to the technologies/users generic components

The model is based on a generic component formalism that represents both water technologies and water users (Figure 5-3). Water technologies are typically drinking water production (DWP), drinking water distribution (DWD), wastewater collection (WWC) and wastewater treatment (WWT). Water users (U) are domestic, industrial, or agricultural users, among others. Because technologies and user components are related to anthropogenic activities, they are located within the technosphere and are in the foreground system.

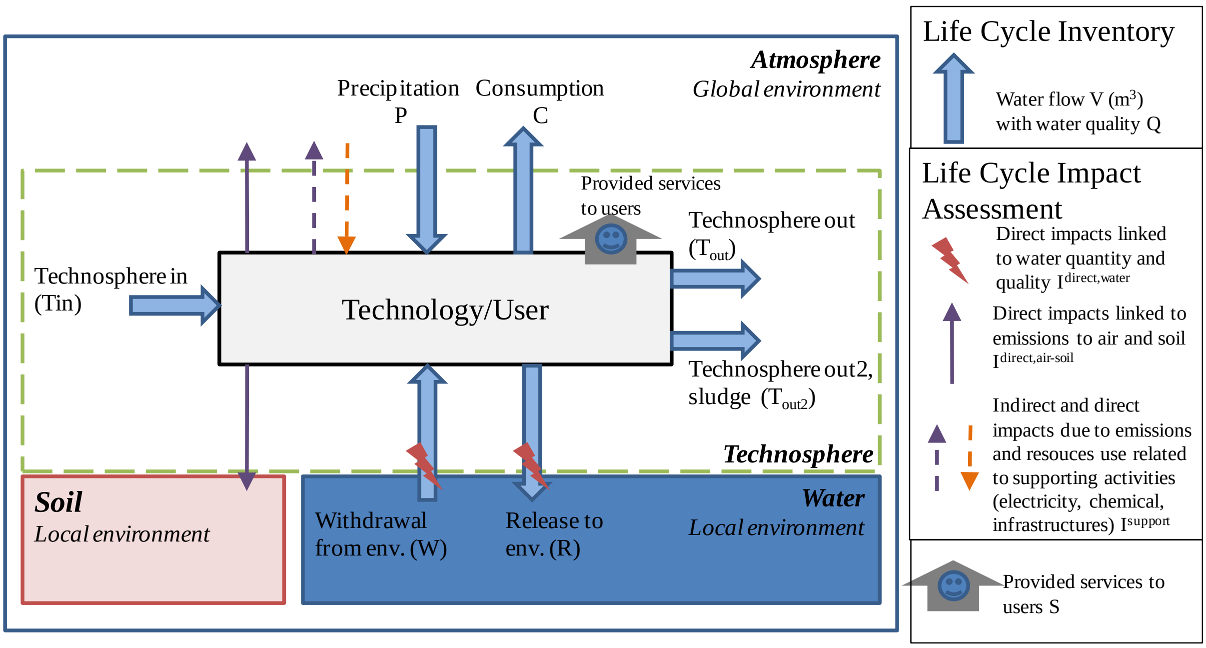


Figure 5-3. Description of water flows and associated impacts/services of the generic component.

Model components exchange water flows with other components and the local or global environment. “Local environment” is defined as the sub-river basin where the component withdraws or releases water, and “global environment” is defined as the atmosphere. Consequently, six types of water flows enter or leave each component as shown in Figure 5-3: (1) input water from the technosphere, T_{in}; (2) output water to the technosphere, T_{out}; (3) input from the local environment, W (withdrawal); (4) output to the local environment, R (release); (5) input from the global environment, P (precipitation); and (6) output to the global environment C (consumption). Two different technosphere flows, termed T_{out} and T_{out2}, are defined for water flow and sludge flow, respectively. Water consumption is the water evaporated, transpired or exported in product (Bayart et al., 2010). According to the international standards (ISO, 2006), within-technosphere flows (T_{in} and T_{out}) are considered intermediate flows, and environment flows (W, R, P, C) are considered elementary flows.

Each flow is characterized by two parameters: the volumetric water flow (**V**), which is expressed in m^3 per unit of time, and the water quality (**Q**), which is expressed by an index. Each flow parameter is defined by the component name to which it is linked (e.g., “DWP” for drinking water production), the flow name (e.g., “T_{out}” for water flow going to the technosphere) and the considered parameter (e.g., **V**) as shown below:

$$\overbrace{\mathrm{DWP}}^{\text{sub-component}}\ \_\ \overbrace{\mathrm{V}}^{\text{parameter}}\ \_\ \overbrace{\mathrm{T_{in}}}^{\text{flow name}} \tag{15}$$

When the water flow crosses the boundary between the technosphere and the local environment, it leads to impacts because of its quantity and quality changes, as shown in Figure 5-3. Estimations of these associated impacts are explained in the sections 5.2.4.3 and 5.2.4.4.

In addition to the direct impacts linked to water flow exchanges, water technologies generate direct impacts due to emissions to the air (e.g., CH_4 in WWT) and soil (e.g., metals in sludge) from the water and sludge lines, defined in section 5.2.4.5.

Other water technology impacts come from supporting activities of the UWS: energy (primarily electricity), chemicals, transportation of sludge and chemicals, and infrastructure (construction, maintenance, and deconstruction). These processes are generally considered background processes because they are typically found in locations other than the UWS territory, as defined by Azapagic et al. (2007). However, some supporting activities can occur in the foreground system, and the related emissions to the air and soil are considered direct. It can concern certain types of energy (e.g., gas to heat water, generator group to produce electricity, methanization of sludge), chemical production (e.g., ozone production) or infrastructure (e.g., civil works). Because these activities are occurring within the UWS territory and are usually under the control of the UWS decision-maker, they should be considered foreground and as generating direct impacts (Frischknecht, 1998; Loiseau et al., 2014). The calculation of these impacts is detailed in section 5.2.4.6.

At the component scale, impacts can be classified according to two different groupings, as shown in Table 5-2: (i) related to foreground (leading to direct impacts) and background activities (leading to indirect impacts) and (ii) related to water and sludge lines or supporting activities. If direct impacts linked to supporting activities are not considered, which should be likely the case, the two groupings are equivalent: direct impacts occur in water and sludge lines, and indirect impacts occur in supporting activities.

Table 5-2. Classification of impacts at the component scale

	Foreground activities (direct impacts)		Background activities (indirect impacts)
Water and sludge lines	- Emissions to water - Water use	- Emissions to air and soil	
Supporting activities	- Production of energy, chemicals and materials for infrastructures occurring in the territory		- Production of energy, chemicals and materials for infrastructures occurring elsewhere the territory

Impacts are stored in a matrix I^{total} of n lines, each of which represents an impact category j, noted I_j, depending on the chosen LCIA method. I^{total} is calculated by adding different contributors to impacts: direct impact due to the exchange of water of various qualities between the technosphere and the environment ($I^{direct,water}$), direct impacts due to emissions to the air and soil from the water and sludge lines ($I^{direct,air\text{-}soil}$), and direct and indirect impacts related to supporting activities ($I^{support}$) for operation (energy I^{ener}, chemicals I^{chem}) and infrastructures (I^{infra}). Each vector has the same number of n lines, but depending on the contributor considered, not all impact categories are necessarily concerned and can be thus set to 0 (e.g., fossil fuel depletion is set to 0 for $I^{direct,water}$ vector). The generic component related to users also generates the services provided to the users, as detailed in the section 5.2.3.

5.2.4.1. Water quantity: Volumetric water flows (LCI)

Volumetric water flows (in m^3 per unit of time) are represented by a vector **V** defined in eq (16). They are balanced for each component to comply with specification S3 regarding the inventory of water quantity. To achieve this water balance, the sum of the volumetric flows that are entering (T_{in}, W, P) must be equal to the volumetric flows that are exiting (T_{out}, T_{out2}, R, C) any component. The volumetric water flows are calculated from a water flow distribution vector, namely **v**, and a known input variable, which is either V_T_{in} or V_W (eq 3). **v** is a vector of intrinsic parameters (m^3/m^3), specific to each component, that defines the distribution of flows entering (T_{in}, W, P) and exiting (T_{out}, T_{out2}, R, C) the component for 1 m^3 at the input (T_{in} or W for DWP).

$$V = \begin{bmatrix} V\_T_{in} \\ V\_T_{out} \\ V\_T_{out2} \\ V\_W \\ V\_R \\ V\_P \\ V\_C \end{bmatrix} \quad \text{where} \quad (V\_T_{in} + V\_W + V\_P) - (V\_T_{out} + V\_T_{out2} + V\_R + V\_C) = 0 \tag{16}$$

$$\begin{array}{ll} \text{For DWD, U, WWC, WWT} & V = V\_T_{in} \cdot v \\ \text{For DWP} & V = V\_W \cdot v \end{array} \qquad \text{where } v = \begin{bmatrix} v\_T_{in} \\ v\_T_{out} \\ v\_T_{out2} \\ v\_W \\ v\_R \\ v\_P \\ v\_C \end{bmatrix} \tag{17}$$

5.2.4.2. Water quality (LCI)

Quality is defined by an index, **Q** that refers to a chemical load in the water. This chemical load is also associated with the potential impacts when water is released to the environment, as explained in the section 5.2.4.4. At this stage of model development, definition of indices is based on Chapter 4: A1, A2, ... are drinking water qualities, B1, B2, ... are water resource qualities, C1, C2, ... are qualities of water effluents from DWP or WWT, D1, D2, ... define raw domestic wastewater qualities and E1, E2, ... define sludge qualities. The chemical composition of each index is given in Chapter 4 and Annex C.3.

For each component, a vector of “intrinsic parameters”, namely **q**, defines the indices **Q** of water flows exiting (eq. 4). To build these vectors, it is first necessary to determine the mass balance of the chemicals considered between the inputs and outputs at the component scale (specification S4).

$$q = \begin{bmatrix} 0 \\ Q\_T_{in} \\ Q\_T_{out2} \\ 0 \\ Q\_R \\ 0 \\ Q\_C \end{bmatrix} \tag{18}$$

Qualities of water entering the component (*Q_T*Q_T_{in}, Q_W, Q_P) are known input variables.

5.2.4.3. Direct impacts associated with water quantity (LCIA)

Withdrawals deprive downstream users from water, whereas releases make the water available again. The water deprivation impacts (I_{WD}) associated with each component are computed based on characterization factors (CF) defined at the sub-river basin scale, as recommended in Chapter 3 (Loubet et al., 2013):

$$I_{WD}^{direct,water} = V\_W \cdot CF_{WD,A} - V\_R \cdot CF_{WD,B} \tag{19}$$

Where I_{WD} is the midpoint impact of water deprivation (m^3 equivalent or m^3 equiv), V_W is the volume of water withdrawn at location A (m^3), V_R is the volume of water released at location B (m^3), and $CF_{WD,A}$ and $CF_{WD,B}$ are the characterization factors for water deprivation at locations A and B, respectively. Water deprivation CFs differentiated at the sub-river basin scale are used to calculate the cascade effects on downstream sub-river basins. The use of this LCIA model is compliant with the spatial scale required for UWS (S7). I_{WD} is then stored within the vector $I^{direct,\ water}$.

5.2.4.4. Direct impacts associated to quality of released water (LCIA)

The different water quality indices introduced in the section 5.2.4.2 refer to the chemical compositions and associated impacts. Impacts are calculated for each component as the difference in the potential impacts associated with water releases and the potential impacts associated with water withdrawals (eq. 5). Indeed, because emitting pollutants affects the environment, it is counted as positive, whereas uptaking the pollutant from the environment is a benefit for the environment and is counted as negative.

$$I^{direct,water} = \begin{bmatrix} I_1^{direct,water} \\ \vdots \\ I_j^{direct,water} \\ \vdots \\ I_n^{direct,water} \end{bmatrix} = V.R \cdot \begin{bmatrix} i_1^{Q\_R} \\ \vdots \\ i_j^{Q\_R} \\ \vdots \\ i_n^{Q\_R} \end{bmatrix} - V.W \cdot \begin{bmatrix} i_1^{Q\_W} \\ \vdots \\ i_j^{Q\_W} \\ \vdots \\ i_n^{Q\_W} \end{bmatrix} \quad (20)$$

where $I^{direct,water}$ is the vector of impact values for each impact category (e.g., freshwater eutrophication, in kg P eq.), due to emissions to water and uptake of a given component from water, and $i_j^{Q.R}$ is the specific impact j (e.g., freshwater eutrophication, in kg P eq./m^3) related to 1 m^3 of a flow, which has a quality index of Q_R. Midpoint impacts and endpoint damages associated with direct emissions due to water exchange depend on the LCIA method and are detailed in Chapter 4.

5.2.4.5. Direct impacts associated with emissions to air and soil (LCIA)

UWS also generate direct impacts associated with emissions to air and soil related to water flows, i.e., pollutants emitted from the water or sludge lines. Emissions to soil occur specifically when spreading sludge from DWP (primarily heavy metals) and WWT (the remaining nutrients and organic compounds as well as heavy metals). The associated impacts are typically eutrophication, ecotoxicity and human toxicity. Air emissions generally occur during WWT, including sludge spreading (mainly CO_2, CH_4, N_2O, NH_4, NO_x, SO_x), and lead to several impacts (Yoshida et al., 2014). Other air emissions can occur in WWC, particularly CH_4 and H_2S. More rarely, air emissions can occur during DWP equipped with membrane processes that require CO_2 stripping to raise the pH (Ventresque and Bablon, 1997).

The direct impacts are considered fixed for each technology and are only dependent on the volumetric flow going through the process. Consequently, $I^{direct,air\text{-}soil}$ is calculated as follows:

$$I^{direct,air\text{-}soil} = \begin{bmatrix} I_1^{direct,air-soil} \\ \vdots \\ I_j^{direct,air-soil} \\ \vdots \\ I_n^{direct,air-soil} \end{bmatrix} = V.T_{in} \cdot \begin{bmatrix} i_1^{direct,air-soil} \\ \vdots \\ i_j^{direct,air-soil} \\ \vdots \\ i_n^{direct,air-soil} \end{bmatrix} \quad (21)$$

where $I_j^{direct,\ air\text{-}soil}$ is the impact j (e.g., climate change, in kg CO_2 eq.) of a component due to its direct emissions to the air and soil and $i_j^{direct,\ air\text{-}soil}$ is the specific impact j occurring for 1

m^3 entering into a component (e.g., climate change, in kg CO_2 eq./m^3). V_T_{in} is replaced by V_T_{out} for DWP technology.

5.2.4.6. Direct and indirect impacts associated with life cycle supporting activities (LCIA)

Iimpacts linked to supporting activities are represented by two vectors, one vector representing direct impacts ($I^{direct,support}$) and the other representing indirect impacts ($I^{indirect,support}$). In the two cases, specific impacts linked to energy, termed i^{ener}, specific impacts linked to chemicals and others, termed i^{chem}, and specific impacts linked to infrastructure, termed i^{infra}, should be defined. These specific impacts are either correlated to the volumetric water flow entering/exiting the system or the quality of water entering/exiting the system or fixed. It is considered that the energy and chemical consumption within the generic component are fully correlated with the volumetric flow entering/exiting the technology. The infrastructure is already built and maintained for the actual volumetric flow rate of the city. Therefore, its associated impacts are fixed, independent of the volumetric water flow. For this situation, the total impacts of the infrastructure (I^{infra}) are considered and are divided by the lifetime (in years or months). Eq (22) and (23) define the direct and indirect impacts for supporting activities.

$$I^{direct,support} = \begin{bmatrix} I_1^{direct,support} \\ \vdots \\ I_j^{direct,support} \\ \vdots \\ I_n^{direct,support} \end{bmatrix} = V.T_{in} \cdot \left(\begin{bmatrix} i_1^{direct,ener} \\ \vdots \\ i_j^{direct,ener} \\ \vdots \\ i_n^{direct,ener} \end{bmatrix} + \begin{bmatrix} i_1^{direct,chem} \\ \vdots \\ i_j^{direct,chem} \\ \vdots \\ i_n^{direct,chem} \end{bmatrix} \right) + \begin{bmatrix} i_1^{direct,infra} \\ \vdots \\ i_j^{direct,infra} \\ \vdots \\ i_n^{direct,infra} \end{bmatrix} \cdot \frac{1}{t} \qquad (22)$$

$$I^{indirect,support} = \begin{bmatrix} I_1^{indirect,support} \\ \vdots \\ I_j^{indirect,support} \\ \vdots \\ I_n^{indirect,support} \end{bmatrix} = V.T_{in} \cdot \left(\begin{bmatrix} i_1^{indirect,ener} \\ \vdots \\ i_j^{indirect,ener} \\ \vdots \\ i_n^{indirect,ener} \end{bmatrix} + \begin{bmatrix} i_1^{indirect,chem} \\ \vdots \\ i_j^{indirect,chem} \\ \vdots \\ i_n^{indirect,chem} \end{bmatrix} \right) + \begin{bmatrix} i_1^{indirect,infra} \\ \vdots \\ i_j^{indirect,infra} \\ \vdots \\ i_n^{indirect,infra} \end{bmatrix} \cdot \frac{1}{t} \qquad (23)$$

The quantification of these impacts are typically well known because the LCA literature on water systems focused on technological impacts (Loubet et al., 2014) and because the ecoinvent database provides data on the background processes for energy and chemical production. Infrastructure-related impacts and associated civil works require further study. Other water technologies that were not typically accounted for in UWS LCA must be added to the model, particularly technologies that are present at the user's place, such as water heating systems, which generate a large proportion of impacts (Arpke and Hutzler, 2006; Fagan et al., 2010).

5.2.4.7. Total impacts (LCIA)

Finally, the total impacts of a component are the sum of all of the above-mentioned impacts.

$$I^{total} = \overbrace{(V\_W \cdot CF_{WD,A} - V\_R \cdot CF_{WD,B}) + (V\_R \cdot i^{Q\_R} - V\_W \cdot i^{Q\_W}) + V\_T_{in} \cdot i^{direct,air-soil}}^{\text{Direct impacts related to water and sludge lines}}$$
$$\overbrace{V\_T_{in} \cdot (i^{elec} + i^{chem}) + i^{infra} \cdot \frac{1}{t}}^{\text{Impacts related to supporting activities}} \tag{24}$$

5.2.4.8. Computation of the impact/service ratio

As stated in section 5.2.3, no unique functions exist due to the multi-functionality of UWS. To refer the total impacts of the system to services (amount of users supplied by water), impact/service ratios (IS ratio, impacts/user) can be computed using eq (25). IS ratio is the inverse of the eco-efficiency ratio (EE) as defined by (Seppäläa and Melanen, 2005).

$$IS_i = \frac{1}{EE} = \frac{I^{total,system}}{S_i} \cdot \frac{DEM_i}{\sum DEM} \tag{25}$$

Where IS_i is the total impact of the system for one user in category i (impact/user), $I^{total,system}$ is the total impact of the system, S_i is the number of users in category i, DEM_i is the water demand from users i (m^3), and ΣDEM is the total water demand from all of the users (m^3).

If the provided service chosen by the stakeholder is the m^3 supplied at the user's place, the impact/service ratio (IS_{m3} ratio, impact/m^3) is computed using eq. 11.

$$IS_{m3} = \frac{I^{total,system}}{\sum DEM} \tag{26}$$

5.2.5. Practical details

The description of the model framework enabled us to show that it was compliant to five out of eight requirements for easily carrying out a LCA based on it. The way its implementation allows us to fit with the three other requirements as described below.

5.2.5.1. Spatial and temporal scales

As stated in specification S6, the monthly scale is adapted to UWS in terms of temporal resolution. Thus, all of the vectors presented in previous sections can be replaced with 12-column matrices that represent monthly characteristics instead of yearly. This is particularly relevant for water deprivation characterization factors that are highly dependent on seasonal effects and, to a lesser extent, water demand and impacts. The operation of water technologies can slightly change during the year because of changes in the water quality at the input (e.g., water withdrawals for DWP). Spatial scales are differentiated for CF_{WD}, as shown in the section 5.2.4.3. Impact assessment related to water quality should also be differentiated at a local scale, but new methods are currently under development.

5.2.5.2. Uncertainty propagation management

Uncertainty in the results both related to volumetric water flows or to impacts can be addressed in two different manners: propagation of the uncertainty or Monte-Carlo analysis. Monte-Carlo simulation, which is widely used in LCA, consists of repeatedly computing the results (water flows and impacts) with parameters that have been randomly sampled from their specified probabilistic distribution. Because the model can be written in a matrix form, analytical calculations of the uncertainties with matrices of variance-covariance of the parameters, as shown in Heijungs and Suh (2002), can be implemented in the proposed model.

Uncertainty management is not implemented here because it is beyond the scope of this chapter, which is focused on introducing the model. Also, the evaluation of the probabilistic distribution is a challenge in LCA.

5.2.6. Implementation of the model within a computer program

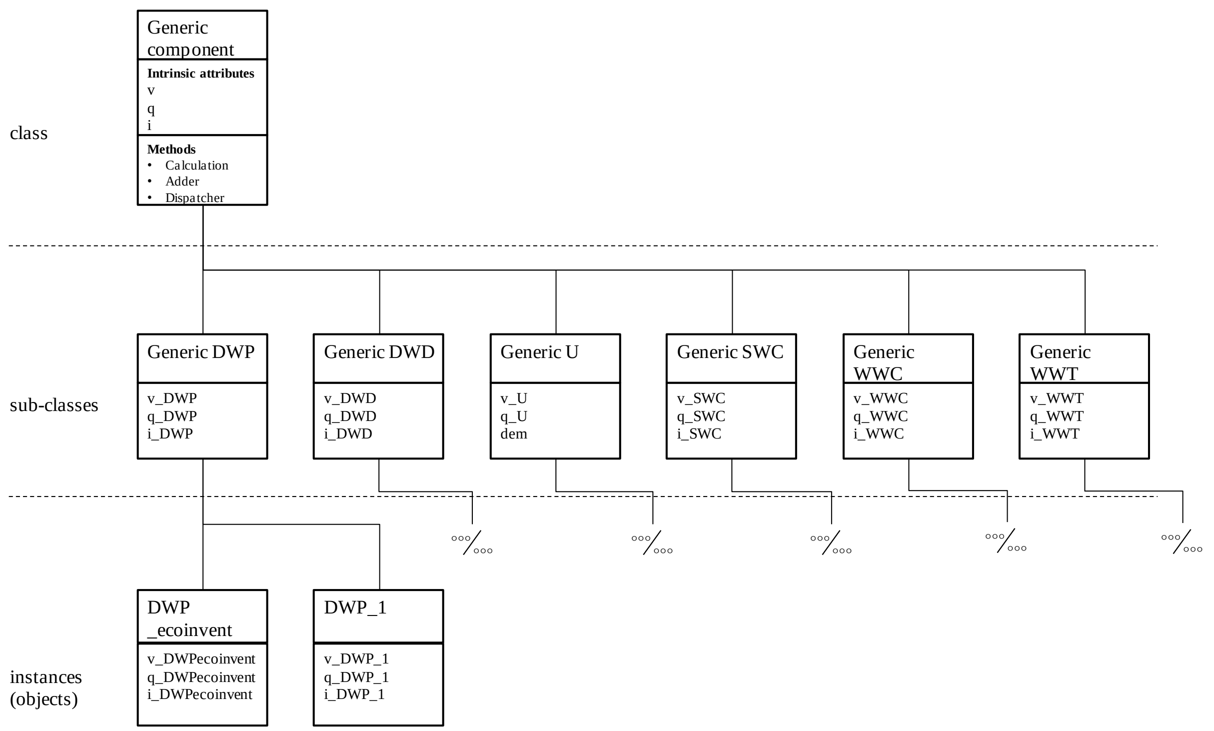
The WaLA model is programmed through a graphical interface built on Matlab/Simulink**,** which enables a practitioner-friendly construction of models using the connection of graphical objects that represent the UWS components.

5.2.6.1. Objects representing technologies and users components

The Object-oriented programming (OOP) approach (Stefik and Bobrow, 1985) is used to implement the model. This approach handles objects which refer to particular instances of a class and interact with each other. In our model, a unique class (superclass) has been built to represent the generic component described in the previous section. It is composed of methods (i.e., functions or routines) and attributes (i.e., parameters). Sub-classes represent the different types of generic technologies (e.g., DWP) and users (e.g., U), and inherit methods and attributes from the unique class. As instances of these sub-classes, the objects are specific technologies and users (Figure 5-4).

Three methods are defined for the class: (i) the "calculation" routine computes the volumetric water flows **V**, water quality **Q**, impacts **I** from eq (16), (17), (24); (ii) the "adder" routine manages the sum of the different flows entering a block; and (iii) the "dispatcher" routine manages the various outputs exiting the block to the technosphere.

Attributes, i.e., variables that allow for the customization of the block, are either "intrinsic" or "extrinsic". The "intrinsic" attributes are defined a priori and are specific to each object (i.e., each technology/user). These are the volumetric water flow distribution vector **v**, the quality distribution vector **q** and the specific impacts matrix **i**. User objects also include the specific water demand dem. The "extrinsic attributes" are defined in the model, either by the practitioner or as a result of the model initialization. These are the number of inputs (in) and outputs (out) from and to the technosphere, the name of the local water resource (Res) to which the object is connected, and, the number of water users (S). Methods and attributes for technologies and users are summarized in Figure 5-4.



v = volumetric water flow distribution vector, q = quality distribution vector, i = specific impacts matrix.

Figure 5-4. Representation of the unique class (superclass) associated with the generic component, its sub-classes associated with each technology/user component, and the instances of each sub-class associated with the specific components.

5.2.6.2. Arrows representing water flow

Arrows represent water flows exchanged by the components in the technosphere. They convey volumetric water flow (**V**) and quality index (**Q**) variables. Several technosphere water flows can go in and/or out of a graphical object. Note that water flows exiting the technosphere to the environment are not represented by an arrow but are directly translated into impacts.

5.2.6.3. Building a specific model: inter-operation of the objects

Each technology and user is represented by a graphical object that has methods and intrinsic attributes. They are stored in a Simulink library and can be selected via “drag and drop” in the graphical Simulink® window. Extrinsic attributes of the objects are defined through the practitioner interface. The different objects are connected with the arrows (technosphere water flows). Figure 5-5 schematizes the entire procedure for the construction of an UWS scenario and the computation of impacts. The structure of the Simulink objects and the associated algorithms are presented in Annex C.1.

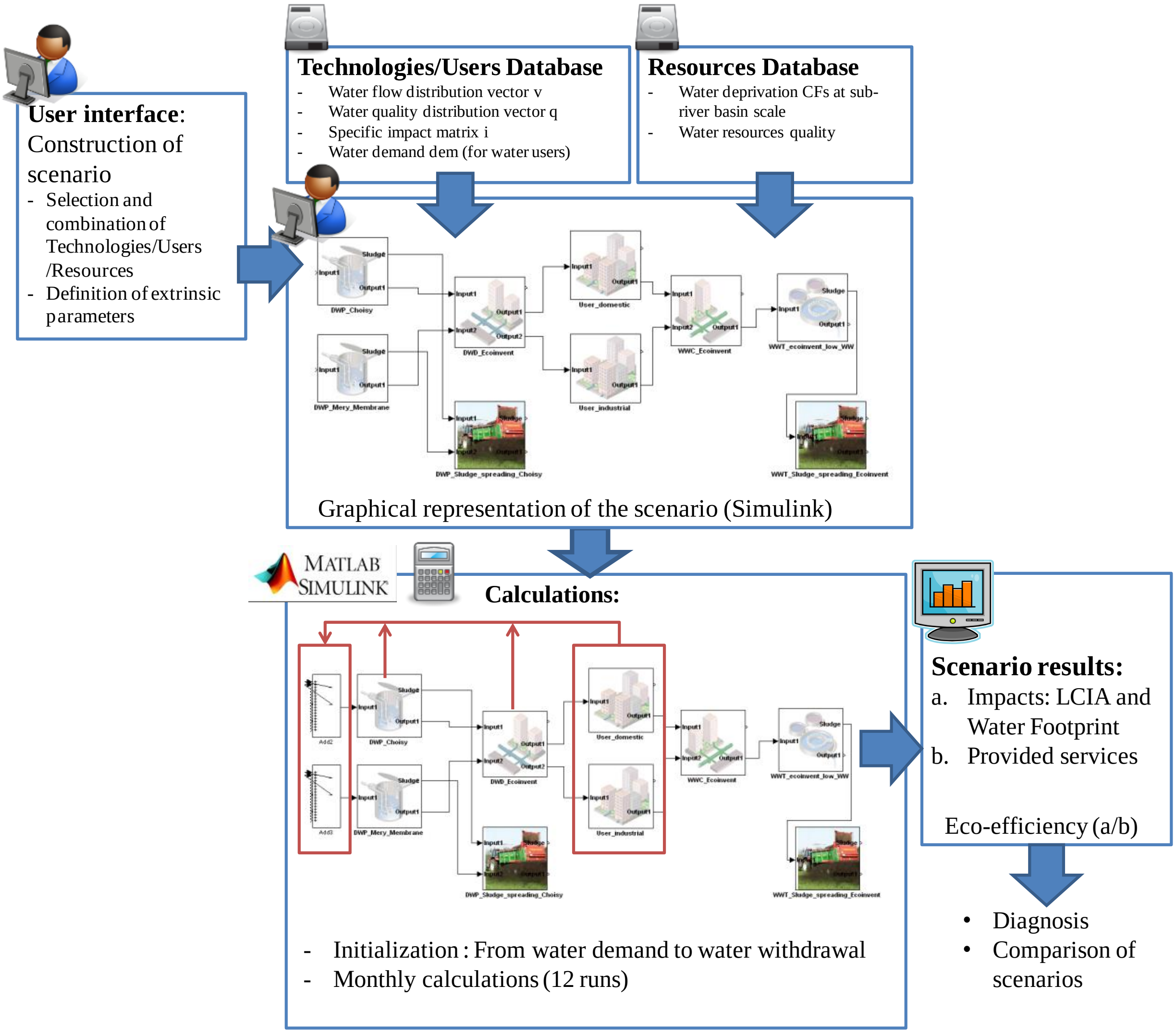


Figure 5-5. Procedure to define an UWS scenario and compute its environmental impacts and impact/service ratios. Practitioners are represented by a character.

5.2.6.4. Computation

The model is computed in two steps: initialization and calculations. The initialization enables the calculation of the initial water withdrawal from DWP plants based on the number and type of water users defined by the scenario (see Annex C.1.4 for details). Once the model is initialized, the variables that drive the updated system are the number of initial water withdrawals. The model is run 12 times to obtain monthly results for the LCIA and the impact/service ratio. The scenario results can be analyzed within Matlab or exported for graphical representations and interpretations in any spreadsheet, such as Excel.

5.2.7. Virtual case study

A virtual case study has been defined to demonstrate the applicability of both the model and the proposed modular approach. The virtual UWS is located in France and covers a territory of 4.325 million inhabitants (S), who have a water demand of 55 m^3/year (dem). In this first case study, the LCIs and the associated assumptions of all of the technologies were adapted from the ecoinvent database. It is made up of:

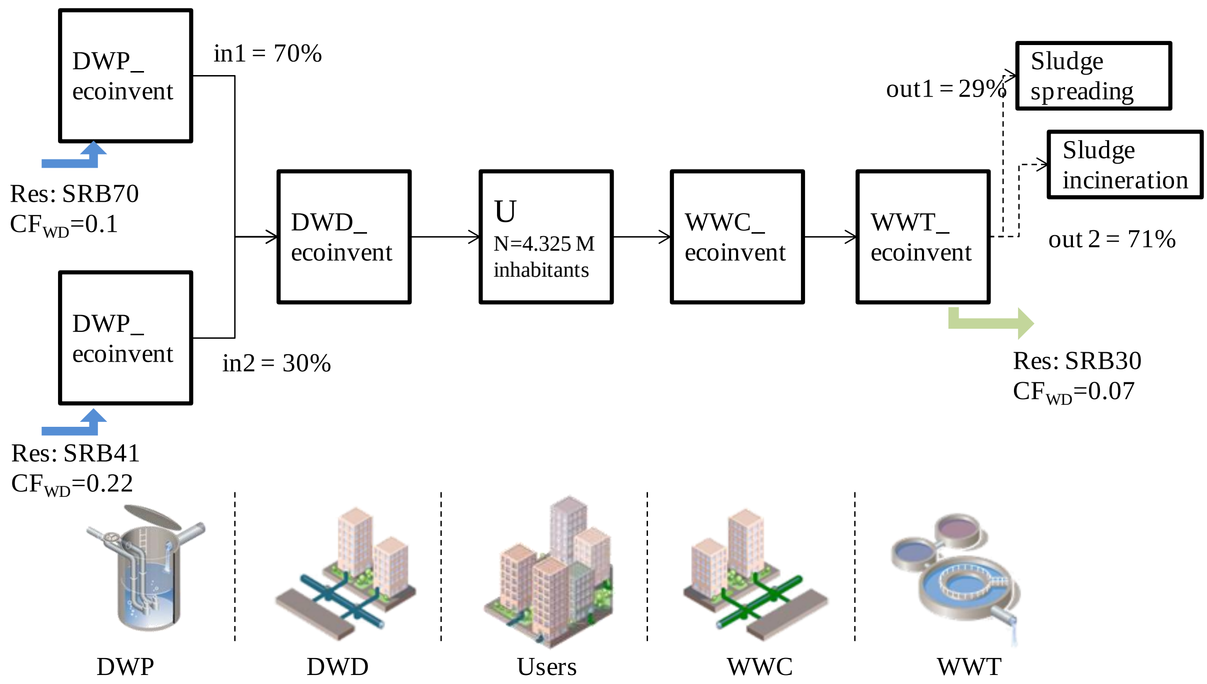
- Two conventional DWP plants, withdrawing water from two different sources (two rivers located in a French river basin but different sub-river basins). For DWP, no sludge end-of-life is considered in ecoinvent.
- One DWD network
- One WWC network (sewer grid)
- One activated sludge WWT plant. We accounted 71.7% of the WWT sludge fate to incineration and 28.3% to sludge spreading, according to ecoinvent assumptions.

Intrinsic parameters, i.e., volumetric water flow distribution vector **v**, water quality distribution vector **q** and specific impacts matrixes **i** for all water users and technologies are detailed in Annex C.2. Impact matrixes **i** are computed with Simapro 8 (Pré Consultants, 2013). All of the supporting activities are occurring in the background system, and the associated impacts are therefore considered indirect impacts, as shown in Table 5-2. The LCIA method is ILCD 2011 v1.03 (EC - JRC - IES, 2010a), and the CFs of the category "water resource depletion" (calculated with Frischknecht et al. (2006)) are replaced by CTA indicators from Hoekstra et al. (2012), which are compatible with the CF_{WD} calculated for the foreground system. To simplify the interpretation of this first application, the data are not differentiated at the monthly scale but are considered on a yearly basis.

5.3. Results and discussion

5.3.1. The graphical representation of the UWS

Because of its object-oriented formalism, the virtual UWS is easily modeled as a graph (Figure 5-6).



N = number of users, Res = sub-river basin connected and its associated CF_{WD}. DWP = drinking water production, DWD = drinking water distribution, WWC = wastewater collection, WWT = wastewater treatment, U = user, SRB = sub-river basin.

Figure 5-6. Graphical representation of the virtual case study and its extrinsic parameters

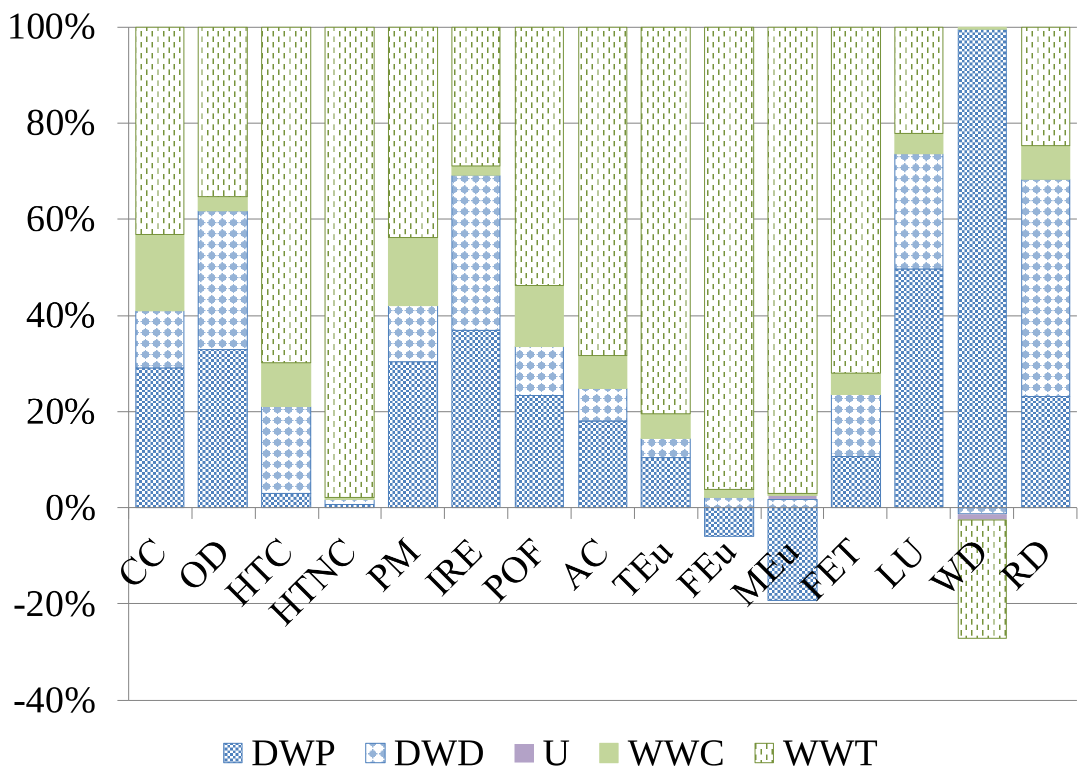
After the computation has been launched, the system returns both computed impacts and services. The aim of the following sections is to display and discuss the general metrics and outputs allowed by the model but not to investigate the results too deeply because they are based on a virtual and simplified case.

5.3.2. Environmental impacts

The results related to impacts are two-fold: contribution of technologies and users (Figure 5-7) and contribution of the direct emissions linked to the foreground activities of the UWS and the indirect emissions linked with three supporting activities (energy, chemicals, and infrastructure) (Figure 5-8). Diverse background activities, such as chemical transportation are included in the chemical group outputs. The model can handle different LCIA methods (Impact 2002+, ReCiPe indicators midpoint and endpoint and Water Impact Index), but only the results related to ILCD midpoints are shown here.

Figure 5-7 shows that the largest shares of impact categories are due to WWT, particularly local impacts, such as eutrophication and ecotoxicity. This is obvious because the majority of pollutants of the UWS are emitted from WWT. One can argue that the majority of these pollutants were initially generated by the users and that the associated impacts could be allocated to them. However, the proposed model allocates the impacts associated with the released flows in each component, and because WWT is the end-of-pipe technology, it is the main contributor. It would be desirable to allocate the impacts associated with released water to the entire UWS. The current mode of representation does not preclude an analysis of pollution sources. Even if most of the pollution comes from the users, other unexpected sources can occur within the system. For example, sulfates emitted at the DWP plants as a result of the use of sulfate-based coagulants (aluminum or ferric) increase the generation of H_2S within the WWC, which is a pollutant and is corrosive to the network (Pikaar et al., 2014).

DWP and DWD also generate a non-negligible share of impacts, primarily in the global impact categories, which are due to background activities. WWC contributes to less than 10% of the impact categories studied because only the impacts of infrastructure are considered for this technology. User contribution is negligible because no technology present at the user place (e.g., water heating systems) was considered in this first theoretical application. Water quantity and quality releases in the environment at the users' places are the only impacts considered. Three impact categories have negative contributions for certain technologies, which means that benefits for the environment occur. First, DWP lowers the impact on freshwater eutrophication and ecotoxicity because it uptakes water and pollutants from the environment and leads to a negative value for impacts, as stated in eq (20). Second, WWT releases water into the local environment and makes it available for the downstream users, thus leading to a negative value for water deprivation. For the water deprivation category, the withdrawn and released water volumes are weighted by CF_{WD}, which is different for the two theoretical water resources, and the release point (lower downstream deprivation). The model also offers a contribution analysis of each technology component, i.e., when there are several plants or networks or when differentiating the sludge end-of-life impacts within WWT or DWP.

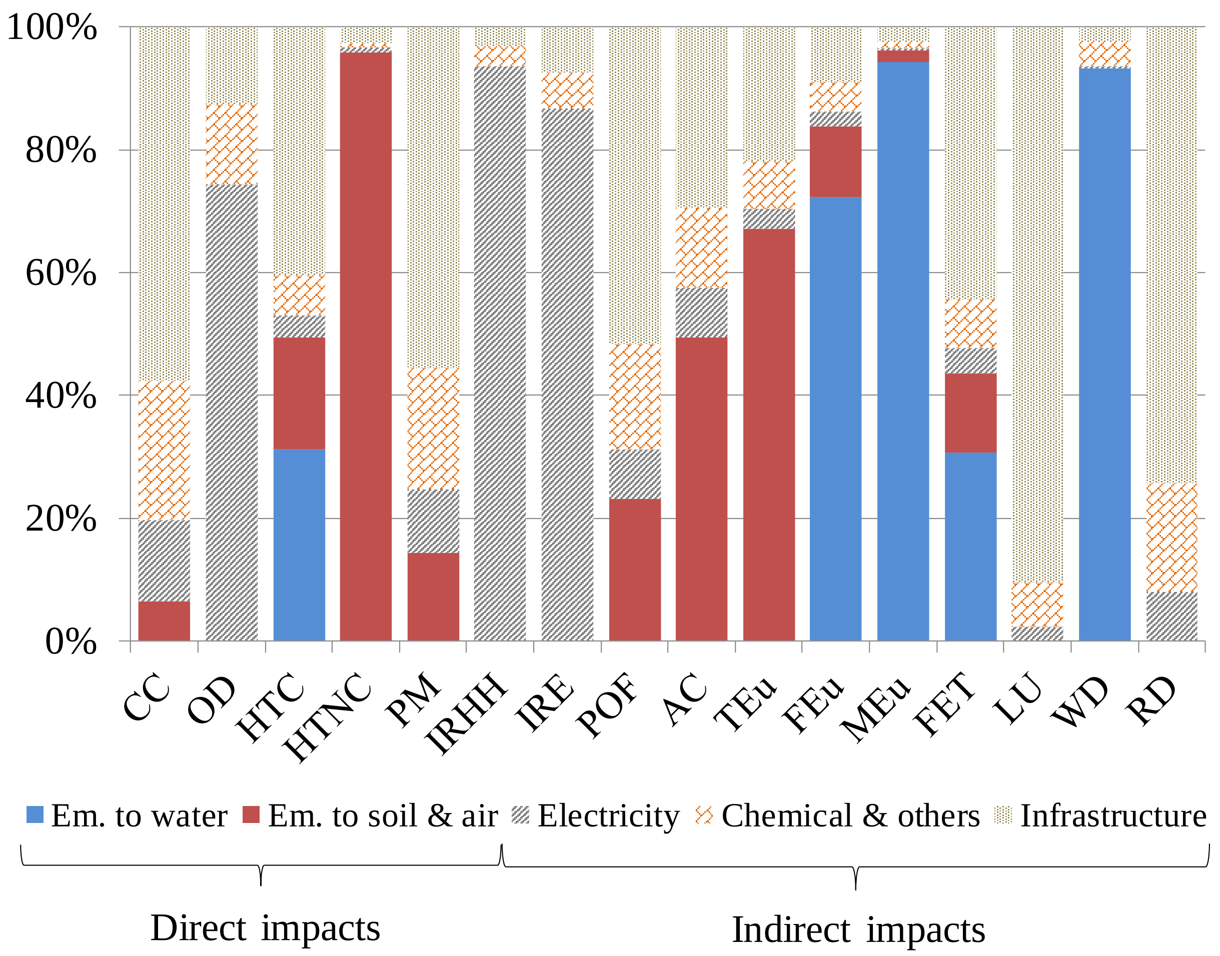


CC = climate change, OD = ozone depletion, HTC = human toxicity cancer effects, HTNC = human toxicity non cancer effects, PM = particulate matter, IRE = ionizing radiation, POF = photochemical ozone formation, AC = acidification, TEu = terrestrial eutrophication, MEu = marine eutrophication, FET = freshwater ecotoxicity, LU = land use, WD = water deprivation, RD = mineral and fossil resource depletion.

Figure 5-7. Relative contributions of technologies and users. The LCIA method is ILCD 1.03.

Figure 5-8 differentiates direct from indirect impacts, thus showing the share of impacts that actually occur on the site. The impacts affecting the water media (marine & freshwater eutrophication, freshwater ecotoxicity and water deprivation) are primarily due to direct interventions in the foreground activities of the UWS. This is also the case for other local impacts (such as human toxicity, acidification and terrestrial eutrophication), which are mainly due to direct emissions to the air and soil from WWT and sludge end-of-life. Land use impacts are primarily generated by infrastructure because of the plants and networks; thus, this category could also have been considered a direct impact. The importance of direct impacts in UWS differs from the results of land planning LCA, which studies all of the activities within a territory where there is a prevalence of indirect impacts (Loiseau et al., 2013). This is because UWS are strongly linked to the local environment through their interactions with water resources, which shows that urban water managers have a key role to play in the environmental management of territories at the local scale.

All global impacts (climate change, ozone depletion, resources depletion) are generated indirectly from background activities, except for climate change, where a low contribution of direct emissions to the air occurs in WWT. Electricity contributes the largest share of ionizing radiation impacts because we considered the French electricity mix. The majority of other indirect impacts are dominated by infrastructure.



CC = climate change, OD = ozone depletion, HTC = human toxicity cancer effects, HTNC = human toxicity non cancer effects, PM = particulate matter, IRE = ionizing radiation, POF = photochemical ozone formation, AC = acidification, TEu = terrestrial eutrophication, MEu = marine eutrophication, FET = freshwater ecotoxicity, LU = land use, WD = water deprivation, RD = mineral and fossil resource depletion.

Figure 5-8. Relative contributions of direct and indirect contributors. The LCIA method is ILCD 1.03.

5.3.3. Provided services and impact/service ratio

In addition to showing the total impacts of the UWS, the model generates an impact/service ratio, which are useful for comparing different scenarios. On the one hand, the values for impacts per user (e.g., capita) allow us to account for user behavior and provide a more complete image of the UWS environmental and social performance. In this case study, only the domestic user has been considered. If we focus on climate change using the above-mentioned scenario, the impact/service ratio results in a value of 30.1 kg CO_2 eq/domestic user/year. On the other hand, the computation of the impacts per m^3 at the user's place pictures the environmental performance regarding the technologies: a value of 0.51 kg CO_2

eq/m^3 is found. This is compliant with the values found in the literature and summarized in Chapter 3 (Loubet et al. 2014), which range from 0.51 to 1.57 kg CO_2 eq./m^3 at the user's place.

5.3.4. Opportunities and limits

This case study is a virtual one and is dedicated to determining whether the WaLA model runs correctly. The data used in this case study are not refined and should be reviewed and improved for future case studies to try to better fit the local characteristics of the UWS.

Regarding water users, this virtual case study has been simplified by only considering domestic users. As stated by Loiseau et al. (2013), the provided services could be calculated according to complex land occupation combining several activities and would thus be a result of the system. This type of model improvement is possible in a future evolution using geographical information system (GIS), for example.

Concerning volumetric water flow distribution vectors, the ecoinvent database has included equilibrated water balances of the various plants since version 3. However, they are neither well documented nor site-specific. Various models in the literature only provide water balances for the whole UWS and not for each component (S. Kenway et al., 2011; Vanham, 2012). Risch et al. (2014)presented water-balanced processes for WWT, but research on water flow inventory remains necessary for other processes. Concerning the water quality distribution vector, it should be noted that the model does not compute a dynamic mass balance of pollutants. The mass balance is defined a priori for each component. Therefore, each component that modifies the water quality (DWP, WWT, U) is to be connected only to a water flow with one specific water quality. As a next step, two options can be explored: either enabling each component to treat different water qualities or computing the mass balance for each component within the model. However, the last option is not straightforward because no current model is able to predict mass balances dynamically and in a consistent way for WWT plants. It could be implemented for DWP technologies (Mery et al., 2013), but it would be time consuming. Additionally, further work on the classification of urban water quality should consider the water quality categories defined by Boulay et al. (2011). Currently, eight types of surface water quality, depending on the usage, can be provided. Therefore, these types should be implemented as water resource qualities and should be only connected to the specific usage they can fulfill.

Regarding specific indirect impacts related to technologies, the results of recent and diverse LCAs should be added to the model's library to free it from ecoinvent generic data. Indeed, the ecoinvent database only includes one technology for DWP (conventional treatment) and one technology for WWT (activated sludge). Other technologies, such as membrane processes (for both DWP and WWT), and wetland systems (e.g., lagoons, polishing pounds, reed bed filter) for WWT should be implemented, based on literature results or local data. However, this is beyond the scope of this chapter. Finally, monthly scale modeling was not used here (i.e., the data were considered constant throughout the year), whereas water cycle conditions may exhibit significant variations during the year. Efforts should be made to gather monthly data and provide monthly characterization factors, particularly for water deprivation.

An inherent problem of stand-alone LCA tools using the impact results calculated from LCA software (Simapro 8) is updating the LCIA results because the LCIA methods and ecoinvent database are modified in different versions. This could be solved by using a database management service within the modeling tool.

Despite these points for improvement, the level of usability and interactivity for a non-specialist is very good for generating scenarios. A great advantage is that despite this first prototype model, which uses proprietary software (Matlab/Simulink), it can be implemented using any other languages or software due to OOP.

5.4. Conclusions

Our objective was to develop a framework to tackle the methodological challenges raising from LCA applied to UWS. A versatile model, WaLA, has been built to consistently and easily determine water flows, related environmental impacts and services within any UWS. The implementation of the model through an object approach and a Matlab/Simulink interface provides a usable and operational tool. Thanks to the proposed framework and OOP implementation, various UWS scenarios can be easily designed and tested by water authorities, industries, or academic institutions that intend to design their own tool for UWS environmental assessment. However, it still can be refined through better management of water quality, the implementation of uncertainty analysis based on error propagation, or Monte-Carlo simulations.

A first virtual case study based on ecoinvent processes has been tested and demonstrated the capacity of the modular approach to easily build the UWS. The results of the contribution analysis based on technologies have shown the predominance of impacts due to WWT. It also reveals the importance of direct impacts on local issues (e.g., eutrophication) and the high contribution of supporting activities (energy, chemicals, infrastructures) on global impacts. The impact/service ratios based on the provided services of the UWS (either users or m^3 delivered to the user) are useful results that can be compared with other systems or scenarios.

The appropriation of such tools by stakeholders was not the objective of this work but will be a great challenge when performing environmental evaluations in a decision-making context. The evaluation of forecasting scenarios is the object of Chapter 6, which focuses on the urban water system of the Paris suburban area (France).

Chapter 6. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 2 – Learning points from the assessment of water management scenarios in Paris suburban area

« Paname

Quand tu t'ennuies tu fais les quais

Tu fais la Seine et les noyés

Ça fait prend' l'air et ça distrait »

Léo Ferré - Paname

The previous chapter presented a model to perform LCA of UWS, and its implementation to graphical user interface. Its application to a virtual case study has been carried out. In this chapter, the WaLA model is applied to a real case study: the urban water system of the Paris suburban area, in France. It aims to verify the capacity of the model to provide environmental insights to stakeholder's issues related to future trends influencing the system (e.g., evolution of water demand, increasing water scarcity) or policy responses (e.g., choices of water resources and technologies). This is achieved by evaluating a baseline scenario for 2012 and several forecasting scenarios for 2022 and 2050. The scenarios are designed through the modeling tool presented in Chapter 5, which is implemented in Simulink/Matlab: it combines components representing the different technologies, users and resources of the UWS. The life cycle inventories of the technologies and users components include water quantity and quality changes, specific operation (electricity, chemicals) and infrastructures data. The methods selected for the LCIA are midpoint ILCD, midpoint water deprivation impacts at the sub-river basin scale, and endpoint Impact 2002+. The results of the baseline scenario show that the majority of impacts occur in wastewater treatment plants, as traditionally encountered in LCA of UWS. Fitting forecasting scenarios into the model suggests its simplicity of use and its capacity to deliver information useful for decision making about future policies, notably with regards to the effects of water deprivation. This chapter refers to the following paper submitted to Water Research: "Loubet, P., Roux, P., Guerin-Schneider L. & Bellon-Maurel, V. WaLA, a versatile model for the life cycle assessment of urban water systems: Part 2 – Learning points from the assessment of water management scenarios in Paris suburban area"

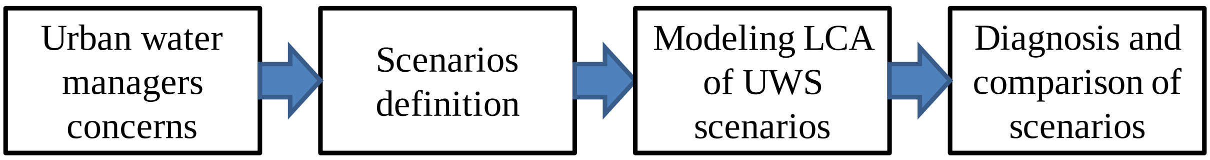


Figure 6-1. Graphical abstract of Chapter 6

Content of Chapter 6

6.1. Introduction

Stakeholders face many challenges related to the management of urban water system (UWS): demographic, water demand and water resource changes (e.g., effects of climate change) (McDonald et al., 2011). The decision making process covering future evolutions should include the environmental evaluation of forecasting scenarios based on the planned modifications of the UWS. Unfortunately, holistic approaches such as life cycle assessment (LCA) are time-intensive and require a high degree of expertise. Chapter 5 proposed a formalism and developed a model, namely WaLA, for applying a LCA to UWS (as a modular approach), to reduce the complexity of UWS environmental evaluation. After showing the feasibility of this method on a virtual case study, this model is applied to a real case study to compare various evolutionary scenarios.

Various types of scenarios have been formalized, differentiating future-trend based scenarios and policy-responsive scenarios (Mahmoud et al., 2009). The first scenarios are based on extrapolations that can be either projective, i.e., using trends experienced over a past period, or prospective, i.e., anticipating upcoming changes that differ from the past. UWS typically include endured changes, indicating that stakeholders of water service institutions do not control the parameters associated with water management, such as urban development or climate change effects on water resources. Conversely, policy-responsive scenarios anticipate events or actions, but with high subjectivity. These scenarios are either based on expert judgment or driven by stakeholders.

A large set of management questions are classified in Table 6-1 according to the type of associated scenario, the questions asked by the stakeholders, the nature of the stakeholder involved and their scale of action. These questions have been identified from a review of UWS LCAs (Loubet et al., 2014), and from water service experts. The model developed in Chapter 5 aims to provide the environmental assessment of these management questions.

Table 6-1. Classification of identified management issues.

Types of scenario: Main UWS components affected	Concerns	Future trend ?	Policy-respon sive ?	Potential questions to be addressed	Stakeholders involved	Relevant scale	Example of LCA literature assessing this concern	UWS model presumed as an appropriate tool ?
Users	Evolution of urban development: population, economic activity, etc.	X		To compare different trends of urban development	Local authority in charge of water services. Local authorities in charge of town planning. Operator.	City	(Lundie et al., 2004)	Yes
	Evolution of water demand	X	X	To compare different trends of water demand reductions	Local authority in charge of water services. Operator.	UWS		Yes
Resources	Evolution of resources and associated infrastructures		X	To compare alternative choices of water resources	Local authority in charge of water services. Operator.	UWS	(Muñoz et al., 2010)	Yes
	Evolution of the stress level on resources	X		To compare different hypotheses of water resource stress because of climate change	Authority in charge of river basin management.	River basin		Yes
	Daily management of resources		X	To select resources used in daily operation	Operator	UWS		No (dynamic tools are needed)
Technol-ogies	Evolution of technologies used		X	To compare different technologies within a specific UWS	Local authority in charge of water services. Operator.	UWS	(Lemos et al., 2013)	Yes
	Modification of processes		X	To evaluate standalone new processes or new technologies	Operator.	Technolo gy	(Mery et al., 2013)	No (process-based LCA is needed)
	Daily management of water technologies		X	To select technologies used in daily operation	Operator	UWS		No (dynamic tools are needed)
All	Contribution of activities to environ-mental impacts			To assess current situation and identify environmental hotspots and contributions	Local authority in charge of water services, operator	UWS	see (Loubet et al 2014)	Yes
	Operation of the service		X	To compare different short-term contractual environmental policies	Operator	UWS	(Barjoveanu et al., 2013)	Yes

The objective of this chapter is to implement the UWS model using a real case study located in suburbs of Paris and to verify the capacity of the model to address identified stakeholders' questions. These objectives achieved by evaluating forecasting scenarios associated with specific concerns covering modifications of users, resources or technologies within the system. After a description of the various scenarios and associated questions they intend to address, the UWS model developed in Chapter 5 is applied. The results for the baseline scenario are the water flows (characterized by their quantities and qualities) the environmental impacts and impact/service ratios. The results provide a comparison of the forecasting scenarios in regards to their environmental impacts. Based on this comparison, conclusions and perspectives on the environmental evaluation of UWS using the proposed model are provided.

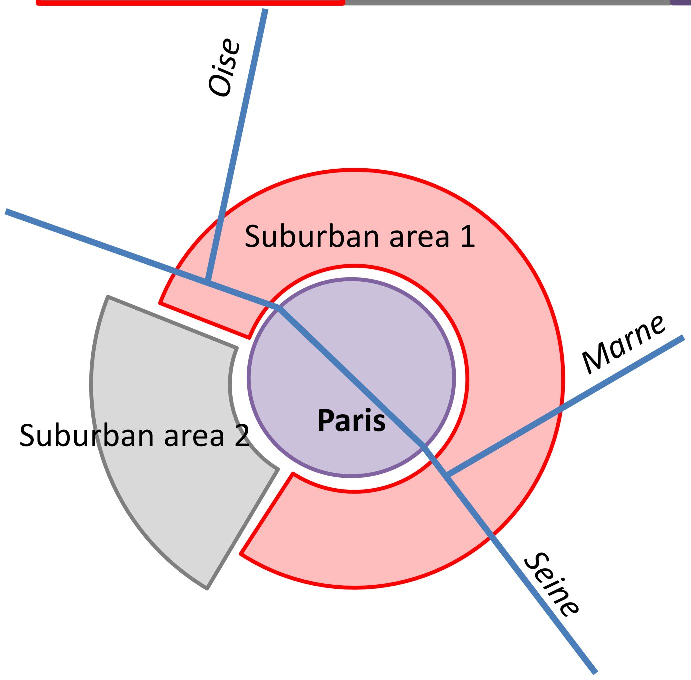
6.2. Material and methods

6.2.1. The greater metropolitan Paris UWS

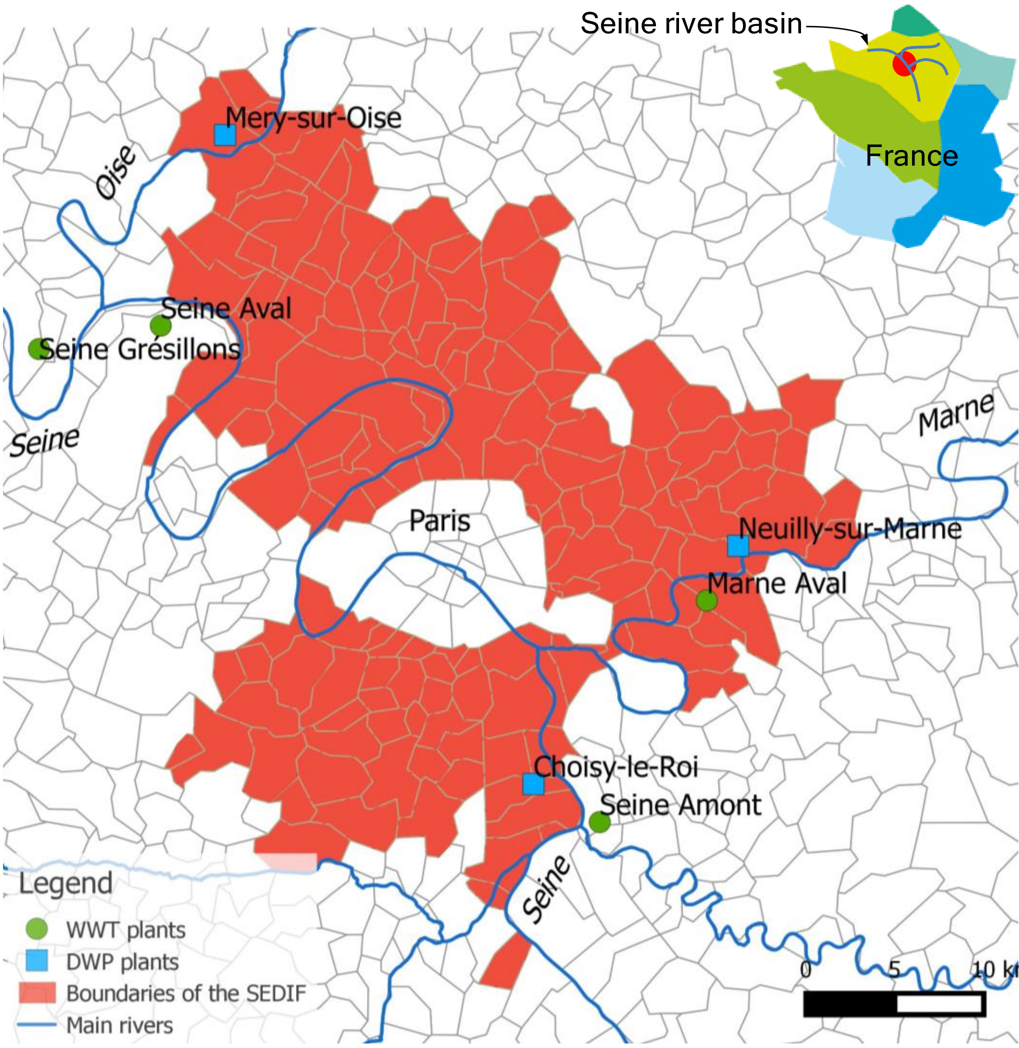
In France, UWS fall under the responsibility of public local authorities at the municipal level. However, this responsibility can be transferred to intermunicipal organizations to take advantage of economies of scale. Such transfers can be full or partial. For instance, drinking water production (DWP) is transferred to the intermunicipal level, whereas wastewater collection (WWC) remains under the responsibility of each municipality. The operation and investment of these water services either remain under direct public management or are delegated to a third party, typically a private operator (Guerin-Schneider et al., 2002). In this context, a water service can be defined as a set of infrastructures and related services that are under the responsibility of one given local authority and under the operation of one given operator. In 2010 more than 14,000 water service and more than 17,200 collective sewerage services operated in France (EauFrance, 2012). Thus water management in the greater metropolitan Paris area is complex as it is composed of many water services. Numerous DWP and drinking water distribution (DWD) services are in operation. The two main services are Eau de Paris (direct management) that covers the city of Paris and Syndicat des Eaux d'Île-de-France (SEDIF, delegated management) that covers more than half of the suburban area. More than ten other DWP and DWD services cover the other cities within the suburban area. WWC is managed at three scales: city collection managed by municipal authorities, departmental transport managed by intermunicipal syndicates and interdepartmental transport managed by the Syndicat Interdépartemental pour l'Assainissement de l'Agglomération Parisienne (SIAAP). Wastewater treatment (WWT) is also managed by the SIAAP. Other small services manage WWT in further sub-urban areas that are not considered in this Chapter. Table 6-2 summarizes the different water services in the Parisian area according to the geographical area they cover and the systems they manage.

Table 6-2. The complexity of water management in the greater Paris metropolitan area: responsibility shares for the different components. Area of the case study is underlined in red.

Components	Greater metropolitan Paris area		
	Paris suburban area 1	Paris suburban area 2	City of Paris
DWP	SEDIF	Several intermunicipal syndicates	Eau de Paris
DWD			
WWC Collection (City scale)		Several municipalities	
WWC Transport (Department scale)	Several intermunicipal syndicates		
WWC Transport (Interdepartmental scale)	SIAAP		
WWT			



Our case study focuses on the urban water system of the SEDIF geographical perimeter, named "Paris suburban area 1" and highlighted in red in Table 6-2 and in Figure 6-2. This focus results from the fact that the main stakeholders involved in the decision making process is the SEDIF and its delegatee, Veolia Eau d'Île-de-France. A map of the case study is provided in Figure 6-2.



The WWT and DWP plants are : CR = Choisy-le-Roi, MO = Mery-sur-Oise, NM = Neuilly-sur-Marne, MA = Marne Aval, SAm = Seine amont, SAv = Seine aval, SC = Seine centre, SG = Seine Grésillons, SM = Seine Morée.

Figure 6-2. General and detailed situation of the case study.

6.2.2. Scenarios investigated and the associated LCA goals and scopes

6.2.2.1. Goal and scope

In a multifunctional system, the goals are complex and can have different dimensions. In this study, a set of stakeholder's questions are selected from Table 6-1 to establish and investigate different scenarios and test the capacity of the model. Table 6-3 presents the different scenarios that are to be assessed and the related questions.

Table 6-3. List of evaluated forecasting scenarios and their key parameters.

Ref	Stakeholder question addressed	Modified parameters related to users		Modified parameters related to resources		Modified parameters related to technologies	
		Policy-responsive	Future-trend	Policy-responsive	Future-trend	Policy-responsive	Future-trend
S1	To compare different short-term environmental policies of the DWP operator	-	↑ Population (+3.3%) ↓ Specific water demand (-5%)	-	-	↑ Water losses in DWD (15%)	-
S2		↓ Water demand (-0.8%)	↑ Population (+3.3%) ↓ Specific water demand (-5%)	-	-	↓ Electricity use DWP&D (-6%) ↓ Chemicals used DWP (-3%)	-
S3		↓ Water demand (-2.7%)	↑ Population (+3.3%) ↓ Specific water demand (-5%)	↑ withdrawals in upstream plants	-	↓ Water losses in DWD (5%) ↓ Electricity use DWP&D (-10%) ↓ Chemicals used DWP (-6%)	-
L1	To compare different trends of urban development and water demand	-	↑ Population (+9.3 %) ↓ Specific water demand (-21 %)	-	Climate change effects	-	-
L2		-	↑ Population (+9.3 %) → Specific water dem	-	Identical to L1	-	-
L3		-	↑↑ Population (+21%) → Specific water dem	-	Identical to L1	-	-
L4	To compare trends in climate change effects	-	Identical to L1	-	No climate change effects	-	-
L5	To compare alternative choices of water resources	-	Identical to L1	Water transfer (42%) from downstream river	Identical to L1	Membrane DWP technology to treat downstream river water (low quality of water)	-
L6		-	Identical to L1	Water transfer (15 %) from upstream source	Identical to L1	Simple DWP technology to treat upstream source water (high quality of water)	-
L7	To compare different technologies within a specific UWS	-	Identical to L1	-	Identical to L1	All DWP are membrane processes	-
L8		-	Identical to L1	-	Identical to L1	All DWP are conventional processes with electricity and chemical consumption of S3	-

As explained in Chapter 5, we follow the "territorial LCA" approach (Loiseau et al., 2013): for each scenario, a "reference flow" is studied, which is the association of the studied UWS within its geographical borders and the implemented scenario. A set of functions is then selected to describe and quantify the multifunctional services provided by the UWS, i.e., supplying different categories of users with water. The indicators associated to these functions are used in the "territorial LCA" to calculate impact/service ratio (inverse of eco-efficiency ratio from Seppäläa and Melanen (2005)) as the ratio between the given impacts and the function indicators (e.g., kg CO_2/hab). Six different types of users and associated indicators that relate the provided service are identified within the urban territory. Domestic users are described by the number of inhabitants in the area. For other users, the five categories of activities, based on INSEE NAF5 classification (INSEE, 2013) are used. Three of these five categories are found in the system: i.e., non-market services (public administration, education, health and social work), market services (commerce, transports, construction and diverse), and industries. These categories were characterized by the number of associated jobs. Because agriculture activity is low in the system (less than 350 jobs), this category has been included in the "other users" category. The "other users" category also includes urban watering, street washing, and firefighting. This category is expressed in total surface of the system (hectares). A last category is defined to encompass all users (i.e., "equivalent inhabitant"), and the associated indicator is the number of inhabitants. Because the selection of the function is dependent on the stakeholder's goal and scope, an alternative and complementary function is defined: one cubic meter delivered to the users.

Boundaries include all components of the UWS, i.e., DWP, DWD, U, stormwater collection (SWC), WWC, WWT. The DWP and WWT components include sludge end-of-life. The operation and infrastructure are considered, but not the associated civil works. The boundaries of the baseline and forecasting scenarios are further explained hereafter.

6.2.2.2. Description of the baseline scenario

In 2012, the perimeter of SEDIF represented 142 towns in the suburbs of Paris. This region comprises a total of 4.3 million inhabitants and an area of 76,198 ha. The specific water demand for each type of user is determined by combining demographic and employment data from French national statistics at the municipal scale (INSEE, 2013) with the customer database from the SEDIF delegatee (namely Veolia Eau d'Île-de-France, confidential source). From the database, the volume of water sold to the different categories of users is known for 2012. The resulting data reported 4,362,705 domestic users (and equivalent inhabitants), 413,251 non-market jobs, 1,051,485 market jobs, 153,208 industry jobs with the average water demand of 39.2 m^3/year/domestic user, 70.2 m^3/year/non-market services job, 23.6 m^3/year/market services job, 43.1 m^3/year/industry job, 83.7 m^3/year/ha for other users, and 54.2 m^3/year/equivalent inhabitant.

Water is produced in four main DWP plants: (i) Choisy-le-Roi (CR), treating Seine river water through a conventional process with a production of 110 Mm^3 in 2012; (ii) Neuilly-sur-Marine (NM), treating Marne river water through a conventional process with a production of 93 Mm^3 in 2012; (iii) Méry-sur-Oise (MO), treating Oise river water through both conventional and membrane processes with a production of 55 Mm^3 in 2012; (iv) Arvigny (A), treating groundwater from the Champigny reservoir through a simplified treatment process with a production of 8 Mm^3 in 2012. A remaining amount of 3 Mm^3 was produced in 2012 from minor plants and imported from other services (SEDIF, 2012). The conventional process for DWP typically consists of coagulation, flocculation, decantation, sand or activated carbon filtration, and disinfection (ozonisation, UV, chlorination). The membrane process for DWP consists of flocculation, decantation, pre-filtration, high pressure filtration, nano-filtration and disinfection (UV, chlorination). DWP sludge is spread in agricultural fields.

Water is then distributed in a 8275-km long network (DWD). The DWD is composed of a 772-km long primary network of pipes having a diameter more than 300 mm and a 7503-km long secondary network of pipes having a diameter less than 300 mm (SEDIF, 2012).

WWC is performed at municipal and departmental levels, with an approximate length of 1.5 m/capita (AESN, 2007). The wastewater transport main network (emissary) has a length of 440 km with pipes ranging from 2.5 to 6 m in diameter (SIAAP, 2014).

WWT is performed in 4 activated sludge plants: (i) Seine Aval (SAv), treating 610 Mm^3; (ii) Seine Amont (SAm), treating 138 Mm^3; (iii) Seine Grésillons (SG) treating 20 Mm^3; (iv) Marne Aval (MA), treating 30 Mm^3 (SIAAP, 2012). All plants perform conventional treatment including pretreatment, primary and secondary decantation (carbon and suspended solids elimination), nitrification/denitrification, dephosphatation and sludge treatment. Sludge is either spread in agricultural fields (as compost or dry sludge) or incinerated. WWT plants do not exclusively treat water from SEDIF perimeter users because SIAAP treats water for the greater Paris area, as shown in Table 6-2. For LCA purposes, an allocation is performed with regard to the volume of the water share treated for users within the perimeter of the case study. Identifying volume shares treated in each WWT plant for the SEDIF perimeter users is not trivial and is explained in Annex C.5. This analysis concludes that 52% of SAv, 67% of SAm, 29% of SG and 100% of MA are used for treating water from the SEDIF perimeter.

Stormwater volumes entering the collection network is independent of the water demand and should be fixed. In this case study, only stormwater which ultimately enters WWT plants is accounted for. This volume has been estimated for 2012 as the difference between the actual volumes treated by WWT plants for the SEDIF perimeter and the expected raw wastewater produced by water users (see Annex C.5). This results in 220.69 Mm^3 of stormwater collected, from which 79% flows to SAv, 16% to SAm, 2% to SG and 3% to MAv. The percentage of stormwater with regard to total water in the four WWC networks associated with the WWT plants has been estimated as follows: 57% in the SAv network, 40% in the SAm network, 56% in the SG network and 26% in the MAm network. This amount of stormwater and these ratios are identical for all forecasting scenarios. This assumption may not be accurate because climate change will affect precipitation in the area.

6.2.2.3. Description of forecasting scenarios

Forecasting scenarios are adapted from the baseline with the modifications detailed in Table 6-3. The proposed policy-responsive scenarios are expert judgment-driven, indicating that they study criteria established by researchers and field experts, but that they do not intend to have a political plausibility, contrary to stakeholder-defined scenarios. Three short term scenarios, for 2022 and eight middle term scenarios, for 2050, are defined through the implementation of parameters related either to users, resources or technologies. They combine the two types of identified scenarios: (i) future trend scenarios (e.g., population evolution) and (ii) policy responsive scenarios (e.g., changes in water resources or technologies).

For the establishment of forecasting scenarios, INSEE (2010) projects a 3.3% and 8.3% increase in population in the region Île-de-France in 2022 and 2040, respectively. Following these trends, we considered an increase of 9.7% for 2050. The high population increase scenario projects a 21.4% increase. As for water demand, a decrease of 0.4% per year for vertical housing and 0.8% per year for suburban houses is estimated in France (BRL Ingénierie, 2012). Considering a proportion of 69% houses and 31% apartments in Île-de-France, the decrease in water demand is of 0.52%/year. All these projections were considered equivalent for all types of users (domestic, market and non-market services, and industries).

Short-term scenarios aim to study operational changes without infrastructures modifications within the drinking water service, which is the main decision making service in this case study. These scenarios follow future trends in the evolution of population. The evolving trend of specific water demand is modulated by different policies towards the behavior of users from water service bodies. The present technologies are used with minor improvements to the processes, i.e., decreased energy and chemicals used. Scenario S1 is a business-as-usual scenario, following forecasting trends in water demand and low maintenance efforts of the DWD network and occasioning an increase of water losses. Scenario S2 is a scenario following contractual actions from the delegatee. The specific water demand per inhabitant is reduced by 0.6%/year (instead of the expected trend of 0.52%) because of a communication program geared towards the users and the resulting increase in awareness. Water losses in DWD decrease, leading to a network performance of up to 90%. Electricity consumption of the DWP plants and DWD networks and the chemical consumption in the plants are reduce by 6% and 3%, respectively, because of the optimization of the processes and pumps. Scenario S3 is an eco-designed scenario aiming to achieve a lower water footprint, i.e., reducing water deprivation impacts. The specific water demand is reduced by 0.8%/year (instead of the expected trend of 0.52%) because of the installation of domestic eco-designed equipment (e.g., tap aerator). The reduction of water losses in DWD allows a performance of the network of 95%. A higher reduction of the electricity and chemical consumed by the plants and networks are considered. Additionally, water is withdrawn preferably from downstream locations to reduce water deprivation impacts.

Long term scenarios aim at modeling large and structural changes for 2050. Scenario L1 is similar to the baseline scenario with the modification of future-trend parameters (increase of population, decrease of water demand, effects of climate change on water scarcity). Scenario L2 and L3 study different hypothesis in the trends regarding two different projections of population (medium and high) and the evolution of the water demand per capita. Scenario L4 is similar to L1 without considering the effect of climate change on water resources. Scenario L5 and L6 study changes in resource withdrawal choices, either transferring low quality water from downstream Seine (L5) or transferring clean water from upstream sources (L6). Scenarios L7 and L8 study changes in DWP technologies, either selecting advanced treatment processes such as membrane processes or selecting low impact technologies.

6.2.3. Customization of the model components: establishing the attribute values

As defined in part 1, the model is based on generic model components, which are customized to represent the different technologies (plants, networks) of the system. Following the object-oriented programming formalism, this selection results in different sub-classes (DWP, DWD, etc. i.e., one per type of technology) and specific instances (DWP_CR, DWP_NM, etc. i.e., one per plant). Each instance is characterized by a set of attributes, namely the volumetric water flow distribution vector **v** (i.e., the way water volumes are partitioned at the output), the quality distribution vector **q** (i.e., the level of quality at the output) and the emission/ consumption inventory associated with impact matrix **i**. Each instance used in the scenarios is specifically detailed in Annex C.2.

6.2.3.1. Volumetric water flow distribution

All flows going in and out of the technologies/users component are computed from volumetric water flow distribution vectors **v**. These values are derived from local measurements, literature data, models, or mass balances. Annex C.2 describes the equations used for each component sub-class of the UWS (e.g., v_DWP), and develops the yearly average water flow distribution vectors for each instance of the sub-class (e.g., v_DWP_CR).

6.2.3.2. Quality water flow distribution

The matrix characterizing the quality level of water flows in the UWS has been adapted for the case study from the water quality classification introduced in Chapter 4. A full composition is provided in Annex C.3.

The drinking water (of type A) composition is obtained from the DWP water service company (SEDIF, 2012). The composition of river water (indices B1 to B6) for different streams located in the case study sub-river basins are retrieved from average composition over 2009-2014 compiled by the Seine river basin authority (AESN, 2014) for all pollutants of the European water framework directive. The composition of DWP release (indices C1 to C4) is retrieved from local data measurement at the three main plants (CR, NM, MO), for which heavy metals (notably aluminum), nitrogen, phosphorus and COD concentrations have been monitored. The raw wastewater composition (index D1) is computed from the definition of the population equivalent (PE) in France, stating that one PE emits 60 g BOD5/day, 135 g DCO/day, and 15 g NTK/day. The concentration of phosphorus is 2.3 g P/day for domestic users and 2 g P/day as an average for all urban users (Stricker and Héduit, 2010). As the total volume of wastewater collected per inhabitant is 133 L/day in 2012, the concentrations of raw wastewater pollutants are 444.5 mg BOD/L, 1000 mg COD/L, 111.1 mg NTK/L and 14.8 mg P/L when considering all water users. The composition of urban runoff, from roofs, roads and yards has been measured in the city of Paris (Gromaire-Mertz, 1998). For stormwater, this runoff provides average values for COD, BOD5, cadmium, copper and zinc. The wastewater quality in the collection network (indices D3 to D6) results from raw wastewater and stormwater mixing according to their different shares explained in section 6.2.2.2. The composition of treated wastewater (indices C5 to C9) is based on local measurements of nitrogen and phosphorus emissions (SIAAP, 2012). Heavy metals and emerging pollutants in wastewater are taken from ecoinvent 3. In each component, mass balances are equilibrated with air and soil emissions as shown hereafter.

6.2.3.3. Direct emissions to air and soil

Emissions to the air during DWP processes are only accounted for in membrane processes with the release of CO_2 during stripping after nanofiltration. Small amounts of ozone emissions to the air can occur when ozone is produced in-site; however, most ozone is treated. Emissions to the air and soil occurring during end-of-life of DWP sludge, especially aluminum emissions, are evaluated from the amount of aluminum sulfate introduced in the coagulation process by equilibrating mass balances.

In WWT processes, the emissions to the air include multiple sources and more complex to estimate. Nitrogen emissions to the air occur during nitrification/denitrification (N_2O released in the air), the digestion of sludge and the incineration of biogas (NO_2, N_2O, and NH_3 released in the air). These emissions are evaluated according to the ecoinvent model (Doka, 2009). Residual nitrogen, calculated as the difference between the WWT plant nitrogen input and the emissions to the water and air during the process, is embedded in the exported sludge. In sludge spreading, emissions of NH_3 and N_2O to the air and uptake of nitrogen by the plants occur (see Annex C.6.3). The phosphorus exported through the sludge is simply the difference between phosphorus contained in raw wastewater and phosphorus released in water. In sludge spreading, the phosphorus that is not taken up by the plants is released into the soil. Heavy metals emissions are modeled following the ecoinvent process in sludge end-of-life. In sludge incineration, ecoinvent emissions for all pollutants have been considered, leading to a non-equilibrated mass balance for nitrogen and phosphorus during this stage.

6.2.4. Inventory linked to operating the UWS components (energy, chemicals)

To compute impacts matrixes **i** of water technologies, LCI related to supporting activities (i.e., energy, chemicals and infrastructures) are needed. They are described hereafter for each technology and full LCI tables are provided in Annex C.4.

DWP technologies of the case study are either conventional or membrane-based ones. Data are based on local measurements, but full inventory details cannot be provided because of confidential information. The energy use for conventional treatment ranges from 0.25 kWh/m^3 (CR and NM plants) to 0.36 kWh/m^3 of water produced (MO conventional plant). The chemicals used include aluminum sulfate, polymers, liquid carbon dioxide, powder activated carbon and sulfuric acid for the clarification step; liquid oxygen for the ozonation step; granulated activated carbon for filtration; phosphoric acid, sodium hypochlorite, sodium chloride, sodium hydroxide and sodium sulfate for disinfection/stabilization; and quicklime for sludge treatment. The energy use for membrane treatment is 0.73 kWh/m^3 of water produced. The chemicals used are similar to conventional technologies with the addition of ethylenediamine-tetracetic acid, polycarboxylates and sodium tripolyphosphate for membrane washing and epoxy resin, glass fibers and polyvinylchloride for membranes replacement. The transport of chemicals is considered. DWP sludge end-of-life operation includes transport and spreading.

DWD operation only considers the electricity consumption for the pumping at the output of plants and all along the network. Electricity data are based on local measurements, resulting in 0.376 kWh/m^3 at the input of the network. WWC is considered to be gravity driven and does not use energy.

WWT plants are conventional designs. Data covering WWT is public (SIAAP, 2012) but is not as detailed as for DWP technologies. The energy sources are diverse (electricity, gas, oil), and part of the energy is auto-produced in the plant. Overall energy use ranges from 0.94 kWh/m^3 (SG plant) to 1.55 kWh/m^3 (SAm plant). Excluding auto-production (from biogas burning), the energy use ranges from 0.69 kWh/m^3 (SAm plant) to 1.25 kWh/m^3 (MAv plant), which are slightly higher than values found in the literature (Loubet et al., 2014). WWT plants use ferric chloride (0.056 kg/m^3), calcium nitrate (0.030 kg/m^3, methanol (0.043 kg/m^3) and polymers (0.015 kg/m^3). The chemical consumptions are considered as identical for all WWT plants in the case study because only overall consumption for all plants is known.

6.2.4.1. Inventory linked to the infrastructure of UWS

The LCI for DWP infrastructures is compiled from local data based on real plants. The materials for each component are considered, including buildings. Pumps are mainly composed of cast iron, steel and copper. Pipes are composed of cast iron or cement. Sand is required for the filters. Based on expert judgment, different lifetimes were adopted: 100 years for buildings, 40 years for pumps, 50 years for pipes and 100 years for sand and anthracite (for filters). The LCI for DWD and WWC infrastructures are built from ecoinvent processes relative to the grid length. The DWD network length is precisely known (SEDIF, 2012), i.e., 8275 km, and the process "water supply network, construction" (Althaus et al., 2007) is selected with a lifespan of 50 years. However, the WWC network is managed by several authorities (Table 6-2), and the precise length of this network is complex to evaluate. A first assumption based on the required length per capita in the suburban Parisian area is adopted, i.e., 1.5 m/capita (AESN, 2007) with an expected lifespan of 100 years. As for WWT, ecoinvent includes five different plants, from class 1 to class 5, with a capacity ranging from 47 to 0.16 Mm^3/yr (Doka, 2009). All case study plants have a total capacity in the range of, or higher than, class 1 levels (27.5 Mm^3/yr to 620.5 Mm^3/yr). Therefore, we assume that the infrastructure needed is correlated to the treatment capacity (in m^3) of the class 1 plant. The lifespan of the plants is considered to be 30 years (ecoinvent assumption). Emission and consumptions because of infrastructure are allocated to the plants according to a volumetric allocation on the volume treated as explained in section 6.2.2.2. The land use dedicated to the plants are computed by adding the plant areas measured on Google Maps Engine (Google, 2014). Forecasting scenarios consider the identical infrastructure as baseline scenario technologies.

6.2.5. Life cycle impact assessment

6.2.5.1. Water deprivation impact

Characterization factors for water deprivation, i.e., $CF_{WD,}$ are computed at the sub-river basin scale according to the methodology introduced by Loubet et al. (2013). CF_{WD} have been refined and updated for the Seine river Basin regarding the basin delineation and the spatial and temporal scale of runoff and water consumption data. The sub-basin delineation is derived from the recent HydroBASINS database, which is based on HydroSHEDS digital elevation model (Lehner and Grill, 2013). This database provides updated and consistent sub-basin boundaries at various scales. The selected scale for the Seine basin study includes 110 nested sub-river basins instead of the 20 in the Chapter 3 (Loubet et al., 2013), as shown inFigure

6-2. Monthly runoff data for 2012 are computed from the GLDAS model (NASA, 2012) to match meteorological and hydrological conditions of the baseline scenario. The data for 2012 was representative of normal conditions in France. Runoff data for forecasting CF_{WD} at the horizon 2050 are based on previsions models accounting for climate change in the Seine river basin (Ducharne et al., 2009). A 20% decrease of runoff during the summer months in 2050 has been considered. Monthly water consumptions are estimated from Hoekstra et al. (2012). Figure 6-3 shows monthly CF_{WD} for November 2012. Resulting CF_{WD} for the baseline and forecasting scenarios are fully presented in Annex C.7.

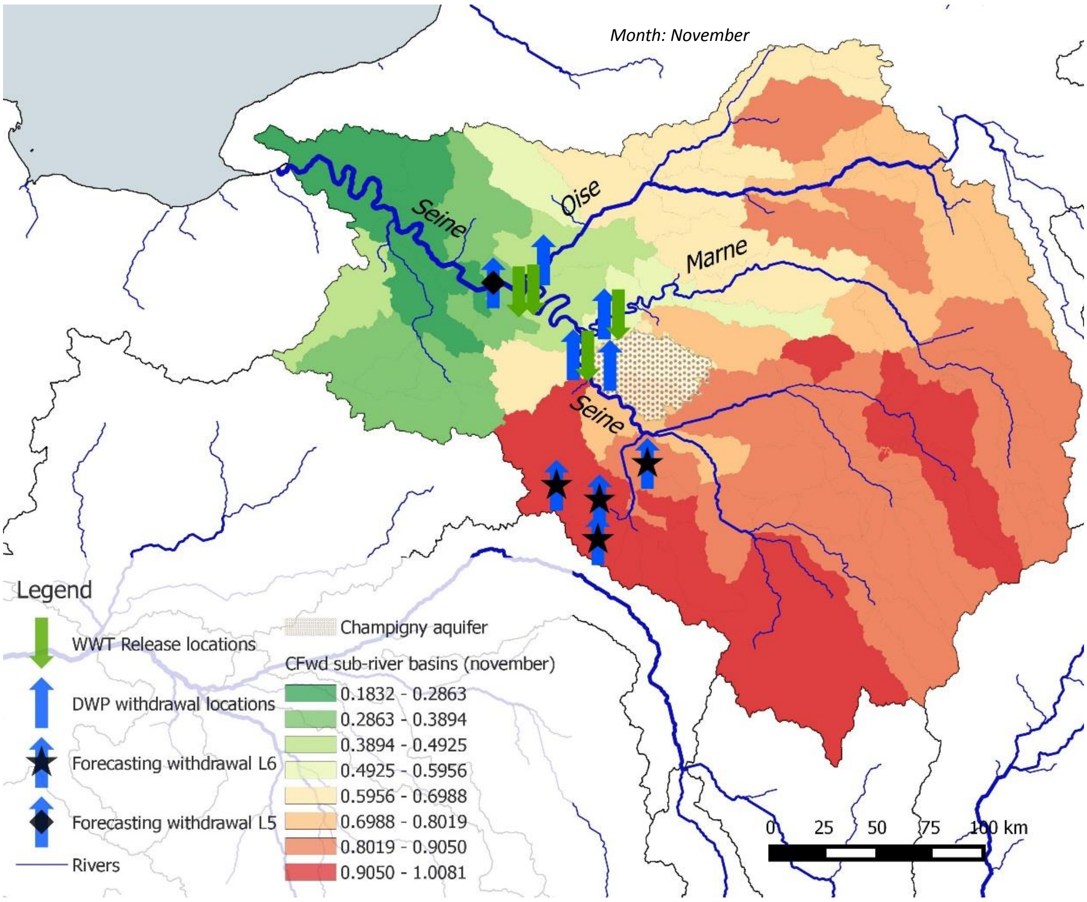


Figure 6-3. CF_{WD} for the Seine river basin (November) and locations of main withdrawals and releases for the baseline and forecasting scenarios.

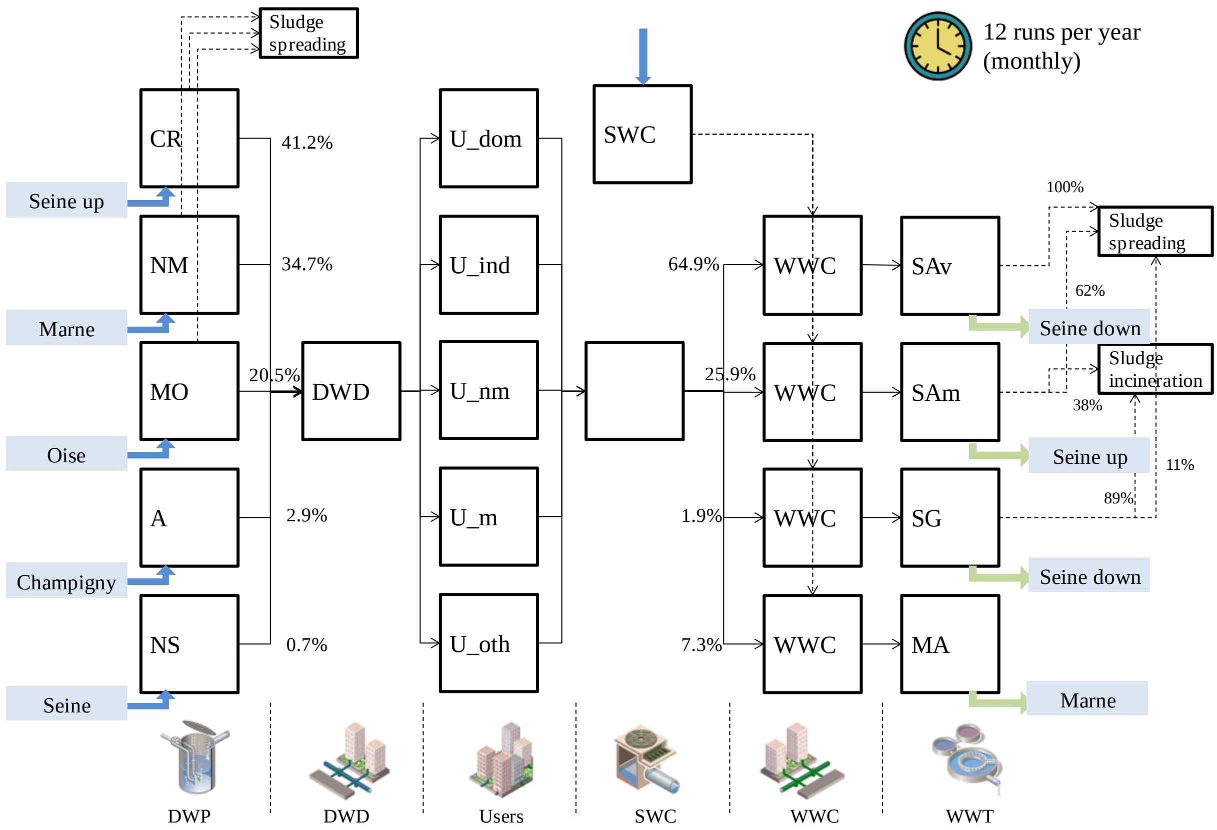
The locations of water withdrawal and water release at the sub-basin scale are diverse. The main locations corresponding to DWP and WWT plants are shown in Figure 6-3. These plants are located in four sub-basins: downstream Marne, downstream Oise, downstream and upstream Seine. Withdrawal locations planned in forecasting scenarios L5 and L6 are also shown in downstream Seine and sources of the Loing river.

6.2.5.2. Other impacts

Others impact categories are evaluated at the midpoint level with ILCD and at the endpoint level with Impact 2002+. The ILCD category "water resource depletion" has been replaced with "water deprivation" as introduced in section 6.2.5.1. Foreground CF_{WD} are derived from Seine sub-river basins at monthly scale, whereas background CF_{WD} are derived from CTA of Hoekstra et al. (2012) which are implemented in ecoinvent 3 at the country scale. All other emissions to air/soil, energy, chemicals and infrastructures are characterized with these two LCIA methods. This is accomplished using Simapro software (Pré Consultants, 2013) and results in impact matrices **i** for each component of the system.

6.2.6. Example of the construction of a scenario using the model

As described in Chapter 5, the volumetric water flow distribution vector **v**, quality distribution vector **q**, and specific impacts matrix **i** are documented for each component of the system and characterize the graphical objects (i.e., instances) of the Simulink library. The baseline scenario is implemented in the Simulink interface by selecting and connecting the graphical objects corresponding to the components of the UWS (Figure 6-4). Extrinsic parameters (i.e., water demand, number of water users, number and ratio of inputs and outputs for each components, connection to water resources) are then defined as shown in Figure 6-4. Each other scenarios are then derived from this representation, with the changes of extrinsic parameters, as summarized in Table 6-4.



CR=Choisy-le-Roi, NM=Neuilly-sur-Marne, MO=Mery-sur-Oise, A=Arvigny, NS=Neuilly-sur-Seine, U_dom=domestic users, U_ind=industrial users, U_nm=non-market users, U_m=market users, U_oth=others users, SAv= Seine Aval, SAm= Seine Amont, SG=Seine Grésillons, MA=Marne Aval.

Figure 6-4. Graphical representation of the baseline scenario with all components, all technosphere flows (black arrows) and major withdrawals (blue arrows) and releases (green arrows).

Table 6-4. List of extrinsic parameters for the construction of each scenario

			B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
Water Users	Domestic users	Number of users (capita)	4362705	4506674	4506674	4506674	4785887	4785887	5296324	4785887	4785887	4785887	4785887	4785887
		Water demand (m3/year/user)	39.2	37.2	36.9	36.2	32.4	39.2	39.2	32.4	32.4	32.4	32.4	32.4
	Non market services users	Number of users (jobs)	413251	426888	426888	426888	453336	453336	501687	453336	453336	453336	453336	453336
		Water demand (m3/year/user)	70.2	66.6	66.1	64.8	58.0	70.2	70.2	58.0	58.0	58.0	58.0	58.0
	Market services users	Number of users (jobs)	1051485	1086184	1086184	1086184	1153479	1153479	1276503	1153479	1153479	1153479	1153479	1153479
		Water demand (m3/year/user)	23.6	22.4	22.2	21.8	19.5	23.6	23.6	19.5	19.5	19.5	19.5	19.5
	Industrial users	Number of users (jobs)	153208	158264	158264	158264	168069	168069	185995	168069	168069	168069	168069	168069
		Water demand (m3/year/user)	43.2	41.0	40.7	39.9	35.7	43.2	43.2	35.7	35.7	35.7	35.7	35.7
	Others users	Surface (ha)	76280	76280	76280	76280	76280	76280	76280	76280	76281	76282	76283	76284
		Water demand (m3/year/ha)	83.7	79.4	78.8	77.2	69.2	83.7	83.7	69.2	69.2	69.2	69.2	69.2
Water resources	Withdrawals from DWP (%)	Seine upstream River - CR (%)	41.2	41.2	41.2	35.5	41.2	41.2	41.2	41.2	0	27.2	42	42
		Seine down River (%)	0	0	0	0	0	0	0	0	42	0	0	0
		Marne River - NM (%)	34.7	34.7	34.7	28.2	34.7	34.7	34.7	34.7	34.7	34.7	36	36
		Oise River - MO (%)	20.5	20.5	20.5	35.6	20.5	20.5	20.5	20.5	20.5	20.5	22	22
		Champigny aquifer (%)	2.9	2.9	2.9	0	2.9	2.9	2.9	2.9	2.9	2.9	0	0
		Albien aquifer (%)	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0	0.7	0	0
		Sources of the Loing River (%)	0	0	0	0	0	0	0	0	0	14	0	0
	Release from WWT (%)	Seine downstream - SAv (%)	64.9	64.9	64.9	64.9	64.9	64.9	64.9	64.9	64.9	64.9	64.9	64.9
		Seine upstream - SAm (%)	26.0	26.0	26.0	26.0	26.0	26.0	26.0	26.0	26.0	26.0	26.0	26.0
		Seine downstream - SG (%)	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85	1.85
		Marne - M (%)	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.26
Water technologies	DWP	Conventional (%)	82.7	82.7	82.7	78.6	82.7	82.7	82.7	82.7	41.5	68.7	0.0	100.0
		Membrane (%)	13.7	13.7	13.7	17.8	13.7	13.7	13.7	13.7	55.7	13.7	100.0	0.0
		Simple treatment (%)	3.6	3.6	3.6	3.6	3.6	3.6	3.6	3.6	2.9	17.6	0.0	0.0
	DWD	Network yield (%)	89	85	90	95	90	90	90	90	90	90	90	95
	WWT	Conventional (%)	100	100	100	100	100	100	100	100	100	100	100	100

6.3. Results and discussion

Results for the baseline scenario are detailed hereafter for water flows, total environmental impacts and impact/service ratios. The ratios are computed by dividing the impacts with the indicators associated with functions of the UWS. For simplicity, only total environmental impacts are shown for forecasting scenario and are compared to the baseline scenario. A sensitivity analysis on the selection of impact/service ratio (based on user or m^3) for the comparison between two scenarios is then discussed.

6.3.1. Baseline scenario

6.3.1.1. Water flows

Figure 6-5 shows the different water flows in the UWS that were computed by the model to check for the water balance of the system. This representation is performed using a Sankey diagram, which is specific type of flow diagram in which the width of the arrows is proportional to the flow quantity. This diagram provides a representation that is easily communicable to stakeholders by mapping the different water flows within the urban water cycle.

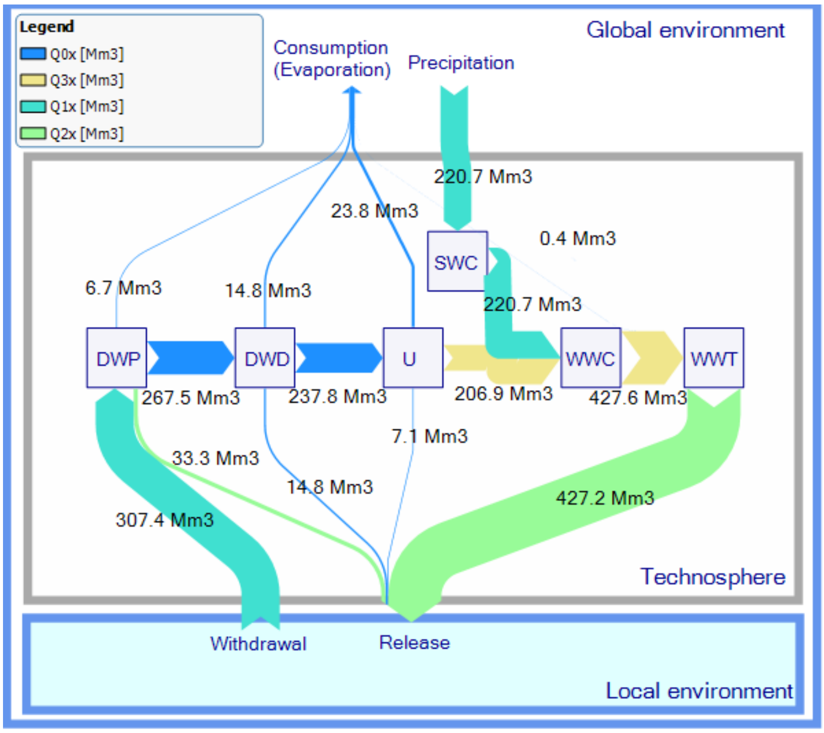
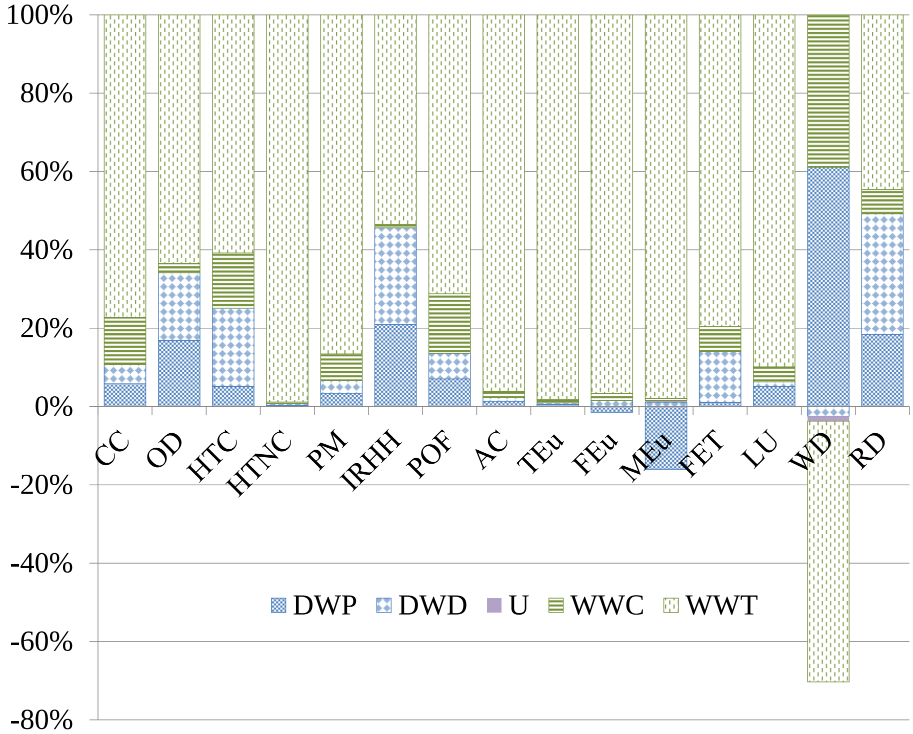


Figure 6-5. Simplified Sankey diagram of water flows within the urban water system of the baseline scenario.

6.3.1.2. Environmental impacts

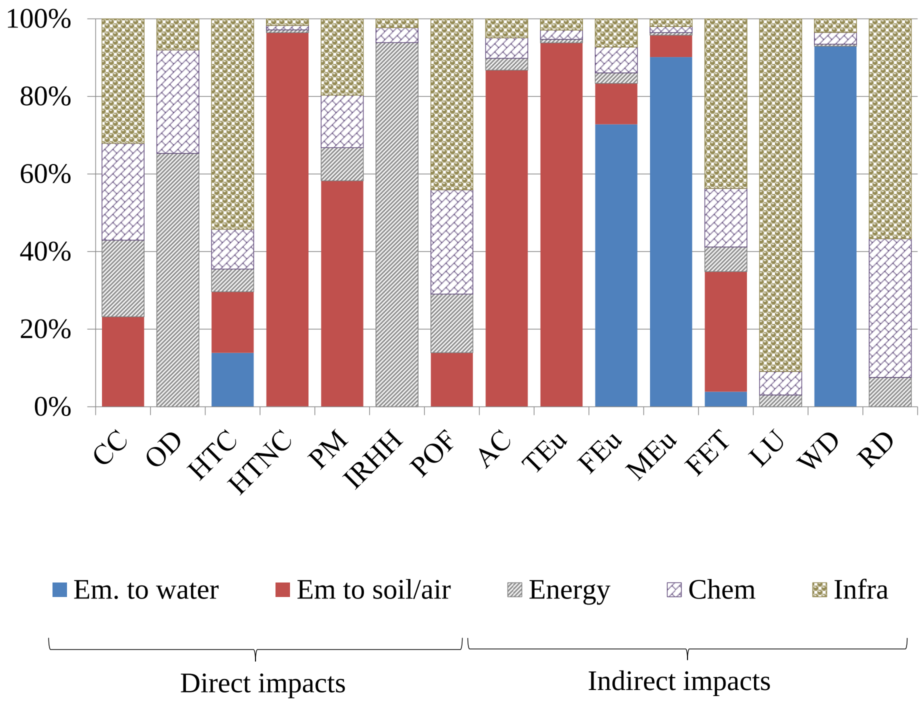
Assessing the impacts of the baseline scenario determines the contribution of each component of the system and the shares between the direct (foreground activities) and indirect impacts (background activities). This assessment provides stakeholders with results for identifying the environmental hotspots of the UWS. The order of magnitude of the baseline scenario contributions is consistent with the previous result found in Chapter 5 that was based on a theoretical model using only ecoinvent. Figure 6-6 shows that the majority of impact categories, particularly those related to water pollution, are dominated by WWT. This result differs from the results of Chapter 5, in which WWT technologies contributed far less. This difference is because WWT plants of this case study use more electricity and chemicals than the ecoinvent ones. WWT plants in this case study include the advanced processes treatment of phosphorus and nitrification/denitrification, which increase the use of energy and chemicals. Additionally, a large amount of stormwater is treated within the case study plants, whereas stormwater treatment was not considered in Chapter 5. Another main difference with the ecoinvent model is the high contribution of WWC (which includes stormwater collection) to water deprivation. This difference is because the collected stormwater (precipitation) is considered as a withdrawal, and we assume that stormwater would runoff to river water in the absence of a collection system.

The contribution of direct and indirect impacts (Figure 6-7) shows results similar to the ecoinvent model, suggesting that this result is generalizable to all UWS.



CC = climate change, OD = ozone depletion, HTC = human toxicity cancer effects, HTNC = human toxicity non cancer effects, PM = particulate matter, IR = ionizing radiation, POF = photochemical ozone formation, AC = acidification, TEu = terrestrial eutrophication, MEu = marine eutrophication, FET = freshwater ecotoxicity, LU = land use, WD = water deprivation, RD = mineral and fossil resource depletion. Chem. = Chemicals and others, Infra. = Infrastructures

Figure 6-6. Relative contributions of UWS components in the baseline scenario. LCIA method: ILCD.



CC = climate change, OD = ozone depletion, HTC = human toxicity cancer effects, HTNC = human toxicity non cancer effects, PM = particulate matter, IR = ionizing radiation, POF = photochemical ozone formation, AC = acidification, TEu = terrestrial eutrophication, MEu = marine eutrophication, FET = freshwater ecotoxicity, LU = land use, WD = water deprivation, RD = mineral and fossil resource depletion. Chem. = Chemicals and others, Infra. = Infrastructures

Figure 6-7. Relative contributions of direct/indirect impacts in the baseline scenario. LCIA method: ILCD.

6.3.1.3. Provided services and impact/service ratios

In addition to the total impact of the UWS, the model details the functions provided by the system, i.e., the amount of each type of user supplied with water. Based on this information, the model computes impact/service ratios that are useful for comparisons with other scenarios or other systems. With the example of the climate change impact category, impact/service ratios found for the different users are the following: 72.55 kg CO_2 eq/year/domestic user, 129.92 kg CO_2 eq/year/non market service job, 43.68 kg CO_2 eq/market service job, 79.95 kg CO_2 eq/year/industry job, 154.9 kg CO_2 eq/year/ha (other uses) and 100.89 kg CO_2/year/equivalent inhabitant. In this study, we considered equivalent water users in terms of input and output water quality. The share of impact relative to each user is only dependent on the water demand and the amount of water users. Therefore, 72% of the impacts result from domestic users, 3% result from industries, 12% result from non-market services, 10% result from market services and 3% result from others users. The low share of impacts resulting from industries is mainly because the low industrial activity in the case study (8.8% of all jobs are in industry in 2012). However, not accounting for the different levels of raw wastewater quality generated by the various users tends to underestimate industries' responsibility in wastewater treatment (the presence of heavy metals and emerging pollutants). A methodological challenge remains to allocate the impacts of a wastewater treatment plant according to the different types of users.

Regarding climate change, the impact/service ratio related to one cubic meter at the user's place results in a value of 1.85 kg CO_2 eq/m^3. This result agrees with values found in the literature and summarized in Loubet et al. (2014), which range from 0.51 to 1.57 kg CO_2 eq./m^3 at the user's place.

6.3.2. Forecasting scenarios

The results of the forecasting scenarios compared to the baseline total impacts are shown in Table 6-5 and discussed hereafter. To simplify the results, only midpoint water deprivation impacts and endpoint damages from Impact 2002+ are considered here.

Table 6-5. Relative evolutions of Impact 2002+ damages and water deprivation impacts for forecasting scenarios compared to baseline scenario.

		Reference flow: Entire urban water system/year											
		Baseline	Short term scenarios horizon 2022			Long term scenarios horizon 2050							
			Operators' scale changes			Users' changes			Resources' changes			Technologies' changes	
		B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
Water deprivation	m3 water eq	100%	4%	-4%	-19%	3%	19%	29%	-8%	-61%	14%	-2%	-7%
Global warming	kg CO2 eq	100%	0%	-1%	-2%	-3%	3%	7%	-3%	1%	-4%	2%	-5%
Human health	DALY	100%	-1%	-1%	-2%	-4%	4%	9%	-4%	-3%	-4%	-3%	-4%
Ecosystem quality	PDF*m2*yr	100%	0%	-2%	-3%	-6%	6%	12%	-6%	-2%	-8%	-4%	-6%
Resources	MJ primary	100%	0%	-3%	-5%	-5%	5%	11%	-5%	8%	-6%	9%	-9%
		Reference flow: DWP and DWD technologies/year											
		B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
Water deprivation	m3 water eq	100%	1%	-4%	-12%	1%	23%	35%	-4%	-31%	7%	-1%	-4%
Global warming	kg CO2 eq	100%	2%	-6%	-10%	-7%	7%	16%	-7%	29%	-12%	33%	-20%
Human health	DALY	100%	2%	-5%	-6%	-6%	6%	14%	-6%	27%	-15%	33%	-15%
Ecosystem quality	PDF*m2*yr	100%	2%	-4%	-8%	-10%	9%	21%	-10%	3%	-19%	-2%	-12%
Resources	MJ primary	100%	2%	-8%	-14%	-9%	9%	20%	-9%	35%	-13%	39%	-23%

6.3.2.1. Short term forecasting scenarios

The short term policies assessed in scenarios S1, S2 and S3 show small variations of damages at the UWS scale, i.e., from 0 to -5% changes. This small variation is because the studied scenario changes only concern DWP & DWD which have a small contribution in the overall system. The fact that damages decrease in all scenarios results from the decrease in the overall water demand. Concerning water deprivation impacts, the policy can have an important effect over short terms by managing the water resource choice; for example, the water deprivation decreases by 19% for scenario S3, which is designed towards high impact/service.

6.3.2.2. Scenarios with changes in water users

Scenarios L1 to L4 aim to study the variability of impacts resulting from contrasted trends in water demand projection of the urban area for 2050 (-10 to +21%). Scores of the "ecosystem quality" range from -6% to +12% in comparison with the baseline. Water deprivation potentially increases from 3% to 29%. This shows the high variability of the system impacts because of different projections of urban development and user's behavior. Knowledge about urban development and water demand is consequently a key point when assessing forecasting scenario because the system is driven by these parameters. These results also demonstrate the capacity of the model to implement easily different water demand scenarios. However, a main limitation of these scenarios is the fact that the wastewater load and WWT plant operation are considered identical to the baseline scenario. This is a strong assumption because a reduced water demand per capita coupled with the identical pollutant load released would increase the concentration of pollutants in the wastewater, thus modifying the residence time in WWC and WWT and the energy and chemical consumption. Nevertheless, determining wastewater pollutant concentrations for future scenarios is complex. Behavior changes of the users might affect the pollutant load released into wastewater because of the decreased use of chemicals and the increasing efficiency of water appliances (Friedler, 2004). Additionally, no LCA model can yet predict impacts of WWT technologies effect on the quality of wastewater.

6.3.2.3. Scenarios with changes in water resources

Scenario L4 studies the water deprivation impact that potentially occurs in 2050 by considering the current state of water scarcity (instead of the water scarcity state that would occur because of climate change, as studied in scenario L1). This scenario results in a -13% decrease of water deprivation impact compared to scenario L1, suggesting a non-negligible effect of climate change on this impact.

Scenarios L5 and L6 study the water transfer from downstream or upstream locations to water withdrawal points and result in contrasted figures in comparison with the baseline. Scenario L5 (42% water transferred from downstream) slightly increases the damages of global warming and on resources because of the water transfer infrastructure and the use of a membrane technology to produce water from lower quality downstream water. Scenario L6 decreases all damage categories with regard to scenario L1 because drinking water from upstream sources is produced with a simple treatment process. Water deprivation impacts are greatly modified, with a reverse trend compared to other damage categories. Scenario L5 decreases the impact (-60%) by withdrawing a large share of water (42%) from a downstream location of the Seine river, whereas scenario L6 increases the water deprivation impact (+14%) by withdrawing a certain amount of water (15%) from upstream sources, thus having a larger downstream impact.

Figure 6-8 shows the monthly evolution of water deprivation impacts for different scenarios. More than 60% of impacts occur during the summer months (July, August, September) for all scenarios. This emphasizes the relevance of a monthly scale model for decision making on water resource choices.

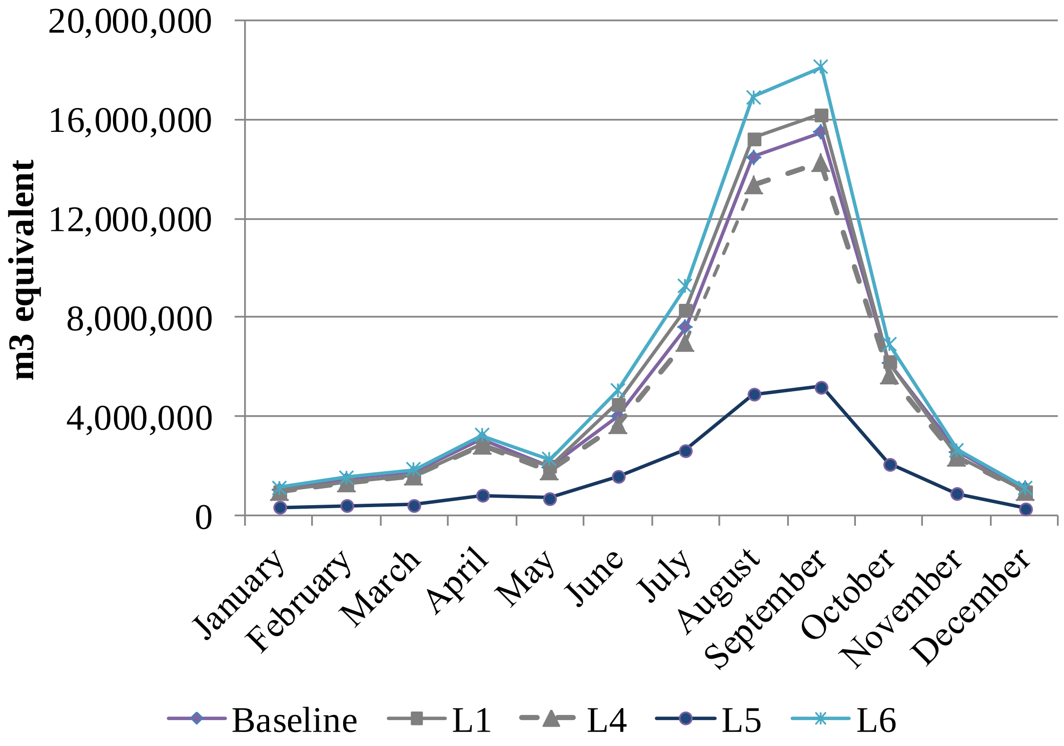


Figure 6-8. Monthly evolution of water deprivation impacts for several scenarios

6.3.2.4. Scenarios with change in water technologies

Changes in DWP technologies do not produce significant impact changes at the UWS scale. Differences are only noted in the DWP and DWD parts of the system. Scenario L7, which studies the implementation of membrane DWP technologies, increases the damages for the "resource" category by +39%, whereas scenario L8, which studies eco-efficient DWP technologies, decreases the damages by -23% ("resource" category). Although membrane processes are substantial consumers of energy and chemicals, they also produce high quality drinking water that could provide more services to users. In particular, these processes decrease the water hardness, thus saving energy and lifespan expectations of water appliances (water heaters, washing machines, etc.) (Godskesen et al., 2012). This case study does not include these appliances because of the lack of data, but the developed framework enables such an inclusion. Research would also be needed on the LCI linked to energy use of water appliances depending on the quality of the drinking water.

6.3.3. Sensitivity analysis on impact/service ratio choices

Impact-services ratio (i.e., inverse of eco-efficiency ratios) are indicators that are useful for communicating with stakeholders. Since the model compute several impact/service ratios that can be interpreted differently by stakeholders, it is important to discuss the relevancy of each ratio, as it is done hereafter.

Figure 6-9 shows the evolution of damages between the baseline scenario and forecasting scenario L1 that models the UWS in 2050 with the expected changes in population (+ 9,3%), water demand (-21%) and water scarcity. Depending on the impact/service ratio, the results radically change. With the impact/service ratio related to the entire urban water system, the damages are expected to decrease with increasing population. This decrease is because the expected reduction in water demand will decrease the overall water demand of the urban area, and therefore the impacts: 215.1 Mm^3 in 2050 versus 237.8 Mm^3 in 2012 (scenario L1). For this scenario, however, the water deprivation is expected to swell because of a higher water scarcity. The impact/service ratio related to users (equivalent inhabitants) decrease (up to 14%) because of additional population and less overall damages in scenario L1 compared to B. Nevertheless, impact/service ratios related to one cubic meter increase because the total water demand is lower in scenario L1 but the infrastructure is considered to be unchanged. Consequently, infrastructure damages related to one cubic meter are more important in scenario L1 than in B. These different results provide useful information for the stakeholder depending on the question. The impact/service ratio related to the entire UWS aims to assess the overall impacts of a territorial system. This assessment is relevant for local authorities that aim to reduce the non-marginal overall impacts on a territory. The impact/service ratio related to users is useful for comparing different types of users' impacts and for analyzing the contribution to an UWS within the total impacts generated by one user in a city (policy-making at a larger scale). The impact/service ratio related to one cubic meter at the user's place is useful for comparing technological efficiencies of different UWS. However, this metric is not relevant in comparing integrated forecasting scenarios of an UWS because it does not account for user's behavior.

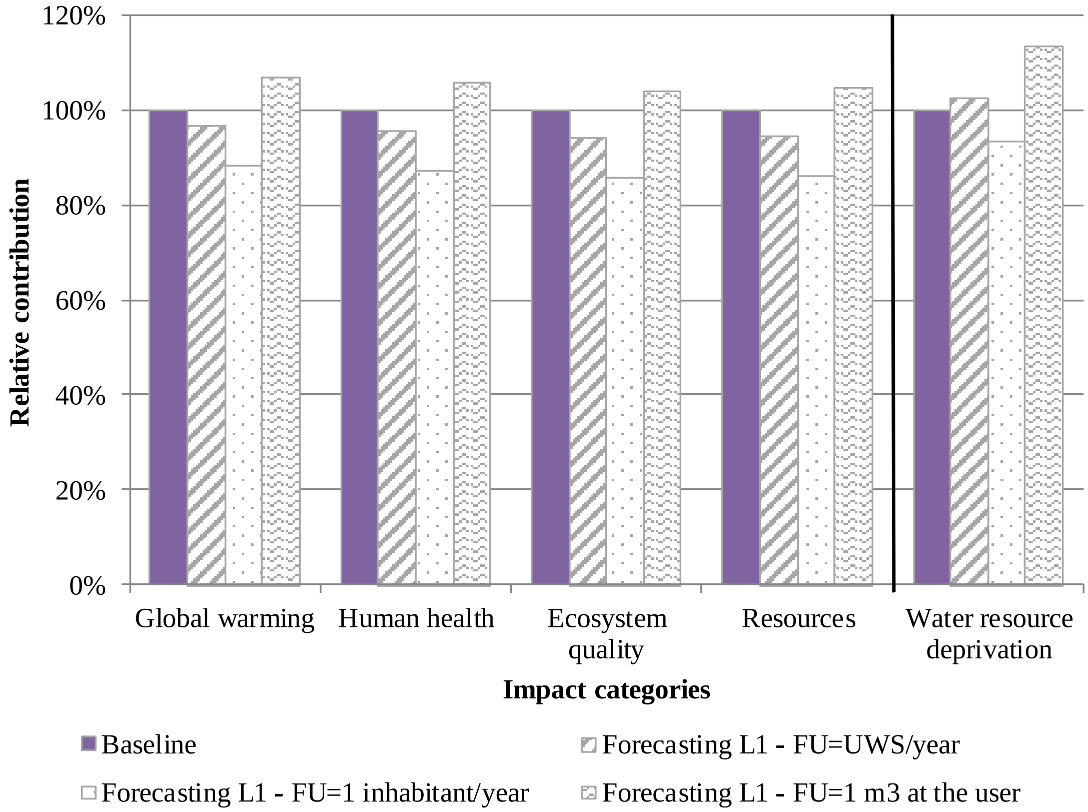


Figure 6-9. Comparison of various impact/service ratios of forecasting scenario L1 to the baseline (set at 100%, whatever the unit). LCIA method: Impact 2002+ endpoint and water deprivation midpoint.

6.3.4. Opportunities and limits

Other scenarios can be easily investigated because of the modularity of the modeling tool and would provide opportunities to address emerging concerns in urban water management. For example, policy-responsive scenarios regarding water technologies could compare different systems of water softening (central softening, such as membrane processes, or household systems), separation of wastewater streams (into blackwater and greywater), wastewater reuse (Meneses et al., 2010) or rainwater harvesting (Angrill et al., 2011). Future-trend scenarios related to technologies were not regarded in this case study but could include a prospective electricity mix.

The fine temporal and geographical scales associated with water deprivation impacts are useful for decision making regarding water resource choices. In this case study, whereas decision makers of DWP have low flexibility for decreasing impacts linked to technologies, they do have a high contribution on water deprivation impacts by selecting withdrawal locations.

These scales could also be important when dealing with water quality associated impacts. The future implementation of refined LCIA methods (e.g., marine and freshwater eutrophication) with a spatial differentiation of the fate and the effect will improve the site-dependent property of the model, and therefore, interest in this method by the stakeholders.

In addition to the generic limits of the model noted in part 1, more specific limits emerge from this case study. The predictions of future wastewater quality and the resulting emissions with an equilibrated mass balance are a real challenge for a LCA. Additionally, uncertainties related to WWT inventory data are high, both in LCI phases - concerning emissions to air and soil at the plant and at the sludge end-of-life and in the LCIA phase concerning in particular the assessment of metals (Rosenbaum et al., 2008).

The integration of stormwater collection within the system also raises concerns and methodological challenges. The assumption under which all stormwater collected is considered as withdrawal is questionable. In the absence of any collection system, not all water would runoff to the river because a portion can evapotranspirate or infiltrate into the soil. This portion of the runoff is dependent on the land use. For urban cover, most of the water would runoff. However, if we consider a natural land reference such as forest, then a much larger portion would evapotranspirate and infiltrate. This question of land use is still a debate in LCA (Núñez et al., 2013b). In addition, a further step would be to consider all stormwater collection and run-off within the area and not only the water collected in combined sewers.

Another challenge is the decision-making process with several stakeholders. As shown in Table 6-2, many water service institutions manage the UWS of the greater metropolitan Paris area. These case study scenarios have been built with one specific stakeholder managing drinking water in a suburban area of the system being related to the environmental performance of the entire UWS. The evaluation of the entire greater Paris area UWS would be an interesting further step because this is a coherent territorial unit. The cooperation of the different water services is also a notable issue in the context of the “Grand Paris” development (Desjardins, 2010).

6.4. Conclusions and outlook

The model for assessing the environmental impacts of UWS developed in Chapter 5 is implemented on a real case study in the suburban areas of Paris with various associated scenarios. This application demonstrates the applicability of the model to assess the environmental impacts of an UWS and the capacity to address several stakeholder’s potential questions related to urban water management. This approach therefore provides useful quantitative information for decision making processes related to policy-response scenarios on water resources and technological choices and related to future-trend scenarios projecting urban development and behavior. The use of this modeling tool and its modular approach greatly facilitates the generation and the evaluation of scenarios.

The results have shown that UWS impacts predominantly result from WWT. The study of the forecasting scenarios results in many findings. A high variability of impacts in forecasting scenarios is noted because of different trends in water demand. Because of their low contribution in the system, changes in DWP and DWD do not produce important modifications of technology-based impacts. However, DWP and DWD have a great effect on water deprivation impacts depending on the choice of water withdrawal location. Further scenarios could be investigated such as the implementation of emerging technologies in DWP and WWT plants.

The different UWS components developed in this case study could be used for the environmental assessment of water management scenarios elsewhere in the world. It would require updated data in the library and the calculations of downstream cascade effect (CF_{WD}) for all sub-river basins because global coverage is not yet available.

Chapter 7. Discussion and conclusion

« Elle a coulé, notre rivière

Depuis ce jour d'antan.

Elle a coulé, notre rivière

Depuis mille et mille ans. »

Hugues Aufray – Notre rivière

Content of Chapter 7

This chapter is split in two complementary sections related to the two main outcomes of the thesis: first methods for improving LCIA of water use (Chapter 3 and Chapter 4), and second, WaLA, a versatile model for LCA of UWS (Chapter 5 and Chapter 6).

7.1. The need to better assess impacts associated to water use

In this section, opportunities and limits of the method for assessing water deprivation at the sub-river basin scale and the perspectives, such as its combination within a mechanistic model for assessing water fate, are presented. Finally, water quality indicators are discussed and the limitation of their integration in water footprint is debated.

7.1.1. Towards appropriate scales for LCA practitioners

The methodology for assessing water deprivation at the sub-river basin scale has proven its worth for LCA of UWS. The evaluation of forecasting scenarios in Chapter 6 has shown that such a method is useful to compare scenarios having chosen different water resources. Even if this methodology was applied to LCA of UWS, its application is wider and could benefit to other LCA application such as agriculture or industries, which are also big water users.

The proposed scale, i.e., the sub-river basin is a step towards site-dependent impact assessment. It would greatly benefit to the appropriation of the methodology by stakeholders who are asking for a consistent assessment of local impacts. In Chapter 6, characterization factors for water deprivation have been calculated at a finer scale than it was originally done in Chapter 3, in order to better address the stakeholder's questions about choices of water resources in the Seine river basin. It is a big output as it makes water-related LCIA methods applicable at any scales dependently to the goal and scope of the study. A further step in this direction would be to use local hydrologic model of a given river basin instead of global water models.

Up to now, in the literature, the scales for LCIA of water use was the river basin, and in the methodology we propose, the sub-river basin scale raises several challenges. LCA practitioners who assess global life cycle of goods or services usually do not know exactly the locations of water withdrawal or release and a sub-river basin scale is not relevant. Therefore, such a fine scale for LCIA methods can be currently used only for foreground processes with a good knowledge of the processes. Also computing the water use impact, at river-basin or finer scales, can only be done outside LCA software, as it is not integrated, yet. These challenges require future development for LCIA methods as well as databases and software developers: (i) as proposed by Mutel et al. (2012), minimization of global spatial autocorrelation should be applied to aggregate small spatial units and build typologies of (sub-)river basins; (ii) the issue of spatial differentiation should also be tackled by LCA software and database. Famous LCA software such as SimaPro (Pré Consultants, 2013), GaBi (PE International, 2011) or Umberto do not allow to select locations of elementary flows. OpenLCA (Ciroth, 2007), which is an open source LCA software, provides the possibility to analyze locations of impacts in a world map. This is a unique feature in LCA software, so far, and a great opportunity to assess water use impacts. LCA database, such as ecoinvent do not differentiate the locations of water elementary flows (withdrawal, release, consumption) at a scale finer than the country one.

7.1.2. Towards the use of consensual hydrological data and models for LCIA developers

It has been shown in Chapter 3 and Chapter 6 that several hydrological data and models can be used to compute CF_{WD}, resulting in different values. There is a high amount of database describing basin topography (e.g., in this thesis: “Hydro1K” and the updated version “HydroBASINS”) and water consumption, as well as of runoff models (e.g., in this thesis: averaged data from 1960-2000 from the “Composite Runoff fields model” and data for the year 2012 from the model “GLDAS”). Data are highly variable, because of the different models used and the time representativeness of the models. These differences emphasize the need for a consensual choice on reliable and recent database at a global scale, which is a question common to all methods related to water use in LCIA. It could be solved by comparing different hydrological models (e.g. runoff model) available in the literature, such as done by Boulay et al. (2014). Outside LCA, the World Resources Institute (WRI) compiled several database for computing the Aqueduct model, which assesses the water risk – ie water scarcity - at the global scale (Gassert et al., 2013). It includes several and recent data sources such as basin delineation, withdrawals, consumption, runoff, etc. that could be used for LCIA of water use.

In addition, the use of a common GIS web-based tool that could compute several LCIA methods according to different database would be an interesting approach for LCIA developers. Such web-based interfaces have already been developed in the hydrogeology or ecology communities, in order to supply users with a large amount of data. Representative examples are the global freshwater biodiversity atlas[1] or the data mapped by the Global Hydrology group of Utrecht University[2]. The latter allows us to calculate data within the web tool, as well as representing data according to specific time representativeness.

Therefore, such tools could be used to compute the downstream cascade effect for all sub-river basins in the world. The algorithms to compute the downstream cascade effect should be automatized and modeling options could be chosen within the proposed GIS web-based tool including (i) choice of database (runoff, water consumption, etc.), (ii) choice of sub-river basin delineation, (iii) choice and combination of weighting parameters for computing the downstream cascade effect (e.g., surface, water volume of rivers, population, etc.).

Site-specific indicators would require local models that would allow to take into account canals, reservoirs, inter- and intra- basins transfers (Rousset, 2004). These kind of human interventions on water resources are not well represented in global models, but are important to the hydrologic regimes in a river basin such as the Seine's one. Also, consideration of groundwater resources should be examined. These challenges can be tackled by the development of mechanistic approaches that is introduced in next section.

7.1.3. Current gap between midpoint indicators based on water stress and the endpoint indicators

UNEP-SETAC Life Cycle Initiative solicited the Water Use in LCA (WULCA) Working Group to undertake the task for a consensual set of methods for water use impact assessment. A focus is placed on three sets of indicators representing: impact pathways leading to damages on human health, impact pathways leading to damages on ecosystem, and a generic stress/scarcity indicator (Boulay et al., 2014), as shown in Figure 7-1. Water scarcity indicators raise concern since they are not in the cause-effect chain of water use impacts, from an LCA perspective. Indeed, water scarcity does not represent actual impacts or damages on ecosystem or human health, as pointed out in Chapter 3. It is rather a characteristic of a basin with regard to risk assessment as it informs us about "the extent to which demand for water compares to the replenishment of water in an area, such as a river basin" (ISO, 2013). However, there is a strong demand from industry to provide such an indicator. This advocates

[1] http://atlas.freshwaterbiodiversity.eu/index.php/maps

[2] http://www.globalhydrology.nl/maps

for the development of a mechanistic model that can be the base for assessing the effects (damages) at steady state of any marginal changes in the water balance.

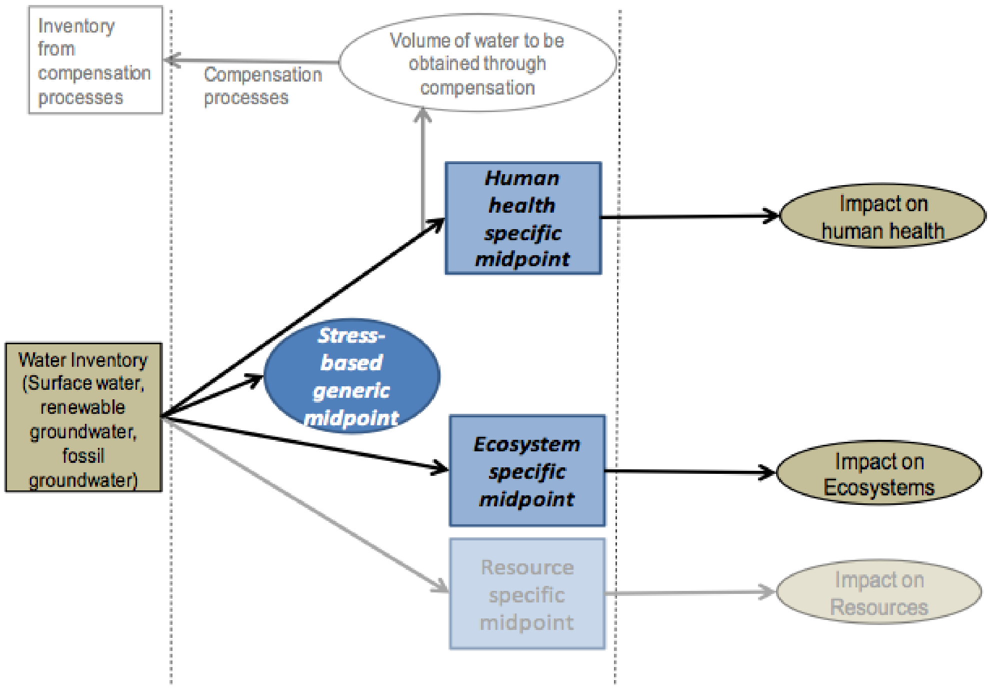


Figure 7-1: illustration of the gap between current mid-point indicators based on stress and damage assessment based on volume deprivation effects (source Boulay, WULCA)

7.1.4. Towards mechanistic approaches in LCIA: combining downstream cascade effect with a consistent water fate model

For damage to human health and to ecosystems, a mechanistic approach should be followed. One proposal is to represent the damage by the product of a fate factor and of an effect factor (and possibly an exposure factor for human health) as it is typically done in other LCIA impact categories such as toxicity or eutrophication. Savard (2013) has reviewed most of pathways of water-use related impacts and discussed in an extended manner on the fate and effect factors found in the literature. Fate factor for water can be defined as the modification of environmental water flows because of a human intervention. Effect factor is the consequence of the modified environmental water flow on ecosystem or human health damages (the same occurs for other water natural resources, such as lakes and groundwater).

$$\mathrm{CF} = \mathrm{FF} \cdot \mathrm{EF} = \frac{\mathrm{dQ}}{\mathrm{dW}} \cdot \frac{\mathrm{dS}}{\mathrm{dQ}} \tag{27}$$

Where dW is the marginal change of the human intervention (e.g., water withdrawal within a river basin), dQ is the marginal modification of the environmental flow (e.g., flow rate of a river), dS is the marginal damage (e.g., damage to species richness).

Effect factors have already been defined for several impact pathways: damages to fish due to decrease flow in river (Hanafiah et al., 2011; Tendall et al., 2014), damages to ecosystems due to decreased wetland volume (Verones et al., 2013), damages to plants due to decrease level of groundwater (van Zelm et al., 2011). However, fate has been mostly disregarded or considered as equal to one, meaning that an intervention has a direct effect on one flow. Also, the difference between inventory and fate of water is still unclear. For instance, Berger et al. (2014) propose to take into account atmospheric evaporation recycling at the inventory phase to compute net consumption, whereas it could be considered as fate. There is a need to bridge the gap between inventory and impact assessment, as it is discussed for example in the pesticides and LCA field (van Zelm et al., 2013).

The development of a multimedia fate model would allow to account for all modifications of environmental water flows because of a human intervention. Figure 7-2 is a representation of the water cycle at the scale of a river basin, i.e., the exchanges of water between environmental compartments, as adapted from the representation of Usetox multimedia model (Rosenbaum et al., 2008, 2007). Human intervention modify the different water flow exchanged by the compartments. For example, water which is withdrawn from a river to be used on an agricultural soil will cause several modifications on: the river flow, the groundwater table, the soil moisture, the evapotranspiration and recirculation of water within the atmosphere, etc. This is shown in Figure 7-3 where the inventory, i.e., the water withdrawal from groundwater, and the fate, i.e., the modifications on environmental flows are represented. Each arrow representing a water flow exchange would be defined by a hydrological water model.

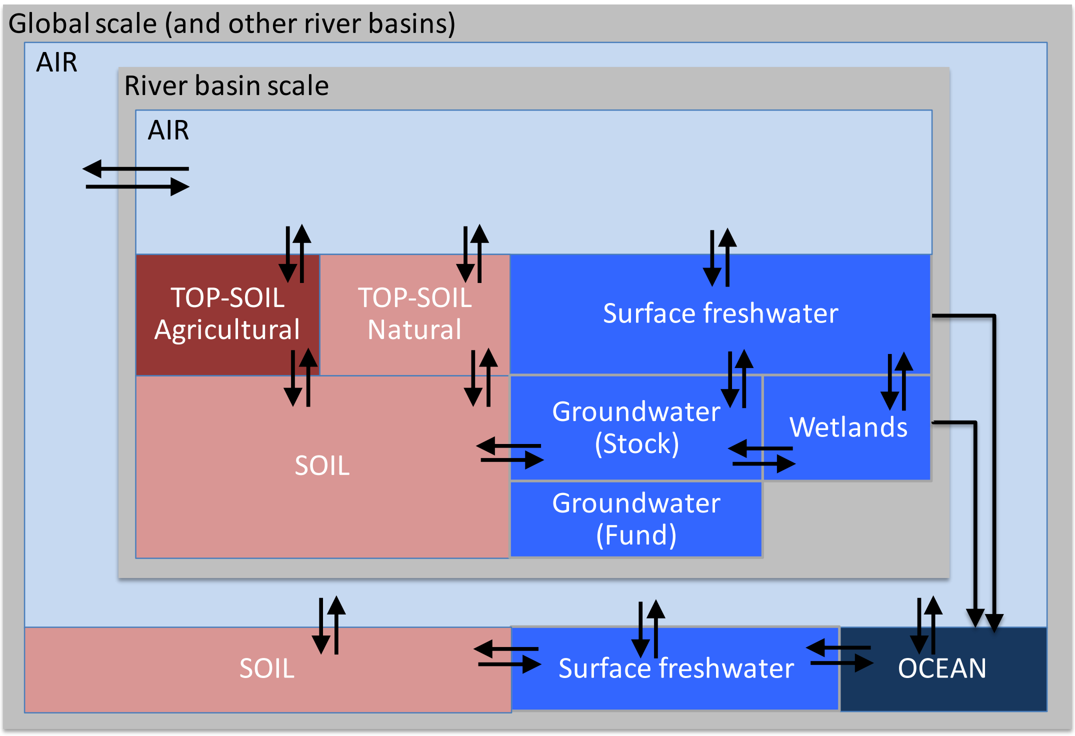


Figure 7-2. Description of the water cycle within a multimedia scheme. Adapted from Usetox multimedia fate model (Rosenbaum et al., 2008).

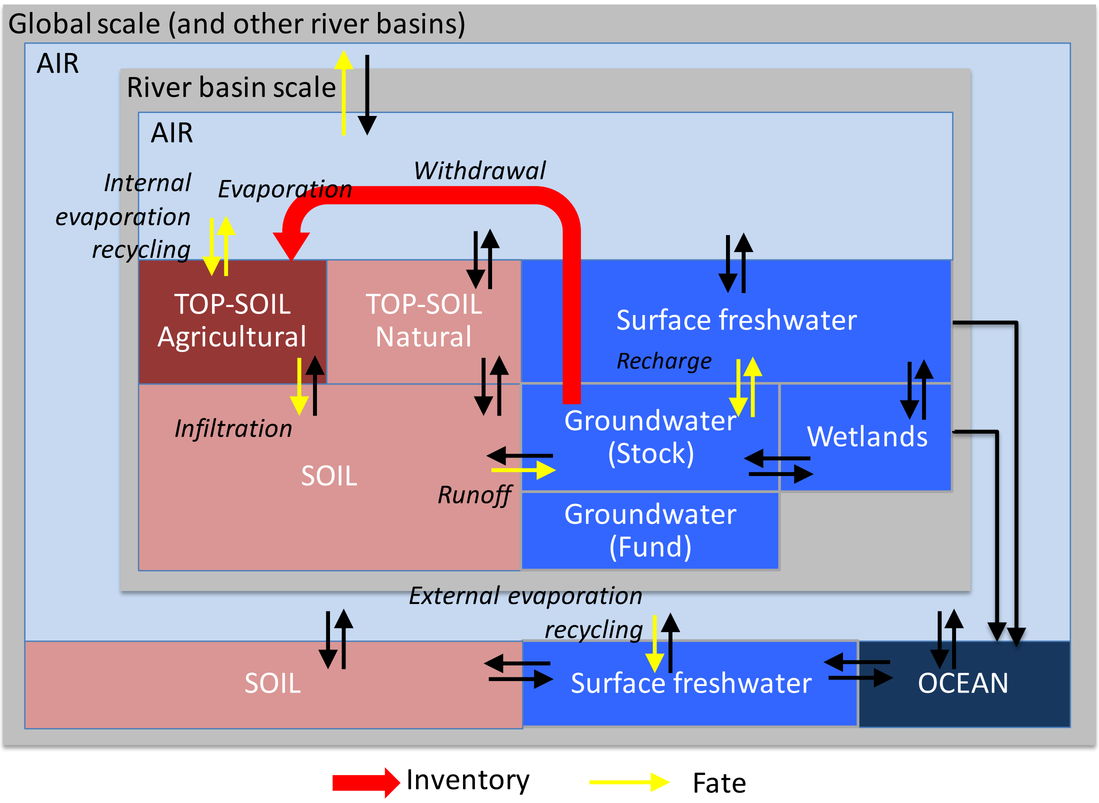


Figure 7-3. Proposed framework of the fate of water flows within a multimedia scheme: modification of environmental water flows (yellow arrows) caused by human interventions (red arrows). Name of water exchange processes are in italic. (source: Roux, P., Nunez, M. Loubet, P., for WULCA group in 2014)

In this context, the "downstream cascade effect" developed in this thesis could be considered within the fate to account for modification of environmental water flows at the sub-river basin scale. This effect can be represented with two nested scales: river basin and sub-river basins. There are n river basins that can exchanges water with ocean and with other river basins through atmospheric recycling of water. Within a river basin termed i, there are m sub-river basins exchanging water from upstream to downstream (downstream cascade effect) and ultimately ocean through surface freshwater flow (i.e., rivers). This is shown in Figure 7-4.

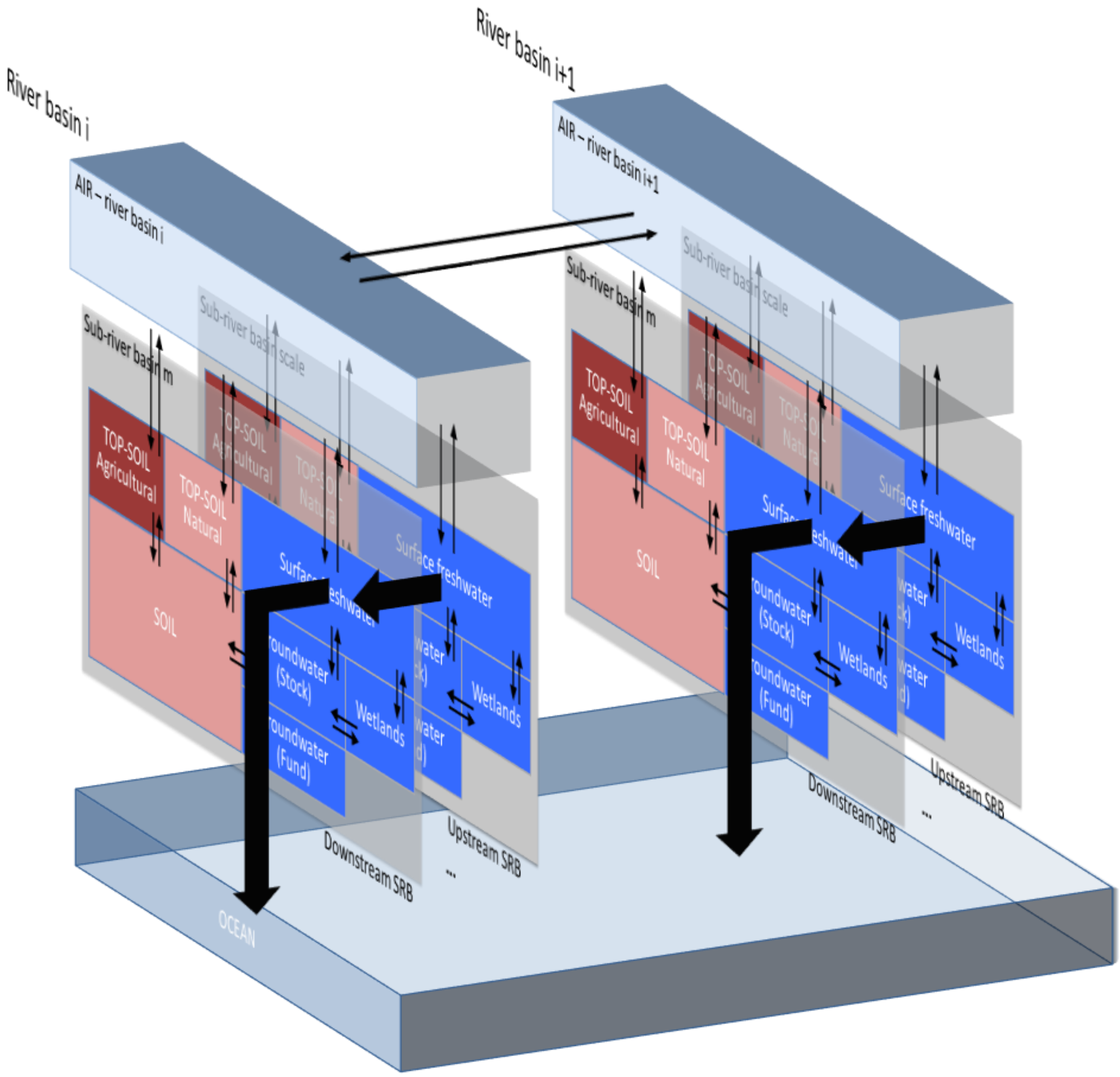


Figure 7-4. Representation of water cycle at the sub-river basin scale. Thick black arrows represent downstream cascade effect

More research is required for presenting the general framework of fate model for water and to integrate hydrological model of water exchanges between compartments. This task is currently undertaken within ELSA-PACT industrial chair and the WULCA group.

7.1.5. Current limits of water footprint related to water quality assessment

Impacts associated to water quality (pollutants) or quantity (water resource consumption) can be presented in a simplified way, in the form of the water footprint profile (ISO, 2013) or a water footprint single score e.g., the Water Impact Index (WIIX) (Bayart et al., 2014), the index of Ridoutt and Pfister (2012) or the one from the Water Footprint Network (Hoekstra et al., 2011). Impacts related to water use have been fully discussed in the previous section. Impacts related to water quality, described in Chapter 4, should be more thoroughly discussed. Indeed, water quality related impacts that should be taken into account in a water footprint profile or single score have not been clearly defined so far. The ISO standard specifies that all life cycle emissions (to air, soil and water) with impact on water quality should be considered (ISO, 2013). Alternatively, the Ridoutt and Pfister (2012) method only takes into account emissions to water. Also, WIIX only quantifies emissions to water and only consider impacts occurring in the water media.

Therefore, qualitative water footprint profiles or single scores differ according to LCI and LCIA choices. In the LCI phase of a water footprint, either only emissions to the water compartment or all emissions to soil, air and water are included. As for the LCIA phase of a water footprint, either only impacts occurring within the water media or all impacts affected by emissions to water (that do not necessarily concern aquatic environment, but terrestrial or human for example) are taken into account. It leads to four different options for considering water quality:

- (1) Direct emissions to water leading to impact categories related or not with water
- (2) Direct emissions to water leading to impact categories related to water
- (3) Direct emissions to air, soil, water, leading to impact categories related to water
- (4) Direct emissions to air, soil, water, leading to impact categories related or not to water

The four different options are illustrated in Figure 7-5. This shows that this issue has not clearly been addressed so far and should be solved for developing standardized method of water footprint profile or single score.

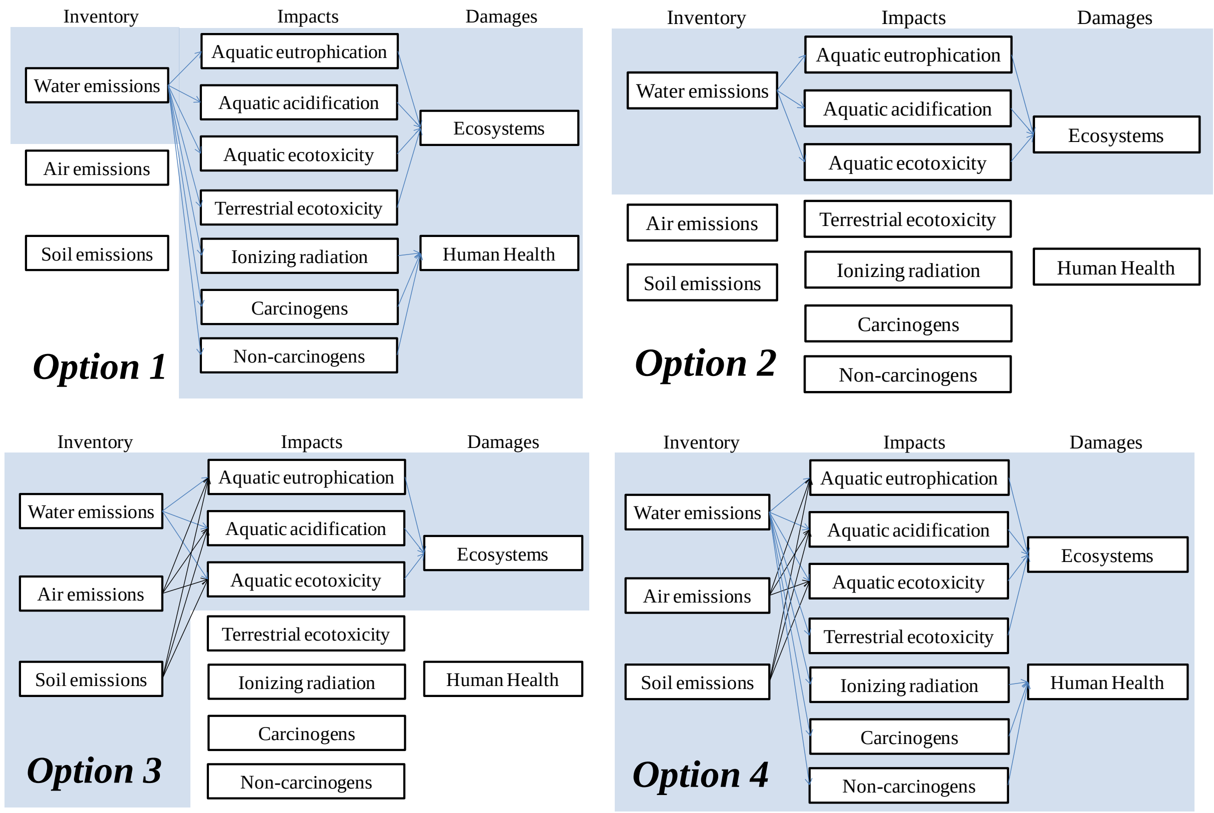


Figure 7-5. Different options for taking into account water quality within a water footprint profile or single score

In conclusion:

- An exhaustive water footprint profile (eg option 4) including the water quality dimension would require numerous inventory data comparable to those needed for a full LCA.
- The fact that four options - presented above - are open in the computation of water footprint index can be confusing for practitioners, and makes result comparison complex or even impossible.

Although footprint approaches appear simpler for communication purposes, it is preferable to opt for full LCA since the efforts to gather data and to assess impacts is similar in both approaches, and LCA enables to avoid pollution shifting between impact categories.

7.2. Perspectives for the WaLA model

7.2.1. Opportunities and limits

The WaLA model opportunities and limits are identified and discussed in chapters 5 and 6. In summary, the main novelties of the model are its modularity that enables the easy construction and assessment of scenarios, and the refined assessment of water related impacts, taking into account fine temporal (monthly) and spatial (sub-basin) scales. Also, the modeling formalism based on the definition of a generic component (for both water users and technologies), represented by an object, allows the appropriation of the model by companies or academics to develop their own tool (with their own data), thanks to the object-oriented programing (OOP). The provision of a large amount of data defining the components of UWS (i.e., volume and quality distribution, impacts of associated activities) could also be used by others practitioners.

Limits are still numerous and require further developments concerning (i) modeling choices and (ii) data collection to conduct case studies:

(i) The management of pollutant mass balance in link with water quality changes at the component scale could be included, as component functions. This requires models able to carry out mass balances for all pollutants (and not only for carbon and nitrogen). It seems possible for DWP (Mery et al., 2013) but requires further research for WWT (including sludge fate). Also, uncertainty management is not implemented in the model but should be dealt with since it is an essential feature for improving the reliability of LCA in the context of decision making. Moreover, the data management within model programmed in Matlab/Simulink interface has not been optimized and should be improved in order to allow update of LCI and LCIA methods.

(ii) The library already includes several components to represent water technologies and water users but still need to be completed: for example, with emerging technologies or water appliances at the user’s place (e.g., water heater). As for water users, they need to be better differentiated in terms of water flows and water quality changes.

7.2.2. Towards scenario assessment in a decision making context

As shown in Chapter 6, the proposed framework and its associated model supply the stakeholder with an integrated tool for decision-making by taking into account a large set of environmental criteria offered by LCA. However, it must be remained that the scenarios assessed in the present study have been built by researchers and field experts, with the aim of assessing the capability of the WaLA model, but that they do not intend to have any political plausibility. Therefore, they have not been assessed in a decision making process where all stakeholders (i.e., all operators of water services, river basin agencies, citizens, etc.) are involved. Contributions of stakeholders to the scenario building process would be beneficial in many aspects. First, it is important that the results are presented to the various stakeholders to assess their level of understanding and adhesion among the public. In this context, the set of chosen indicators is an important issue for communication purposes. In addition, stakeholders can provide relevant analytical elements in the definition and evaluation functions of the systems. Also, they have a crucial role in the construction of policy responses scenarios.

7.2.3. Towards a tool for benchmarking

In addition to comparing different scenarios, the use of the proposed framework could also be used to compare impact/service ratios of different cities, and therefore compare the sustainability of diverse urban water system. This could be done by using a limited set of indicators such as the WIIX+ or, the endpoint damage scores. The use of the WaLA model for the UWS of another megacity has not been done yet, but the modularity of the model would allow it without much effort except for collecting ad hoc data. In addition, there are still remaining challenges to gather data from contrasted UWS, for example, with the presence of decentralized systems, such as wells for drinking water, or on-site sanitation. In developing countries, challenges related to UWS are different from those in developed countries. The level of services provided by the systems is way lower, with a limited access to safe drinking water, a non-continuous service, low sanitation (Montgomery and Elimelech, 2007). It also raises concerns about the LCA methodology for taking into account damages to human health occurring because of non-potable water use and non-sanitation (Harder et al., 2014; Heimersson et al., 2014).

Also, this approach could be integrated within a full assessment of a territory, as proposed by Loiseau et al. (2013), in order to compare environmental impacts of the urban water sector with other activities within the territory. More specifically regarding water use, consumption activities related to food lead to high impacts in other river basins of the world because of imported food, agriculture being an important water consumer. It is more likely that this induced water use would lead to higher impacts than the ones related to the urban water system (Hoekstra, 2012). Therefore stakeholders involved with water management in the territory have low room of maneuver for mitigating these impacts. However, such comparison of activities would be useful as it would raise awareness of the public. In the context of competition on water resources, using water for an activity instead of another would lead to consequential effects at different upper scales (neighboring territories, national, regional, continental) that should be studied.

Following the concept of carbon neutrality, there is a growing trend towards the water neutrality of industries (Hoekstra, 2008). In this context, there is a concern of the water services to mitigate water use impacts. This can be done by eco-design actions, in order to lower water footprint as studied in scenarios of Chapter 6. Also, compensation for negative impacts can be done for example by investing in improved watershed management - in the same hydrological unit - for example to reduce the upstream pollution. This is a step to go from integrated urban water management (IUWM) to the larger scale of integrated water resource management (IWRM). Investing in other river basins for compensating a water footprint is highly questionable since water impacts are local (ISO, 2013), on the contrary of carbon footprint which is a global impact. A first operational benefit of this thesis is that such a process to go water neutral has been engaged by Veolia Eau d'Île-de-France, delegatee of the SEDIF, in collaboration with the consulting company Quantis, based on the outputs of this thesis. However, this process must be completed by a full LCA-based multi criteria analysis for avoiding pollution shifting between impact categories.

7.3. General conclusion

The work done in this thesis sought to develop a method for the multi-criteria environmental assessment of urban water system seen as a whole (i.e., including water technologies, water users and water resources), in order to evaluate forecasting scenarios. The research hypothesis was: "a methodology can be developed in order to easily and consistently assess scenarios of urban water systems in megacities, within the framework of LCA." This global issue has been approached through two axes, each one related to a crucial phase of LCA. In the goal & scope and LCI phases, the question was: "how to model the UWS of big cities, in order to be at the same time, simple to implement, representative of a given UWS scenario, and compliant to LCA specifications?" In the LCIA phase, the question was: "regarding the fact that UWS will have major qualitative and quantitative effects on the water compartment, how to better take this effects into account?" These axes have led us to identify five sub-objectives.

The first sub-objective (Chapter 2) was to show that LCA is a worthy methodology in the environmental evaluation of UWS. The literature review revealed that LCA has been increasingly used to assess forecasting scenarios. However, there are still methodological challenges, such as the multi-functionality of the UWS, the need for better accounting of water quantity and quality.

The second and third sub-objectives (Chapter 3 and Chapter 4) are linked to the axe related to water-related impacts. They are of prime interest in the development of a model for LCA to be applied to UWS. A methodology for assessing water deprivation at the sub-river basin scale integrating "downstream cascade effect" has been developed in Chapter 3. The proposed scale, i.e., the sub-basin instead of the whole basin, is crucial for assessing water deprivation impacts of UWS since there are multiple choices of water withdrawal sources within a same river basin, and since locations of water release can be far from withdrawal ones. This approach proposes to go beyond the assessment of water scarcity at a finer scale by taking into account the impacts that a water withdrawal or consumption will have on downstream users and ecosystems, i.e., the extent to which it will deprive water in downstream sub-basins. Chapter 4 aims at accounting for water quality of urban water flows in order to manage the issue of water quality within the model. A classification of urban water flows is done, according to the damage scores of the different water types.

Fourth and fifth sub-objectives deal with the creation and implementation of a model, for representing UWS and assessing their environmental performance through LCA. The fourth sub-objective (Chapter 5) was the development of a versatile model for the LCA of UWS, namely the WaLA model. A framework has been proposed in order to tackle main methodological challenges related to the application of LCA to UWS, as identified in Chapter 2, and to integrate the developments presented in Chapter 3 and Chapter 4. This framework allows to easily build up scenarios of UWS composed of water users, water technologies and water resources, through a user-friendly graphical interface. It relies on the definition of a formalism based on a “generic component” that represents the water technologies and the water users. This generic component consistently manages the water quantity and quality going in and out, as well as the associated impacts related to water flows and to supporting activities (i.e., energy, chemicals, infrastructures, etc.). The model follows the territorial LCA approach, introduced by Loiseau et al. (2013), by computing environmental impacts and provided services for a UWS scenario.

The last sub-objective was to apply the WaLA model on a real case study, the urban water system of a suburban Parisian area, in order to address potential stakeholder’s questions and evaluate environmental impacts of associated forecasting scenarios. This application has shown the capacity of the model to easily implement scenarios, including changes in water users, technologies and resources within the UWS, and to provide indicators, i.e., environmental impacts and impact/service ratio. Based on this case study, opportunities and limits of the WaLA model have been identified. The main novelties of this model are its modularity and the possibility to define various types of water users and of water resources. However, remaining limits related to the management of pollutant mass balances and uncertainties require further developments. Finally, gathering LCI data for other water technologies and water users, as well as applying new LCIA methods related to water quality that are site-dependent, would increase the reliability and the completeness of the model.

The perspectives of the methodologies developed in this thesis are numerous. First, concerning the methodology for assessing water deprivation at the sub-basin scale, its application to the world basins still has to be done. Next step would be to implement the proposed methodology to multimedia fate model for assessing the impacts of water use within a mechanistic approach. Second, regarding the WaLA model, its application to other case studies in different context would be an interesting process to demonstrate its applicability. The appropriation of results by the stakeholders, and their contribution to the decision-making process are important challenges to be met with such tools.

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Annex A. Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature

This annex corresponds to the Supplementary Material of the publication presented in Chapter 2 and published in Water Research (Loubet et al. 2014).

Table A-1. Complete list of the 116 compiled LCA papers dealing with water technologies

Citation	Year	Full Citation	Journal	Technology	City	Country
Amores2013	2013	Amores, M.J., Meneses, M., Pasqualino, J., Antón, A., Castells, F., 2013. Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach. J. Clean. Prod. 43, 84–92.	Journal of Cleaner Production	UWS	Tarragona	Spain
Arpke2006	2006	Arpke, A., Hutzler, N., 2006. Domestic Water Use in the United States. Environ. Eng. 10, 169–184.	Environmental Engineering	UWS		United States$
Barjoveanu2013	2013	Barjoveanu, G., Comandaru, I.M., Rodriguez-Garcia, G., Hospido, A., Teodosiu, C., 2013. Evaluation of water services system through LCA. A case study for Iasi City, Romania. Int. J. Life Cycle Assess. 19, 449–462.	The International Journal of Life Cycle Assessment	UWS	Iasi City	Romania
Barrios2008	2008	Barrios, R., Siebel, M., Vanderhelm, A., Bosklopper, K., Gijzen, H., 2008. Environmental and financial life cycle impact assessment of drinking water production at Waternet. J. Clean. Prod. 16, 471–476.	Journal of Cleaner Production	DWP	Amsterdam	Netherlands
Basupi2013	2013	Basupi, I., Kapelan, Z., Butler, D., 2013. Reducing life-cycle carbon footprint in the (re)design of water distribution systems using water demand management interventions. Urban Water J. 1–17.	Urban Water Journal	DWD	New York	United States
Beavis2003	2003	Beavis, P., Lundie, S., 2003. Integrated environmental assessment of tertiary and residuals treatment--LCA in the wastewater industry. Water Sci. Technol. 47, 109–16.	Water Science and Technology	WWT	Sydney	Australia
Benetto2009	2009	Benetto, E., Nguyen, D., Lohmann, T., Schmitt, B., Schosseler, P., 2009. Life cycle assessment of ecological sanitation system for small-scale wastewater treatment. Sci. Total Environ. 407, 1506–16.	The Science of the total environment	WWT	Luxembourg	Luxembourg
Bonton2011	2011	Bonton, A., Bouchard, C., Barbeau, B., Jedrzejak, S., 2011. Comparative life cycle assessment of water treatment plants. Desalination.	Desalination	DWP	Lebel	Canada
Bravo2011	2011	Bravo, L., Ferrer, I., 2011. Life Cycle Assessment of an intensive sewage treatment plant in Barcelona (Spain) with focus on energy aspects. Water Sci. Technol. 64, 440.	Water Science and Technology	WWT	Barcelona	Spain
Brix1999	1999	Brix, H., 1999. How “green”are aquaculture, constructed wetlands and conventional wastewater treatment systems? Water Sci. Technol.	Water Science and Technology	WWT		
Buckley2011	2011	Buckley, C., Friedrich, E., Blottnitz, H. von, 2011. Life-cycle assessments in the South African water sector : A review and future challenges. WaterSA 37, 719–726.	Water SA	UWS		South Africa
Clauson-Kaas2004	2004	Clauson-Kaas, J., Poulsen, T.S., Neergaard-Jacobsen, B., Guildal, T., Thirsing, C., 2004. Economic and environmental optimization of phosphorus removal. Water Sci. Technol. 50, 243–8.	Water Science and Technology	WWT	Damhusaen	Denmark
Coats2011	2011	Coats, E.R., Watkins, D.L., Kranenburg, D., 2011. A Comparative Environmental Life-Cycle Analysis for Removing Phosphorus from Wastewater: Biological versus Physical/Chemical Processes. Water Environ. Res. 83.	Water Environment Research	WWT		United States
Corominas2013a	2013	Corominas, L., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S., Shaw, A., 2013. Life cycle assessment applied to wastewater treatment: state of the art. Water Res. 47, 5480–92.	Water research	WWT		
Crettaz1999	1999	Crettaz, P., Jolliet, O., Cuanillon, J.M., Orlando, S., 1999. Life cycle assessment of drinking water and rain water for toilets flushing. Aqua 48.	Aqua	DWP		Swiss
Dennison1998	1998	Dennison, F.J., Azapagic, A., 1998. Assessing management options for wastewater treatment works in the context of life cycle assessment. Water Sci. Technol. 38, 23–30.	Water Science and Technology	WWT	London	United Kingdom
Dixon2003	2003	Dixon, A., Simon, M., Burkitt, T., 2003. Assessing the environmental impact of two options for small-scale wastewater treatment: comparing a reedbed and an aerated biological filter using a life cycle approach. Ecol. Eng. 20, 297–308.	Ecological Engineering	WWT		United Kingdom
Mahgoub2010	2010	Mahgoub, M.E., van der Steen, N.P., Abu-Zeid, K., Vairavamoorthy, K., 2010. Towards sustainability in urban water: a life cycle analysis of the urban water system of Alexandria City, Egypt. J. Clean. Prod. 18, 1100–1106.	Journal of Cleaner Production	UWS	Alexandria	Egypt
EMMERSON1995	1995	Emmerson, R.H.C., Morse, G.K., Lester, J.N., Edge, D.R., 1995. The Life-Cycle Analysis of Small-Scale Sewage-Treatment Processes. Water Environ. J. 9, 317–325.	Water and Environment Journal	WWT		
Fagan2010	2010	Fagan, J.E., Reuter, M. a., Langford, K.J., 2010. Dynamic performance metrics to assess sustainability and cost effectiveness of integrated urban water systems. Resour. Conserv. Recycl. 54, 719–736.	Resources, Conservation and Recycling	UWS	Aurora	Australia

Citation	Year	Full Citation	Journal	Technology	City	Country
Filion2004	2004	Filion, Y.R., 2008. Impact of Urban Form on Energy Use in Water Distribution Systems. J. Infrastruct. Syst. 14, 337–346.	Journal of Infrastructure Systems	DWD	New York	United States
Filion2008	2008	Filion, Y.R., MacLean, H.L., Karney, B.W., 2004. Life-Cycle Energy Analysis of a Water Distribution System. J. Infrastruct. Syst. 10, 120.	Journal of Infrastructure Systems	DWD		
Flores-Alsina2010	2010	Flores-Alsina, X., Gallego, A., Feijoo, G., Rodriguez-Roda, I., 2010. Multiple-objective evaluation of wastewater treatment plant control alternatives. J. Environ. Manage. 91, 1193–201.	Journal of environmental management	WWT		
Foley2010	2010	Foley, J., de Haas, D., Hartley, K., Lant, P., 2010. Comprehensive life cycle inventories of alternative wastewater treatment systems. Water Res. 44, 1654–66.	Water research	WWT		Australia
Foley2010b	2010	Foley, J.M., Rozendal, R. a, Hertle, C.K., Lant, P. a, Rabaey, K., 2010. Life cycle assessment of high-rate anaerobic treatment, microbial fuel cells, and microbial electrolysis cells. Environ. Sci. Technol. 44, 3629–37.	Environmental science & technology	DWP		
Friedrich2002	2002	Friedrich, E., 2002. Life-cycle assessment as an environmental management tool in the production of potable water. Water Sci. Technol. 46, 29–36.	Water Science and Technology	DWP	Durban	South Africa
Friedrich2007	2007	Friedrich, E., Pillay, S., 2007. The use of LCA in the water industry and the case for an environmental performance indicator. Water SA 33, 443–452.	Water SA	UWS	eThekwini	South Africa
Friedrich2009	2009	Friedrich, E., Pillay, S., Buckley, C., 2009a. Carbon footprint analysis for increasing water supply and sanitation in South Africa: a case study. J. Clean. Prod. 17, 1–12.	Journal of Cleaner Production	UWS		
Friedrich2009a	2009	Friedrich, E., Pillay, S., Buckley, C., 2009b. Environmental life cycle assessments for water treatment processes a South African case study of an urban water cycle. Water SA 35, 73–84.	Water SA	UWS	eThekwini	South Africa
Fuchs2011	2011	Fuchs, V.J., Mihelcic, J.R., Gierke, J.S., 2011. Life cycle assessment of vertical and horizontal flow constructed wetlands for wastewater treatment considering nitrogen and carbon greenhouse gas emissions. Water Res. 45, 2073–81.	Water research	WWT		United States
Gallego2008	2008	Gallego, A., Hospido, A., Moreira, M.T., Feijoo, G., 2008. Environmental performance of wastewater treatment plants for small populations. Resour. Conserv. Recycl. 52, 931–940.	Resources, Conservation and Recycling	WWT	Coruna	Spain
Glick2013	2013	Glick, S., Guggemos, A.A., 2013. Rethinking Wastewater-Treatment Infrastructure: Case Study Using Life-Cycle Cost and Life-Cycle Assessment to Highlight Sustainability Considerations. J. Constr. Eng. Manag. A5013002.	Journal of Construction Engineering and Management	WWC		United States
Godin2012	2012	Godin, D., Bouchard, C., Vanrolleghem, P. a., 2012. Net environmental benefit: introducing a new LCA approach on wastewater treatment systems. Water Sci. Technol. 65, 1624.	Water Science and Technology	WWT	Quebec	Canada
Godskesen2011	2011	Godskesen, B., Hauschild, M., Rygaard, M., Zambrano, K.C., Albrechtsen, H.-J., 2013. Life-cycle and freshwater withdrawal impact assessment of water supply technologies. Water Res. 47, 2363–2374.	Water Science and Technology	DWP DWD	Copenhagen	Denmark
Godskesen2013a	2013	Godskesen, B., Zambrano, K.C., Trautner, A., Johansen, N.-B., Thiesson, L., Andersen, L., Clauson-Kaas, J., Neidel, T.L., Rygaard, M., Kløverpris, N.H., Albrechtsen, H.-J., 2011. Life cycle assessment of three water systems in Copenhagen--a management tool of the future. Water Sci. Technol. 63, 565–72.	Water Research	UWS	Copenhagen	Denmark
Hancock2011	2011	Hancock, N.T., Black, N.D., Cath, T.Y., 2011. A comparative life cycle assessment of hybrid osmotic dilution desalination and established seawater desalination and wastewater reclamation processes. Water Res. 46, 1145–1154.	Water Research	DWP		
Herstein2009	2009	Herstein, L., Filion, Y., Hall, K., 2011. Evaluating the Environmental Impacts of Water Distribution Systems by Using EIO-LCA-Based Multiobjective Optimization. J. Water Resour. Plan. Manag. 137, 162.	Journal of Infrastructure Systems	DWD		
Herstein2011	2011	Herstein, L.M., Filion, Y.R., Hall, K.R., 2009. Evaluating Environmental Impact in Water Distribution System Design. J. Infrastruct. Syst. 15, 241.	Journal of Water Resources Planning and Management	DWD		
Herz2002	2002	Herz, R., Lipkow, A., 2002. Life cycle assessment of water mains and sewers. Water Sci. Technol. water supply 51–58.	Water Science and Technology	DWD WWC	Dresden	Germany
Hoibye2008	2008	Høibye, L., Clauson-Kaas, J., Wenzel, H., Larsen, H.F.,	Water Science	WWT		Denmark

Citation	Year	Full Citation	Journal	Technology	City	Country
		Jacobsen, B.N., Dalgaard, O., 2008. Sustainability assessment of advanced wastewater treatment technologies. Water Sci. Technol. 58, 963–8.	and Technology			
Hospido2004	2004	Hospido, A., Moreira, M., 2004. Environmental performance of a municipal wastewater treatment plant. Int. J. Life Cycle Assesment 9, 261–271.	The International Journal of Life Cycle Assessment	WWT		Spain
Hospido2007	2007	Hospido, A., Moreira, M.T., Feijoo, G., 2007. A comparison of municipal wastewater treatment plants for big centres of population in Galicia (Spain). Int. J. Life Cycle Assess. 13, 57–64.	The International Journal of Life Cycle Assessment	WWT	Coruna	Spain
Igos2013	2013	Igos, E., Benetto, E., Baudin, I., Tiruta-Barna, L., Mery, Y., Arbault, D., 2013. Cost-performance indicator for comparative environmental assessment of water treatment plants. Sci. Total Environ. 443, 367–74.	The Science of the total environment	DWP	Paris	France
Igos2014	2014	Igos, E., Dalle, A., Tiruta-Barna, L., Benetto, E., Baudin, I., Mery, Y., 2014. Life Cycle Assessment of water treatment: what is the contribution of infrastructure and operation at unit process level? J. Clean. Prod. 65, 424–431.	Journal of Cleaner Production	DWP		France
Kalbar2013	2013	Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2013. Assessment of wastewater treatment technologies: life cycle approach. Water Environ. J. 27, 261–268.	Water and Environment Journal	WWT		India
Karrman2001	2001	Kärrman, E., Jönsson, H., 2001. Normalising impacts in an environmental systems analysis of wastewater systems. Water Sci. Technol. 43, 293–300.	Water Science and Technology	WWT		Sweden
Klaversma2013	2013	Klaversma, E., van der Helm, A.W.C., Kappelhof, J.W.N.M., 2013. The use of life cycle assessment for evaluating the sustainability of the Amsterdam water cycle. J. Water Clim. Chang. 4, 103.	Journal of Water and Climate Change	UWS	Amsterdam	Netherlands
Landu2007	2007	Landu, L., Brent, A.C., 2007. Environmental life cycle assessment of water supply in South Africa: The Rosslyn industrial area as a case study. Water SA 32, 249–256.	Water SA	DWP	Rosslyn	South Africa
Lassaux2006a	2006	Lassaux, S., Renzoni, R., Germain, A., 2006. Life Cycle Assessment of Water: From the pumping station to the wastewater treatment plant (9 pp). Int. J. Life Cycle Assess. 12, 118–126.	The International Journal of Life Cycle Assessment	UWS	Bruxelles	Belgium
Lemos2013a	2013	Lemos, D., Dias, A.C., Gabarrell, X., Arroja, L., 2013. Environmental assessment of an urban water system. J. Clean. Prod. 54, 157–165.	Journal of Cleaner Production	UWS	Aveiro	Portugal
Lim2007	2007	Lim, S., Park, J., 2007. Environmental and economic analysis of a water network system using LCA and LCC. AIChE J. 53, 3253–3262.	AIChE Journal	DWP DWD		
Lim2008	2008	Lim, S.-R., Park, J.M., 2008. Synthesis of an Environmentally Friendly Water Network System. Ind. Eng. Chem. Res. 47, 1988–1994.	Industrial & Engineering Chemistry Research	DWD		
Lim2009	2009	Lim, S.-R., Park, J.M., 2009. Environmental impact minimization of a total wastewater treatment network system from a life cycle perspective. J. Environ. Manage. 90, 1454–62.	Journal of environmental management	WWC WWT		
Lundie2004	2004	Lundie, S., Peters, G., Beavis, P., 2005. Quantitative systems analysis as a strategic planning approach for metropolitan water service providers. Water Sci. Technol. 52, 11–20.	Environmental science & technology	UWS	Sydney	Australia
Lundie2005	2005	Lundie, S., Peters, G.M., Beavis, P.C., 2004. Life cycle assessment for sustainable metropolitan water systems planning. Environ. Sci. Technol. 38, 3465–73.	Water Science and Technology	WWT		
Lundin2000	2000	Lundin, M., Bengtsson, M., Molander, S., 2000. Life Cycle Assessment of Wastewater Systems: Influence of System Boundaries and Scale on Calculated Environmental Loads. Environ. Sci. Technol. 34, 180–186.	Environmental science & technology	WWT	Lulea	Sweden
Lundin2002	2002	Lundin, M., Morrison, G.M., 2002. A life cycle assessment based procedure for development of environmental sustainability indicators for urban water systems. Urban Water 4, 145–152.	Urban Water Journal	UWS	Goteborg	Sweden
Machado2007	2007	Machado, a. P., Urbano, L., Brito, A.G., Janknecht, P., Salas, J.J., Nogueira, R., 2007. Life cycle assessment of wastewater treatment options for small and decentralized communities. Water Sci. Technol. 56, 15.	Water Science and Technology	WWT		
Mels1999	1999	Mels, A., van Nieuwenhuijzen, A.F., van der Graaf, J.H.J.M., Klapwijk, B., de Konin, J., Rulkens, W.H., 1999. Sustainability criteria as a tool in the development of new sewage treatment methods. Water Sci. Technol.	Water Science and Technology	WWT		Netherlands
Meneses2010	2010	Meneses, M., Pasqualino, J.C., Castells, F., 2010.	Chemosphere	WWT		Spain

Citation	Year	Full Citation	Journal	Technology	City	Country
		Environmental assessment of urban wastewater reuse: treatment alternatives and applications. Chemosphere 81, 266–72.				
Mery2013	2013	Mery, Y., Tiruta-Barna, L., Benetto, E., Baudin, I., 2013. An integrated “process modelling-life cycle assessment” tool for the assessment and design of water treatment processes. Int. J. Life Cycle Assess.	The International Journal of Life Cycle Assessment	DWP	Paris	France
Mo2010	2010	Mo, W., Nasiri, F., Eckelman, M.J., Zhang, Q., Zimmerman, J.B., 2010. Measuring the embodied energy in drinking water supply systems: a case study in the Great Lakes region. Environ. Sci. Technol. 44, 9516–21.	Environmental science & technology	DWP DWD	Kalamazoo	United States
Mo2011	2011	Mo, W., Zhang, Q., Mihelcic, J.R., Hokanson, D.R., 2011. Embodied energy comparison of surface water and groundwater supply options. Water Res. 45, 5577–5586.	Water research	DWP DWD	Kalamazoo \& Tampa	United States
Mohamed-Zine2013	2013	Mohamed-Zine, M.-B., Hamouche, A., Krim, L., 2013. The study of potable water treatment process in Algeria (boudouaou station) -by the application of life cycle assessment (LCA). J. Environ. Heal. Sci. Eng. 11, 37.	Journal of environmental health science & engineering	DWP		Algeria
Mohapatra2002	2002	Mohapatra, P., Siebel, M., Gijzen, H., Van der Hoek, J., Groot, C., 2002. Improving eco-efficiency of Amsterdam water supply: A LCA approach. Aqua 51, 217–227.	Aqua	DWP	Amsterdam	Netherlands
Moore2013	2013	Moore, T.L.C., Hunt, W.F., 2013. Predicting the carbon footprint of urban stormwater infrastructure. Ecol. Eng. 58, 44–51.	Ecological Engineering	WWC		United States
Munoz2008	2008	Muñoz, I., Fernández-Alba, A.R., 2008. Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. Water Res. 42, 801–11.	Water research	DWP	Almeria	Spain
Munoz2010b	2010	Muñoz, I., Milà-i-Canals, L., Fernández-Alba, A.R., 2010. Life Cycle Assessment of Water Supply Plans in Mediterranean Spain. J. Ind. Ecol. 14, 902–918.	Journal of Industrial Ecology	UWS		Spain
Nogueira2009	2009	Nogueira, R., Brito, A.G., Machado, A.P., Janknecht, P., Salas, J.J., Vera, L., Martel, G., 2009. Economic and environmental assessment of small and decentralized wastewater treatment systems. Desalin. Water Treat. 4, 16–21.	Desalination and Water Treatment	WWT	Minho	Portugal
Ortiz2007	2007	Ortiz, M., Raluy, R., Serra, L., 2007. Life cycle assessment of water treatment technologies: wastewater and water-reuse in a small town. Desalination 204, 121–131.	Desalination	WWT	Tauste	Spain
Pasqualino2009	2009	Pasqualino, J.C., Meneses, M., Abella, M., Castells, F., 2009. LCA as a decision support tool for the environmental improvement of the operation of a municipal wastewater treatment plant. Environ. Sci. Technol. 43, 3300–7.	Environmental science & technology	WWT	Tarragona	Spain
Pasqualino2011a	2011	Pasqualino, J.C., Meneses, M., Castells, F., 2011. Life Cycle Assessment of Urban Wastewater Reclamation and Reuse Alternatives. J. Ind. Ecol. 15, 49–63.	Journal of Industrial Ecology	WWT	Barcelona	Spain
Piratla2012	2011	Piratla, K., Ariaratnam, S.T., Cohen, A., 2011. Estimation of CO 2 emissions from the life cycle of a potable water pipeline project. J. Manag. … 22–30.	Urban Water Journal	DWD	Phoenix	United States
Poussade2011	2011	Poussade, Y., Vince, F., Robillot, C., 2011. Energy consumption and greenhouse gases emissions from the use of alternative water sources in South East Queensland. Water Sci. Technol. Water Supply 11, 281.	Water Science and Technology	DWP	South East Queensland	Australia
Qi2012	2012	Qi, C., Chang, N.-B., 2012. Integrated carbon footprint and cost evaluation of a drinking water infrastructure system for screening expansion alternatives. J. Clean. Prod. 27, 51–63.	Journal of Cleaner Production	UWS	Manatee County	United States
Racoviceanu2007a	2007	Racoviceanu, A.I., Karney, B.W., Kennedy, C.A., Colombo, A.F., 2007. Life-Cycle Energy Use and Greenhouse Gas Emissions Inventory for Water Treatment Systems. J. Infrastruct. Syst. 13, 261–270.	Journal of Infrastructure Systems	WWT	Toronto	Canada
Raluy2004	2004	Raluy, R.G., Serra, L., Uche, J., 2004. Life Cycle Assessment of Water Production Technologies - Part 1: Life Cycle Assessment of Different Commercial Desalination Technologies (MSF, MED, RO) (9 pp). Int. J. Life Cycle Assess. 10, 285–293.	The International Journal of Life Cycle Assessment	DWP		
Raluy2004a	2004	Raluy, R.G., Serra, L., Uche, J., Valero, A., 2004. Life Cycle Assessment of Water Production Technologies - Part 2: Reverse Osmosis Desalination versus the Ebro River Water Transfer (9 pp). Int. J. Life Cycle Assess. 10, 346–354.	The International Journal of Life Cycle Assessment	DWP		Spain
Remy2008	2008	Remy, C., Jekel, M., 2008. Sustainable wastewater management: life cycle assessment of conventional and	Water Science	WWT		Germany

Citation	Year	Full Citation	Journal	Technology	City	Country
		source-separating urban sanitation systems. Water Sci. Technol. 58, 1555–62.	and Technology			
Remy2012	2012	Remy, C., Jekel, M., 2012. Energy analysis of conventional and source-separation systems for urban wastewater management using Life Cycle Assessment. Water Sci. Technol. 65, 22–9.	Water Science and Technology	UWS	Berlin	Germany
Renou2008	2008	Renou, S., Thomas, J.S., Aoustin, E., Pons, M.N., 2008. Influence of impact assessment methods in wastewater treatment LCA. J. Clean. Prod. 16, 1098–1105.	Journal of Cleaner Production	WWT		France
Ribera2013	2013	Ribera, G., Clarens, F., Martínez-Lladó, X., Jubany, I., V Martí, Rovira, M., 2013. Life cycle and human health risk assessments as tools for decision making in the design and implementation of nanofiltration in drinking water treatment plants. Sci. Total Environ. 466-467C, 377–386.	The Science of the total environment	DWP	Manresa	Spain
Risch2014	2014	Risch, E., Loubet, P., Núñez, M., Roux, P., 2014. How environmentally significant is water consumption during wastewater treatment? : Application of recent developments in LCA to WWT technologies used at 3 contrasted geographical locations. Water Res.	Water Research	WWT		France
Rodriguez-Garcia2011	2011	Rodriguez-Garcia, G., Molinos-Senante, M., Hospido, A., Hernandez-Sancho, F., Moreira, M.T., Feijoo, G., 2011. Environmental and economic profile of six typologies of wastewater treatment plants. Water Res. 5.	Water Research	WWT		Spain
Roeleveld1997	1997	Roeleveld, P., Klapwijk, A., Eggels, P.G., Rulkens, W.H., van Starkenburg, W., 1997. Sustainability of municipal waste water treatment. Water Sci. Technol.	Water science and Technology	WWT		Netherlands
Rowley2009	2009	Rowley, H. V., Lundie, S., Peters, G.M., 2009. A hybrid life cycle assessment model for comparison with conventional methodologies in Australia. Int. J. Life Cycle Assess. 14, 508–516.	The International Journal of Life Cycle Assessment	UWS		Australia
Sahely2005	2005	Sahely, H.R., Kennedy, C.A., Adams, B., 2005. Developing sustainability criteria for urban infrastructure systems. Can. J. Civ. Eng. 32, 72–85.	Canadian Journal of Civil Engineering	UWS	Toronto	Canada
Sharaai2009b	2009	Sharaai, A.H., Mahmood, N.Z., Sulaiman, A.H., 2009. Life Cycle Impact Assessment (LCIA) of Potable Water Treatment Process in Malaysia: Comparison Between Dissolved Air Flotation (DAF) and Ultrafiltration (UF) Technology. Aust. J. Basic Appl. Sci. 3, 3625–3632.	Australian Journal of Basic and Applied Sciences	DWP		
Sharaai2010	2010	Sharaai, A.H., Mahmood, N.Z., Sulaiman, A.H., 2010. Life Cycle Impact Assessment (LCIA) in Potable Water Production in Malaysia : Potential Impact Analysis Contributed from Production and Contruction Phase Using Eco-indicator 99 Evaluation Method. World Appl. Sci. J. 11, 1230–1237.	World Applied Sciences Journal	DWP		Malaysia
Sharaai2011	2011	Sharaai, A.H., Mahmood, N.Z., Sulaiman, A.H., 2011. Life cycle impact assessment (LCIA) using EDIP 97 method: An analysis of potential impact from potable water production. Sci. Res. Essays 6, 5658–5670.	Scientific Research and Essays	DWP		Malaysia
Slagstad2013	2013	Slagstad, H., Brattebø, H., 2013. Life cycle assessment of the water and wastewater system in Trondheim, Norway – A case study. Urban Water J. 1–12.	Urban Water Journal	UWS	Trondheim	Norway
Sombekke1997	1997	Sombekke, H., Voorhoeve, D., Hiemstra, P., 1997. Environmental impact assessment of groundwater treatment with nanofiltration. Desalination 113, 293–296.	Desalination	DWP	Hammerflier	Netherlands
Stokes2005	2005	Stokes, J., Horvath, A., 2005. Life Cycle Energy Assessment of Alternative Water Supply Systems (9 pp). Int. J. Life Cycle Assess. 11, 335–343.	The International Journal of Life Cycle Assessment	DWP DWD	California	United States
Stokes2006	2006	Stokes, J., Horvath, A., 2006. LCA Methodology and Case Study Life Cycle Energy Assessment of Alternative Water Supply Systems. Int. J. Life Cycle Assess. 11, 335 – 343.	The International Journal of Life Cycle Assessment	DWP DWD		
Stokes2009	2009	Stokes, J., Horvath, A., 2009. Energy and air emission effects of water supply. Environ. Sci. Technol. 43, 2680–2687.	Environmental science & technology	DWP DWD	California	United States
Stokes2011	2011	Stokes, J., Horvath, A., 2011. Life-Cycle Assessment of Urban Water Provision: Tool and Case Study in California. J. Infrastruct. Syst. 17, 15.	Journal of Infrastructure Systems	DWP DWD	California	United States
Tarnacki2011	2011	Tarnacki, K.M., Melin, T., Jansen, a. E., van Medevoort, J., 2011. Comparison of environmental impact and energy efficiency of desalination processes by LCA. Water Sci. Technol. Water Supply 11, 246.	Water Science and Technology	DWP		
Tillman1998	1998	Tillman, A.M., Svingby, M., Lundström, H., 1998. Life cycle assessment of municipal waste water systems. Int. J. Life Cycle Assess. 3, 145–157.	The International Journal of Life	UWS	Bergsjon	Sweden

Citation	Year	Full Citation	Journal	Technology	City	Country
			Cycle Assessment			
Uche2013	2013	Uche, J., Martínez-Gracia, A., Carmona, U., 2013. Life cycle assessment of the supply and use of water in the Segura Basin. Int. J. Life Cycle Assess.	The International Journal of Life Cycle Assessment	DWP DWD	Alicante	Spain
VanderHelm2008	2008	Van der Helm, a. W.C., Rietveld, L.C., Bosklopper, T.G.J., Kappelhof, J.W.N.M., van Dijk, J.C., 2008. Objectives for optimization and consequences for operation, design and concept of drinking water treatment plants. Water Sci. Technol. Water Supply 8, 297.	Water Science and Technology	DWP	Amsterdam	Netherlands
Venkatesh2009	2009	Venkatesh, G., Brattebø, H., 2011. Energy consumption, costs and environmental impacts for urban water cycle services: Case study of Oslo (Norway). Energy 36, 792–800.	Journal of Industrial Ecology	WWC	Oslo	Norway
Venkatesh2011	2011	Venkatesh, G., Brattebø, H., 2011. Environmental impact analysis of chemicals and energy consumption in wastewater treatment plants: case study of Oslo, Norway. Water Sci. Technol. 63, 1018–31.	Energy	UWS	Oslo	Norway
Venkatesh2011a	2011	Venkatesh, G., Brattebø, H., 2012a. Assessment of Environmental Impacts of an Aging and Stagnating Water Supply Pipeline Network. J. Ind. Ecol. 00, no–no.	Water Science and Technology	WWT	Oslo	Norway
Venkatesh2012	2012	Venkatesh, G., Brattebø, H., 2012b. Environmental analysis of chemicals and energy consumption in water treatment plants: case study of Oslo, Norway. Water Sci. Technol. Water Supply 12, 200.	Journal of Industrial Ecology	DWD	Oslo	Norway
Venkatesh2012a	2012	Venkatesh, G., Hammervold, J., Brattebø, H., 2009. Combined MFA-LCA for Analysis of Wastewater Pipeline Networks. J. Ind. Ecol. 13, 532–550.	Water Science and Technology	DWP	Oslo	Norway
Vidal2002	2002	Vidal, N., Poch, M., Marti, E., Rodriguez-Roda, I., 2002. Evaluation of the environmental implications to include structural changes in a wastewater treatment plant. J. Chem. Technol. Biotechnol. 77, 1206–1211.	Journal of Chemical Technology& Biotechnology	WWT		Spain
Vince2008	2008	Vince, F., Aoustin, E., Breant, P., Marechal, F., 2008. LCA tool for the environmental evaluation of potable water production. Desalination 220, 37–56.	Desalination	DWP		
Vlasopoulos2006	2006	Vlasopoulos, N., Memon, F. a, Butler, D., Murphy, R., 2006. Life cycle assessment of wastewater treatment technologies treating petroleum process waters. Sci. Total Environ. 367, 58–70.	The Science of the total environment	WWT		
Wang2012	2012	Wang, X., Liu, J., Ren, N.-Q., Duan, Z., 2012. Environmental profile of typical anaerobic/anoxic/oxic wastewater treatment systems meeting increasingly stringent treatment standards from a life cycle perspective. Bioresour. Technol. 126C, 31–40.	Bioresource technology	WWT		China
Wenzel2008	2008	Wenzel, H., Larsen, H.F., Clauson-Kaas, J., Høibye, L., Jacobsen, B.N., 2008. Weighing environmental advantages and disadvantages of advanced wastewater treatment of micro-pollutants using environmental life cycle assessment. Water Sci. Technol. 57, 27–32.	Water Science and Technology	WWT		
Wu2010	2010	Wu, J.-G., Meng, X.-Y., Liu, X.-M., Liu, X.-W., Zheng, Z.-X., Xu, D.-Q., Sheng, G.-P., Yu, H.-Q., 2010. Life cycle assessment of a wastewater treatment plant focused on material and energy flows. Environ. Manage. 46, 610–7.	Environmental management	WWT	Bengdu City	China
Wu2010a	2010	Wu, W., Maier, H.R., Simpson, A.R., 2010a. Single-Objective versus Multiobjective Optimization of Water Distribution Systems Accounting for Greenhouse Gas Emissions by Carbon Pricing. J. Water Resour. Plan. Manag. 136, 555–565.	Journal of Water Resources Planning and Management	DWD		Australia
Wu2010b	2010	Wu, W., Simpson, A.R., Maier, H.R., 2010b. Accounting for Greenhouse Gas Emissions in Multiobjective Genetic Algorithm Optimization of Water Distribution Systems. J. Water Resour. Plan. Manag. 136, 146–155.	Journal of Water Resources Planning and Management	DWD		
Yldrm2012	2012	Yıldırım, M., Topkaya, B., 2012. Assessing Environmental Impacts of Wastewater Treatment Alternatives for Small-Scale Communities. CLEAN - Soil, Air, Water 40, 171–178.	CLEAN - Soil, Air, Water	WWT		
Zhang2000	2000	Zhang, Q.H., Wang, X.C., Xiong, J.Q., Chen, R., Cao, B., 2010. Application of life cycle assessment for an evaluation of wastewater treatment and reuse project--case study of Xi'an, China. Bioresour. Technol. 101, 1421–5.	Water and Environment Journal	WWT		Singapore
Zhang2010a	2010	Zhang, Z., 2000. Life-Cycle Assessment of a Sewage-Treatment Plant in South-East Asia. Water Environ. J. 14, 51–56.	Bioresource technology	WWT	Xi'an	China

Annex B. Assessing water deprivation at the sub-river basin scale in LCA integrating downstream cascade effects

This annex corresponds to the Supplementary Material of the publication presented in Chapter 3 and published in Environmental Science & Technology (Loubet et al. 2013).

B.1. Illustration of the Consumption-to-Availability ratio at the sub-river basin scale

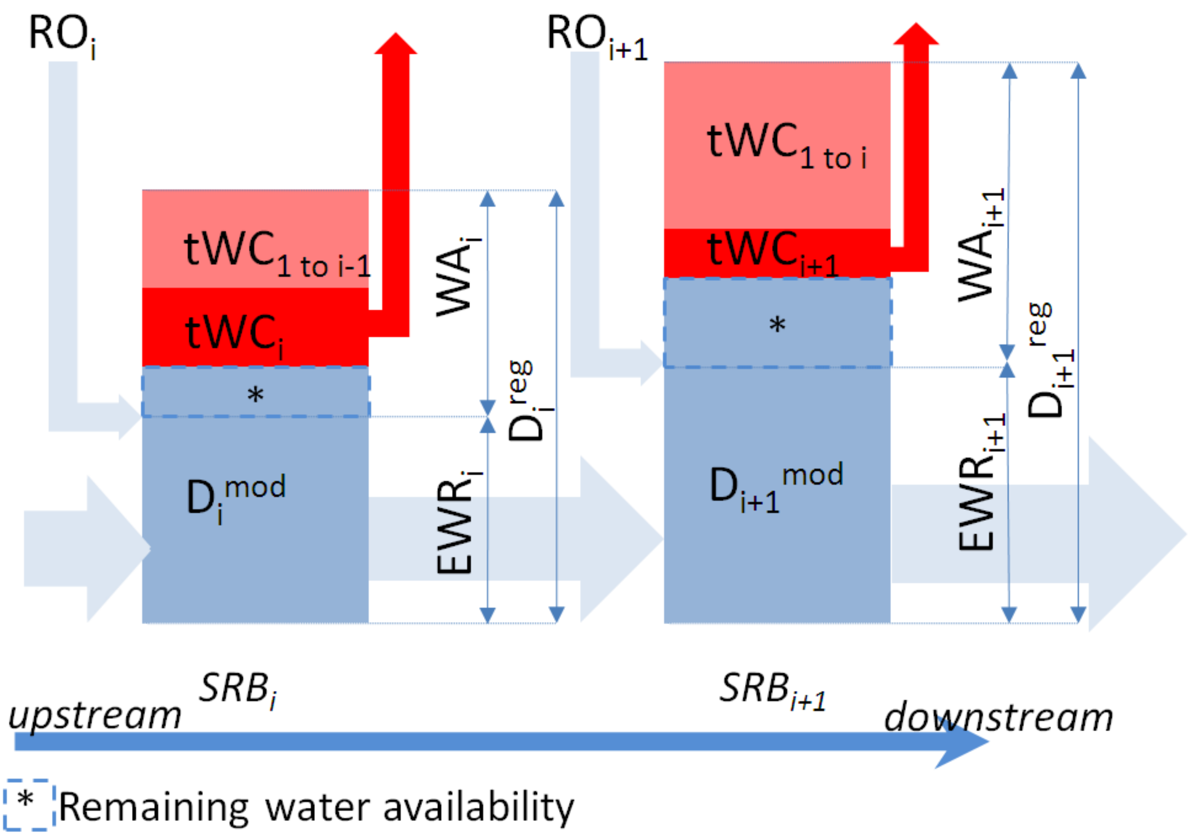


Figure B-1. Water consumption and water availability propagation through a river basin. *Subscripts:* $_i$=assessed SRB, $_{1\ to\ i}$=all upstream SRBs, $_{i+1}$=first downstream SRB; *Superscripts:* ^{mod}=modified, ^{reg}=regulated; RO=Runoff, D=Discharge, tWC=total Water Consumption, WA=Water Availability, EWR=Environmental Water Requirements.

B.2. Description of the Pfafstetter topologic navigation system

The identification of the upstream and downstream sub-river basins for each SRB has been made from the Pfafstetter sub-river basin coding system provided by the HYDRO1k database. The Pfafstetter system is hierarchal, and sub-river basins are delineated from junctions on a river network. Level 1 basins correspond to continental scale basins. Higher levels (levels 2, 3, 4, etc.) represent ever-finer tessellations of the land surface into smaller basins. Each basin is assigned a specific Pfafstetter Code based on its location within the overall drainage system and on the total drainage area upstream of the watershed's outlet. In the Hydro1K database, each smallest sub-river basin unit has 6 Pfafstetter digits, each one corresponding to a level of basin.

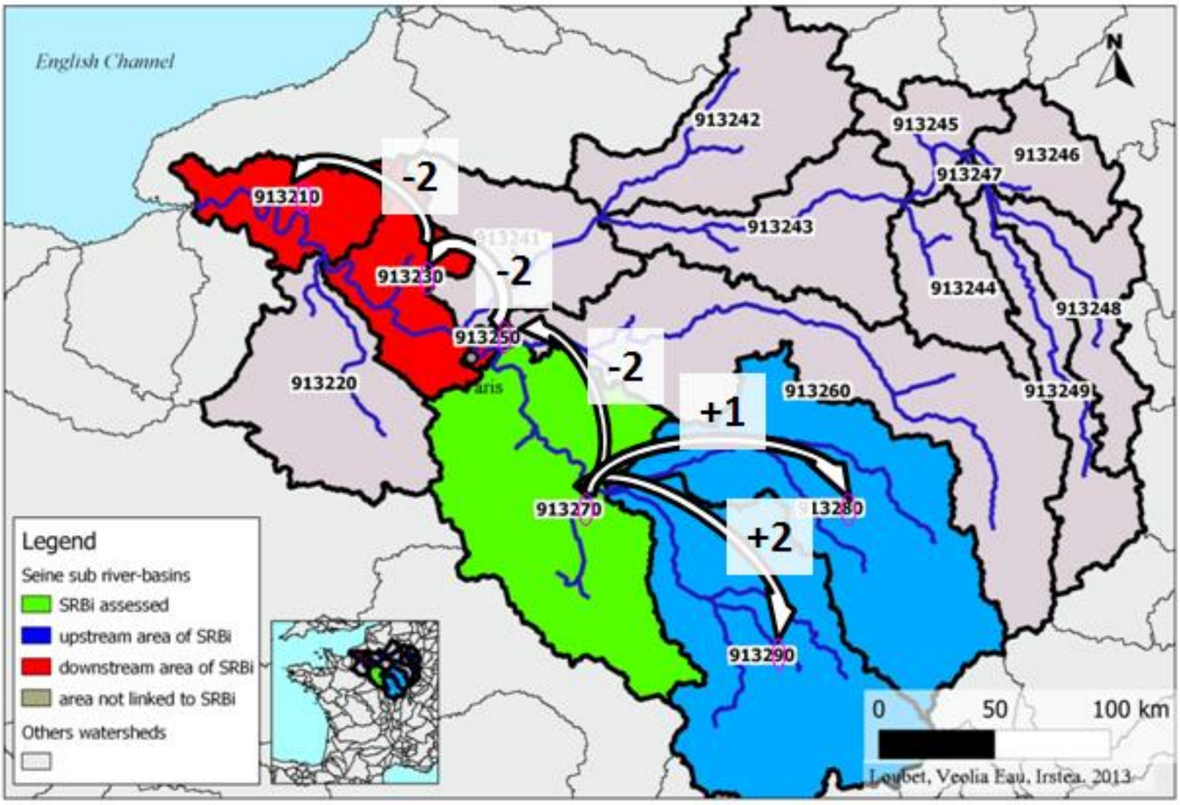


Figure B-2. Downstream and upstream SRBs identification according to the Pfafstetter system.

In Figure B-2. **Downstream and upstream SRBs identification according to the Pfafstetter system.**Figure B-2 is presented the original ID of each sub-river basin of the Seine river basin according to Hydro1k database. The four first digits (i.e., 9132) are common for each SRB and characterize the Seine river basin. The two last digits are those presented in the paper and are different for each SRB.

The last digit which is not zero gives information on the position of the sub-river basin: if it is an even digit (2, 4, 6, 8), the SRB has no upstream SRB, if it is an odd digit (1, 3, 5, 7, 9), the SRB has two upstream SRBs, excepted the last upstream SRB. The most downstream SRB has the digit 1 and the two most upstream have the digit 8 and 9. Consequently, a SRB which has an odd last digit k will have two upstream SRBs k+1 and k+2 and one downstream SRB k-2. For example, in the Figure B-2, basin 913270 have two direct upstream basins (913280 and 913290) and one direct downstream SRB (913250). It has also two other downstream SRBs (k-4 and k-6), i.e., 913230 and 913210.

Since the routine made available by Furnans and Olivera (2001) gives automatically the two direct upstream and the direct downstream basins of each basin, we built a routine which navigates within the SRBs data table to locate all the downstream and upstream SRBs of each

SRB. All the upstream and downstream SRBs of each SRB are identified by writing new routines under Matlab (e.g., in Figure 3-4, id42, id43, id45, id47, id44, id46, id48, id49 are the upstream SRBs of id 41 and id30 and id10 are the downstream SRBs of id41). From this identification and local data of RO_i and tWC_i, CTA_i and $CF_{WD,i}$ are respectively computed with eq (5), (7) and (8).

B.3. Step-by-step reproducible procedure

As to apply the proposed methodology at the sub-river basin scale, one needs the local data (tWC and RO) of each SRB, and the identification of each upstream and downstream SRBs. It can be done with the paper's data or any other chosen data

The procedure is summarized in Figure B-3. Blue boxes refer to data processing routines, which are detailed below.

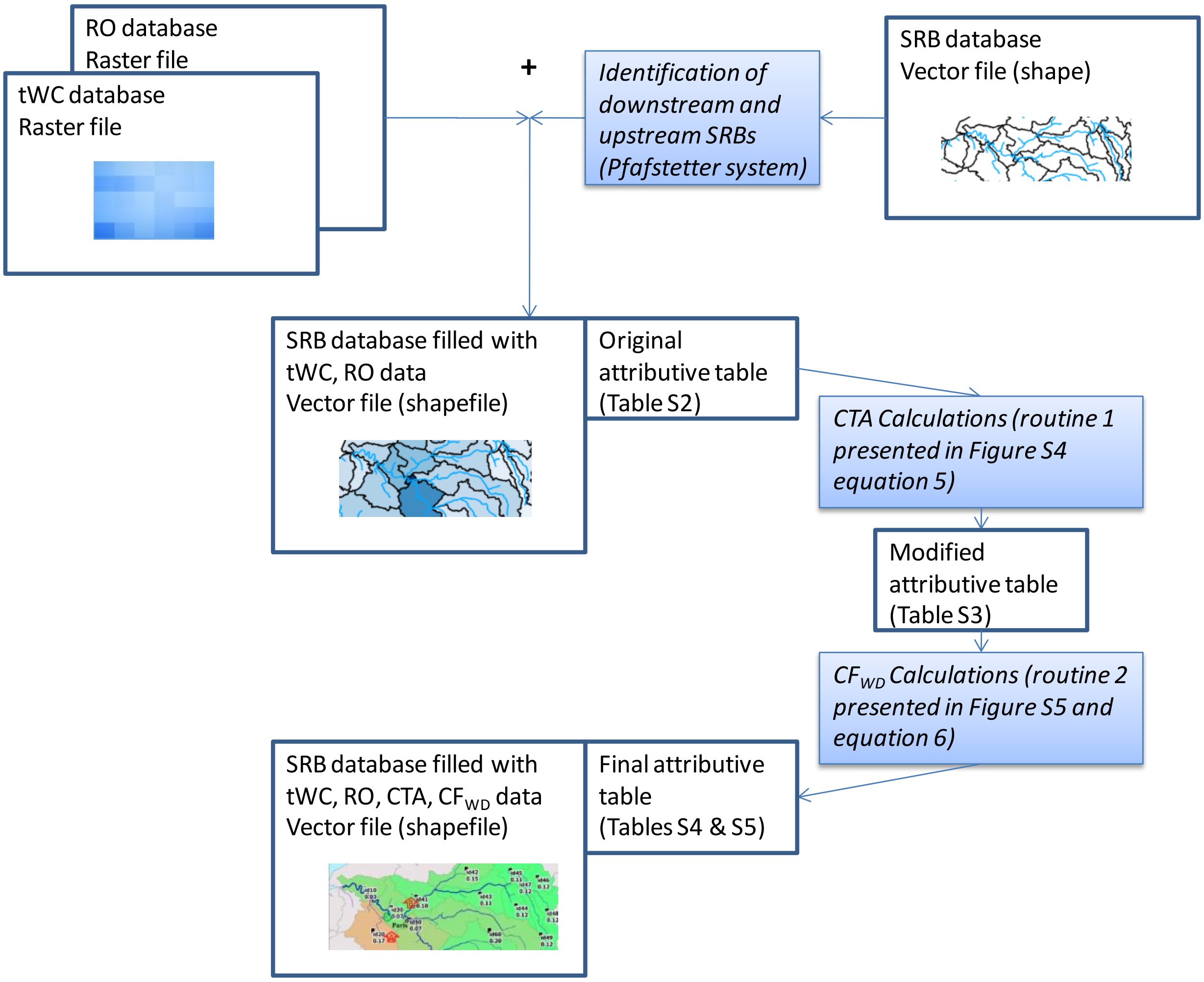


Figure B-3. Summary of the procedure to reproduce the proposed methodology for calculating CF_{WD} at the sub-river basin scale

B.3.1. Identification of downstream and upstream SRBs.

One method (Pfafstetter) is explained in the previous part of SI. From this identification and the merging of SRBs vector file and tWC, RO and p (area, population, volume) raster files, the following attributive table is built:

Table B-1. Attributive table filled with hydrologic parameters and identification of upstream and downstream SRBs. a is the index of the SRB, id is the simplified Pfafstetter identifier of SRB_i, down is the simplified Pfafstetter identifier of the downstream SRB of SRB_i, up1 and up2 are the simplified Pfafstetter identifiers of the two upstream SRBs of SRB_i. Displayed data are not real and are shown as an example.

a	id $=SRB_i$	down $=SRB_{i+1}$	up1 $=SRB_{i-1}$	up2 $=SRB_{i-2}$	tWC_i (m^3)	RO_i (m^3)	p (area, population, …)
5 (n)	10	0	20	30	10	50	
4	20	10	0	0	10	50	
3	30	10	40	50	10	100	
2	40	30	0	0	10	100	
1	50	30	0	0	10	100	

B.3.2. CTA calculations – upstream parameters

First, $tWC_{1\ to\ i}$ and $RO_{1\ to\ i}$ are calculated for each SRBs, taking into account upstream data. From the Table B-1 data and the Figure B-4, the following routine is applied to calculate the upstream sum of a parameter (tWC or RO).

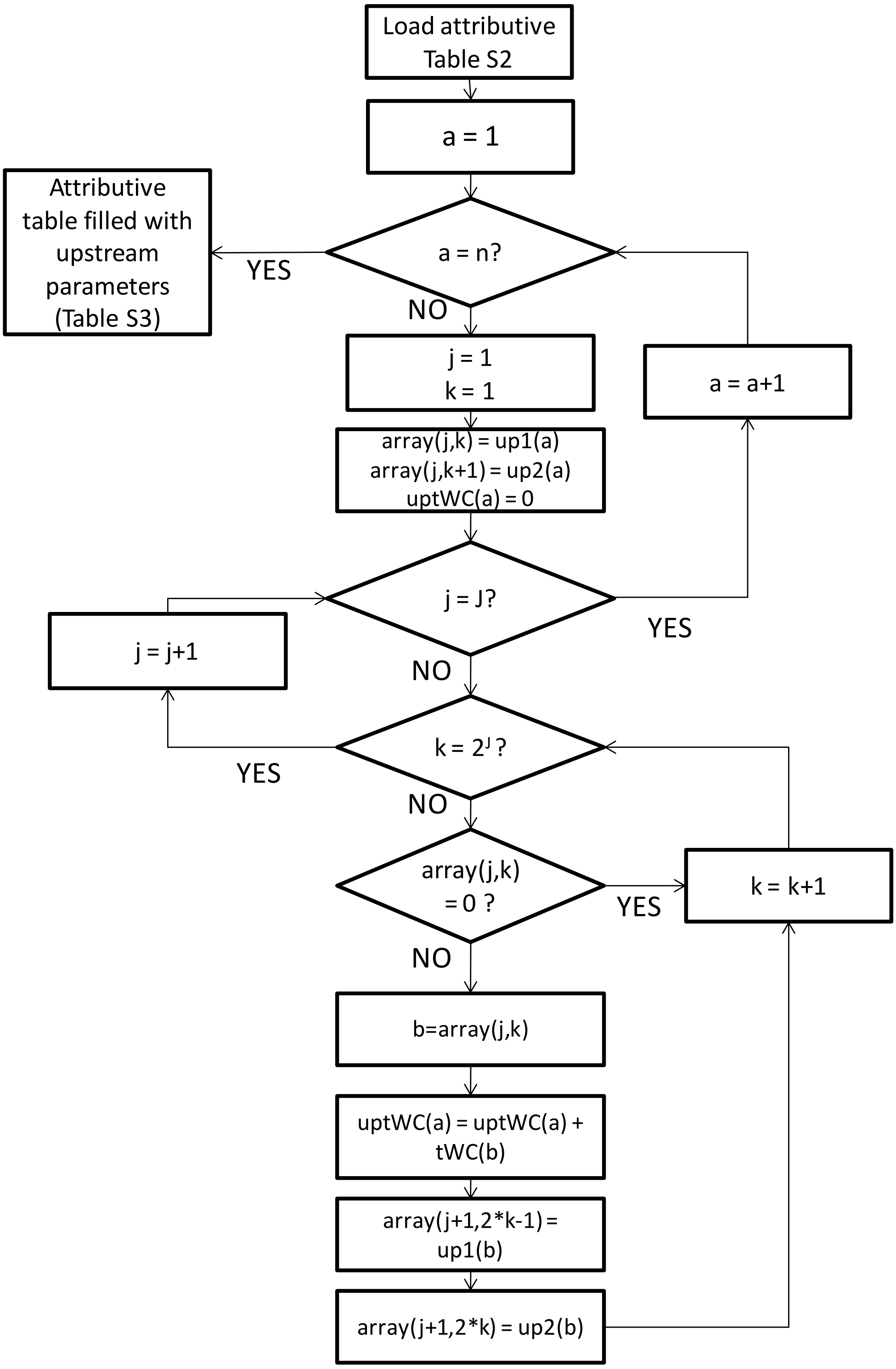


Figure B-4. Routine that calculates upstream parameters (e.g. tWC, here). uptWC(i) represents $tWC_{1\ to\ i}$. J is assigned from **Figure B-5**.

The array that contains j and k indices stocks all the upstream SRBs for each SRB_i, according to Figure B-5.

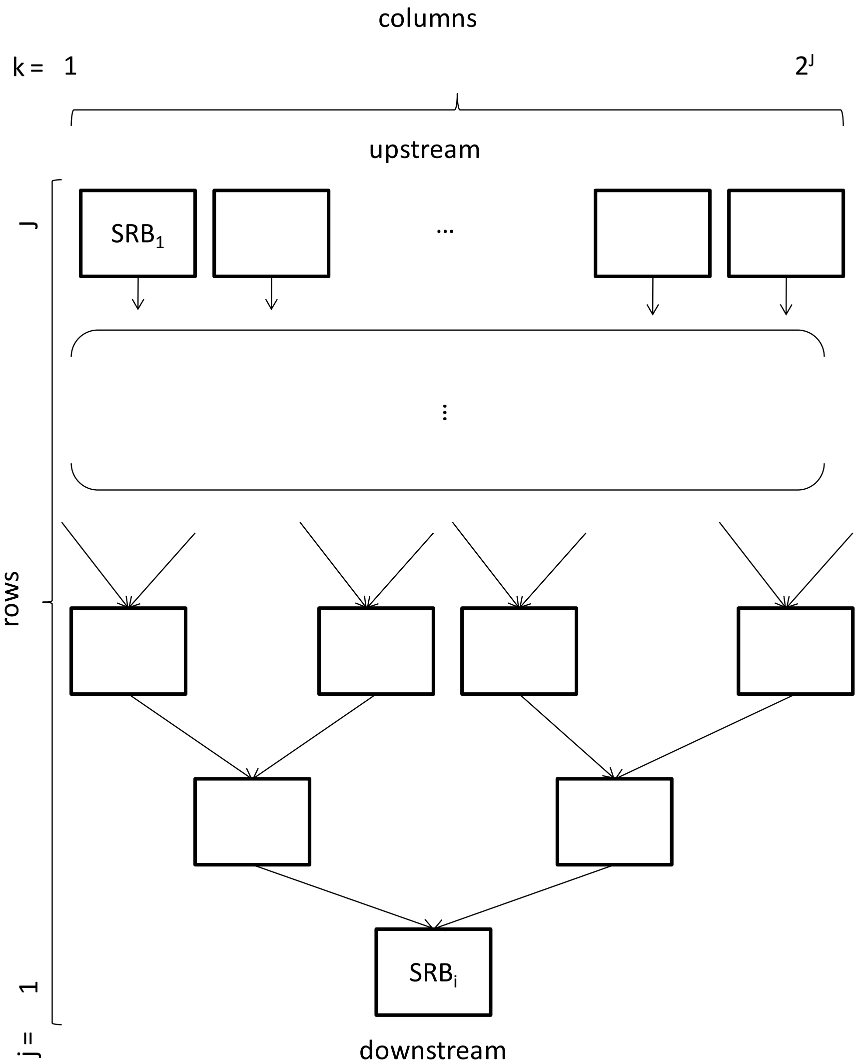


Figure B-5. Scheme representing the possible upstream SRBs of a SRB_i

Then, WA and CTA are simply calculated for each SRBs according to eq (5) and (7). From there, the attributive table is modified as shown in Figure B-2.

Table B-2. Modified attributive table filled with upstream parameters and CTA. Displayed data are not real and are shown as an example.

a	id $=SRB_i$	down $=SRB_{i+1}$	up1 $=SRB_{i-1}$	up2 $=SRB_{i-2}$	**tWC_i (m^3)**	**RO_i (m^3)**	**p (area, population)**	$uptWC_i$	WA_i	CTA
5	10	0	20	30	10	50		50	80	0.625
4	20	10	0	0	10	50		10	10	1
3	30	10	40	50	10	100		30	60	0.5
2	40	30	0	0	10	100		10	20	0.5
1	50	30	0	0	10	100		10	20	0.5

B.3.3. CF_{WD} calculations

From the Figure B-2 data, the following routine (Figure B-6) is applied to calculate CF_{WD}.

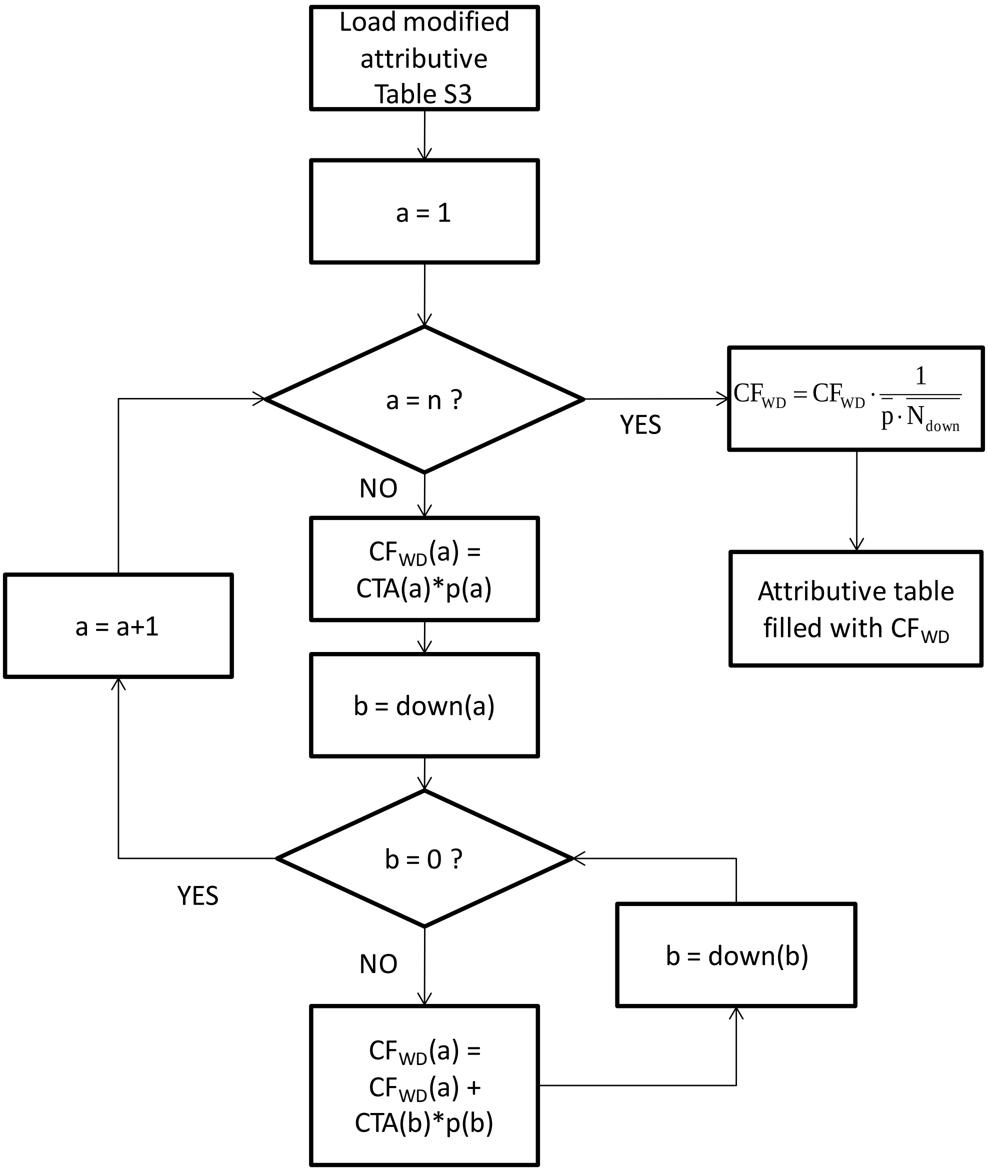


Figure B-6. Routine that calculates CF_{WD} from downstream parameters

B.4. Raw data and results for the Seine and the Guadalquivir river basins

Table B-3. Data and results for the Seine river-basin

id	tWC_i	RO_i	D_i	WA_i	$Area_i$	Pop_i	Volume	CTA_i	$CF_{WD,i}$	$CF_{WD,i}$	$CF_{WD,i}$
	hm^3/yr	hm^3/yr	hm^3/yr	hm^3/yr	km^2	hab	hm^3		p=area	p=population	p=water volume
10	21	598	17128	3607	2920	755935	15.67	0.251	0.034	0.045	0.206
20	67	481	481	109	5875	606219	0.36	0.615	0.200	0.135	0.217
30	47	321	16049	3373	3341	1049897	16.37	0.242	0.071	0.106	0.413
41	58	479	6508	1335	4292	1035695	5.91	0.126	0.096	0.138	0.452
42	43	1050	1050	219	5786	451303	0.75	0.195	0.148	0.159	0.460
43	39	932	4978	1009	5219	477129	6.75	0.067	0.112	0.145	0.476
44	4	566	566	114	2721	38583	0.17	0.031	0.116	0.146	0.476
45	6	674	3481	701	1998	195184	1.77	0.035	0.115	0.147	0.479
46	6	766	766	154	2291	240409	0.18	0.042	0.120	0.149	0.480
47	0.5	71	2041	411	235	6424	0.44	0.030	0.115	0.147	0.480
48	8	904	904	182	3277	184326	0.51	0.041	0.122	0.149	0.481
49	4	1066	1066	214	3595	125556	0.96	0.020	0.119	0.148	0.481
50	38	18	9220	1964	277	1159787	2.07	0.307	0.075	0.191	0.447
60	101	2176	2176	455	12385	1683128	4.19	0.221	0.201	0.281	0.495
70	365	985	7026	1498	10297	7702009	10.87	0.310	0.222	0.762	0.623
80	65	2537	2537	520	10585	414182	3.47	0.125	0.282	0.774	0.645
90	34	3505	3505	708	10974	418085	5.26	0.048	0.246	0.767	0.636

Table B-4. Data and results for the Guadalquivir river-basin

id	tWC_i	RO_i	D_i	WA_1	$Area_i$	Pop_i	Volume	CTA_i	$CF_{WD,i}$	$CF_{WD,i}$	$CF_{WD,i}$
	hm^3/yr	hm^3/yr	hm^3/yr	hm^3/yr	km^2	hab	hm^3		p=area	p=population	p=water volume
41	369	129	8889	2562	2159	532758	2.97	1.53	0.169	0.612	0.449
42	191	856	856	209	2752	552924	0.41	0.91	0.294	0.991	0.487
43	77	17	7904	2253	260	84397	0.96	1.49	0.189	0.706	0.591
44	168	1150	1150	264	3083	203093	0.45	0.64	0.286	0.803	0.620
45	559	2118	6737	1971	6999	184664	10.62	1.58	0.760	0.925	2.253
46	644	1310	1310	391	8363	915249	1.47	1.65	1.481	2.056	2.492
47	922	1956	3309	1045	17705	952451	9.84	1.83	2.424	2.234	4.037
48	273	509	509	157	5232	94280	0.44	1.75	2.896	2.358	4.113
49	720	844	844	313	9941	204959	1.07	2.30	3.615	2.588	4.281

B.5. Comparison with other water scarcity indicators found in the literature

Table B-5. Water deprivation/scarcity indicators calculated for two river basins using different available methods

	Seine	**Guadalquivir**
Yearly CF_{WD} at sub-river basin scale (m^3 equivalent/m^3)		
1 -Sub-river basin CF_{WD} p=area: average (min - max) [Paper data]	0.18 (0.03 – 0.28)	1.89 (0.16 – 3.46)
1bis -Sub-river basin CF_{WD} p=population: average (min - max) [Paper data]	0.47 (0.04 – 0.77)	1.62 (0.61 – 2.59)
1ter -Sub-river basin CF_{WD} p=volume: average (min - max) [Paper data]	0.44 (0.20 – 0.64)	2.70 (0.45 – 4.28)
Yearly CTA (or WTA) at river basin scale (m^3 equivalent/m^3)		
2- River basin CTA [Paper data or Hoekstra et al. (2012)]	0.25	1.58
3- River basin WTA [WaterGap2: Alcamo et al. (2003)] – used by Water Stress Index (WSI) of Pfister et al. (2009)	0.52	1.37
4- River basin WTA [Smakhtin et al. (2004)] – used by Water Stress Indicator (WSI) of (Milà i Canals et al., 2008)	0.53	1.77
5- River basin CTA – used by "Water Use Scarcity" (Frischknecht et al., 2006)	0.58	1.50
Average of monthly CTA at river basin scale (m^3 equivalent/m^3)		
6- River basin CTA (Hoekstra et al., 2012)	0.83	2.38

Table B-5 provides a comparison between the different methods used to assess water scarcity at the river basin scale. Mean CF_{WD} values, as well as river basin scale CTA values, were calculated with this paper data. In both river basins, the difference between the minimum and maximum values of the CF_{WD} is greater than one order of magnitude. This significant variation underscores the interest and need for taking into account downstream effects. CF_{WD} values at the SRB scale have a high variation in both examples and thus provide useful information that cannot be shown by previous indicators. It should be noted that the yearly CTA (line 2) is different from the arithmetic average of monthly CTA values (line 6), even though the same databases (water consumption (Hoekstra et al., 2012) and runoff (Fekete, 2002)) were used in both cases. The ratio of the yearly values is not equal to the average of the monthly ratios.

B.6. Adaptation of the framework to existing endpoint CFs (from Hanafiah et al.)

Hanafiah et al. (2011) developed following endpoint CF for water use impacts on freshwater species richness as following:

$$CF_{endpoint,river} = \frac{0{,}4}{D_{mouth,river}} \cdot V_{river} \qquad \textbf{(S1)}$$

Where $CF_{endpoint,\ river}$ is expressed in PDF.m^3.yr.m^{-3}, $D_{mouth,\ river}$ is the discharge (m^3.yr^{-1}) of the river at the mouth, $V_{river\ basin}$ is the estimated volume (m^3) of the river.

Using our framework, this CF can be adapted at the sub-river basin scale as per eq S2:

$$CF_{endpoint,i} = \sum_{j=i}^{n} \frac{0{,}4}{D_j} \cdot V_j \qquad \textbf{(S2)}$$

where D_j is the discharge (m^3.yr^{-1}) of the SRB_j, V_j is the estimated volume (m^3) of the SRB_j and D_j is the discharge (m^3.yr^{-1}) of the SRB_j. The CF is the sum of effects on downstream SRBs. Each effect on SRB is calculated with eq S1 adapted according data at the SRB scale. We can note that no weighted parameter is needed since volume is already taken into account in the effect factor.

Hanafiah et al. calculated the volume of a river as following:

$$V_{river} = 0.47 \cdot \left(\frac{D_{mouth,river}}{2} \right)^{0.90} \cdot L_{river} \qquad \textbf{(S3)}$$

where V_{river} is in m^3, $D_{mouth,\ river}$ is in m^3.s^{-1} and L_{river} is the length of the river within the river basin (m).

This volume calculation is adapted to a SRB. D/2 is supposed to be the mean discharge within a river basin. At the sub-river basin scale, this hypothesis is adapated: the mean discharge would be the upstream runoff plus half of the local runoff. Eq S4 shows the evaluation of the volume within a SRB_i.

$$V_i = 0.47 \cdot \left(D_{i-1} + \frac{RO_i}{2} \right)^{0.90} \cdot L_i \qquad \textbf{(S4)}$$

Where V_i is the volume of the rivers contained in SRB_i (m^3) , D_{i-1} is the water discharge coming in SRB_i ($m^3.s^{-1}$), RO_i is the local runoff within SRB_i ($m^3.s^{-1}$) and L_i is the length of the rivers which are within SRB_i. Length of the streams come from the Hydro1k database (U.S. Geological Survey Center for Earth Resources Observation and Science, 2004).

$CF_{endpoint}$ were calculated for each sub-river basin of the Guadalquivir river basin with eq S2 and $CF_{endpoint}$ for the entire river basin with eq S1. We consider that the estimated volume of the river basin is the sum of the estimated volume of the streams in each sub-river basin. This is different from Hanafiah et al. who only takes one stream of a river basin in their assessment.

Results are shown in Figure B-7. $CF_{endpoint}$ are obviously decreasing in downstream locations. The difference between minimum and maximum values are greater than one order of magnitude. The average of SRB $CF_{endpoint}$ (0.0011 $PDF.m^3.yr.m^{-3}$) is similar to the river basin $CF_{endpoint}$ (0.0013 $PDF.m^3.yr.m^{-3}$).

id	$CF_{endpoint}$ ($PDF.m^3.yr.m^{-3}$)
41	0.000134
42	0.000327
43	0.000182
44	0.000340
45	0.000812
46	0.001261
47	0.002001
48	0.002346
49	0.002510

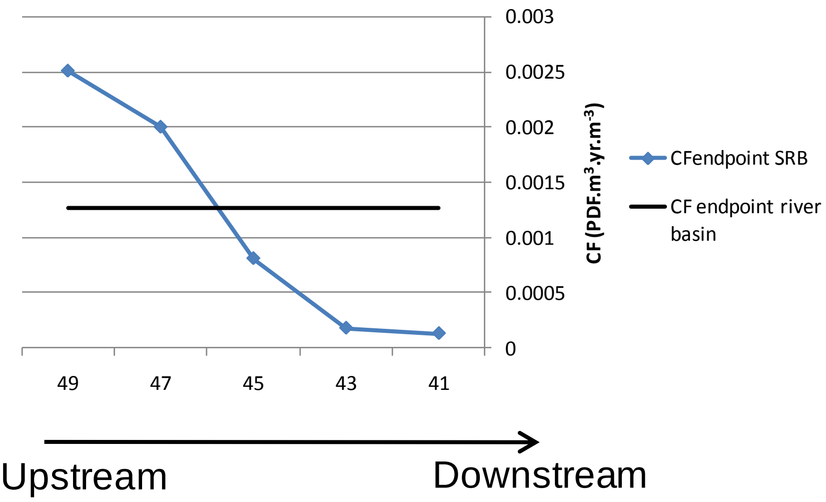


Figure B-7. CF for water consumption on freshwater fish species calculated at the SRB scale for the Guadalquivir river basin.

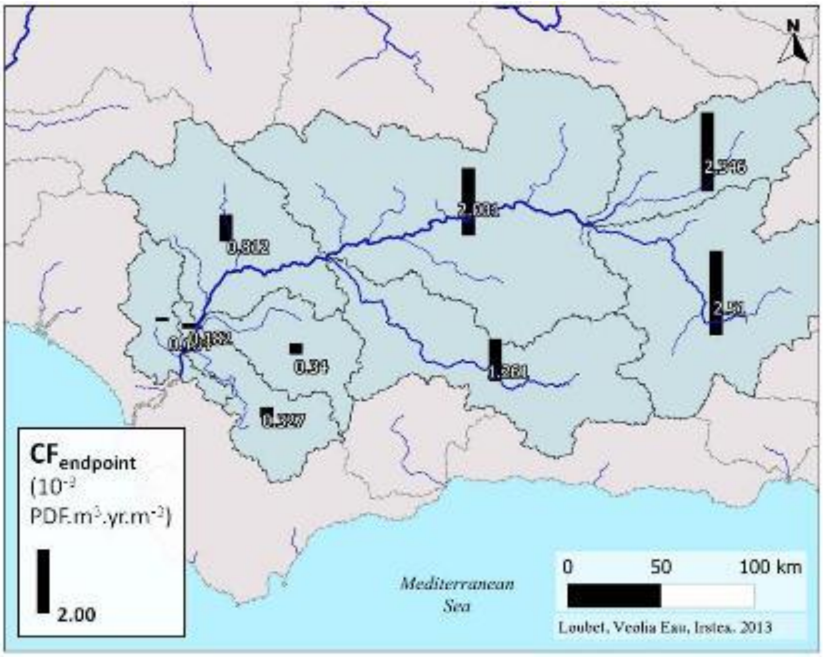


Figure B-8. Sub-river basins endpoint CF for water consumption on freshwater species of the Guadalquivir river basin (Spain)

This example is only a first approach and further work is needed to adapt river basin equations at the sub-river basin scale. However, it shows that the methodology can be adapted for calculating endpoint CFs based on existing methodology.

Annex C. WaLA, a versatile model for the life cycle assessment of urban water systems

This annex corresponds to the Supplementary Material of the publication presented in Chapter 5 and 6 and submitted for publication in "Water Research" as part 1 and part 2.

C.1. Structure of the Matlab/Simulink computer program

C.1.1. General structure of the program

The WaLA model is run through a Matlab program called “MAIN.m” that runs the following tasks:

- 1. A Matlab window is opened where the practitioner can load or create a UWS scenario file (e.g., ecoinvent.mdl) as shown in Figure C-2.
- 2. Once this is done, the model and the library of components are opened in a Simulink window, as explained in section C.1.2. The structure of the library and its associated components are developed in section C.1.3.
- 3. Through the Simulink interface, the practitioner select and combine the different water technologies and users components (drag and drop of the components from the library to the model window). The extrinsic attributes of each component can be customized by the user through the graphical mask of each component. Once the UWS model is ready, the practitioner can run the simulation from the Matlab interface (Figure C-2).
- 5. The model is initialized thanks to the function “ini_model.m”, as explained in section C.1.4.
- 6. The model is run on Simulink with 12 steps of time representing the 12 months
- 7. Results of the model are stored in the Matlab workspace as matrixes results and rearranged thanks to a Matlab script “Formating_Results.m”

This procedure is summarized in Figure C-1. The different scripts, functions and Simulink libraries are not provided here but can be provided upon request to the author.

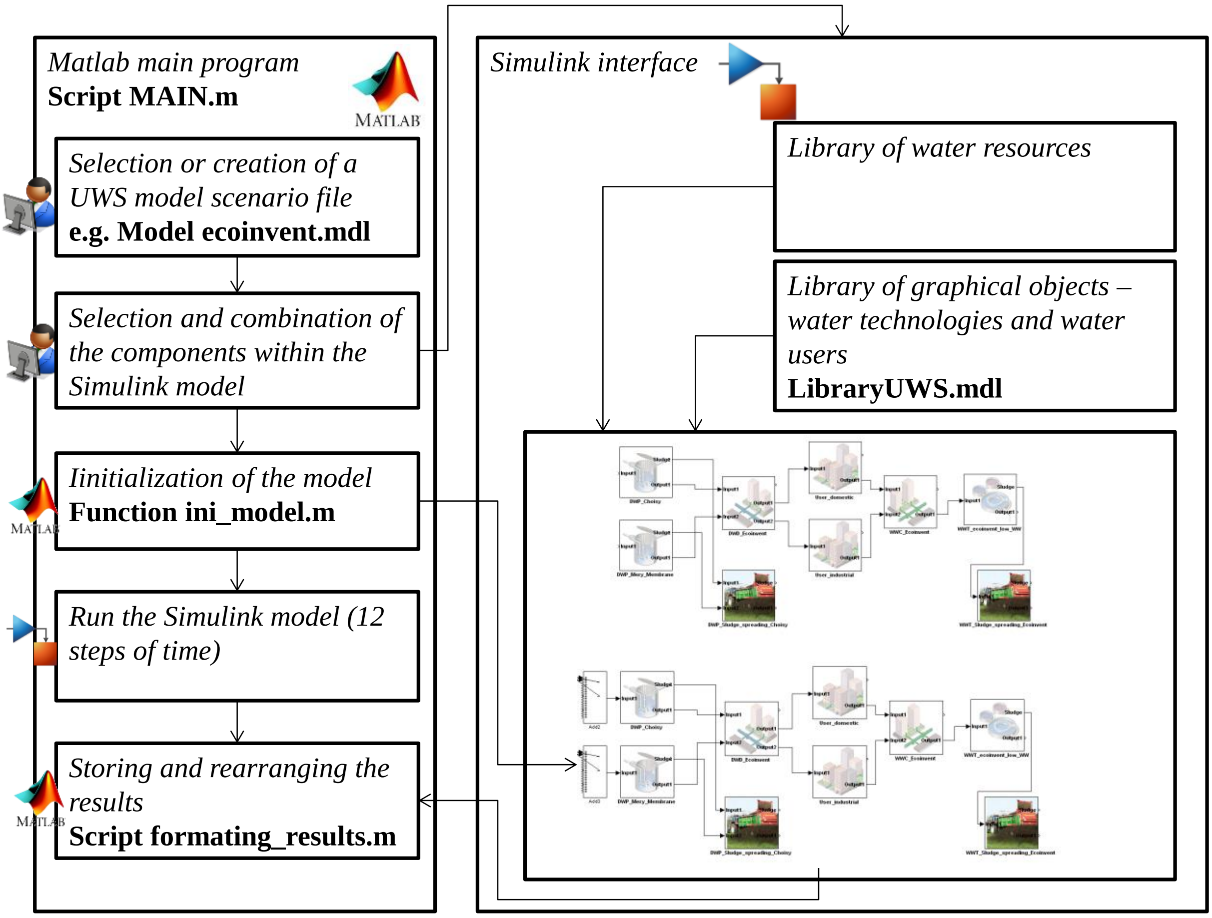


Figure C-1. Schematic explanation of the Matlab/Simulink tool. Icons depict who/which runs the different tasks: either the practitioner, Matlab or Simulink.

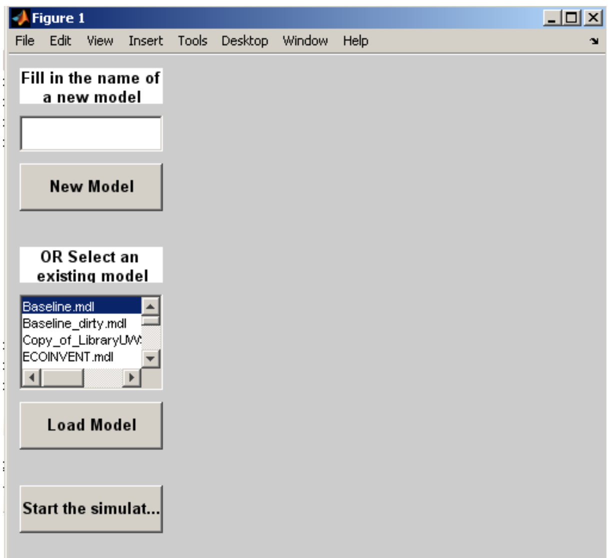


Figure C-2. Matlab interface that enables to select or create a new model

C.1.2. Structure of the Simulink library for UWS components

The components are stored within a library (LibraryUWS.mdl) with different folders for each sub-class of components (e.g., DWP, DWD, etc.).

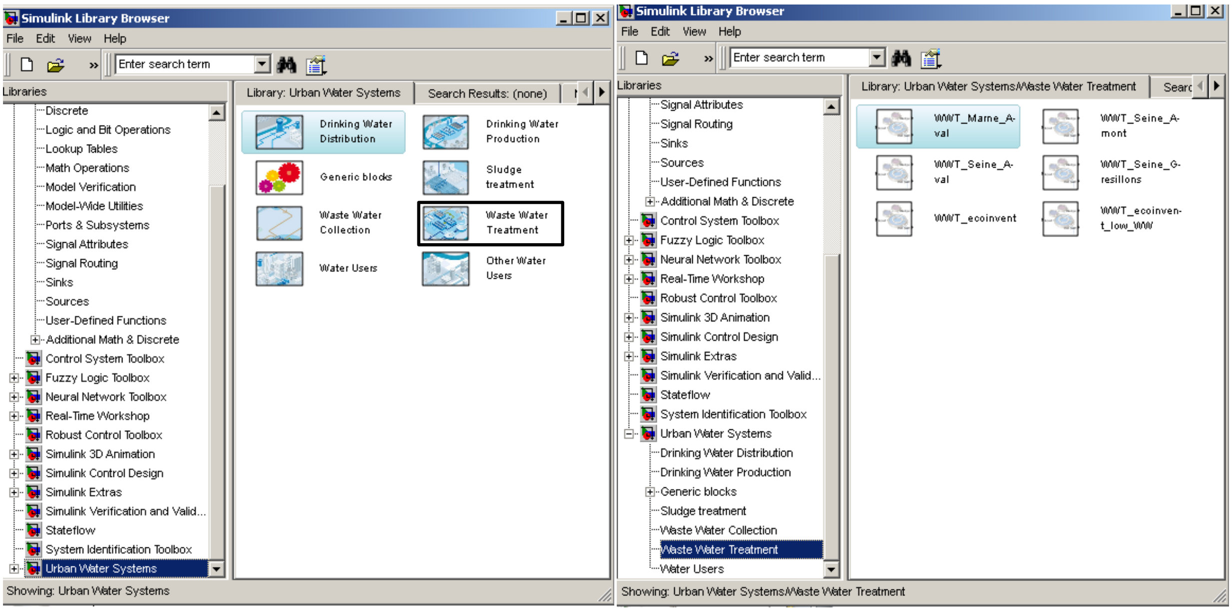


Figure C-3. Library of components

C.1.3. Structure of Simulink graphical objects

We describe here the structure of the graphical objects. The graphical object is a Simulink subsystem composed of built-in objects that represent the methods (i.e., functions: calculator, dispatcher, adder) and the intrinsic attributes of the components (Figure C-4). The extrinsic attributes related to each component are defined within the masking of the block where the user can select the parameters through an interface (Figure C-5). This structure is the same for all graphical objects representing technologies and users.

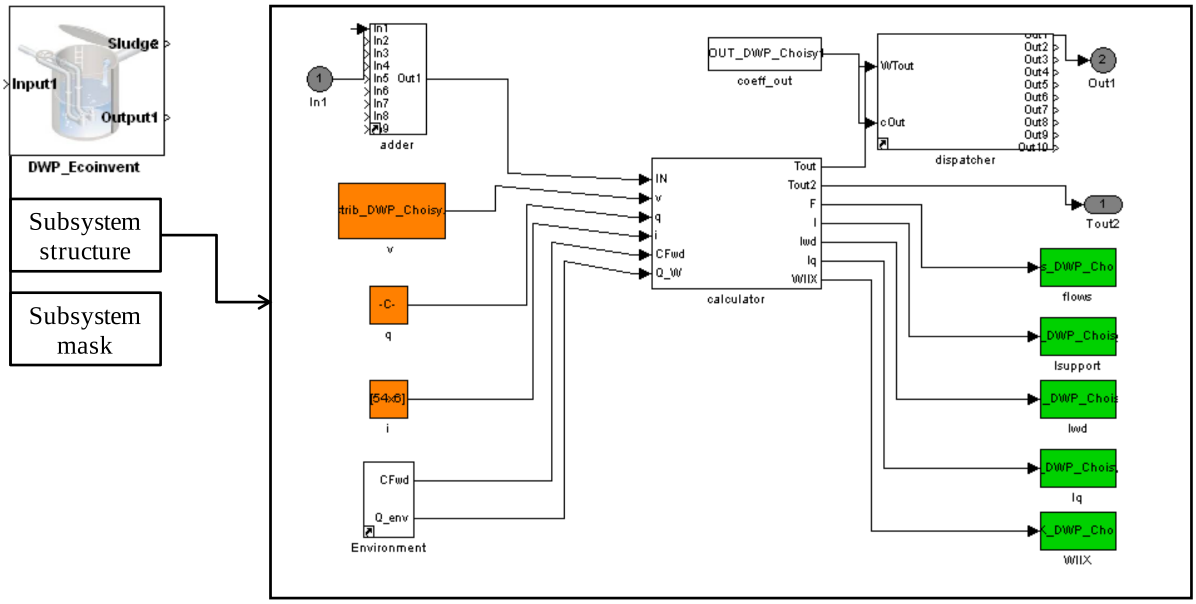


Figure C-4. Content of a generic block representing a technology or a user in Simulink. Methods are white blocks, intrinsic attributes are defined in orange blocks, and results are stored in green blocks.

- Methods (i.e., functions) are stored in "Embedded matlab function" blocks (EMF) represented in white (calculator, adder, dispatcher). EMF blocks are built from Matlab scripts.
- Intrinsic attributes are stored in databases blocks represented in orange (**v**, **q**, **i**). These blocks are either "Constant" for parameters that do not change within the year (q and i) or "From file" that refer to a time series matrix with monthly value (for the parameter v).The Environment block includes parameters (CF_{WD}, Q) of the different water resources within the local environment.
- Results (V, I, WIIX) are stored in vectors thanks to "To workspace" blocks, in green. Three kind of signals enter and leave the block and are represented by the three grey ports: "In" (Technosphere in or withdrawal depending on the technology), "Out1" (liquid technosphere out), "Out2" (sludge technosphere out). There can be several In et Out signals entering and leaving the blocks, that are managed with adder and dispatcher methods.
- Extrinsic parameters of the block are defined by the practitioner through the "Function block parameter" designed using the "mask" editor of Simulink. The interface is shown for a component in Figure C-5. The practitioner can select the number of inputs/outputs as well as the volumetric share between inputs and outputs. The connection to resources is also done within this interface (Location in and out). The practitioner also can select whether or not he wants to inform monthly data. The

dynamic management of ports at the input and the output of the block is done through the use of an initialization function built as a Matlab function and named "ini_block".

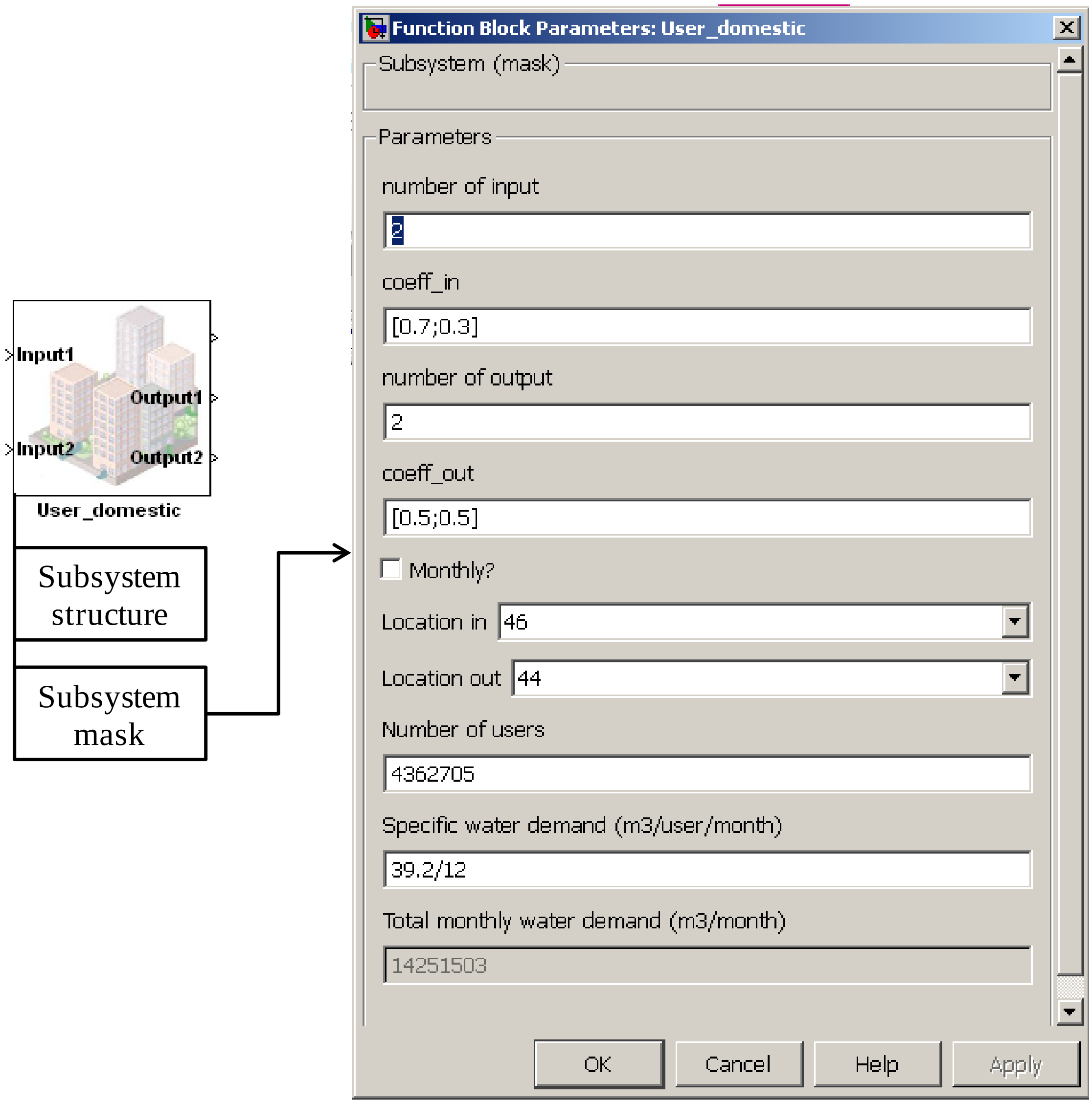


Figure C-5. Function block parameter of a User block

C.1.4. Initialization of the model

When the graphical construction of the model using Simulink interface is completed, it has to be initialized in order to calculate the water withdrawals from DWP for each month, i.e., the variables that runs the model. This initialization is run through a Matlab script that looks up at all connections of the Simulink model. This script first get water demand from all water users, then ascend the graph to calculate how much water each DWP plant must withdraw to satisfy this water demand. This is done with a loop programmed in a Matlab function "ini_model.m".

C.2. Description of components attributes: sub-classes and their instances

Following sub-sections C.2.1 to C.2.6 aim at describing volumetric water flow distribution vector **v**, quality distribution **q**, and sources of data for LCI of operation and infrastructures (for impacts matrix **i**) of the main components found in UWS. One form is provided for each generic sub-class defined in Chapter 5: drinking water production, drinking water distribution, user, stormwater collection, wastewater collection, wastewater treatment. Specific instances (i.e., objects) from the sub-classes are also developed. Figure C-6 summarizes the main instances available.

Each following sub-section includes:

- A schematic representation of the flows going in and out the technology/user per m^3
- A description of the flows.
- The calculations needed for estimating flows
- A table summarizing attributes **v** and **q** for the generic sub-class and for specific instances (ecoinvent and case study), as well as source of data for computing impacts matrixes **i** for each instance.

The different flows going in and out are characterized for $1m^3$ at the input of the technology (i.e., V_W = 1 m^3 for drinking water production, V_P = 1 m^3 for stormwater collection and V_Tin = 1 m^3 for all others technologies). Thus they refer to the parameter **v** introduced in Chapter 5. Water flow distribution **v** can be estimated from: (i) measurement from flow meter (V, m^3), (ii) calculation from an external model, (iii) literature data, (iv) result of a mass balance when all other flows are known. Quality of these flow refer to the indices introduced in Chapter 4. The chemical composition of each index is provided in section C.3.

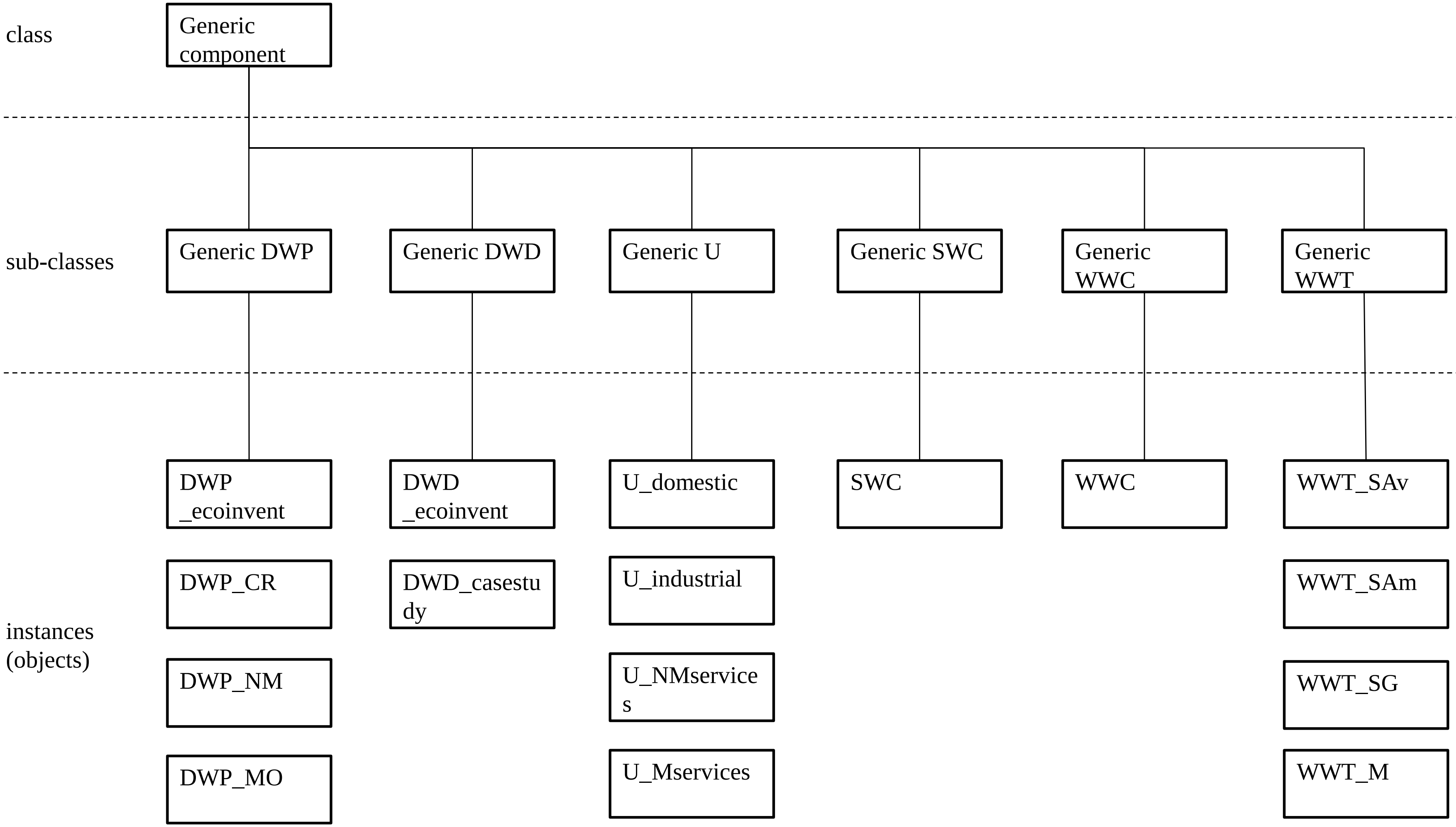
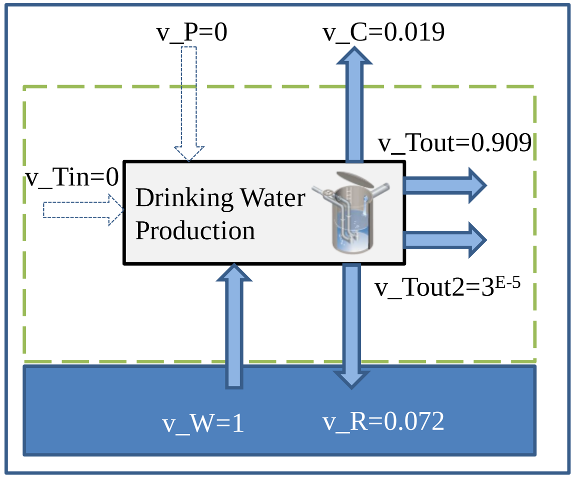


Figure C-6. Declination of the unique class (superclass) associated with the generic component, into sub-classes associated with each technology/user component, and into instances of each sub-class associated with the specific components. (note: practitioners can customize any instances or create any new ones from sub-classes)

C.2.1. Sub-class: drinking water production (DWP)



DWP_CR

Description

DWP plant usually withdraws (W) water directly from the environment (surface, ground or seawater). However, water can be conveyed to the plants through aqueducts. In this specific case, water going in DWP comes from the technosphere (Tin). Precipitation is not considered for this process since they directly runoff to the river. Water releases (R) to the environment (river or sea water) are dependent on the process (conventional, membrane) and include backwash water from filters and membranes, overflows and reservoirs washing. Evaporation (C) occurs in reservoirs and during sludge drying. Two flows go out to the technosphere: Tout that is the drinking water produced and Tout2 that is the water incorporated in sludge.

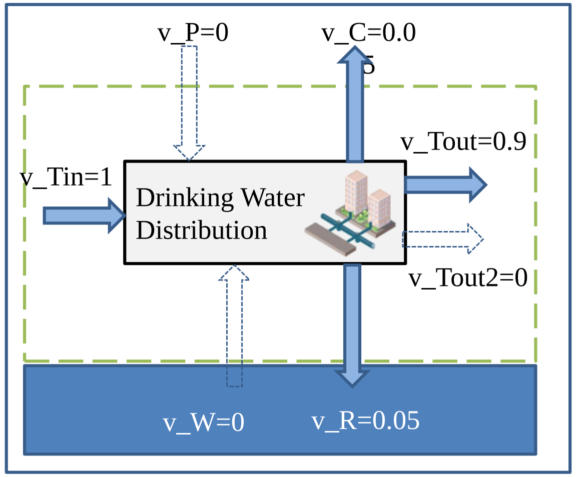
Estimation of water flows (models and/or data from literature)

v_Tout and v_R are obtained from local measurement of yearly water flows (V_Tout, V_W V_R). v_Tout2 is computed from the sludge dry content d and its total mass m_s (kg) ($v\_Tout2 = m_s \cdot (1-d_S)/V\_W$). v_C is computed from mass balance of water flows.

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class DWP		Specific instances (objects) available							
				DWP_ecoinvent		DWP_CR (conventional)				DWP_MO (membrane)	
Volume and quality	Flow name	v_DWP	q_DWP	v_DWP-ecoinvent	q_DWP-ecoinvent	v_DWP-CR	q_DWP-CR	v_DWP-NM	q_DWP-NM	v_DWP-membrane	q_DWP-membrane
	T_{in}	0	-	0	-	0	-		-	0	-
	T_{out}	V_Tout/V_W	A	0.933	A1	0.909	A1		A1	0.798	A1
	T_{out2}	m_S*(1-S)/V_W	E	0	E1	3.18E-5	E1		E1	3.25E-5	E1
	W	V_W/V_W	B	1	B1	1	B2		B5	1	B6
	R	V_R/V_W	C	0.057	C1	0.072	C3		C2	0.198	C4
	P	0	-	0	-	0	-		-	0	-
	C	v_W-(v_Tout+v_Tout2+v_R)	-	0.01	-	0.019	-		-	0.003	-
Impacts	operation	-		"tap water, at user {CH}| tap water production and supply"		Based on local data from Choisy-le-Roi plant		Based on local data from Neuilly-sur-Marne plant		Based on local data from Mery-sur-Oise plant	
	infra.	-		"water works {CH}| construction"							

C.2.2. Sub-class: drinking water distribution (DWD)



DWD_casestudy90

Description

DWD transfers water within the technosphere. Input comes from the technosphere (Tin). Largest part of the input goes out to the technosphere (Tout). Water losses either drain to local environment (release, R), drain away via the wastewater collection system (to technosphere, Tout), or is intercepted and used by vegetation (evaporation, C) (Mitchell et al. 2001).

Estimation of water flows (models and/or data from literature)

Tout is computed from the network performance η. $v\_Tout = \eta$

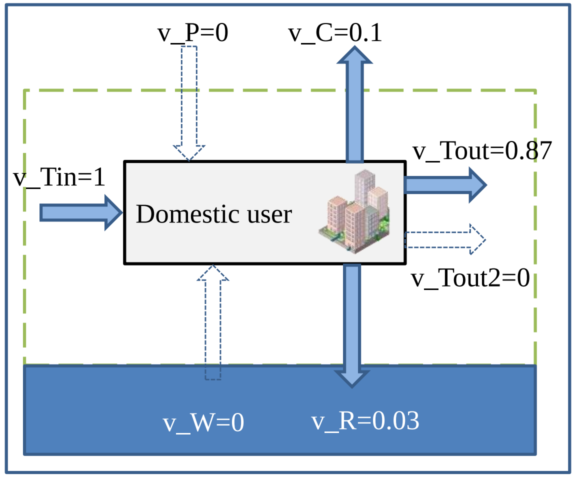
Part of losses that returns to the local environment and (noted α) and part that is evaporated (noted β) depend on many parameters (climate, nature of soil, urbanization, etc.) and is complex to estimate. Ecoinvent estimates that 85% of the losses come back to the environment whereas 15% of the losses are evaporated without citing any reference. In the case study, we chose a more conservative assumption (50% evaporated and 50% run-off).

$$v\_R = \alpha(1-\eta) \quad \text{and} \quad v\_C = \beta(1-\eta) \qquad \text{where} \qquad \alpha + \beta = 1$$

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class DWP		Specific instances (objects) available					
				DWD_ecoinvent		DWD_casestudy90		DWD_casestudy95	
Volume and quality	Flow name	v_DWD	q_DWD	v_DWD-ecoinvent	q_DWD-ecoinvent	v_DWD_casestudy90	q_DWD_casestudy90	v_DWD_casestudy95	q_DWD_casestudy95
	T_{in}	V_Tin/V_Tin=1	-	1	-	1	-	1	-
	T_{out}	η	A	0.95	A1	0.9	A1	0.95	A1
	T_{out2}	0	-	0	-	0	-	0	-
	W	0	-	0	-	0	-	0	-
	R	α(1- η)	A	0.043	A1	0.05	A1	0.025	A1
	P	0	-	0	-	0	-	0	-
	C	β(1- η)	-	0.007	-	0.05	-	0.025	-
Impacts	operation	-		"tap water, at user {CH}| tap water production and supply"		Based on local data from SEDIF		Based on local data from SEDIF	
	infra.	-		"water supply network {GLO}"		Based on ecoinvent ("water supply network {GLO}")		Based on ecoinvent ("water supply network {GLO}")	

C.2.3. Sub-class: user (U)



U_domestic

Description

Most of water supplied to water users is released as wastewater, which is collected and stays in the technosphere (Tout). The other part which is not collected (watering, car washing, etc.) is either released to the local environment (R) or evaporated (C). It largely depends on the type of housing (apartment, house with or without garden and swimming pool, etc.).

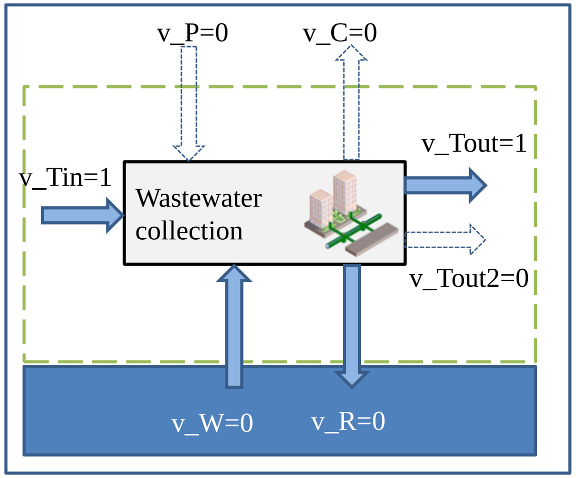
Estimation of water flows (models and/or data from literature)

Evaporation (v_C) and release (v_R) are estimated from the literature.

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class DWP		Specific instances (objects) available	
				U_domestic	
Volume and quality	Flow name	v_DWD	q_DWD	v_U_domestic	q_U_domestic
	T_{in}	1	-	1	-
	T_{out}	v_Tin – v_R – v_C	D	0.87	D1
	T_{out2}	0	-	0	-
	W	0	-	0	-
	R	v_R	A	0.03	A1
	P	0	-	0	-
	C	v_C	-	0.10	-
Impacts	operation	-		-	
	infra.	-		-	

C.2.4. Sub-class: stormwater collection (SWC)



Description

SWC collects precipitation (P) and outputs water in the technosphere (Tout) to a combined sewer system or a retention basin. Evapo(transpi)ration (C) related to stormwater system should follow the framework for green water in LCI (Núñez et al., 2013b). In this case, we need to consider the net change in the evapo(transpi)ration of the stormwater collection system compared to a reference situation. The identification of the reference situation is complex: it is more relevant to consider as a reference city without stormwater collection than natural vegetation. Indeed urbanization increases runoff and decreases evapotranspiration compared to natural vegetation (Haase, 2009), and this would lead to a benefit from the city system compared to vegetation if the time scale considered is one year. However, this benefit is a bias: rainwater runoff actually increases occurrence of flood whereas natural vegetation enables to stock water in soil for the dry season. In the case of an urban area without stormwater collection, largest part of water would run-off to streams

Estimation of water flows (models and/or data from literature)

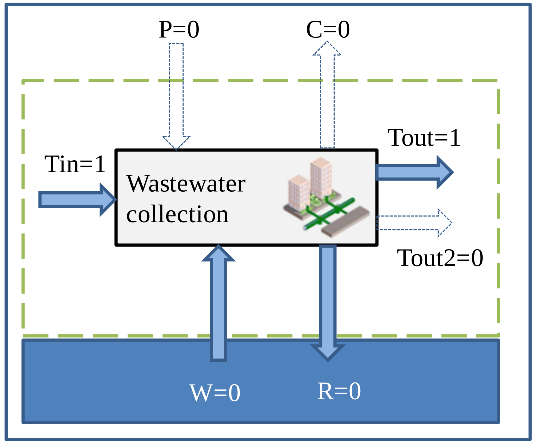
$$C_{net} = C_{system} - C_{ref}$$

$$W_{total} = W + (P - C_{net})$$

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class DWP		Specific instances (objects) available	
				SWC	
Volume and quality	Flow name	V_SWC	q_SWC	v_SWC	q_SWC
	T_{in}	0	-	0	-
	T_{out}	1	B	1	-
	T_{out2}	0	-	0	-
	W	0	-	0	-
	R	0	-	0	-
	P	1	B	1	-
	C	C_{system}-C_{ref}	-	0	-
Impacts	operation	-		-	
	infra.	-		-	

C.2.5. Sub-class: wastewater collection (WWC)



Description

Unitary sewers collect from the technosphere (Tin) wastewater from the users. In the case of combined sewers, rainwater collected from stormwater system also comes from the technosphere (Tin). In the case of draining system, part of water from the ground can infiltrate within the system, and it is considered as a withdrawal from the local environment (W). There is also exfiltration of wastewater from the system that is considered as release to the local environment (R). More rarely, water losses from DWD system can inflow within the sewers (Tin). Resulting wastewater (more or less diluted) is then transported to a WWT plant within the technosphere (Tout).

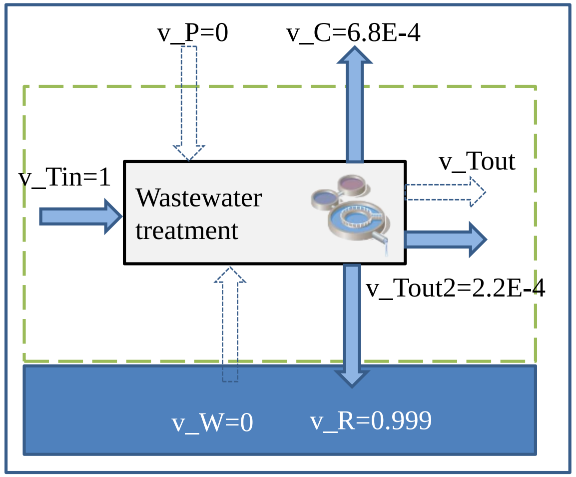
Estimation of water flows (models and/or data from literature)

Infiltration (In) and exfiltration (Ex) rates can be estimated according to literature values. In the case study, they were not considered.

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class WWC		Specific instances (objects) available			
				WWC_ecoinvent		WWC_casestudy	
Volume and quality	Flow name	v_DWD	q_DWD	v_DWD-ecoinvent	q_DWD-ecoinvent	v_WWC	q_WWC
	T_{in}	1	-	1	-	1	-
	T_{out}	v_Tin-v_W-v_R	D	1	D2	1	D3, D4, D5, D6
	T_{out2}	0	-	0	-	0	-
	W	In	-	0	-	0	-
	R	Ex	D	0	-	0	-
	P	0	-	0	-	0	-
	C	0	-	0	-	0	-
Impacts	operation	-		-		-	
	infra.	-		“water supply network {GLO}”		Based on ecoinvent (“water supply network {GLO}”)	

C.2.6. Sub-class: wastewater treatment (WWT)



WWT_SAv

Description

Wastewater plant treats water coming from collection system (Tin). Depending on the technology, part of water is evaporated (C) from basins (activated sludge) and/or from lagoons. Water from sludge is evaporated when dried (C), and the remaining amount is exported within the technosphere (Tout2) to incineration, agricultural spreading or landfill. Treated water is finally released to the local environment (R).

Estimation of water flows (models and/or data from literature)

Evapo(transpi)ration in WWT plants can be estimated according to Risch et al. (2014) that compute evaporation from open water surfaces in the activated sludge basins (E_{AS}, in m/year) and polishing ponds and evapotranspiration from planted vertical reed bed filters (ET_{plants}, in m/year), depending on the location. Total consumption (C) is from the surfaces of basins (S_{AS}, in m^2) and planted ponds (S_{plants}, in m^2).

$$v\_C = (E_{AS} \cdot S_{AS} + ET_{plants} \cdot S_{plants}) / V\_Tin$$

Amount exported in sludge is estimated according to the mass of sludge exported and its dryness. Finally release to local environment result from mass balance between inputs and outputs.

Attributes: volumetric water flow distribution v, quality distribution q, and sources of data for impacts i

		Generic sub-class WWT		Specific instances (objects) available									
				WWT_ecoinvent		WWT_SAv		WWT_SAm		WWT_SG		WWT_M	
	Flow name	v_WWT	q_WWT	v_WWT-ecoinvent	q_WWT-ecoinvent	v	q	v	q	v	q	v	q
Volume and quality	T_{in}	1	-	1	-	1	-	1	-	1	-	1	-
	T_{out}	0	-	0	-	0	-	0	-	0	-	0	-
	T_{out2}	m_S*(1-S)/V_W	E	1.85E-3	-	2.2E-4	-	2.5E-4	-	7.8E-4	-	0	-
	W	0	-	0	-	0	-	0	-	0	-	0	-
	R	1-v_$Tout_2$-v_C	C	0.9	C5	0.99	C6	0.99	C7	0.99	C8	0.99	C9
	P	0	-	0	-	0	-	0	-	0	-	0	-
	C	v_C	-	0.099	-	6.8E-4	-	1.2E-3	-	5.4E-4	-	1.1E-3	-
Impacts	operation	-		"wastewater, average {CH}| treatment of, capacity 4.7E10l/year effluent "		Based on local data from SIAAP							
	infra.	-		"wastewater treatment facility, capacity 4.7E10l/year"		Based on ecoinvent							

C.3. LCI for the quality indexes

In the sub-sections C.2.1 to C.2.6, vectors **q** are presented for each instance in order to give water quality indices of water at the output. Each water quality index (ex. A1, B1, etc.) refers to a type of water presented in Chapter 4. Chemical composition of each index is presented in Table C-1.

Table C-1. Composition of water flows from the case studies. Concentrations highlighted in grey are not known and taken equal to very good state

		A	B – Resources water						C – Releases from plants			
	Indiex	A1	B1	B2	B3	B4	B5	B6	C1	C2	C3	C4
	Description →	Drinking water SEDIF	Vanne river at Molinons	Seine river at Orly	Seine river at Paris	Seine river at poissy	Marne river at joinville	Oise river at Meriel	Release DWP_ ecoinvent	Release DWP_NM	Release DWP_CR	Release MO
Pollutants	CAS	(SEDIF, 2012)	(AESN, 2014)						(SEDIF, 2012)			
COD	-	3,50E-01	7,01E+00	1,64E+01	1,58E+01	1,77E+01	1,64E+01	1,86E+01	1,17E+01	1,17E+01	5,33E+01	2,50E+01
BOD	-	3,00E+00	9,63E-01	1,56E+00	1,36E+00	2,09E+00	1,60E+00	1,47E+00	3,00E+00	3,00E+00	3,00E+00	3,00E+00
Phosphorus total (Pt)	7723140	1,00E-02	4,00E-02	6,30E-02	1,06E-01	1,57E-01	8,56E-02	1,16E-01	5,00E-01	5,00E-01	5,00E-02	3,80E-01
Ion ammonium (NH4+)	14798039	3,00E-02	6,28E-02	7,15E-02	1,21E-01	8,75E-01	1,10E-01	1,32E-01	1,00E-01	1,00E-01	1,00E-01	8,60E-02
Nitrate (NO3-)	14797650	1,81E+01	2,52E+01	1,96E+01	2,02E+01	2,45E+01	1,67E+01	1,93E+01	3,18E+00	3,18E+00	2,00E+01	1,34E+01
Nitrite (NO2-)	14797650	1,00E-02	4,69E-02	7,81E-02	1,03E-01	5,16E-01	1,09E-01	1,12E-01	6,00E-02	6,00E-02	1,00E-01	1,00E-01
Cadmium (Cd)	7440439	1,00E-08	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05	7,50E-05
Mercury (Hg)	7439976	1,00E-08	5,00E-04	1,60E-05	2,50E-05	1,63E-05	1,68E-05	1,63E-05	2,50E-05	1,50E-04	2,50E-05	2,50E-05
Arsenic (As)	7440382	1,00E-08	1,74E-03	9,67E-04	2,10E-03	1,01E-03	8,98E-04	9,84E-04	2,10E-03	2,10E-03	2,10E-03	2,10E-03
Aluminum (Al)	7429905	1,00E-08	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,29E+00	1,36E+00	6,00E+00	5,10E-01
Iron (Fe)	7439896	1,00E-08	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	1,00E-01
Chromium (Cr)	7440473	1,00E-08	2,18E-03	5,83E-04	1,70E-03	7,49E-04	6,80E-04	9,32E-04	1,70E-03	1,70E-03	1,70E-03	1,70E-03
Copper (Cu)	7440508	1,00E-08	1,01E-03	1,46E-03	2,17E-03	2,25E-03	1,79E-03	1,58E-03	7,00E-04	3,82E-03	7,00E-04	7,00E-04
Lead (Pb)	7439921	1,00E-08	3,60E-03	3,60E-03	3,60E-03	3,60E-03	3,60E-03	3,60E-03	3,60E-03	8,10E-04	3,60E-03	3,60E-03
Zinc (Zn)	7440666	1,00E-08	4,89E-03	4,69E-03	3,90E-03	8,01E-03	5,22E-03	5,81E-03	3,90E-03	9,80E-03	3,90E-03	3,90E-03

Table C 1 continued. Composition of water flows from the case studies. Concentrations highlighted in grey are not known and taken equal to very good state

		C – Release from plants					D - Wastewater					
	Index	C5	C6	C7	C8	C9	D1	D2	D3	D4	D5	D6
	Description →	Release WWT_ecoinvent	Release WWT_SAv	Release WWT_SAm	Release WWT_SG	Release WWT_M	Raw waste water from France	Input WWT_ ecoinvent	Input WWT_SAv	Input WWT_SAm	Input WWT_SG	Input WWT_M
Pollutants	CAS	(SIAAP, 2012)			(Stricker and Héduit, 2010)			(Doka, 2009)			(SIAAP, 2012)	
COD	-	2,75E+01	5,50E+01	2,70E+01	2,60E+01	4,50E+01	2,00E+01	1,55E+02	4,58E+02	6,75E+02	3,71E+02	6,43E+02
BOD	-	8,15E+00	1,30E+01	3,00E+00	5,00E+00	7,90E+00	3,00E+00	1,04E+02	1,86E+02	3,00E+02	1,67E+02	2,63E+02
Phosphorus total (Pt)	7723140	8,49E-01	9,00E-01	7,00E-01	6,00E-01	5,00E-01	5,00E-02	3,07E+00	5,81E+00	7,00E+00	6,00E+00	7,14E+00
Ion ammonium (NH4+)	14798039	1,10E+01	9,51E+00	2,06E+00	2,31E+00	4,61E+00	1,00E-01	1,92E+01	1,00E-01	1,00E-01	1,00E-01	1,00E-01
Nitrate (NO3-)	14797650	4,83E+01	4,25E+01	8,10E+01	2,75E+01	2,88E+01	2,00E+01	4,65E+00	2,00E+01	2,00E+01	2,00E+01	2,00E+01
Nitrite (NO2-)	14797650	6,44E-01	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,31E+00	1,00E-01	1,00E-01	1,00E-01	1,00E-01
Cadmium (Cd)	7440439	2,81E-04	2,81E-04	7,50E-05	7,50E-05	7,50E-05	7,50E-05	2,81E-04	2,54E-04	2,54E-04	2,54E-04	2,54E-04
Mercury (Hg)	7439976	2,00E-04	2,00E-04	2,50E-05	2,50E-05	2,50E-05	2,50E-05	2,00E-04	5,36E-04	5,36E-04	5,36E-04	5,36E-04
Arsenic (As)	7440382	4,20E-03	4,20E-03	2,10E-03	2,10E-03	2,10E-03	2,10E-03	9,00E-04	1,49E-03	1,49E-03	1,49E-03	1,49E-03
Aluminum (Al)	7429905	1,04E+00	1,04E+00	1,00E-01	1,00E-01	1,00E-01	1,00E-01	1,04E+00	1,20E+00	1,20E+00	1,20E+00	1,20E+00
Iron (Fe)	7439896	7,09E+00	7,09E+00	5,00E-02	5,00E-02	5,00E-02	5,00E-02	7,09E+00	1,60E+00	1,60E+00	1,60E+00	1,60E+00
Chromium (Cr)	7440473	1,22E-02	1,22E-02	1,70E-03	1,70E-03	1,70E-03	1,70E-03	1,22E-02	1,35E-02	1,35E-02	1,35E-02	1,35E-02
Copper (Cu)	7440508	3,74E-02	3,74E-02	7,00E-04	7,00E-04	7,00E-04	7,00E-04	3,74E-02	8,49E-02	8,49E-02	8,49E-02	8,49E-02
Lead (Pb)	7439921	8,63E-03	8,63E-03	3,60E-03	3,60E-03	3,60E-03	3,60E-03	8,63E-03	2,31E-02	2,31E-02	2,31E-02	2,31E-02
Zinc (Zn)	7440666	3,24E-02	3,24E-02	3,90E-03	3,90E-03	3,90E-03	3,90E-03	1,09E-01	1,88E-01	1,88E-01	1,88E-01	1,88E-01

C.4. Water users database

Table C-2. Database of numbers of users in each city of the SEDIF perimeter for the year 2012. Source (INSEE, 2013)

Cities of the SEDIF perimeter	Post code	Area (ha)	Popu-lation (capita)	Non market services (jobs)	Market services (jobs)	Indus-tries (jobs)	Agri-culture (jobs)	Total
TOTAL		761.48	4 362 705	413 251	1 164 261	153 028	335	1 730 875
Brou-sur-Chantereine	77177	4.28	4 306	340	171	14	1	526
Villeparisis	77270	8.29	24 296	1 196	2 577	235	2	4 010
Vaires-sur-Marne	77360	6.02	12 459	463	1 048	248	0	1 759
Chelles	77500	15.90	53 238	3 379	7 439	1 169	3	11 990
Vélizy-Villacoublay	78140	8.93	20 348	2 085	25 229	13 659	0	40 973
Viroflay	78220	3.49	16 224	1 406	2 671	72	0	4 149
Jouy-en-Josas	78350	10.14	8 316	1 531	2 303	125	55	4 014
LesLoges-en-Josas	78351	2.48	1 596	148	584	115	0	847
Sartrouville	78500	8.46	51 504	2 908	5 764	1 286	0	9 958
LeMesnil-le-Roi	78600	3.25	6 543	309	657	33	37	1 036
Houilles	78800	4.43	31 849	1 548	2 369	209	1	4 127
Palaiseau	91120	11.51	31 175	4 177	6 067	840	0	11 084
Athis-Mons	91200	8.56	30 845	3 760	3 673	157	0	7 590
Juvisy-sur-Orge	91260	2.24	14 756	1 672	2 239	116	0	4 027
Massy	91300	9.43	43 006	4 684	16 132	4 957	0	25 773
Wissous	91320	9.11	5 965	271	11 297	939	1	12 508
Verrières-le-Buisson	91370	9.91	15 830	894	3 132	218	0	4 244
Igny	91430	3.82	10 878	487	1 558	311	1	2 357
Bièvres	91570	9.69	4 747	663	2 017	233	21	2 934
Boulogne-Billancourt	92100	6.17	115 264	9 887	75 787	3 833	11	89 518
Clichy	92110	3.08	59 228	7 108	32 918	3 533	1	43 560
Montrouge	92120	2.07	48 983	3 029	17 682	1 703	10	22 424
Issy-les-Moulineaux	92130	4.25	65 178	5 127	45 717	4 846	0	55 690
Clamart	92140	8.77	53 113	4 927	7 132	3 398	0	15 457
Antony	92160	9.56	62 644	6 189	14 080	2 027	1	22 297
Vanves	92170	1.56	27 314	2 699	5 344	129	0	8 172
Meudon	92190	9.90	45 834	4 259	13 604	1 801	0	19 664
Neuilly-sur-Seine	92200	3.73	62 565	7 778	39 242	2 243	0	49 263
Bagneux	92220	4.19	38 384	2 597	5 241	1 467	0	9 305
Malakoff	92240	2.07	31 325	1 999	12 106	282	0	14 387
Fontenay-aux-Roses	92260	2.51	23 603	1 871	4 862	47	0	6 780
Châtenay-Malabry	92290	6.38	32 573	3 762	2 871	81	6	6 720
Levallois-Perret	92300	2.41	64 757	5 377	53 701	4 501	0	63 579
Sèvres	92310	3.91	23 412	2 375	6 621	467	2	9 465
Châtillon	92320	2.92	32 947	2 037	10 965	956	0	13 958
Sceaux	92330	3.60	19 986	2 553	1 920	172	0	4 645
Bourg-la-Reine	92340	1.86	20 303	1 654	2 917	75	0	4 646
LePlessis-Robinson	92350	3.43	27 931	2 363	4 147	4 946	2	11 458
Chaville	92370	3.55	18 887	980	1 529	53	0	2 562
Puteaux	92800	3.19	45 093	8 412	80 859	7 005	19	96 295
Bobigny	93000	6.77	47 855	26 327	10 726	1 265	0	38 318

Cities of the SEDIF perimeter	Post code	Area (ha)	Popu-lation (capita)	Non market services (jobs)	Market services (jobs)	Indus-tries (jobs)	Agri-culture (jobs)	Total
Montreuil	93100	8.92	103 675	15 791	27 190	2 338	0	45 319
Rosny-sous-Bois	93110	5.91	41 431	3 338	9 125	868	0	13 331
LaCourneuve	93120	7.52	38 361	2 365	7 385	2 677	0	12 427
Noisy-le-Sec	93130	5.04	39 949	2 919	5 472	1 037	0	9 428
Bondy	93140	5.47	53 934	4 504	7 167	377	0	12 048
Noisy-le-Grand	93160	12.95	63 526	7 569	15 208	2 127	0	24 904
Bagnolet	93170	2.57	34 232	3 802	9 264	931	1	13 998
Livry-Gargan	93190	7.38	42 060	2 717	4 693	363	0	7 773
Saint-Denis	93200	12.36	107 959	19 488	52 167	7 110	7	78 772
Gagny	93220	6.83	39 350	2 942	1 861	119	0	4 922
Romainville	93230	3.44	26 025	1 714	2 888	1 019	0	5 621
Stains	93240	5.39	34 048	2 652	3 613	582	0	6 847
Villemomble	93250	4.04	28 257	1 811	2 538	182	0	4 531
LesLilas	93260	1.26	22 410	1 444	3 109	123	0	4 676
Sevran	93270	7.28	50 225	2 800	2 642	512	6	5 960
Aubervilliers	93300	5.76	76 728	6 208	19 301	1 722	0	27 231
LePré-Saint-Gervais	93310	0.70	18 171	969	1 668	236	0	2 873
LesPavillons-sous-Bois	93320	2.92	21 972	1 368	3 678	220	0	5 266
Neuilly-sur-Marne	93330	6.86	33 781	4 219	4 865	1 030	0	10 114
LeRaincy	93340	2.24	14 194	1 714	2 132	55	0	3 901
LeBourget	93350	2.08	14 943	1 187	4 090	354	0	5 631
Neuilly-Plaisance	93360	3.42	20 683	1 242	3 610	318	0	5 170
Montfermeil	93370	5.45	25 499	3 432	3 118	197	0	6 747
Pierrefitte-sur-Seine	93380	3.41	28 076	1 395	1 597	398	0	3 390
Clichy-sous-Bois	93390	3.95	29 998	1 371	1 891	196	0	3 458
Saint-Ouen	93400	4.31	47 604	3 555	30 481	5 626	0	39 662
Vaujours	93410	3.78	6 601	644	651	500	0	1 795
Villetaneuse	93430	2.31	12 662	2 049	1 757	195	0	4 001
Dugny	93440	3.89	10 735	644	359	127	0	1 130
L'Ile-Saint-Denis	93450	1.77	7 070	303	1 380	135	0	1 818
Gournay-sur-Marne	93460	1.68	6 457	278	469	49	0	796
Coubron	93470	4.14	4 795	207	359	18	0	584
Pantin	93500	5.01	54 464	6 264	16 905	3 133	0	26 302
Aulnay-sous-Bois	93600	16.20	82 778	6 131	13 933	5 454	0	25 518
Drancy	93700	7.76	67 202	3 681	6 175	540	1	10 397
Epinay-sur-Seine	93800	4.57	54 775	2 822	5 104	326	0	8 252
Arcueil	94110	2.33	19 964	1 773	14 677	707	2	17 159
Fontenay-sous-Bois	94120	5.58	53 667	4 547	22 141	1 415	7	28 110
Nogent-sur-Marne	94130	2.80	31 975	2 978	3 359	212	6	6 555
Alfortville	94140	3.67	44 439	2 099	6 323	864	0	9 286
Rungis	94150	4.20	5 729	688	21 788	3 357	10	25 843
Saint-Mandé	94160	0.92	22 666	3 106	2 389	213	0	5 708
LePerreux-sur-Marne	94170	3.96	32 799	1 689	2 797	190	0	4 676
Ivry-sur-Seine	94200	6.10	58 189	5 661	29 058	2 497	0	37 216
Charenton-le-Pont	94220	1.85	29 664	2 165	13 514	619	0	16 298
Cachan	94230	2.74	28 550	4 407	4 142	446	0	8 995

Cities of the SEDIF perimeter	Post code	Area (ha)	Popu-lation (capita)	Non market services (jobs)	Market services (jobs)	Indus-tries (jobs)	Agri-culture (jobs)	Total
L'Hay-les-Roses	94240	3.90	30 588	1 608	2 257	391	0	4 256
Gentilly	94250	1.18	17 222	2 043	5 096	562	0	7 701
Fresnes	94260	3.56	26 446	3 311	3 666	178	0	7 155
LeKremlin-Bicêtre	94270	1.54	26 267	7 197	5 723	365	0	13 285
Villeneuve-le-Roi	94290	8.40	18 568	1 238	4 225	1 024	0	6 487
Vincennes	94300	1.91	48 955	3 855	10 302	1 122	1	15 280
Orly	94310	6.69	21 691	2 841	17 232	505	0	20 578
Thiais	94320	6.43	29 949	1 983	9 082	301	0	11 366
Joinville-le-Pont	94340	2.30	17 990	1 092	1 751	376	0	3 219
Villiers-sur-Marne	94350	4.33	27 568	1 638	3 190	186	0	5 014
Bry-sur-Marne	94360	3.35	15 825	2 484	4 966	123	0	7 573
Vitry-sur-Seine	94400	11.67	86 210	5 714	15 985	3 407	11	25 117
Saint-Maurice	94410	1.43	14 647	3 858	2 132	151	0	6 141
Chennevières-sur-Marne	94430	5.27	18 227	1 040	4 235	776	0	6 051
Ablon-sur-Seine	94480	1.11	5 198	239	149	8	0	396
Champigny-sur-Marne	94500	11.30	76 235	4 969	9 828	1 905	0	16 702
Chevilly-Larue	94550	4.22	18 659	1 617	10 703	1 195	0	13 515
Choisy-le-Roi	94600	5.43	41 275	2 431	5 251	978	0	8 660
Maisons-Alfort	94700	5.35	53 513	3 124	9 800	1 781	0	14 705
Villejuif	94800	5.34	55 879	10 170	8 095	577	3	18 845
Argenteuil	95100	17.22	104 843	9 848	14 571	5 074	9	29 502
Sannois	95110	4.78	26 659	1 649	2 814	264	0	4 727
Ermont	95120	4.16	27 713	2 472	2 613	64	0	5 149
Franconville	95130	6.19	33 324	2 031	3 057	337	0	5 425
LePlessis-Bouchard	95131	2.69	7 812	319	663	218	1	1 201
Taverny	95150	10.48	26 440	2 053	3 362	1 143	0	6 558
Montmorency	95160	5.37	21 475	2 423	1 114	234	1	3 772
Deuil-la-Barre	95170	3.76	21 741	1 427	1 227	78	0	2 732
Sarcelles	95200	8.45	59 204	5 560	6 741	1 365	0	13 666
Saint-Gratien	95210	2.42	20 326	833	2 405	146	0	3 384
Herblay	95220	12.74	26 533	1 511	4 368	752	4	6 635
Soisy-sous-Montmorency	95230	3.98	17 670	842	3 033	111	1	3 987
Cormeilles-en-Parisis	95240	8.48	23 318	1 685	2 122	245	13	4 065
Beauchamp	95250	3.02	8 834	388	2 288	739	0	3 415
Saint-Leu-la-Forêt	95320	5.24	14 962	814	1 328	236	0	2 378
Domont	95330	8.33	15 075	1 168	1 867	291	1	3 327
Piscop	95350	4.08	778	37	147	24	9	217
SaintBrice-sous-Forêt	95351	6.00	14 487	532	2 121	130	0	2 783
Montmagny	95360	2.91	14 423	771	913	204	1	1 889
Montigny-lès-Cormeilles	95370	4.07	19 296	935	2 142	170	0	3 247
Saint-Prix	95390	7.93	7 464	401	431	52	1	885
Villiers-le-Bel	95400	7.30	27 004	2 917	1 191	243	8	4 359
Groslay	95410	2.95	8 601	365	699	64	14	1 142

Cities of the SEDIF perimeter	Post code	Area (ha)	Popu-lation (capita)	Non market services (jobs)	Market services (jobs)	Indus-tries (jobs)	Agri-culture (jobs)	Total
Auvers-sur-Oise	95430	12.69	6 953	223	339	108	8	678
Ecouen	95440	7.59	7 515	414	489	435	5	1 343
Pierrelaye	95480	9.21	8 122	352	2 479	216	12	3 059
LaFrette-sur-Seine	95530	2.02	4 621	114	170	25	0	309
Méry-sur-Oise	95540	11.17	9 410	443	1 159	217	14	1 833
Bessancourt	95550	6.39	7 090	346	234	55	3	638
Andilly	95580	2.70	2 570	356	745	235	1	1 337
Margency	95581	0.72	2 891	477	249	8	0	734
Eaubonne	95600	4.42	24 386	3 420	2 528	186	0	6 134
Montlignon	95680	2.84	2 685	236	288	365	0	889
Bezons	95870	4.16	28 277	2 163	12 838	1 494	0	16 495
Enghien-les-Bains	95880	1.77	11 959	1 391	2 927	109	0	4 427

Table C-3. Database of water volumes used and WWT plant connected to each city in the SEDIF perimeter for the year 2012. Sources: (SEDIF, 2012; SIAAP, 2012)

Cities of the SEDIF perimeter	WWT plant connected	Water use - total (m3)	Water use - Domestic (m3)	Water use - Non market services (m3)	Water use- Market services (m3)	Water use - Industries (m3)	Water use - Others (m3)
TOTAL		240 773 566	171 083 928	29 017 675	56 083 455	6 606 222	6 374 549
Brou-sur-Chantereine	MAv	200 057	151 213	29 714	47 102	256	1 406
Villeparisis	SAv	1 055 641	871 121	81 242	157 993	15 814	8 899
Vaires-sur-Marne	MAv	574 683	408 740	75 253	107 828	57 130	956
Chelles	MAv	2 468 016	1 886 696	327 784	518 317	21 692	32 874
Vélizy-Villacoublay	SAv	1 639 038	727 328	153 195	716 687	155 777	37 843
Viroflay	SAv	708 611	608 499	62 055	92 909	2 788	4 075
Jouy-en-Josas	SAm	518 591	270 795	124 742	228 820	1 622	15 566
Les Loges-en-Josas	SAm	93 714	74 617	6 962	16 185	728	2 184
Sartrouville	SAv	2 416 245	2 091 503	174 114	277 449	18 236	24 923
Le Mesnil-le-Roi	SAv	327 337	265 895	38 849	49 929	303	10 101
Houilles	SAv	1 393 932	1 183 758	97 831	191 101	9 461	9 026
Palaiseau	SAm	1 768 190	960 571	508 331	693 859	79 742	24 085
Athis-Mons	SAv	1 472 957	1 055 681	246 893	382 501	26 136	4 521
Juvisy-sur-Orge	SAm	799 813	593 756	100 913	185 506	6 998	11 717
Massy	SAm	2 537 858	1 640 583	256 278	649 012	142 262	57 826
Wissous	SAm	430 978	259 201	26 948	144 187	20 050	6 031
Verrières-le-Buisson	SAm	789 005	673 980	69 493	102 864	7 183	2 666
Igny	SAm	471 660	390 051	47 234	74 847	5 696	1 066
Bièvres	SAm	266 386	177 102	34 100	77 984	6 901	4 276
Boulogne-Billancourt	SAv	7 417 658	5 767 334	659 175	1 470 730	63 746	105 179
Clichy	SAv	3 909 549	2 322 507	494 353	1 288 762	173 169	122 245
Montrouge	SAv	2 662 926	2 068 572	226 616	535 411	14 486	35 152
Issy-les-Moulineaux	SAv	3 810 102	2 758 140	320 675	936 874	37 213	55 908
Clamart	SAv	2 952 341	1 765 828	615 328	822 547	347 140	11 619
Antony	SAv	3 338 945	2 372 719	539 595	842 261	66 132	46 706
Vanves	SAv	1 572 818	1 179 689	171 566	350 227	11 567	26 128
Meudon	SAv	2 494 060	1 950 640	288 543	484 208	36 910	20 838
Neuilly-sur-Seine	SAv	4 957 470	3 986 554	459 306	852 633	21 755	93 227
Bagneux	SAv	1 841 075	1 370 412	175 238	407 112	13 192	47 286
Malakoff	SAv	1 774 582	1 044 858	161 025	521 415	22 510	185 621
Fontenay-aux-Roses	SAv	1 236 716	867 256	123 832	183 559	174 809	11 038
Châtenay-Malabry	SAv	1 723 582	1 135 421	447 717	547 021	5 851	32 548
Levallois-Perret	SAv	4 431 928	2 896 836	300 442	834 061	24 775	664 022
Sèvres	SAv	1 287 450	994 445	195 089	254 208	21 479	16 952
Châtillon	SAv	1 842 245	1 219 896	257 510	592 298	16 184	10 548
Sceaux	SAv	1 091 802	808 624	158 089	208 818	4 188	69 282
Bourg-la-Reine	SAv	1 014 203	858 604	74 418	119 699	4 633	31 203
Le Plessis-Robinson	SAv	1 518 925	1 105 379	230 593	344 399	19 235	46 101
Chaville	SAv	921 876	804 353	67 925	108 759	1 401	5 856
Puteaux	SAv	3 930 420	1 901 061	485 822	1 734 203	18 504	275 014
Bobigny	SAv	2 879 857	1 795 471	542 037	968 866	62 350	43 189
Montreuil	SAv	5 784 086	4 142 436	497 934	1 163 896	97 714	352 734
Rosny-sous-Bois	MAv	2 309 396	1 538 912	236 557	736 482	11 024	21 876
La Courneuve	SAv	2 682 157	1 661 524	170 688	494 199	405 556	72 301

Cities of the SEDIF perimeter	WWT plant connected	Water use - total (m3)	Water use - Domestic (m3)	Water use - Non market services (m3)	Water use- Market services (m3)	Water use - Industries (m3)	Water use - Others (m3)
Noisy-le-Sec	SAv	2 006 649	1 646 275	167 824	307 765	34 234	15 731
Bondy	SAv	2 441 899	1 914 324	309 077	486 555	13 950	25 652
Noisy-le-Grand	MAv	3 312 035	2 491 839	248 883	585 340	162 297	61 976
Bagnolet	SAv	2 092 540	1 281 581	233 034	600 728	25 654	183 445
Livry-Gargan	SAv	1 941 148	1 567 493	186 420	331 028	12 219	25 440
Saint-Denis	SAv	6 995 314	4 295 105	900 362	2 347 419	204 372	112 296
Gagny	MAv	1 719 903	1 474 649	152 715	217 120	12 922	14 458
Romainville	SAv	1 381 657	989 599	95 546	229 723	130 885	26 327
Stains	SAv	1 880 518	1 515 859	188 443	299 747	47 222	17 392
Villemomble	MAv	1 389 776	1 117 077	133 670	235 649	8 189	26 830
Les Lilas	SAv	1 225 172	936 650	105 084	230 281	9 751	48 372
Sevran	Morée	2 221 340	1 788 252	246 179	389 067	22 240	15 291
Aubervilliers	SAv	4 799 518	3 485 810	414 599	1 112 379	101 364	93 196
Le Pré-Saint-Gervais	SAv	899 640	758 379	67 130	119 234	13 124	8 818
Les Pavillons-sous-Bois	SAv	1 005 769	838 186	67 033	135 519	16 099	14 782
Neuilly-sur-Marne	MAv	1 951 387	968 916	779 707	921 161	20 100	38 607
Le Raincy	MAv	757 630	584 438	120 147	153 325	14 733	4 778
Le Bourget	SAv	968 387	667 880	141 548	264 571	28 739	6 627
Neuilly-Plaisance	MAv	980 223	808 495	72 647	146 332	13 564	10 532
Montfermeil	MAv	1 164 256	857 068	247 459	292 687	2 004	11 590
Pierrefitte-sur-Seine	SAv	1 427 188	1 181 391	66 756	180 316	19 339	37 019
Clichy-sous-Bois	SAv	1 309 001	1 080 240	137 477	208 748	15 653	3 300
Saint-Ouen	SAv	3 154 866	2 087 140	244 633	830 874	104 516	125 970
Vaujours	Morée	334 001	237 393	32 377	59 436	30 841	6 331
Villetaneuse	SAv	723 734	444 301	166 754	246 362	7 845	24 734
Dugny	SAv	602 968	175 812	41 795	410 311	8 161	7 936
L'Ile-Saint-Denis	SAv	380 521	314 282	34 268	62 990	1 977	1 103
Gournay-sur-Marne	MAv	291 059	252 322	19 210	33 783	761	4 193
Coubron	MAv	195 462	174 812	8 694	19 437	1 138	0
Pantin	SAv	3 385 035	2 275 692	314 739	830 601	122 947	146 482
Aulnay-sous-Bois	Morée	4 294 164	2 985 217	456 969	1 115 945	141 341	44 751
Drancy	SAv	3 076 348	2 433 174	280 880	559 803	47 394	31 761
Epinay-sur-Seine	SAv	2 880 314	2 333 450	224 369	484 319	17 639	41 069
Arcueil	SAm	1 179 551	794 168	83 603	311 075	11 988	59 497
Fontenay-sous-Bois	SAm	3 011 774	1 956 552	370 234	747 088	228 231	77 303
Nogent-sur-Marne	SAm	1 762 839	1 374 134	238 718	321 635	15 340	48 513
Alfortville	SAm	2 256 148	1 848 393	140 039	333 486	26 387	42 621
Rungis	SAm	671 978	342 179	26 166	149 068	4 474	175 056
Saint-Mandé	SAm	1 404 752	748 036	310 852	363 047	32 680	260 182
Le Perreux-sur-Marne	SAm	1 581 343	1 334 018	155 325	224 887	7 762	13 849
Ivry-sur-Seine	SAm	3 511 007	2 306 384	452 800	1 032 952	101 316	59 452
Charenton-le-Pont	SAm	1 808 904	1 261 835	156 875	384 435	33 849	121 356
Cachan	SAm	1 567 558	869 269	403 564	647 098	7 677	42 303
L'Hay-les-Roses	SAm	1 528 073	1 171 131	210 215	298 038	9 838	45 492
Gentilly	SAm	937 577	687 323	108 573	210 853	6 988	27 931
Fresnes	SAm	1 628 900	553 538	921 694	1 032 914	14 566	26 537

Cities of the SEDIF perimeter	WWT plant connected	Water use - total (m3)	Water use - Domestic (m3)	Water use - Non market services (m3)	Water use- Market services (m3)	Water use - Industries (m3)	Water use - Others (m3)
Le Kremlin-Bicêtre	SAm	1 664 265	780 955	603 526	806 889	8 729	67 468
Villeneuve-le-Roi	SAm	979 745	657 605	101 487	200 801	87 013	24 255
Vincennes	SAm	2 805 741	1 982 800	158 540	522 982	13 079	283 737
Orly	SAm	1 278 839	990 292	166 671	255 395	8 055	23 913
Thiais	SAm	1 859 459	1 425 850	184 447	383 306	8 300	38 043
Joinville-le-Pont	SAm	943 521	707 531	54 465	178 986	21 605	29 843
Villiers-sur-Marne	SAm	1 365 578	1 018 817	241 001	311 504	6 323	27 786
Bry-sur-Marne	SAm	978 445	623 360	157 897	320 144	7 279	27 662
Vitry-sur-Seine	SAm	5 256 487	3 129 416	381 278	998 767	1 013 369	100 406
Saint-Maurice	SAm	861 722	411 798	292 110	335 067	1 669	112 510
Chennevières-sur-Marne	SAm	1 010 223	788 962	102 110	193 426	15 142	12 442
Ablon-sur-Seine	SAm	233 687	189 986	21 695	33 153	910	9 296
Champigny-sur-Marne	SAm	3 528 050	2 843 687	339 557	569 818	26 425	66 123
Chevilly-Larue	SAm	1 486 200	1 166 011	133 456	226 897	50 970	39 875
Choisy-le-Roi	SAm	1 999 044	1 610 325	151 795	326 621	28 540	31 785
Maisons-Alfort	SAm	3 198 462	1 917 743	384 816	877 735	345 835	48 560
Villejuif	SAm	3 372 994	2 190 302	802 388	1 026 711	22 571	132 590
Argenteuil	SAv	5 179 867	3 521 709	676 076	1 228 079	297 790	62 016
Sannois	SAv	1 192 164	961 593	121 038	207 094	9 311	13 147
Ermont	SAv	1 406 077	1 143 204	162 607	230 934	7 453	20 486
Franconville	SG	1 504 759	1 176 873	219 132	333 614	16 632	16 394
Le Plessis-Bouchard	SAv	328 345	326 999	0	187	0	1 159
Taverny	SG	1 296 403	937 686	205 901	281 817	42 825	22 065
Montmorency	SAv	1 129 634	879 160	160 412	205 322	3 988	41 030
Deuil-la-Barre	SAv	1 061 729	947 626	56 401	96 550	4 519	12 220
Sarcelles	SAv	3 230 235	2 463 115	444 680	693 286	53 356	15 410
Saint-Gratien	SAv	1 110 527	909 511	83 847	178 478	2 792	19 416
Herblay	SAv	1 256 968	969 136	136 465	254 154	11 419	18 628
Soisy-sous-Montmorency	SAv	900 892	729 365	88 897	151 892	1 688	17 947
Cormeilles-en-Parisis	SAv	993 827	789 740	127 167	180 651	3 996	10 088
Beauchamp	SG	427 896	313 096	21 737	78 854	28 546	7 400
Saint-Leu-la-Forêt	SAv	689 103	572 331	74 202	105 829	3 945	5 661
Domont	SAv	642 909	536 592	51 896	87 901	3 593	14 240
Piscop	SAv	38 762	29 251	3 550	9 459	179	414
Saint Brice-sous-Forêt	SAv	670 639	497 509	57 474	157 557	4 873	3 889
Montmagny	SAv	590 600	503 348	52 093	76 123	5 354	5 234
Montigny-lès-Cormeilles	SG	880 117	689 308	91 859	178 267	6 177	5 202
Saint-Prix	SAv	353 749	289 878	45 808	58 605	280	4 986
Villiers-le-Bel	SAv	1 459 290	1 045 254	275 177	393 904	10 388	8 164
Groslay	SAv	369 504	304 343	38 524	52 530	5 813	6 039
Auvers-sur-Oise	SAv	281 565	232 151	16 945	44 081	2 036	2 871
Ecouen	SAv	312 642	258 184	33 012	51 657	1 244	1 557
Pierrelaye	SAv	408 589	268 072	36 276	116 561	18 937	4 123
La Frette-sur-Seine	SAv	190 573	172 296	10 124	15 441	1 726	1 110

Cities of the SEDIF perimeter	WWT plant connected	Water use - total (m3)	Water use - Domestic (m3)	Water use - Non market services (m3)	Water use- Market services (m3)	Water use - Industries (m3)	Water use - Others (m3)
Méry-sur-Oise	SAv	395 265	339 693	19 686	44 116	3 848	6 286
Bessancourt	SG	320 229	217 648	18 363	97 786	903	2 494
Andilly	SAv	146 474	105 672	25 555	34 992	2 041	3 321
Margency	SAv	140 231	114 829	22 446	24 418	159	825
Eaubonne	SAv	1 230 852	865 788	292 046	336 291	14 060	12 946
Montlignon	SAv	142 224	118 744	18 071	19 707	1 882	1 891
Bezons	SAv	1 575 793	1 027 539	170 196	340 201	193 802	1 736
Enghien-les-Bains	SAv	733 964	572 623	35 683	126 056	6 245	28 365

C.5. Calculation of the stormwater collected in the system and the volumetric allocation for WWT

Table C-4. Calculation of the stormwater collected in each WWT plant and the volumetric allocation used to allocate the total impacts of each WWT plant for the SEDIF perimeter.

Parameters	Row	Equation	Sources of data	Seine Aval	Seine Amont	Marne Aval	Seine Grésillons
Total volume treated (10^3 m^3)	a	data	(SIAAP, 2012)	610 932	138 272	19 728	30 191
Volume of wastewater & stormwater / eq inhabitant / year (m^3)	b	data	(SIAAP, 2012)	112	81	66	110
theoretical volume of wastewater / eq inhabitant / year (m^3)	c	data		49	49	49	49
% of wastewater at the input of WWT	d	c/b		43%	60%	74%	44%
% of stormwater at the input of WWT	e	1-d		57%	40%	26%	56%
Theoretical volume of stormwater collected (10^3 m^3)	f	e*a		345 182	55 269	5 122	16 766
Theoretical volume of wastewater collected from cities of SEDIF (10^3 m^3)	g	data	Table C-3	138 138	55 225	15 451	3 942
Theoretical volume of wastewater & stormwater collected from cities of SEDIF (10^3 m^3)	h	g/d		317 567	91 998	20 870	8 866
Theoretical volume of wastewater & stormwater collected from cities of SEDIF (10^3 m^3)	i	g-h		179 428	36 773	5 419	4 924
% of the plant used for SEDIF perimeter (volumetric allocation)	j	h/a		52%	67%	100%	29%
% of total stormwater collected in the SEDIF system	k	i / sum (i)		79%	16%	3%	2%

C.6. LCI for computing impact matrices i

In the sub-sections C.2.1 to C.2.6, sources of LCI for computing impact matrices **i** are presented. LCI data for processes based on local data are developed hereafter. LCI data corresponding to ecoinvent processes are not presented here. The resulting matrices **i** obtained from these LCI data are computed within Simapro 8 (Pré Consultants, 2013). As a matter of simplicity, only one example of matrix **i** is presented in section C.6.4.

C.6.1. Drinking water production and distribution

Data for life cycle inventory of drinking water production and distribution are confidential. They have been gathered during an internship done in the context of this thesis (Catel, L. 2012. Analyse du cycle de vie de trois usines de production d'eau potable. Confidential report). The report can be requested upon request.

C.6.2. Wastewater treatment

Table C-5. LCI data for WWT operation and infrastructures

Technosphere Input/Emissions (ecoinvent names)	Seine Aval	Seine Amont	Seine Gré.	Marne Amont	Unit	Source of data
Operation For V_Tin = 1 m3						
Materials/fuels						
Iron (III) chloride, without water, in 40% solution state {GLO}| market for | Alloc Def, U	0.0559	0.0559	0.0559	0.0559	kg	Local data
Methanol {GLO}| market for | Alloc Def, U	0.0428	0.0428	0.0428	0.0428	kg	Local data
Calcium nitrate {RER}| production | Alloc Def, U	0.0301	0.0301	0.0301	0.0301	kg	Local data
Polymer	0.0150	0.0150	0.0150	0.0150	kg	Local data
Electricity/heat						
Electricity, medium voltage {FR}| market for | Alloc Def, U	0.382	0.687	0.612	1.0757	kWh	Local data
Heat, central or small-scale, natural gas {Europe without Switzerland}| market for heat, central or small-scale, natural gas | Alloc Def, U	0.24	0	0.333	0.1744	kWh	Local data
Heat, district or industrial, other than natural gas {Europe without Switzerland}| heat production, heavy fuel oil, at industrial furnace 1MW | Alloc Def, U	0.24	0.00261	0	0	kWh	Local data
Emissions to air						
Methane, biogenic	1.71E-03	1.15E-03	1.67E-03	1.20E-04	kg	Local data
Cadmium	6.20E-14	6.20E-14	6.20E-14	6.20E-14	kg	ecoinvent
Ammonia	7.27E-05	7.27E-05	7.27E-05	7.27E-05	kg	ecoinvent
NMVOC, non-methane volatile organic compounds, unspecified origin	2.28E-06	2.28E-06	2.28E-06	2.2E-06	kg	ecoinvent
Arsenic	2.53E-10	2.53E-10	2.53E-10	2.53E-10	kg	ecoinvent
Lead	2.82E-13	2.82E-13	2.82E-13	2.82E-13	kg	ecoinvent
Dinitrogen monoxide	7.76E-05	7.83E-05	7.79E-05	4.34E-05	kg	Local data
Carbon dioxide, biogenic	4.01E-01	4.01E-01	4.01E-01	9.46E-02	kg	Local data
Magnesium	5.53E-07	5.53E-07	5.53E-07	5.53E-07	kg	ecoinvent
Carbon monoxide, biogenic	1.52E-04	1.52E-04	1.52E-04	1.52E-04	kg	ecoinvent
Nitrogen oxides	6.54E-04	6.54E-04	6.54E-04	6.54E-04	kg	ecoinvent
Tin	3.36E-13	3.36E-13	3.36E-13	3.36E-13	kg	ecoinvent
Mercury	3.33E-13	3.33E-13	3.33E-13	3.33E-13	kg	ecoinvent
Infrastructures For 1 year (total plant)						
Materials/fuels						
Wastewater treatment facility, capacity 4.7E10l/year without land use {FR}	13.2	4.66	2.33	0.58	p	Local data
Transformation, from pasture to industrial area	8	0.8	0.198	0.0026	km2	Local data
Occupation, industrial area	240	24	5.94	0.777	km2a	Local data
Occupation, industrial site	8	0.8	0.198	0.0026	km2a	Local data

Table C-6. LCI data for WWT sludge spreading

Technosphere Input/Emissions (ecoinvent names)	Seine Aval	Seine Amont	Seine Gré.	Unit	Source of data
Operation For V_Tin = 1 m3 (at the WWT plant)					
Mass of sludge	0.221	0.253	0.776		Local data
Materials/fuels					
Liquid manure spreading, by vacuum tanker {CH} | processing | Alloc Def, U	0.221	0.253	0.776	L	Local data
Transport, freight, lorry 16-32 metric ton, EURO3 {RER} | Alloc Def, U	0.0122	0.0139	0.0427	kg	Local data
Emissions to air					
Ammonia	1.13E-2	1.31E-2	1.25E-14	kg	Local data
Dinitrogen monoxide	6.71E-4	7.75E-4	7.42E-05	kg	Local data
Emissions to agricultural soil					
Carbon	1.73E-2	1.73E-2	1.73E-2	kg	ecoinvent
Sulfur	1.54E-3	1.54E-3	1.54E-3	kg	ecoinvent
Arsenic	1.94E-7	1.94E-7	1.94E-7	kg	ecoinvent
Cadmium	4.46E-7	4.46E-7	4.46E-7	kg	Local data
Cobalt	7.97E-7	7.97E-7	7.97E-7	kg	ecoinvent
Chromium	5.87E-6	5.87E-6	5.87E-6	kg	Local data
Copper	5.95E-5	5.95E-5	5.95E-5	kg	Local data
Mercury	2.66E-7	2.66E-7	2.66E-7	kg	Local data
Manganese	2.61E-5	2.61E-5	2.61E-5	kg	ecoinvent
Molybdenum	4.72E-7	4.72E-7	4.72E-7	kg	ecoinvent
Nickel	3.06E-6	3.06E-6	3.06E-6	kg	Local data
Lead	1.66E-5	1.66E-5	1.66E-5	kg	Local data
Tin	1.98E-6	1.98E-6	1.98E-6	kg	ecoinvent
Zinc	1.86E-4	7.56E-5	7.56E-5	kg	Local data
Silicon	2.93E-3	2.93E-3	2.93E-3	kg	ecoinvent
Iron	1.33E-2	1.33E-2	1.33E-2	kg	ecoinvent
Calcium	4.99E-3	4.99E-3	4.99E-3	kg	ecoinvent
Aluminum	1.47E-3	1.47E-3	1.47E-3	kg	ecoinvent
Magnesium	5.60E-3	5.60E-3	5.60E-3	kg	ecoinvent
Phosphorus	1.26E-4	1.26E-4	1.26E-4	kg	ecoinvent

C.6.3. Nitrogen mass balance in WWT

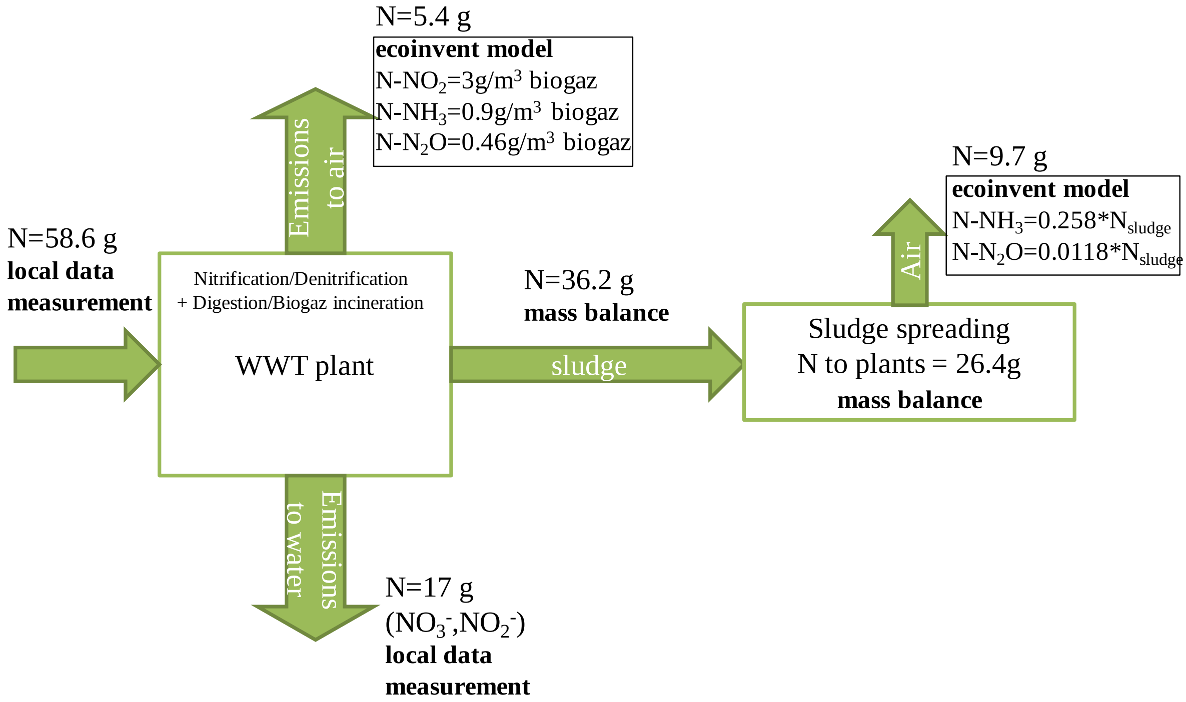


Figure C-7. Nitrogen mass balance in WWT

C.6.4. Example of impact matrix i

Table C-7. Matrix of specific impacts i, for the instance WWT_Marne-Aval

LCIA Method	Impact category	Unit	direct, air and soil	indirect, energy	indirect, chemicals & others (for 1m^3)	indirect, infrastructure (total)
Impact 2002+	Carcinogens	DALY	8,828E-13	6,43E-09	4,08E-08	2,79E-01
	Non-carcinogens	DALY	1,925E-11	7,11E-09	1,19E-08	3,06E-01
	Respiratory inorganics	DALY	6,45E-08	7,89E-08	1,36E-07	3,03E+00
	Ionizing radiation	DALY	0	1,85E-08	5,29E-10	4,29E-03
	Ozone layer depletion	DALY	0	1,05E-10	3,37E-11	1,48E-04
	Respiratory organics	DALY	2,92E-12	6,65E-11	1,79E-10	4,15E-03
	Aquatic ecotoxicity	PDF*m2*yr	1,495E-08	1,89E-03	7,87E-04	1,08E+04
	Terrestrial ecotoxicity	PDF*m2*yr	6,488E-06	3,49E-02	2,46E-02	6,04E+05
	Terrestrial acid/nutri	PDF*m2*yr	0,0048667	1,91E-03	4,59E-03	8,43E+04
	Land occupation	PDF*m2*yr	0	2,43E-03	2,33E-03	8,00E+04
	Aquatic acidification	PDF*m2*yr	5,247E-06	5,40E-06	1,10E-05	1,80E+02
	Aquatic eutrophication	PDF*m2*yr	0	1,97E-04	6,13E-04	1,05E+04
	Global warming	kg CO2 eq	0,0073617	1,48E-01	1,86E-01	3,97E+06
	Non-renewable energy	MJ primary	0	1,27E+01	3,54E+00	3,51E+07
	Mineral extraction	MJ primary	0	6,73E-03	2,09E-02	8,72E+05
	Human health	DALY	6,45E-08	1,11E-07	1,90E-07	3,63E+00
	Ecosystem quality	PDF*m2*yr	4,88E-03	4,13E-02	3,29E-02	7,90E+05
	Resources	MJ primary	0,00E+00	1,28E+01	3,56E+00	3,60E+07
ILCD	Climate change	kg CO2 eq	0,0156269	1,56E-01	2,22E-01	4,05E+06
	Ozone depletion	kg CFC-11 eq	0	9,96E-08	3,21E-08	1,41E-01
	HT, cancer effects	CTUh	1,098E-13	7,49E-09	8,88E-09	5,81E-01
	HT, non-cancer effects	CTUh	4,61E-12	5,78E-08	8,28E-08	1,87E+00
	Particulate matter	kg PM2.5 eq	9,942E-06	6,66E-05	1,33E-04	2,86E+03
	Ionizing radiation HH	kBq U235 eq	0	8,65E-01	2,49E-02	2,02E+05
	Ionizing radiation E (interim)	CTUe	0	1,06E-06	5,73E-08	6,91E-01
	Photochemical ozone formation	kg NMVOC eq	0,000658	3,37E-04	6,48E-04	1,65E+04
	Acidification	molc H+ eq	0,0007039	7,34E-04	1,55E-03	2,29E+04
	Terrestrial eutrophication	molc N eq	0,0037698	1,11E-03	3,00E-03	5,55E+04
	Freshwater eutrophication	kg P eq	0	2,88E-05	6,07E-05	1,20E+03
	Marine eutrophication	kg N eq	0,0002613	1,09E-04	2,47E-04	5,08E+03
	Freshwater ecotoxicity	CTUe	4,287E-06	9,55E-01	2,14E+00	9,78E+07
	Land use	kg C deficit	0	8,57E-02	1,78E-01	1,13E+07
	CTA Hoekstra	m3 water eq	0	5,43E-04	3,84E-03	47832,654
	resource depletion	kg Sb eq	0	5,71E-06	2,10E-05	1,61E+02

C.7. Updated CF_{WD} for baseline and forecasting scenarios

Updated monthly CF_{WD} computed in Chapter 6 are shown hereafter for 2012 (Table C-8) and 2050 (Table C-9, considering climate change effects). Only the CF_{WD} from 12 sub-river basins (on a total of 110 within the Seine river basin), are shown as a matter of simplicity, and because these are the concerned SRBs in the case study. Since the boundaries and numbers of SRBs in the updated model are different from Chapter 3, id of each sub-river basins are different from those defined in Figure 3-4. The updated ids are from HydroSHEDS. We also defined simplified ids to easily locate the different SRBs. Locations of SRBs are shown in Figure C-8.

Table C-8. Updated CF_{WD} at the monthly scale for 2012

simplified id	Id (20804-)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	55490	0,06	0,08	0,10	0,17	0,11	0,23	0,42	0,80	0,87	0,32	0,14	0,05
2	55180	0,08	0,11	0,12	0,22	0,16	0,32	0,58	1,13	1,20	0,47	0,19	0,07
3	55290	0,08	0,11	0,13	0,24	0,16	0,31	0,57	1,10	1,18	0,44	0,19	0,08
4	59800	0,09	0,13	0,15	0,26	0,17	0,35	0,65	1,23	1,33	0,52	0,22	0,09
5	59960	0,12	0,17	0,21	0,36	0,23	0,46	0,88	1,67	1,80	0,68	0,28	0,12
6	64550	0,13	0,19	0,23	0,40	0,25	0,51	0,99	1,90	2,03	0,76	0,31	0,13
7	64540	0,20	0,27	0,32	0,56	0,34	0,66	1,17	2,21	2,46	0,92	0,38	0,18
8	70760	0,14	0,19	0,23	0,41	0,25	0,51	0,99	1,92	2,05	0,77	0,31	0,14
9	70920	0,15	0,21	0,25	0,44	0,27	0,56	1,08	2,08	2,24	0,86	0,35	0,15
10	81100	0,16	0,22	0,28	0,48	0,31	0,61	1,19	2,28	2,47	0,97	0,42	0,17
11	77390	0,18	0,24	0,29	0,50	0,32	0,63	1,20	2,27	2,44	0,98	0,45	0,19
12	C-A	0,13	0,18	0,22	0,38	0,24	0,48	0,93	1,79	1,92	0,72	0,29	0,13

Table C-9. Updated CF_{WD} at the monthly scale for 2050

simplified id	id (20804-)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	55490	0,06	0,08	0,10	0,17	0,13	0,28	0,50	0,92	0,98	0,35	0,14	0,05
2	55180	0,08	0,11	0,12	0,22	0,18	0,38	0,70	1,29	1,37	0,52	0,19	0,07
3	55290	0,08	0,11	0,13	0,24	0,17	0,38	0,68	1,25	1,34	0,48	0,19	0,08
4	59800	0,09	0,13	0,15	0,26	0,19	0,42	0,77	1,41	1,51	0,57	0,22	0,09
5	59960	0,12	0,17	0,21	0,36	0,25	0,56	1,05	1,91	2,04	0,74	0,28	0,12
6	64550	0,13	0,19	0,23	0,40	0,28	0,62	1,18	2,18	2,31	0,84	0,31	0,13
7	64540	0,20	0,27	0,32	0,56	0,38	0,80	1,39	2,52	2,77	1,00	0,38	0,18
8	70760	0,14	0,19	0,23	0,41	0,28	0,62	1,19	2,19	2,33	0,84	0,31	0,14
9	70920	0,15	0,21	0,25	0,44	0,30	0,68	1,29	2,38	2,55	0,94	0,35	0,15
10	81100	0,16	0,22	0,28	0,48	0,34	0,75	1,42	2,62	2,80	1,06	0,42	0,17
11	77390	0,15	0,22	0,27	0,46	0,32	0,71	1,36	2,53	2,71	1,01	0,38	0,16
12	C-A	0,13	0,18	0,22	0,38	0,26	0,59	1,11	2,05	2,18	0,79	0,29	0,13

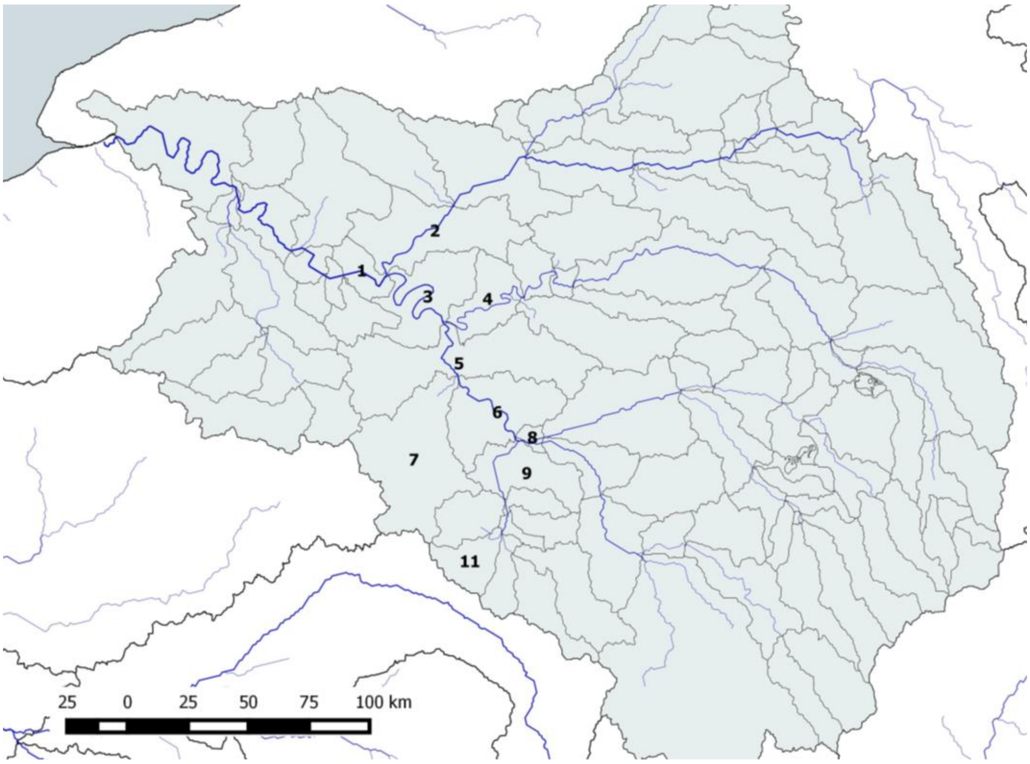


Figure C-8. Locations of the SRBs concerned in Chapter 6 within the Seine river basin.

C.8. Full results of environmental impacts for the baseline and forecasting scenarios

Table C-10. Full results of the baseline and the forecasting scenarios: impacts for the entire UWS

			B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
Impact 2002+	Carcinogens	DALY	5,17E+01	5,14E+01	5,12E+01	5,09E+01	5,01E+01	5,33E+01	5,53E+01	5,01E+01	5,10E+01	4,99E+01	5,12E+01	4,97E+01
	Non-carcinogens	DALY	8,24E+02	8,17E+02	8,13E+02	8,09E+02	7,88E+02	8,60E+02	9,03E+02	7,88E+02	7,91E+02	7,85E+02	7,93E+02	7,88E+02
	Respiratory inorganics	DALY	6,18E+02	6,14E+02	6,10E+02	6,06E+02	5,95E+02	6,42E+02	6,70E+02	5,95E+02	6,08E+02	5,93E+02	6,09E+02	5,90E+02
	Ionizing radiation	DALY	7,30E+00	7,35E+00	6,97E+00	6,72E+00	6,80E+00	7,80E+00	8,40E+00	6,80E+00	8,36E+00	6,68E+00	8,50E+00	6,31E+00
	Ozone layer depletion	DALY	6,27E-02	6,28E-02	6,05E-02	5,89E-02	5,90E-02	6,64E-02	7,08E-02	5,90E-02	6,85E-02	5,82E-02	6,91E-02	5,61E-02
	Respiratory organics	DALY	3,26E-01	3,25E-01	3,22E-01	3,21E-01	3,18E-01	3,33E-01	3,42E-01	3,18E-01	3,28E-01	3,17E-01	3,29E-01	3,15E-01
	Aquatic ecotoxicity	PDF*m2 *yr	2,21E+08	2,22E+08	2,16E+08	2,10E+08	2,07E+08	2,35E+08	2,52E+08	2,07E+08	2,10E+08	1,97E+08	2,05E+08	2,04E+08
	Terrestrial ecotoxicity	PDF*m2 *yr	1,06E+10	1,06E+10	1,04E+10	1,03E+10	1,00E+10	1,12E+10	1,19E+10	1,00E+10	1,04E+10	9,76E+09	1,02E+10	9,94E+09
	Terrestrial acid/nutri	PDF*m2 *yr	7,54E+07	7,47E+07	7,44E+07	7,39E+07	7,21E+07	7,86E+07	8,24E+07	7,21E+07	7,25E+07	7,21E+07	7,25E+07	7,20E+07
	Land occupation	PDF*m2 *yr	7,76E+06	7,76E+06	7,66E+06	7,48E+06	7,61E+06	7,91E+06	8,09E+06	7,61E+06	7,84E+06	7,55E+06	7,45E+06	7,17E+06
	Acidification	PDF*m2 *yr	8,92E+04	8,84E+04	8,79E+04	8,74E+04	8,54E+04	9,29E+04	9,73E+04	8,54E+04	8,66E+04	8,53E+04	8,68E+04	8,51E+04
	Eutrophication	PDF*m2 *yr	2,78E+07	2,75E+07	2,76E+07	2,75E+07	2,68E+07	2,89E+07	3,01E+07	2,68E+07	2,60E+07	2,66E+07	2,70E+07	2,69E+07
	Global warming	kg CO2 eq	3,86E+08	3,85E+08	3,81E+08	3,78E+08	3,74E+08	3,98E+08	4,12E+08	3,74E+08	3,90E+08	3,72E+08	3,92E+08	3,69E+08
	Non-renewable energy	MJ primary	8,43E+09	8,44E+09	8,16E+09	7,97E+09	7,98E+09	8,86E+09	9,39E+09	7,98E+09	9,12E+09	7,89E+09	9,22E+09	7,62E+09
	Mineral extraction	MJ primary	4,58E+07	4,58E+07	4,54E+07	4,51E+07	4,50E+07	4,67E+07	4,77E+07	4,50E+07	4,61E+07	4,48E+07	4,62E+07	4,46E+07
	Human health	DALY	1,50E+03	1,49E+03	1,48E+03	1,47E+03	1,44E+03	1,56E+03	1,64E+03	1,44E+03	1,46E+03	1,44E+03	1,46E+03	1,44E+03
	Ecosystem quality	PDF*m2 *yr	1,09E+10	1,09E+10	1,07E+10	1,06E+10	1,03E+10	1,16E+10	1,23E+10	1,03E+10	1,07E+10	1,01E+10	1,05E+10	1,03E+10

			B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
	Resources	MJ primary	8,47E+09	8,49E+09	8,20E+09	8,01E+09	8,03E+09	8,91E+09	9,43E+09	8,03E+09	9,17E+09	7,93E+09	9,26E+09	7,67E+09
ILCD	Climate change	kg CO2 eq	4,40E+08	4,39E+08	4,34E+08	4,30E+08	4,25E+08	4,55E+08	4,73E+08	4,25E+08	4,42E+08	4,23E+08	4,44E+08	4,19E+08
	Ozone depletion	kg CFC-11 eq	5,89E+01	5,90E+01	5,68E+01	5,53E+01	5,54E+01	6,24E+01	6,66E+01	5,54E+01	6,44E+01	5,46E+01	6,50E+01	5,26E+01
	Human toxicity, cancer effects	CTUh	5,54E+01	5,52E+01	5,48E+01	5,44E+01	5,41E+01	5,67E+01	5,82E+01	5,41E+01	5,52E+01	5,36E+01	5,50E+01	5,32E+01
	Human toxicity, non-cancer effects	CTUh	3,58E+03	3,54E+03	3,53E+03	3,51E+03	3,42E+03	3,73E+03	3,92E+03	3,42E+03	3,43E+03	3,42E+03	3,43E+03	3,42E+03
	Particulate matter	kg PM2.5 eq	5,01E+05	4,98E+05	4,94E+05	4,91E+05	4,82E+05	5,20E+05	5,43E+05	4,82E+05	4,94E+05	4,81E+05	4,95E+05	4,78E+05
	Ionizing radiation HH	kBq U235 eq	3,41E+08	3,43E+08	3,26E+08	3,14E+08	3,17E+08	3,64E+08	3,92E+08	3,17E+08	3,90E+08	3,12E+08	3,97E+08	2,95E+08
	Ionizing radiation E (interim)	CTUe	4,53E+02	4,55E+02	4,34E+02	4,19E+02	4,23E+02	4,82E+02	5,18E+02	4,23E+02	5,13E+02	4,16E+02	5,21E+02	3,95E+02
	Photochemical ozone formation	kg NMVOC	1,28E+06	1,28E+06	1,26E+06	1,25E+06	1,24E+06	1,32E+06	1,37E+06	1,24E+06	1,30E+06	1,23E+06	1,30E+06	1,22E+06
	Acidification	molc H+ eq	1,53E+07	1,52E+07	1,51E+07	1,50E+07	1,46E+07	1,59E+07	1,67E+07	1,46E+07	1,48E+07	1,46E+07	1,48E+07	1,46E+07
	Terrestrial eutrophication	molc N eq	6,35E+07	6,29E+07	6,27E+07	6,23E+07	6,07E+07	6,62E+07	6,95E+07	6,07E+07	6,09E+07	6,07E+07	6,09E+07	6,07E+07
	Freshwater eutrophication	kg P eq	4,69E+05	4,65E+05	4,63E+05	4,60E+05	4,50E+05	4,88E+05	5,12E+05	4,50E+05	4,50E+05	4,44E+05	4,65E+05	4,49E+05
	Marine eutrophication	kg N eq	8,30E+06	8,20E+06	8,22E+06	8,18E+06	7,99E+06	8,60E+06	8,97E+06	7,99E+06	7,67E+06	7,98E+06	7,99E+06	8,02E+06
	Freshwater ecotoxicity	CTUe	7,05E+09	7,02E+09	6,97E+09	6,93E+09	6,86E+09	7,23E+09	7,46E+09	6,86E+09	6,98E+09	6,83E+09	7,01E+09	6,81E+09
	Land use	kg C deficit	1,54E+09	1,54E+09	1,54E+09	1,52E+09	1,53E+09	1,55E+09	1,56E+09	1,53E+09	1,54E+09	1,53E+09	1,50E+09	1,50E+09
	Water resource depletion	m3 water eq	6,53E+07	6,81E+07	6,29E+07	5,27E+07	6,72E+07	7,80E+07	8,45E+07	5,60E+07	2,54E+07	7,45E+07	6,42E+07	6,05E+07
	Mineral, fossil & ren resource depletion	kg Sb eq	2,99E+04	2,98E+04	2,95E+04	2,86E+04	2,92E+04	3,06E+04	3,15E+04	2,92E+04	2,97E+04	2,89E+04	2,78E+04	2,69E+04
WIIX+	WIIX+	m3eq	4,94E+07	4,90E+07	4,89E+07	4,25E+07	4,73E+07	6,50E+07	7,48E+07	4,73E+07	1,07E+07	4,99E+07	5,16E+07	4,76E+07

Table C-11. Full results of the baseline and the forecasting scenarios: impacts for 1 m^3 a the user

			B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
Impact 2002+	Carcinogens	DALY	2,18E-07	2,21E-07	2,21E-07	2,23E-07	2,33E-07	2,05E-07	1,92E-07	2,33E-07	2,37E-07	2,32E-07	2,38E-07	2,31E-07
	Non-carcinogens	DALY	3,47E-06	3,51E-06	3,52E-06	3,55E-06	3,66E-06	3,30E-06	3,14E-06	3,66E-06	3,68E-06	3,65E-06	3,69E-06	3,66E-06
	Respiratory inorganics	DALY	2,60E-06	2,64E-06	2,64E-06	2,66E-06	2,76E-06	2,46E-06	2,33E-06	2,76E-06	2,83E-06	2,76E-06	2,83E-06	2,74E-06
	Ionizing radiation	DALY	3,07E-08	3,15E-08	3,02E-08	2,95E-08	3,16E-08	3,00E-08	2,92E-08	3,16E-08	3,89E-08	3,11E-08	3,95E-08	2,94E-08
	Ozone layer depletion	DALY	2,64E-10	2,70E-10	2,62E-10	2,58E-10	2,74E-10	2,55E-10	2,46E-10	2,74E-10	3,18E-10	2,70E-10	3,21E-10	2,61E-10
	Respiratory organics	DALY	1,37E-09	1,40E-09	1,40E-09	1,41E-09	1,48E-09	1,28E-09	1,19E-09	1,48E-09	1,52E-09	1,47E-09	1,53E-09	1,46E-09
	Aquatic ecotoxicity	PDF*m2 *yr	9,29E-01	9,51E-01	9,33E-01	9,23E-01	9,61E-01	9,02E-01	8,76E-01	9,61E-01	9,78E-01	9,14E-01	9,54E-01	9,49E-01
	Terrestrial ecotoxicity	PDF*m2 *yr	4,46E+01	4,55E+01	4,50E+01	4,50E+01	4,65E+01	4,31E+01	4,15E+01	4,65E+01	4,82E+01	4,54E+01	4,74E+01	4,62E+01
	Terrestrial acid/nutri	PDF*m2 *yr	3,17E-01	3,21E-01	3,22E-01	3,24E-01	3,35E-01	3,02E-01	2,87E-01	3,35E-01	3,37E-01	3,35E-01	3,37E-01	3,35E-01
	Land occupation	PDF*m2 *yr	3,26E-02	3,33E-02	3,32E-02	3,28E-02	3,54E-02	3,04E-02	2,81E-02	3,54E-02	3,65E-02	3,51E-02	3,46E-02	3,33E-02
	Acidification	PDF*m2 *yr	3,75E-04	3,80E-04	3,80E-04	3,83E-04	3,97E-04	3,57E-04	3,39E-04	3,97E-04	4,02E-04	3,96E-04	4,03E-04	3,95E-04
	Eutrophication	PDF*m2 *yr	1,17E-01	1,18E-01	1,19E-01	1,21E-01	1,25E-01	1,11E-01	1,05E-01	1,25E-01	1,21E-01	1,24E-01	1,26E-01	1,25E-01
	Global warming	kg CO2 eq	1,62E+00	1,65E+00	1,65E+00	1,66E+00	1,74E+00	1,53E+00	1,43E+00	1,74E+00	1,81E+00	1,73E+00	1,82E+00	1,71E+00
	Non-renewable energy	MJ primary	3,54E+01	3,62E+01	3,53E+01	3,50E+01	3,71E+01	3,40E+01	3,27E+01	3,71E+01	4,24E+01	3,67E+01	4,28E+01	3,54E+01
	Mineral extraction	MJ primary	1,93E-01	1,97E-01	1,96E-01	1,98E-01	2,09E-01	1,79E-01	1,66E-01	2,09E-01	2,14E-01	2,08E-01	2,15E-01	2,07E-01
	Human health	DALY	6,32E-06	6,40E-06	6,42E-06	6,47E-06	6,70E-06	6,01E-06	5,70E-06	6,70E-06	6,79E-06	6,68E-06	6,81E-06	6,68E-06
	Ecosystem quality	PDF*m2 *yr	4,60E+01	4,69E+01	4,64E+01	4,64E+01	4,80E+01	4,44E+01	4,28E+01	4,80E+01	4,96E+01	4,68E+01	4,89E+01	4,76E+01
	Resources	MJ primary	3,56E+01	3,64E+01	3,55E+01	3,52E+01	3,73E+01	3,42E+01	3,28E+01	3,73E+01	4,26E+01	3,69E+01	4,31E+01	3,56E+01

			B	S1	S2	S3	L1	L2	L3	L4	L5	L6	L7	L8
ILCD	Climate change	kg CO2 eq	1,85E+00	1,88E+00	1,88E+00	1,89E+00	1,98E+00	1,75E+00	1,65E+00	1,98E+00	2,05E+00	1,97E+00	2,06E+00	1,95E+00
	Ozone depletion	kg CFC-11 eq	2,48E-07	2,53E-07	2,46E-07	2,43E-07	2,57E-07	2,40E-07	2,32E-07	2,57E-07	2,99E-07	2,54E-07	3,02E-07	2,44E-07
	Human toxicity, cancer effects	CTUh	2,33E-07	2,37E-07	2,37E-07	2,39E-07	2,51E-07	2,18E-07	2,02E-07	2,51E-07	2,57E-07	2,49E-07	2,56E-07	2,47E-07
	Human toxicity, non-cancer effects	CTUh	1,50E-05	1,52E-05	1,53E-05	1,54E-05	1,59E-05	1,43E-05	1,36E-05	1,59E-05	1,59E-05	1,59E-05	1,59E-05	1,59E-05
	Particulate matter	kg PM2.5 eq	2,11E-03	2,14E-03	2,14E-03	2,15E-03	2,24E-03	2,00E-03	1,89E-03	2,24E-03	2,30E-03	2,24E-03	2,30E-03	2,22E-03
	Ionizing radiation HH	kBq U235 eq	1,43E+00	1,47E+00	1,41E+00	1,38E+00	1,48E+00	1,40E+00	1,36E+00	1,48E+00	1,81E+00	1,45E+00	1,85E+00	1,37E+00
	Ionizing radiation E (interim)	CTUe	1,90E-06	1,95E-06	1,88E-06	1,84E-06	1,97E-06	1,85E-06	1,80E-06	1,97E-06	2,39E-06	1,93E-06	2,42E-06	1,84E-06
	Photochemical ozone formation	kg NMVOC eq	5,39E-03	5,49E-03	5,47E-03	5,50E-03	5,77E-03	5,07E-03	4,76E-03	5,77E-03	6,02E-03	5,74E-03	6,04E-03	5,69E-03
	Acidification	molc H+ eq	6,43E-02	6,51E-02	6,53E-02	6,58E-02	6,80E-02	6,13E-02	5,82E-02	6,80E-02	6,88E-02	6,80E-02	6,89E-02	6,78E-02
	Terrestrial eutrophication	molc N eq	2,67E-01	2,70E-01	2,71E-01	2,73E-01	2,82E-01	2,54E-01	2,42E-01	2,82E-01	2,83E-01	2,82E-01	2,83E-01	2,82E-01
	Freshwater eutrophication	kg P eq	1,97E-03	2,00E-03	2,00E-03	2,02E-03	2,09E-03	1,88E-03	1,78E-03	2,09E-03	2,09E-03	2,07E-03	2,16E-03	2,09E-03
	Marine eutrophication	kg N eq	3,49E-02	3,52E-02	3,56E-02	3,59E-02	3,72E-02	3,30E-02	3,12E-02	3,72E-02	3,56E-02	3,71E-02	3,72E-02	3,73E-02
	Freshwater ecotoxicity	CTUe	2,96E+01	3,01E+01	3,02E+01	3,04E+01	3,19E+01	2,78E+01	2,60E+01	3,19E+01	3,24E+01	3,18E+01	3,26E+01	3,16E+01
	Land use	kg C deficit	6,47E+00	6,61E+00	6,64E+00	6,68E+00	7,12E+00	5,94E+00	5,42E+00	7,12E+00	7,15E+00	7,11E+00	6,99E+00	6,97E+00
	Water resource depletion	m3 water eq	2,75E-01	2,92E-01	2,72E-01	2,31E-01	3,12E-01	3,00E-01	2,94E-01	3,12E-01	1,18E-01	3,46E-01	2,98E-01	2,81E-01
	Mineral, fossil & ren resource depletion	kg Sb eq	1,26E-04	1,28E-04	1,28E-04	1,25E-04	1,36E-04	1,18E-04	1,10E-04	1,36E-04	1,38E-04	1,35E-04	1,29E-04	1,25E-04
WIIX+	WIIX+	m3eq	2,08E-01	2,06E-01	2,12E-01	1,86E-01	2,20E-01	2,50E-01	2,60E-01	2,20E-01	4,98E-02	2,32E-01	2,40E-01	2,21E-01

Résumé étendu

Chapitre 1 : Introduction

Depuis les années 1970, l'Homme a pris conscience du caractère vulnérable des milieux naturels. Le rapport du club de Rome, publication majeure marquant l'apparition des préoccupations environnementales, donne l'alerte sur la finitude des ressources naturelles dans un contexte de croissance démographique. Cette prise de conscience a encouragé la construction d'un nouveau paradigme environnemental et a favorisé l'émergence du concept de développement durable. Dans ce contexte, les villes, grandes consommatrices de ressources naturelles, ont un rôle essentiel à jouer. Les projections montrent que plus de 60% de la population mondiale résidera dans des zones urbaines en 2030 augmentant encore la pression sur les ressources naturelles dont les ressources en eau. Ces dernières sont déjà rares et la concurrence entre les différents usagers (domestiques, agricoles, industriels) s'intensifie (World Water Assessment Programme UN, 2009). La gestion de l'eau en milieu urbain est une réelle problématique d'un point de vue environnemental (Global Water Partnership Technical Committee, 2012).

Les décideurs ont besoin d'outils pour évaluer la performance environnementale des systèmes d'eau urbains (comprenant les technologies, les usagers de l'eau et des ressources en eau). Dans ce contexte, l'analyse du cycle de vie (ACV) est un outil d'évaluation environnementale normalisé (ISO, 2006) et largement reconnu au niveau mondial. Cet outil quantifie les impacts d'un produit ou d'un service tout au long de son cycle de vie (de l'extraction des matières premières, à sa production, distribution, utilisation et jusqu'à la gestion de sa fin de vie). A la différence d'autres outils d'évaluation environnementale (par exemple, l'empreinte carbone ou le bilan énergétique), l'ACV est une approche multicritère qui prend en compte toutes les étapes du « cycle de vie » d'un bien ou d'un service. Ce caractère holistique de l'ACV permet d'identifier les transferts de pollution entre catégories d'impacts, entre étapes du cycle de vie et/ou entre lieux géographiques. Alors que l'ACV a été initialement conçue pour des approches orientées « produit/service », son application à des systèmes territoriaux émerge avec le concept d'ACV territoriale (Loiseau et al., 2013). Le territoire est une échelle pertinente pour évaluer les impacts environnementaux associés aux systèmes d'eau urbains. Cependant les études ACV appliquées à des systèmes conséquents tels que les systèmes d'eau des mégapoles urbaines nécessite une importante quantité de données et des efforts de modélisation, notamment lorsque de multiples scénarios doivent être étudiés (Schulz et al., 2012). Par conséquent, il y a un besoin pressent de développer des outils simplifiés afin de fournir aux décideurs des indicateurs sur la performance environnementale des systèmes d'eau urbains existants ou issus de scénarios prospectifs.

Au-delà des besoins méthodologiques en termes de modélisation du système d'eau urbain, un autre défi scientifique fait l'objet de nombreux travaux dans la communauté ACV : l'évaluation des impacts associés à l'utilisation de la ressource en eau. L'eau a cette propriété d'être à la fois une ressource et un habitat environnemental, deux raisons qui expliquent les nombreuses préoccupations portées sur cet « or bleu ». Bien entendu l'eau n'est pas aussi rare que l'or. Bien au contraire puisque c'est une ressource renouvelable l'eau se déplace continuellement sur Terre à travers un cycle global. Mais les ressources en eau sont très mal distribuées dans le monde et les activités humaines exacerbent cette situation. Plus de 2.5 milliards de personnes font face à la rareté de l'eau pendant au moins un mois de l'année, ce qui signifie qu'ils ne disposent pas assez d'eau disponible pour répondre aux demandes (Hoekstra et al., 2012). En plus de la quantité, l'accès limité à l'eau est lié à des problèmes de qualité affectant la santé de population vulnérable. Par ailleurs, l'eau étant aussi un habitat environnemental, la rareté de l'eau et ses pollutions impactent de nombreux écosystèmes sensibles.

Afin de tenir compte des problématiques liées à la prise en compte de la ressource en eau dans l'évaluation environnementale des activités humaines, les concepts d'eau virtuelle et d'empreinte de l'eau ont été développés. Ces méthodes permettent de quantifier les mètres cubes équivalents nécessaires pour produire des biens ou des services, en prenant en compte l'eau bleue (eau de surface), l'eau verte (eau evapotranspirée) et l'eau grise (eau polluée). Par exemple, un kilogramme de viande de bœuf représente 15 400 L d'eau ou un kilogramme de café près de 19 000 L. Cependant, l'interprétation de ces approches volumétriques pose des problèmes car elles ne prennent pas en compte les impacts potentiels associés à l'utilisation et à la pollution de l'eau sur les écosystèmes, sur la santé humaine et sur les ressources. Au contraire, l'ACV caractérise les données d'inventaire afin de quantifier tous les impacts potentiels sur l'environnement. A l'origine, l'ACV évaluait seulement les dommages dus à la pollution de l'eau (aspects qualitatifs), à travers les catégories d'impact eutrophisation, et écotoxicité. L'évaluation des impacts liés à l'utilisation de la ressource en eau (aspect quantitatif) est plus récente et est à un stade de développement précoce, mais de nouvelles méthodes sont en cours de développement et certaines sont opérationnelles (Kounina et al., 2012). L'application et le raffinement de ces approches d'évaluation environnementale pour les systèmes d’eau urbains sont nécessaires car ces systèmes jouent un rôle clé dans la gestion de l'eau à l'échelle de bassins versants.

A partir de ces éléments de contexte, i.e. (i) la nécessité de concevoir des outils simplifiés permettant aux décideurs de disposer d’indicateurs fiables sur la performance environnementale des systèmes d'eau urbains et (ii) une meilleure prise en compte des impacts liés à l’utilisation de la ressource en eau dans les méthodes d’évaluation environnementale, la question de recherche de cette thèse peut être formulée ainsi: "Est-il possible de modéliser un système d’eau urbain dans sa globalité afin d'évaluer ses impacts environnementaux et les services rendus aux usagers, en utilisant le cadre conceptuel de l'ACV?" Pour répondre à cette question de recherche, cinq sous-objectifs ont été définis :

- Identifier les principaux verrous méthodologiques liés à l'application de l’ACV aux systèmes d'eau urbains et justifier la nécessité d'une approche standardisée. Ce sous objectif est traité à travers une revue bibliographique dans le chapitre 2, qui correspond à une publication scientifique publiée dans *Water Research* (Loubet et al., 2014).
- Evaluer l'impact de la privation d'eau à une échelle appropriée et pertinente pour être applicable aux systèmes d'eau urbains. Ce sous objectif est traité dans le chapitre 3, qui correspond à une publication scientifique publiée dans *Environmental Science & Technology* (Loubet et al., 2013).
- Caractériser et comptabiliser la qualité des flux d’eau urbain grâce aux méthodes d’évaluation d’impacts du cycle de vie et d’empreinte eau. Ce sous objectif est traité dans le chapitre 4.
- Développer un cadre conceptuel, un formalisme associé et un modèle pour évaluer les impacts environnementaux des scénarios prospectifs de systèmes d’eau urbains. Ce modèle, nommé WaLA (pour « Water systems Life cycle Assessment ») réduit la complexité du système tout en étant représentatif pour l’ACV. Il s’agit de la partie centrale de la thèse qui intègre les besoins méthodologiques identifiés et développés dans les chapitres précédents. Ce sous-objectif est traité dans le chapitre 5, qui correspond à une publication soumise dans *Water Research*.
- Démontrer l’applicabilité et la capacité du modèle à répondre à des questions de gestion de l’eau avec l'évaluation des scénarios prospectifs. Le modèle est ainsi appliqué au système d’eau urbain correspondant au périmètre géographique du syndicat des eaux d’Île-de-France (SEDIF), en banlieue parisienne. Ce sous objectif

est traité dans le chapitre 6, qui correspond à une publication soumise dans *Water Research*.

Chapitre 2. Analyse comparative des publications sur l'ACV des systèmes d'eau urbains

L'analyse du cycle de vie (ACV) a été largement utilisée pour évaluer les performances environnementales des technologies liées à l'eau depuis les 20 dernières années. Une revue de la littérature a été réalisée afin de compiler toutes les publications traitant des ACV de ces technologies, c'est-à-dire la production et la distribution d'eau potable et la collecte et le traitement des eaux usées. 130 publications ont été inventoriées, dont 18 qui traitent des systèmes d'eau urbains dans leur globalité. Une attention particulière a été portée sur ces 18 publications qui ont été analysés selon des critères définis pour chacune des quatre phases de l'ACV : définition des objectifs et du champ d'étude, inventaire du cycle de vie, évaluation des impacts du cycle de vie et interprétation.

Les résultats de l'étude comparative montrent que les cas d'étude partagent un objectif similaire en apportant des informations quantitatives aux gestionnaires sur les impacts environnementaux des systèmes d'eau urbains et de leurs scénarios prospectifs. Néanmoins, les études existantes sont basées sur des objectifs et des champs différents : les unités fonctionnelles (UF) différent ainsi que les frontières des systèmes étudiés. Trois UF sont relevées : distribuer et traiter 1 m^3 chez l'usager (en résumé, « 1m^3 »), distribuer et traiter l'eau nécessaire pour un usager pendant un an (en résumé, « 1 usager/an ») ou le fonctionnement du système d'eau urbain pendant un an (en résumé « système/an »). Les données d'inventaire disponibles (utilisation d'électricité et les flux d'eau) et les résultats d'évaluation des impacts (changement climatique, eutrophisation et le score unique) sont comparés quantitativement. Cette revue de littérature apporte ainsi des données et résultats synthétiques sur l'ACV des systèmes d'eau urbains.

La revue formule des recommandations sur la manière de conduire les ACV des systèmes d'eau urbain et identifie des verrous méthodologiques :

- Pour l'évaluation environnementale d'un système d'eau urbain, la définition de l'unité fonctionnelle devrait inclure l'usager car la fonction de ce système est de satisfaire ses besoins (en termes de qualité et de quantité).
- La multifonctionnalité des systèmes d'eau urbains devrait profiter de l'adaptation du cadre de l'ACV à l'évaluation territoriale (Loiseau et al., 2013).

- L'évaluation de scénarios prospectifs par l'ACV devrait permettre de différencier et combiner les changements de technologies, d'usagers de l'eau et de ressources en eau.
- Les frontières du système devraient inclure toutes les étapes (construction, opération, déconstruction) du cycle de vie des systèmes d'eau urbains. Une attention particulière devrait être apportée sur le génie civil lors des travaux sur les réseaux.
- Un inventaire approprié de tous les flux d'eau devrait être fourni, en différenciant les flux dans la technosphère, les eaux prélevées et rejetées dans l'environnement local et l'eau évapotranspirée vers l'atmosphère (i.e. eau consommée) (Bayart et al., 2010).
- Le bilan de matière des polluants (en particulier l'azote, le phosphore et le carbone) devrait être équilibré tout au long du système.
- Les méthodes d'évaluation des impacts actuelles permettent des évaluations multicritères des systèmes d'eau urbains. Les approches monocritères telles que l'empreinte carbone devraient donc être évitées afin de limiter les transferts de pollution, en particulier vers les catégories d'impacts reliées à l'eau telles que l'eutrophisation, l'écotoxicité et la privation d'eau.
- Les recherches récentes sur les méthodes d'évaluation des impacts de l'usage de l'eau devraient être implémentées. La différentiation spatiale et temporelle à des échelles appropriées devrait permettre une évaluation site-dépendant qui est très utile à l'évaluation des systèmes d'eau urbains.

Cette revue a permis d'identifier un certain nombre d'axes de recherche qui ont été investis au cours de cette thèse, dans le but de développer un modèle pour l'évaluation de la performance environnementale des systèmes d'eau urbains.

Chapitre 3. Evaluation de la privation d'eau à l'échelle du sous bassin versant en ACV : intégration des effets cascades en aval

L'une des principales problématiques identifiées est la nécessité de prendre en compte la privation de l'eau dans les ACV des systèmes d'eau urbains. La privation d'eau au niveau midpoint est actuellement étudiée dans les méthodes d'impacts d'ACV en utilisant des indicateurs de stress hydrique à l'échelle du bassin versant (Kounina et al., 2012; Pfister et al., 2009). Bien que ces indicateurs représentent un grand pas en avant dans l'évaluation des impacts liés à l'utilisation des ressources en eau en ACV, d'importants points restent encore à améliorer concernant leur précision et leur pertinence. Plus précisément, dans le cadre de l'ACV, la définition de la privation d'eau associée à une activité humaine devrait être reliée aux effets que celle-ci pourrait occasionner en aval de la dite activité. En effet, la consommation à la source d'une rivière prive plus d'usagers et d'écosystèmes qu'une consommation à l'embouchure (Vörösmarty et al., 2000) et le fait de considérer les impacts à l'échelle du bassin versant entier ne permet pas cette différentiation. Pour les systèmes d'eau urbains qui disposent de ressources en eau variées au sein même d'un bassin versant, cette différentiation est nécessaire.

Une méthodologie quantifiant des facteurs de caractérisation midpoint de privation d'eau à l'échelle du sous bassin versant a été développée en tenant compte des effets cascades. Le cadre proposé est basé sur une approche en deux étapes. Tout d'abord, le stress hydrique est quantifié à l'échelle du sous-bassin versant par un ratio entre la quantité d'eau consommée et la quantité d'eau disponible (ratio plus communément appelé « consumption-to-availability » ou CTA). D'autre part, les facteurs de caractérisation de privation d'eau (notés CF_{WD}) sont calculés en sommant et en pondérant les CTA des sous bassins en aval. Ainsi, le CF_{WD} d'un sous bassin donné mesure l'impact d'un prélèvement ou d'une consommation d'eau sur la privation d'eau dans les sous-bassins en aval, comme le montre la Figure a.

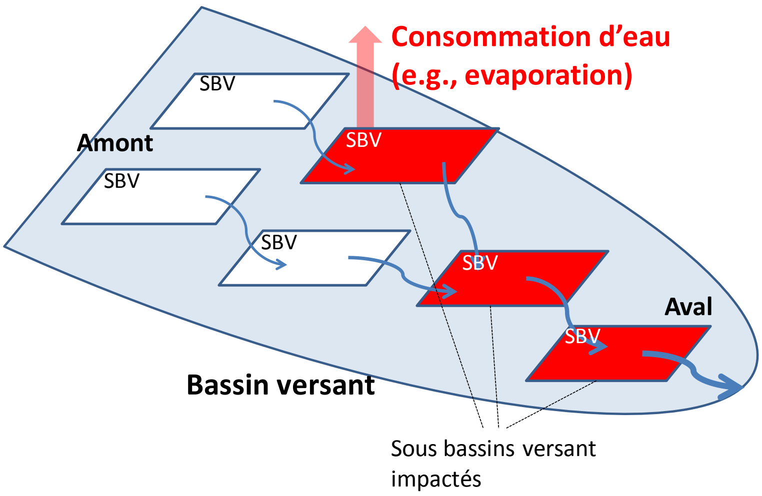


Figure a. Illustration de l'effet cascade. SBV : Sous bassin versant

Les CTA et les CF_{WD} ont été calculés grâce à un bilan d'eau à l'échelle du sous bassin versant, et avec des bases de données hydrologiques sur les écoulements d'eau et la consommation d'eau. Ces calcules ont été menés pour les bassins versant de la Seine (France) et du Guadalquivir (Espagne, Figure b).

Les résultats montrent des différences significatives entre les CF_{WD} calculés au sein d'un même bassin versant (un ordre de grandeur), en fonction de la position en amont ou en aval.

En effet, plus un sous bassin est en amont, plus un prélèvement va impacter la quantité d'eau disponible pour les sous-bassins en aval et plus le CF_{WD} sera important. Les CF_{WD} sont appliqués à un cas d'étude théorique démontrant leur applicabilité pour étudier des scénarios de gestion de l'eau. Cette méthodologie démontre qu'il est essentiel de localiser les points de prélèvement et de rejet d'eau dans un bassin versant.

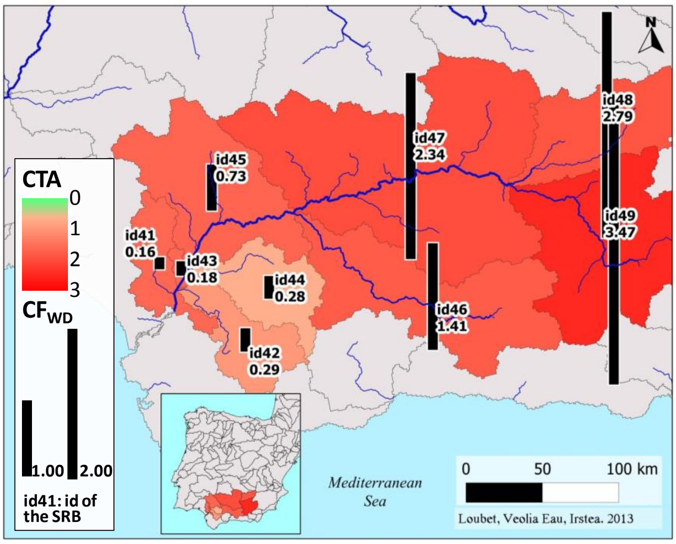


Figure b. CTA et CF_{WD} du bassin versant Guadalquivir (Espagne)

Chapitre 4. Evaluation de la qualité des flux d'eau urbains avec les méthodes existantes d'évaluation des impacts du cycle de vie et d'empreinte eau

En plus des problèmes de quantité d'eau présentés dans le chapitre 3, les impacts associés à la qualité de l'eau doivent être pris en compte dans les ACV de systèmes d'eau urbains. Le chapitre 4 propose une revue des méthodes d'évaluation des impacts du cycle de vie (EICV) et d'empreinte eau pour évaluer la qualité des flux d'eau de systèmes urbains. Ainsi, des scores de dommages sur les écosystèmes et la santé humaine sont calculés pour différents types de flux d'eau (ex. ressources en eau, eaux usées, rejets d'usines, etc.) à partir de concentrations en polluants. Les polluants caractérisés sont ceux de la directive cadre sur l'eau, permettant de définir l'état physico-chimique et chimique de l'eau. Les méthodes d'impacts utilisées sont Impact 2002+, ReCiPe, ILCD, permettant d'évaluer les impacts sur l'eutrophisation, l'écotoxicité, l'acidification, et les dommages sur les écosystèmes et la santé humaine. Les scores de dommages sont aussi utilisés pour calculer un indicateur simplifié d'empreinte eau le Water Impact Index (WIIX).

Les résultats permettent de comparer les scores de dommages avec les états physico-chimiques et chimiques des ressources en eau. Cette comparaison montre l'intérêt de l'ACV d'agréger plusieurs polluants en un score de dommage par rapport à la définition des états de la directive cadre sur l'eau qui est basée sur des valeurs de seuils de polluants à ne pas dépasser. Ils permettent aussi de classer les différents types de flux d'eau urbains selon leurs scores de dommage et leurs indices de qualité calculés avec la méthode WIIX avancée. Cette classification est utilisée par ailleurs dans le chapitre suivant pour gérer la qualité des flux d'eau dans un système d'eau urbain.

Chapitre 5. WaLA, un modèle pour l'analyse du cycle de vie des systèmes d'eau urbains : cadre conceptuel et formalisme pour une approche modulaire

Ce chapitre représentant le cœur de la thèse vise à élaborer un cadre conceptuel, un formalisme et un modèle associé pour réaliser l'ACV de systèmes d'eau urbains et de leurs scénarios prospectifs. Le modèle, nommé WaLA pour « Water systems Lifecycle Assessment », a pour but de résoudre les questions méthodologiques identifiées dans le chapitre 2 et d'intégrer les développements méthodologiques des chapitres 3 et 4. Comme la construction de modèles de systèmes d'eau urbains est complexe si plusieurs scénarios sont à évaluer, le modèle proposé réduit la complexité du système, tout en assurant une bonne représentation du point de vue de l'ACV.

Le cadre proposé est basé sur la définition d'un composant générique qui peut représenter les technologies et les usagers, et qui sont connectés à des ressources en eau spécifiques (cf. Figure c).

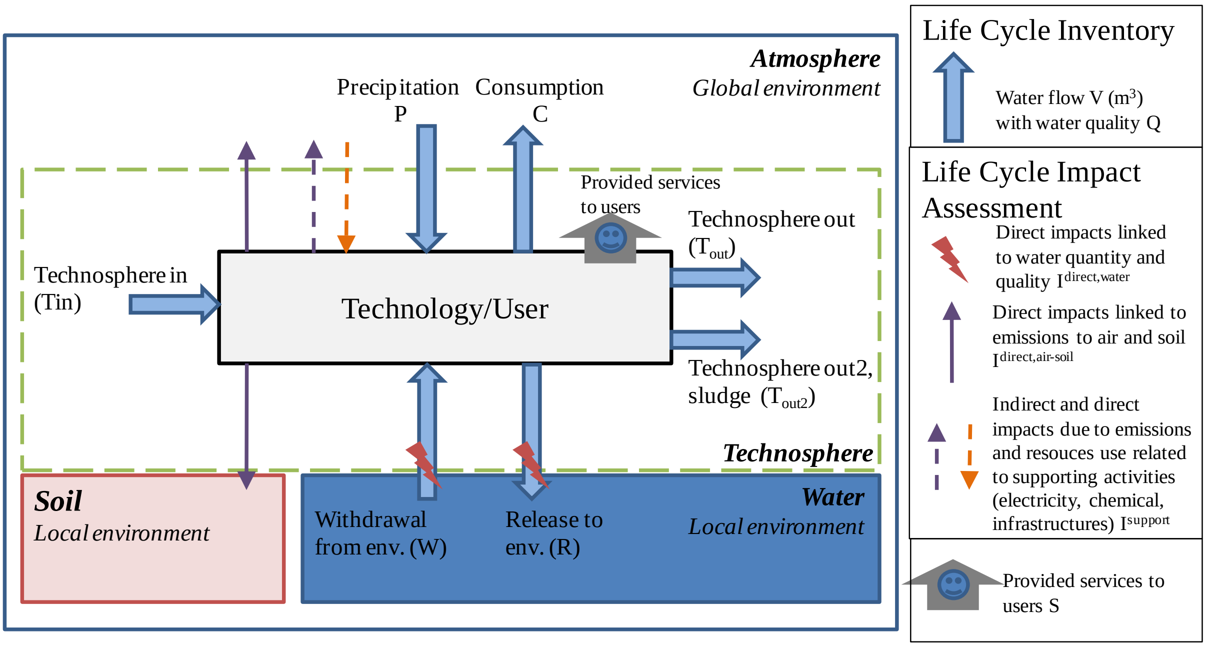
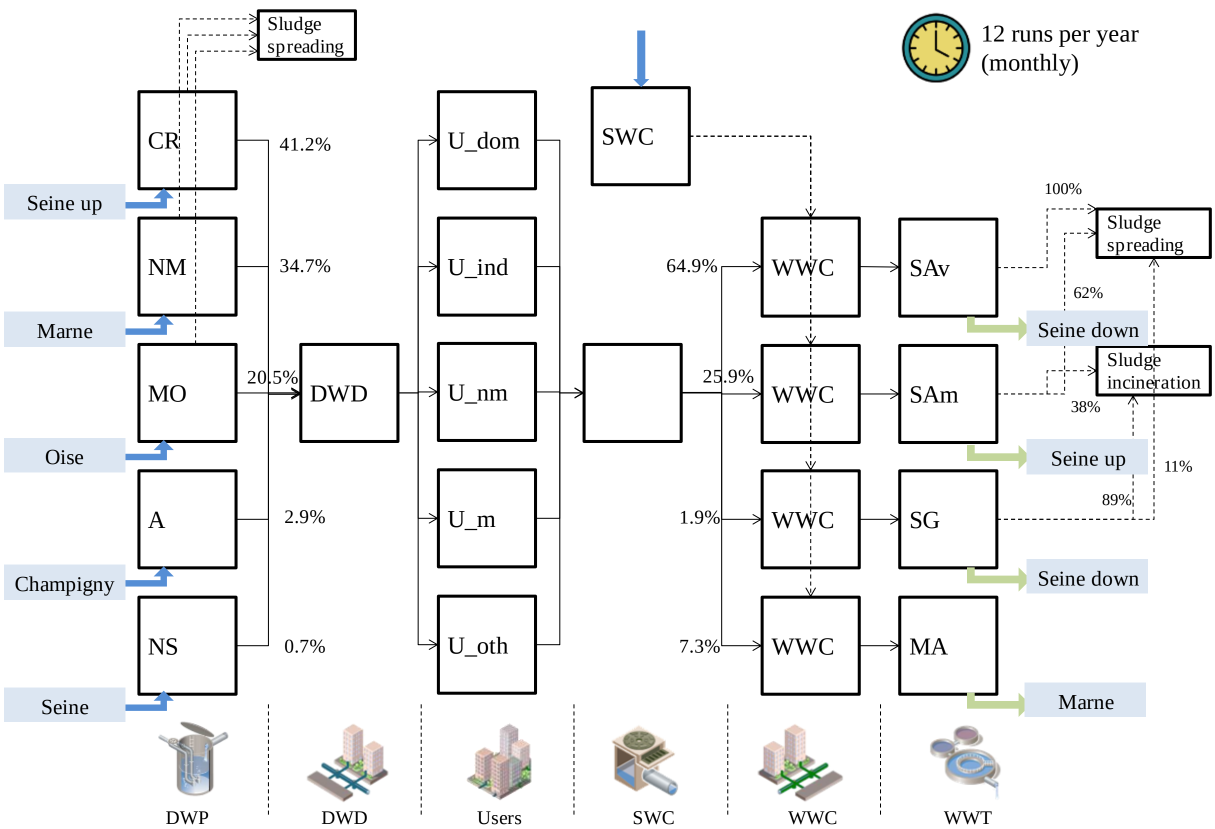


Figure c. Description des flux d'eau et des impacts et services associés à un composant générique

Ces composants permettent de calculer les flux d'eau en entrée et en sortie (quantité et qualité), ainsi que les impacts associés dus aux prélèvements et rejets d'eau directs et aux activités de support (énergie, produits chimiques, infrastructures). Ces composants peuvent être reliés entre eux d'une manière modulaire, afin de construire un scénario de système d'eau urbain. Le modèle calcule les impacts du cycle de vie et les services fournis aux usagers, tels que définis par le scénario, et pour un pas de temps mensuel. En effet, le système étudié est multifonctionnel selon le cadre de l'ACV territoriale : plusieurs types d'usagers sont pris en compte (ex. usagers domestiques, industriels etc.). Ceci permet de calculer des ratios d'impacts sur services rendus (ex. impact/habitant) qui sont utiles pour le diagnostic ou la comparaison de différentes alternatives. Le modèle est mis en œuvre dans une interface Matlab/Simulink grâce à la programmation orientée objet. L'applicabilité du modèle est démontrée en utilisant une étude de cas virtuelle basée sur des processus ecoinvent (Doka, 2009).

Chapitre 6. WaLA, un modèle pour l'analyse du cycle de vie des systèmes d'eau urbain : mise en œuvre pour l'évaluation de scénarios de gestion de l'eau dans la banlieue parisienne

Le modèle WaLA est appliqué à un cas d'étude: le système d'eau urbain de la banlieue parisienne (périmètre géographique du syndicat des eaux d'Île-de-France – SEDIF), en France. Ce cas d'étude vise à vérifier la capacité du modèle à évaluer les impacts environnementaux des scénarios de gestion de l'eau et à fournir des indicateurs appropriés aux décideurs. Les scénarios étudiés prennent en compte certaines tendances futures qui influent sur le système (ex., l'évolution de la demande en eau ou l'augmentation du stress hydrique) ou à des réponses de décisions sur la gestion de l'eau (par exemple, le choix des ressources en eau et des technologies). Un scénario de référence pour l'année 2012 est établi, décrivant l'état actuel du système d'eau urbain. Ce système comprend environ 4,3 millions d'habitants pour une demande en eau globale de 236 millions de mètre cube. Cinq types d'usagers sont considérés (domestiques, industries, services marchands, services non marchands, et autres). Quatre usines d'eau potable prélevant l'eau dans trois rivières différentes (Seine, Marne et Oise) et une nappe (Champigny), ainsi que quatre stations de traitement des eaux usées constituent les technologies du système. La Figure d représente graphiquement le scénario actuel, tel qu'il est implémenté dans l'outil Matlab/Simulink.



CR=Choisy-le-Roi, NM=Neuilly-sur-Marne, MO=Mery-sur-Oise, A=Arvigny, NS=Neuilly-sur-Seine, U_dom=usagers domestiques, U_ind=usagers industriels, U_nm=services non marchands, U_m=services marchands, U_oth=autres usagers, SAv= Seine Aval, SAm= Seine Amont, SG=Seine Grésillons, MA=Marne Aval. DWP : Production d'eau potable, DWD : Distribution d'eau potable, SWC : Stormwater collection WWC : Wastewater collection, WWT : Wastewater treatment

Figure d. Représentation graphique du scenario actuel, avec les composants du systems, les flux d'eau dans la technosphère (flèches noires) et les prélèvements et rejets principaux (flèches bleues et vertes).

Trois scénarios sont définis à l'horizon 2022 afin d'évaluer des choix de gestion à court terme. Huit scénarios sont étudiés à l'horizon 2050 étudiant des changements importants concernant les usagers (évolution de la population, de la demande en eau), les ressources (choix des ressources, effets du changement climatique sur le stress hydriques) et les technologies. Tous les scénarios sont construits facilement dans une interface Matlab/Simulink, tel que présenté dans le chapitre 5. Les méthodes choisies pour l'évaluation des impacts du cycle de vie sont Impact 2002+ endpoint, ILCD midpoint et les impacts de la privation de l'eau à l'échelle du bassin versant (défini dans le chapitre 3). Les résultats d'impact du scénario actuel permettent d'évaluer les contributions par rapport aux technologies, montrant que la majorité des impacts sont générés par les usines de traitement des eaux usées. L'analyse des contributions entre impacts directs et indirects (associés aux activités de support), montrent que les impacts locaux (ex. eutrophisation, écotoxicité) sont dominés par les impacts directs du système, alors que les impacts globaux (tels que le réchauffement global et l'épuisement des ressources fossiles) sont dus aux activités de support. L'analyse des résultats mensuels de la privation d'eau montre que la majorité des impacts a lieu pendant les mois d'été. L'étude des scénarios prospectifs démontre la capacité du modèle à fournir des informations pertinentes et utiles quant aux politiques futures. Les scénarios proposés étudient principalement des changements au niveau de la production et de la distribution d'eau potable et montrent que les gestionnaires de ces services ont peu d'influence sur la majorité des impacts environnementaux (en comparaison au traitement des eaux usées), mais ont une grande influence sur l'impact de privation d'eau du système en choisissant les ressources en eau dans le bassin versant.

Sur la base de cette étude de cas, les apports et les limites du modèle WaLA sont identifiés. Les principales nouveautés de ce modèle sont sa modularité et la prise en compte des usagers de l'eau et des ressources en eau (à travers l'évaluation affinée des impacts liés à l'eau). Aussi, le formalisme du modèle, qui est programmé selon une méthode orientée objet, permet son appropriation par de futurs développeurs et son implémentation dans des logiciels autres que Matlab/Simulink. Toutefois, en l'état actuel de son développement, certaines limites du modèle demeurent, notamment sur la gestion du bilan équilibré des polluants à l'échelle des composants, et sur la gestion des incertitudes. Ces limites nécessitent de nouveaux développements. Enfin, la collecte de données d'inventaires pour des technologies émergentes et pour d'autres usagers, ainsi que l'application de nouvelles méthodes d'EICV spatialisées et liées à la qualité de l'eau, pourraient améliorer la fiabilité et l'exhaustivité du modèle.

Perspectives et conclusion

L'objectif principal de la thèse a consisté à développer un modèle d'évaluation environnementale multicritère de scénarios de gestion (actuels ou prospectifs) de systèmes d'eau urbains. L'hypothèse de recherche, i.e. une méthode peut être développée afin d'évaluer facilement et régulièrement des scénarios de gestion de systèmes d'eau urbains dans le cadre de l'ACV, a été validée à travers les développements méthodologiques menés dans les cinq sous objectifs de la thèse détaillés ci-dessus.

Les résultats de cette thèse débouchent sur des perspectives de travaux à la fois scientifiques et opérationnels. D'un point de vue scientifique, des développements méthodologiques doivent encore être menés pour évaluer les facteurs de devenir liés à l'utilisation des ressources en eau selon un modèle mécaniste et consensuel. La méthodologie d'évaluation de la privation de l'eau à l'échelle des sous-bassins conçue au cours de la thèse pourrait alors être intégrée dans ce type d'approche. D'un point de vue plus opérationnel, cette méthodologie basée sur les effets cascades reste encore à déployer sur les bassins versants du monde entier pour pouvoir être utilisée en routine dans les ACV. En ce qui concerne le modèle WaLA, son application à d'autres études de cas dans des contextes différents serait un pas de plus pour démontrer sa faisabilité et son intérêt. Enfin, l'appropriation des résultats par les parties prenantes et les décideurs, et leur contribution dans un processus de prise de décision restent des défis importants pour les sciences de gestion.

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Abstract

To improve water management at the scale of large cities is a real challenge. However, the quantification of flows and environmental impacts linked to water use are not yet sufficiently developed. This is the objective of the thesis: “how to model complex urban water system of a megacity for assessing its environmental impacts in relation to the provided services to water users, within the life cycle assessment (LCA) framework?” The core of the thesis is the development of a generic framework defining water flows and environmental impacts associated with 3 categories of items – i.e., water technologies, water users and water resources – from a LCA point of view. The UWS model (termed WaLA) is built through a modular approach allowing the interoperation of these three components in an integrated way. The model provides indicators of impacts on services which may be useful to decision makers and stakeholders. It simplifies the evaluation of forecasting scenarios and decreases the complexity of the urban water system while ensuring its good representation from a LCA perspective. In addition to this main objective, the thesis also aims at refining water use impact indicators at a relevant scale for UWS. A methodology that assesses water deprivation at the sub-river basin scale in life cycle impact assessment (LCIA) integrating downstream cascade effects has been developed. It allows differentiating the withdrawal and release locations within a same river basin. The WaLA model and its associated indicators are applied to assess the environmental impacts of the water system of a Paris suburban area (perimeter of Syndicat des Eaux d’Île-de-France). It shows the interest and the applicability of the model for assessing and comparing baseline and forecasting scenarios.

Key words: life cycle assessment (LCA), environmental impacts, urban water system, water use, water footprint, drinking water, wastewater, modeling, suburbs of Paris

Résumé

La gestion intégrée de l'eau à l'échelle des grandes villes est un réel défi. Cependant, la quantification des flux et des impacts environnementaux liés à l'utilisation de l'eau n'est pas encore suffisamment développée. Dans ce contexte, la question de recherche de la thèse est: "comment modéliser le système d'eau urbain complexe d'une mégapole pour l'évaluation de ses impacts sur l'environnement et des services fournis aux usagers de l'eau, dans le cadre de l'analyse du cycle de vie (ACV)?" Le cœur de la thèse est le développement d'un cadre général définissant les flux d'eau et les impacts environnementaux associés aux trois composants principaux du système d'eau urbain, à savoir, les technologies de l'eau, les usagers de l'eau et les ressources en eau. Le modèle proposé de système d'eau urbain (nommé WaLA) se construit à travers une approche modulaire permettant l'interopérabilité des trois composants. Le modèle fournit des indicateurs d'impacts et de services rendus qui peuvent être utiles aux décideurs et aux parties prenantes. Il simplifie l'évaluation des scénarios et diminue la complexité du système tout en assurant sa bonne représentation du point de vue de l'ACV. En plus de cet objectif principal, la thèse vise à raffiner les indicateurs d'impact sur la privation d'eau afin qu'ils soient pertinents pour les systèmes d'eau urbains. Une méthode qui permet d'évaluer la privation d'eau à l'échelle du sous bassin versant en intégrant les effets en aval a ainsi été développée. Cette méthode permet de différencier les impacts selon les points de prélèvements et de rejets dans un même bassin versant. Enfin, le modèle WaLA et les indicateurs associés sont mis en œuvre pour évaluer les impacts environnementaux du système d'eau urbain de la banlieue parisienne (périmètre du Syndicat des Eaux d'Ile-de-France). L'intérêt et l'applicabilité du modèle pour évaluer et comparer des scénarios actuels et prévisionnels sont ainsi démontrés.

Mots clés : analyse du cycle de vie (ACV), impacts environnementaux, système d’eau urbain, utilisation de l’eau, empreinte eau, eau potable, eaux usées, modélisation, banlieue parisienne