WATER-ENERGY-CARBON NEXUS: A SYSTEM DYNAMICS APPROACH FOR ASSESSING URBAN WATER SYSTEMS

by

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Abstract

Water, energy, and carbon emissions of Urban Water Systems (UWSs) are intertwined and have complex interactions forming a water-energy-carbon (WEC) nexus. A comprehensive methodology to quantify dynamic WEC nexus is required. The main objective of this research is to develop a decision support system (DSS) for assessing the WEC nexus for sustainable planning and management of UWSs.

This research has been accomplished in five distinct steps. In the first step, key Sustainability Performance Indicators (SPIs) of small to medium-sized UWSs have been identified. The SPIs related to water consumption, energy use, carbon emissions, and cost were used for developing the DSS. In the second step, a WEC DSS has been developed for an operational phase of an UWS using system dynamics and then applied to the City of Penticton. The highest energy consumer was found to be indoor hot water use in the city. In the third step, a framework has been developed to study the impacts of neighbourhood densification on the WEC nexus. A higher net residential density will result in lower per capita water demand, energy use, net carbon emissions, and life cycle cost of water distribution system. The proposed framework provides an optimal residential density and energy intensity of water distribution, which can be used as inputs to the WEC DSS. In the fourth step, microbial water quality guidelines for reclaimed water have been developed for various non-potable urban reuses. Moreover, the FitWater tool has been developed for evaluating fit-for-purpose wastewater treatment and reuse potentials based on cost, health risk, and the WEC nexus. The outputs of FitWater can be used as inputs to the WEC DSS. In the last step, the economics of the WEC nexus of net-zero water communities has been analyzed using the WEC model.

The DSS developed based on this research is capable of quantifying dynamic water consumption, energy use, carbon emissions, and the cost of UWSs. The DSS can analyze different WEC-based interventions. The DSS can be used by utilities, urban developers, and policy makers for long-term planning of urban water in communities.

Preface

I, Gyan Kumar Chhipi Shrestha, conceived and developed all the contents in this thesis under the supervision of Dr. Kasun Hewage and Dr. Rehan Sadiq. I wrote all the manuscripts and both supervisors have reviewed them and provided feedback to improve the manuscripts and the thesis. Altogether eight journal papers and one conference paper have been published or submitted for publication in peer-reviewed scientific journals and a conference proceeding as follows:

1. A version of Chapter 3 has been published in *Clean Technologies and Environmental Policy* journal entitled "'Socializing' sustainability: A critical review on current development status of social life cycle impact assessment method" (Chhipi-Shrestha et al. 2015a).
2. A version of Chapter 4 has been published in *Water Environment Research* journal entitled "Sustainability performance indicators for small to medium sized urban water systems: A selection process using Fuzzy-ELECTRE method" (Chhipi-Shrestha et al. 2017a).
3. A version of Chapter 5 has been published in the *ASCE Journal of Water Resources Planning and Management* entitled "Water-Energy-Carbon nexus modelling for an urban water system: A system dynamics approach" (Chhipi-Shrestha et al. 2017b).
4. A version of Chapter 6 has been published in the *Journal of Cleaner Production* entitled "Impacts of neighbourhood densification on water-energy-carbon nexus: Investigating water distribution and residential landscaping system" (Chhipi-Shrestha et al. 2017c).
5. A version of Chapter 7 has been published in the *Science of the Total Environment* journal entitled " Probabilistic risk-based investigation on microbial quality of reclaimed water for urban reuses" (Chhipi-Shrestha et al. 2017d).
6. A version of Chapter 8 has been submitted and is under review in the *Science of the Total Environment* journal entitled "Fit-for-purpose wastewater treatment: Conceptualization and development of decision support tool (I)" for possible publication (Chhipi-Shrestha et al. 2017e).

7. A version of Chapter 8 has been submitted and is under review in the *Science of the Total Environment* journal entitled “Fit-for-purpose wastewater treatment: Testing and implementation of decision support tool (II)” for possible publication (Chhipi-Shrestha et al. 2017f).
8. A version of Chapter 9 has been published in the proceeding of Canadian Society for Civil Engineering (CSCE) conference entitled “System dynamics modelling for an urban water system: net-zero water analysis for Peachland (BC)” (Chhipi-Shrestha et al. 2015b).
9. A version of Chapter 9 has been submitted and is under review in the *ASCE Journal of Sustainable Water in the Built Environment* entitled “Economic and energy efficiency of net-zero water communities: A system dynamics analysis” for possible publication (Chhipi-Shrestha et al. 2017g).

Table of Contents

List of Tables

List of Figures

List of Symbols

α	Safety factor (in pumping power equation)
α and r	Parameters referring to pathogen infectivity constant (in dose-response relationships)
$\tilde{A}$	Tilde (~) on "A "meaning a fuzzy number "A"
A_p and A_q	A pair of alternative with p, q = 1, 2, ..., m and, $p \neq q$
Cd	Cadmium
C^*_{pq}	Concordance index
Cu	Copper
d	Pathogen dose;
D^*_{pq}	Discordance index
dy/dx	Differentiate y with respect to x
E	Energy consumed
f	Friction loss
f_s	Susceptibility fraction
Hg	Mercury
l	Lowermost or lower value
m	Most probable or middle value
N ~ (μ, σ)	Normal distribution with mean (μ) and standard deviation (σ)
N_{50}	Median infective dose, i.e., the dose required to infect 50% of population
$P_{ill\|inf}$	Risk of disease given infection, i.e., morbidity;
P_{inf} (d)	Probability or risk of infection to an individual exposed to a single pathogen dose "d"
$P_{inf(A)}$ (d)	Annual probability or risk of an infection from "n" exposures per year due to a single pathogen dose "d"
Pb	Lead
T~ (l, m, u)	Triangular distribution with lower most, most probable, and uppermost value
u	uppermost or upper value
yr	Year
γ	Specific weight of water
η	Efficiency
η_t	Mechanical transmission efficiency
ρ	Spearman's rank correlation coefficient
$	Dollar
$\int_0^t f(t)\,dt$	Definite integral of $f(t)$ from time 0 to t

List of Abbreviations

2-D MCA	Two-dimensional Monte Carlo Analysis
ABM	Agent-based Modelling
Alt.	Alternative
AS	Activated sludge
Avg.	Average
BC	British Columbia
BNR	Biological nutrient removal
BWA	Boil water advisory
CDF	Cumulative density functions
CE	Carbon emissions
CF	Coagulation and flocculation (in wastewater process in FitWater)
CF	Carbon footprint (in WEC model)
CI	Commercial and institutional
CII	Commercial, institutional, and industrial
Cl_2	Chlorination
CLD	Causal loop diagram
CO_2e	Carbon dioxide equivalent
Consvn.	Conservation
CoP	City of Penticton
DALYs	Disability-Adjusted Life-Years
DBPC	Disease burden per case
DBPs	Disinfection Byproducts
DMB	Decision Maker's Boundary
DSS	Decision Support System
DST	Decision Support Tool
DU	Dwelling units
DWS	Drinking Water System
E. coli	*Escherichia coli*
EC	Economic sustainability performance indicator
EE	Embodied energy
EF	Ecological footprint
EI	Energy intensity
ELECTRE	ELimination Et Choix Traduisant la REalite´ (Elimination and Choice Translating Reality)
EN	Environmental sustainability performance indicator
F-AHP	Fuzzy-Analytical Hierarchical Process
FBR	Fed Batch Reactor
g	Gram

gha	Global hectare
GHG	Greenhouse gases
GIS	Geographic Information System
ha	Hectare
IN	Institutional sustainability performance indicator
Insti.	Institutional
kg	Kilogram
kWh	Kilowatt-hour
L	Litre
L/p/day	Litres/person/day
LCA	Life cycle assessment
LCC	Life cycle cost
LCCA	Life cycle cost analysis
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCSA	Life cycle sustainability assessment
MBR	Membrane Bioreactor
MCS	Monte Carlo Simulation
MF	Microfiltration (in wastewater process)
MF	Multi-family (in neighbourhood densification)
ML	Million Litres
Mm^3	Million cubic metre
MWh	Meghwatt-hour
MWR	Municipal Wastewater Regulation
ND	Not Detected
NPV	Net present value
NPW	Net-Positive Water
NR	Non-residential
NWWBI	National Water and Wastewater Benchmarking Initiative
NZW	Net-Zero Water
O_3	Ozonation
Oper.	Operating
PPCPs	Pharmaceuticals and Personal Care Products
PS	Primary Sedimentation
PSR	Pressure-State-Response (framework)
Q	Discharge
R	Rank
r^2	Coefficient of determination
REUM	Residential End-Use Model
RME	Reasonable Maximum Estimate

RO	Reverse osmosis
RW	Reclaimed water
SBR	Sequencing Batch Reactor
SDM	System dynamics model
SF	Surface filtration (in wastewater process)
SF	Single-family (in neighbourhood densification)
S-LCA	Social Life Cycle Assessment
SMUWS	Small to Medium-sized Urban Water System
SO	Social sustainability performance indicator
SP	Sludge processing
SPI	Sustainability Performance Indicator
sqft	Square feet
SWS	Storm Water System
tCO_2e	Ton carbon dioxide equivalent
TE	Technical sustainability performance indicator
TFN	Triangular Fuzzy Number
UF	Ultrafiltration
UK	United Kingdom
US EPA	United States Environment Protection Agency
US	United States
UV	Ultraviolet disinfection
UWS	Urban Water System
W	Water
WDRLS	Water Distribution and Residential Landscaping System
WDS	Water Distribution System
WEC nexus	Water-Energy-Carbon nexus
WECCo	Water-Energy-Carbon-Cost (triangle)
WF	Water footprint
WHO	World Health Organization
WQ	Water quality
WRCC	Water Resource Carrying Capacity
WT	Water treatment
WW	Wastewater
WWS	Wastewater System
WWT	Wastewater Treatment
WWTP	Wastewater treatment plant

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My special thanks to the dedicated library staff of the UBC Okanagan Library for helping me to get the necessary documents on time. Even in the age of e-library, they proved themselves invaluable in searching and getting rare documents on time through inter-library loans. This service is highly praiseworthy and was very helpful in my research. I would also like to thank Patty Wellborn and Chris Bowerman for disseminating my research findings across the globe through UBC media.

Finally, I would like to acknowledge my exceptional parents for their patience and constant support throughout my studies including this research work. Special thanks are owed to my beloved wife Ambika Shrestha for her constant support, caring, and inspiration to overcome the stressful environment in order to accomplish this work.

Dedication

To my father Jagat Hari,
mother Sagat Maya, and
wife Ambika

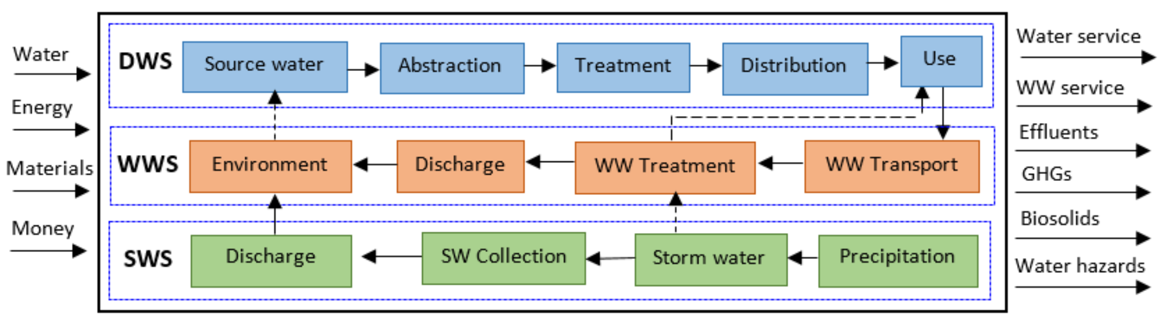
Chapter 1 Introduction

1.1 Background and motivation

The world's urban population is more than half (~54%) of the total population and is rapidly increasing (UN DESA 2014). In Canada, the urban population is very high (~ 81%) and is continuing to grow (Statistics Canada 2014a). The growing population requires a large volume of water served by urban water supply systems. Urban water processes, such as water abstraction, treatment, distribution, wastewater treatment, disposal, and storm water drainage are essential in any urban area. These processes are necessary for the human consumption of safe water and reduction of environmental impacts due to wastewater discharge (Termes-Rifé et al., 2013). These human regulated urban water processes constitute a *human hydrologic cycle* (Bagley et al. 2005), or simply an urban water system (UWS).

1.1.1 UWS metabolism

An urban water system consists of a drinking water system (DWS), a wastewater system (WWS), and a storm water system (SWS) (Figure 1.1).



Note: DWS: Drinking Water System, WWS: Wastewater System, SWS: Storm Water System

Figure 1.1 Major inputs and outputs of urban water systems

UWSs consume resources (inputs), such as water, energy, materials (e.g., water treatment chemicals), and financial resources to produce outputs like water and wastewater services. Similar to any other built infrastructure, UWSs release greenhouse gases (GHGs), discharge effluents (potentially containing heavy metals and other harmful materials), generate solid waste (biosolids), and pose hazards like storm water flooding in some instances. This phenomenon is referred to as the urban metabolism (Novotny 2012) for a water sector.

Globally, total freshwater withdrawals are estimated to have increased by around 1% per year from 1987 to 2000 based on the FAO AQUASTAT database, and the present rate is expected to be the same considering a similar overall trend (UN-WWAP 2014). Municipal water accounts for 12% of the total withdrawals with industrial and agricultural water withdrawals accounting for 19% and 69% respectively (FAO 2014). Similar to the global outlook, Canada has 13% of total water withdrawals by municipal sector; however, the thermal power generation, manufacturing, agriculture, mining, and oil and gas account for 69%, 10%, 6%, 2%, and 1% of total water withdrawals respectively (Environment Canada 2014a). Residential water use is a major component of municipal water use and accounts for more than 50% of all municipal water use in Canada (Environment Canada 2004).

Water consumption is highly dependent on the geography and the development status of a country. For instance, domestic water consumption varies largely from 135 L/p/day in Israel to 343 L/p/day in Canada, 490 L/p/day in British Columbia (Canada) and 675 L/p/day in the Okanagan Valley (BC, Canada) as shown in Table 1.1. Even domestic water use reaches upto 1,000 L/p/day in the summer in the Okanagan Valley (OBWB 2011). Canada has a very high domestic water consumption rate that is also referred to as overconsumption (Renzetti, 1999; Ma, 2014). Although Canada overall has abundant freshwater supplies, water supply shortages exist due to water quality and/or quantity issues in some communities. Approximately 26% of municipalities with water supply systems in place experienced water supply shortages from 1994 to 1999 due to droughts, infrastructure problems, and increased consumption (Environment Canada 2004).

Table 1.1 Domestic water consumption in the developed countries

Country	Consumption (L/p/day)
Israel	135
France	150
Sweden	200
Italy	250
United States	382
Canada	343
British Columbia (BC) (Canada)	490
Okanagan average (BC, Canada)	675
Okanagan summer average (BC)	1,000

Source: (Environment Canada 2014b; OBWB 2011)

1.1.2 UWS sustainability

Modern urban water systems are designed to provide clean drinking water, remove wastewater, and manage storm water without posing harm to the environment. Although UWSs achieve these first three fundamental requirements to a high degree, this sector is criticized from the sustainability perspective (Hellstro 2000). Moreover, Canadian communities have been facing pressing concerns of carbon mitigation and adaptation to global climate change in different sectors including urban water management (Maas 2009; Environment Canada 2014c; Government of BC 2012).

Current UWSs are usually linear in terms of urban metabolism, sometimes called the “take, make, waste approach”. This approach discourages water reuse and has become unsustainable (Daigger 2009). Linear systems result in a higher level of pollution, and also consume a greater amount of resources, such as water, air, and soil for the dilution and assimilation of residuals compared to closed systems based on recycling (Novotny 2012; Joustra and Yeh 2014a). Water reuse provides additional water that would otherwise be discarded from the system. Closed or semi-closed UWSs can be developed for achieving net-zero water with some water input from rainfall (Englehardt et al., 2013). Net-zero water refers to the balance of water demand and supply within a given areal boundary (Holtzhower et al., 2014). “Net-zero water limits the consumption of freshwater resources and returns water back to the same watershed so not to deplete the groundwater and surface water resources of that region in quantity or quality over the course of a year" (US Army 2011). However, the reclaimed water use in net-zero water may result in the exposure to various pathogens, such as virus, bacteria, and parasitic protozoa posing increased health risks to reclaimed water consumers (Schoen et al., 2014).

Urban water systems consume a significant amount of energy. The energy consumption of the treated water supply and wastewater management is about 3% of city energy use in the USA, but it can be as high as 20% in some states, for example in California (Novotny 2012). Energy is required in almost all stages of UWSs: water abstraction, treatment, distribution, use, wastewater collection, treatment, disposal and/or reuse. Usually, water and wastewater utilities have the largest expenditure in energy cost (Tuladhar et al., 2014). Moreover, energy use directly contributes to GHG emissions, or simply, carbon emissions. Energy use in an UWS can be significantly reduced by employing the use of efficient water appliances, water conservation strategies, and efficient water and wastewater management practices (Barry, 2007; Novotny, 2011). These strategies can be combined with renewable energy use, including wastewater energy recovery and onsite solar and wind energy generation for achieving net-zero energy and carbon emissions (Novotny 2011).

Urban water systems are financially sustainable when their revenues equal or exceed expenses (Rehan et al. 2011). At the minimum, financial resources (revenues) for operation and maintenance costs should be available to make UWSs functional (World Bank 2003). An UWS, particularly water reuse and energy efficiency improvement projects for net-zero water and net-zero energy should be economically sustainable. However, there are a number of challenges for the economic sustainability of these projects. These challenges can be classified as (a) high capital and operation costs, (b) unavailable or inadequate incentives associated with the conservation of water resources and the reduction of pollution, (c) no reward for avoided headwork in water abstraction, (d) relatively long payback period, and (e) low level of revenue from recycled water services (Listowski et al., 2013). Also, current water reuse strategies do not consider important social and environmental benefits and costs traditionally considered as intangible (Novotny 2012).

1.2 Research gap

Conventional centralized UWSs, specifically wastewater systems, have less flexibility in associated facilities to adapt to changes (e.g., high population fluctuation), a high and long-term capital investment (Bieker et al. 2010), and longer distance between a recovery station and potential users (Wang et al. 2008). On the other hand, household level wastewater treatment might be appropriate in low-density households, but not in densely populated urban areas due to operational risk and limited space availability (Bieker et al. 2010). To overcome the limitations of centralized wastewater treatment and decentralized household-level wastewater treatment, an intermediate scale of UWS could be appropriate (Asano et al. 2007; Bieker et al. 2010; CCME 2002; Zarski and Ancel 2012). This level is referred to as the small to medium-sized urban water system (SMUWS). Moreover, the number of studies defining sustainability performance indicators (SPIs) for an entire UWS is very limited and are most studies are confined to a performance assessment (of service) of individual water and wastewater utilities (CWWA 2009; Sydney Water 2013; Water UK 2011). On the other hand, available SPIs are established mainly for large UWSs (Foxon et al. 2002; Van Leeuwen et al. 2012; Van Leeuwen and Marques 2013) and cannot be adopted as is for a sustainability assessment of SMUWSs.

The decision makers of UWSs can have multiple alternatives for each stage in urban water planning and management. These alternatives may vary in their sustainability performance. In addition, the decision makers face increasing challenges as UWSs are under pressure from increasing populations (Lallana et al. 2001), lower household occupancy (Inman and Jeffrey 2006) , increasingly severe droughts and floods (IPCC 2014), higher prices of water and energy (Fagan et al., 2010), lifestyle changes related to technology, personal habits and affluence (Princen, 1999; Lallana et al., 2001), and lower overall sustainability (Hellstro 2000). Moreover, climate change will increase the variability of water availability in many regions of the world (IPCC 2014). These factors ultimately affect the sustainability of UWSs.

In particular, water, energy, and carbon emissions can be considered as major elements of urban water sustainability. These elements are intertwined and have complex interactions (Nair et al., 2014; PMSEIC, 2010; Maas, 2009; Kenway, 2013). This interconnection results in a complex web called the water-energy-carbon (WEC) nexus. Because of the tight linkages in a WEC nexus, decisions for one area could have inadvertent consequences on the other. These pervasive interactions require integrated solutions (PMSEIC, 2010; Nair et al., 2014). However, a comprehensive methodology and decision support system (DSS) to quantify the WEC nexus and its dynamic behavior in an UWS at the community level is lacking (Nair et al., 2014; Rothausen & Conway, 2011; Arora et al., 2013; Kenway, 2013; Kenway et al., 2011).

Water Distribution Systems (WDS) can consume significant amounts of energy and release GHGs (Hellstro 2000). Neighbourhood densification can reduce per capita water distribution infrastructure, land resources, and water demand (Duncan 1989; Filion 2008; Frank 1989; Gleick et al. 2003). Reduced water demand itself is associated with decreased upstream energy use. These interlinkages suggest that higher residential densities can have reduced water consumption, energy use, and carbon emissions per capita. However, a lesser amount of landscaping in dense residences results in reduced carbon sequestration too. Furthermore, the life cycle cost of WDSs may be lower for neighbourhoods of high residential density (Speir and Stephenson 2002). An integrated study of the WEC dynamics of water distribution and residential landscaping under neighbourhood densification could not be found in the published literature.

Reclaimed water use reduces freshwater withdrawal, enhancing water sustainability. However, reclaimed water use pose human health risks. These risks are primarily associated with pathogenic microorganisms (EPHC/NHMRC/NRMMC 2008). Unlike drinking water, no globally accepted standard guidelines exist for reclaimed water. National guidelines for many urban reuses are yet to be developed in numerous countries, including Canada. Based on the long-term goal of the Canadian federal government (Health Canada 2010), and the recommendations of WHO (WHO 2006a), further research is required for investigating and developing reclaimed water use guidelines for specific reuses in non-potable purposes other than toilet and urinal flushing in Canada.

In addition, several decision support tools (DST) are available and are in practice for the planning and operation of wastewater treatment plants, such as ECAM tool (GIZ/MENCBNS/IWA 2015), WEST tool (Stokes et al. 2011), QMRAspot (Schijven et al. 2011), and QMRAcatch (Schijven et al. 2015). However, no DSS is flexible and capable enough to evaluate the potential of wastewater treatment and reuse for different purposes simultaneously based on cost, health risk, energy use, carbon emissions, and the amount of reclaimed water production. Furthermore, decision support systems for analysing the site-specific potential of net-zero water incorporating cost are limited due to a newer concept (Joustra and Yeh 2014a). Net-zero water development may result in higher costs (Englehardt et al. 2013; Gassie et al. 2016; Wang and Zimmerman 2015) and energy use (Vieira et al. 2014; Wang and Zimmerman 2015).

1.3 Research objectives

The overall goal of this research is to improve the sustainability of urban water systems by developing a WEC-based decision support system (DSS). The proposed DSS can assist municipalities, urban developers, and policy makers to optimize UWSs in terms of water consumption, energy use, carbon emissions, health risk, and cost. In this research, a WEC-based DSS has been demonstrated using SMUWSs. The specific objectives of the research are as follows:

1. Conduct a state-of-the-art review of the existing sustainability performance indicators (SPIs) of small to medium-sized urban water systems (SMUWSs) and identify key SPIs.

2. Conceptualize, build, and validate a model and decision support system for optimizing the water-energy-carbon (WEC) nexus of SMUWSs.

3. Develop a framework to assess the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system.

4. Develop microbial water quality guidelines for reclaimed water use in various non-potable urban reuses and propose a tool to optimize life cycle cost and human health risk in conjunction with the WEC nexus for fit-for-purpose wastewater treatment.

5. Assess the economic and energy efficiency of SMUWSs in developing net-zero water in different climatic and topographic regions based on the developed WEC model.

1.4 Thesis organization

Research Objectives 1, 2, 3, 4, and 5 have been achieved in Chapters 3, 4, 5, 6, 7, 8, and 9. The conclusions and recommendations of the thesis are provided in Chapter 10. The organization of the chapters is shown in Figure 1.2.

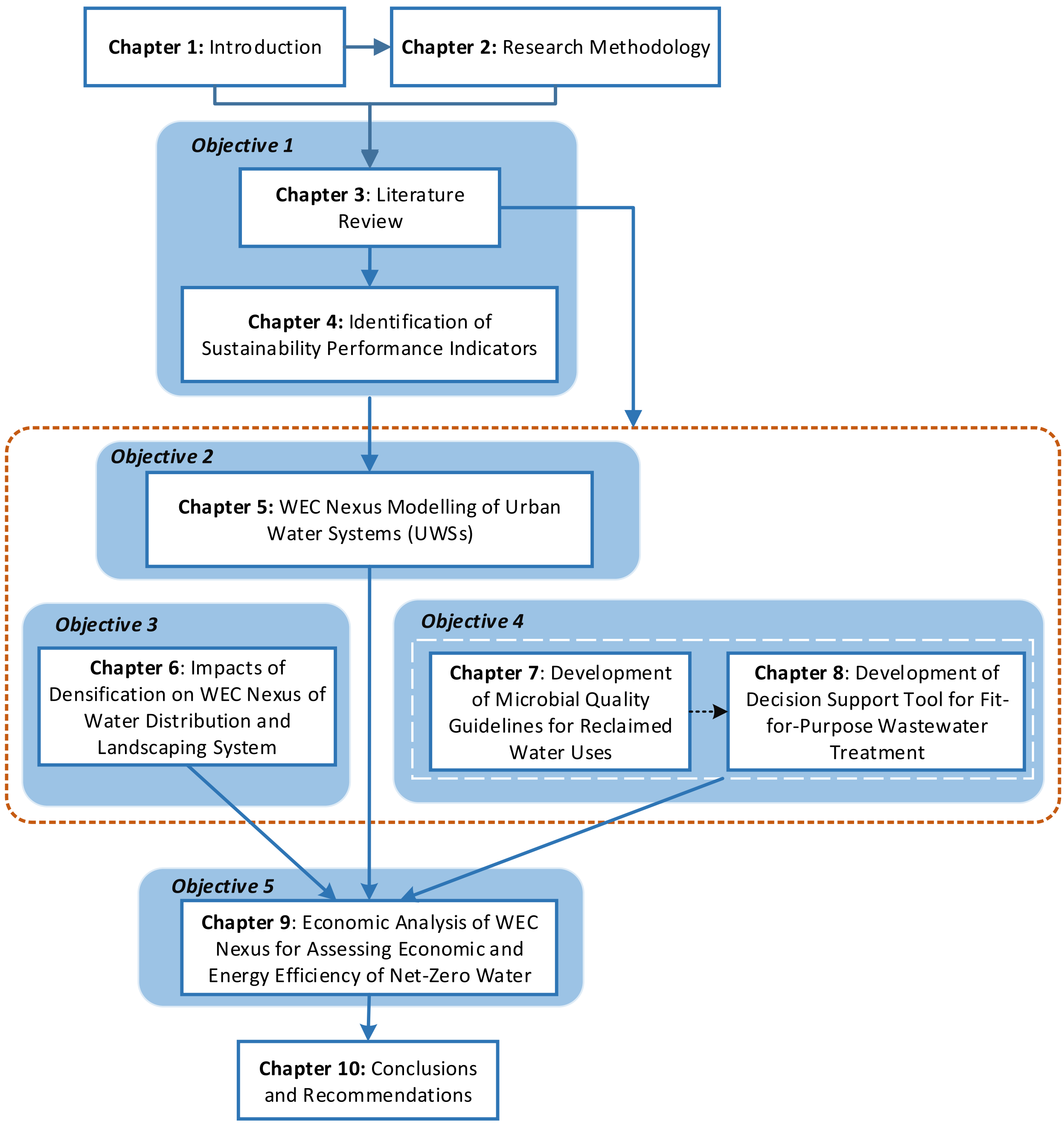


Figure 1.2 Thesis structure and organization

1.5 Meta language

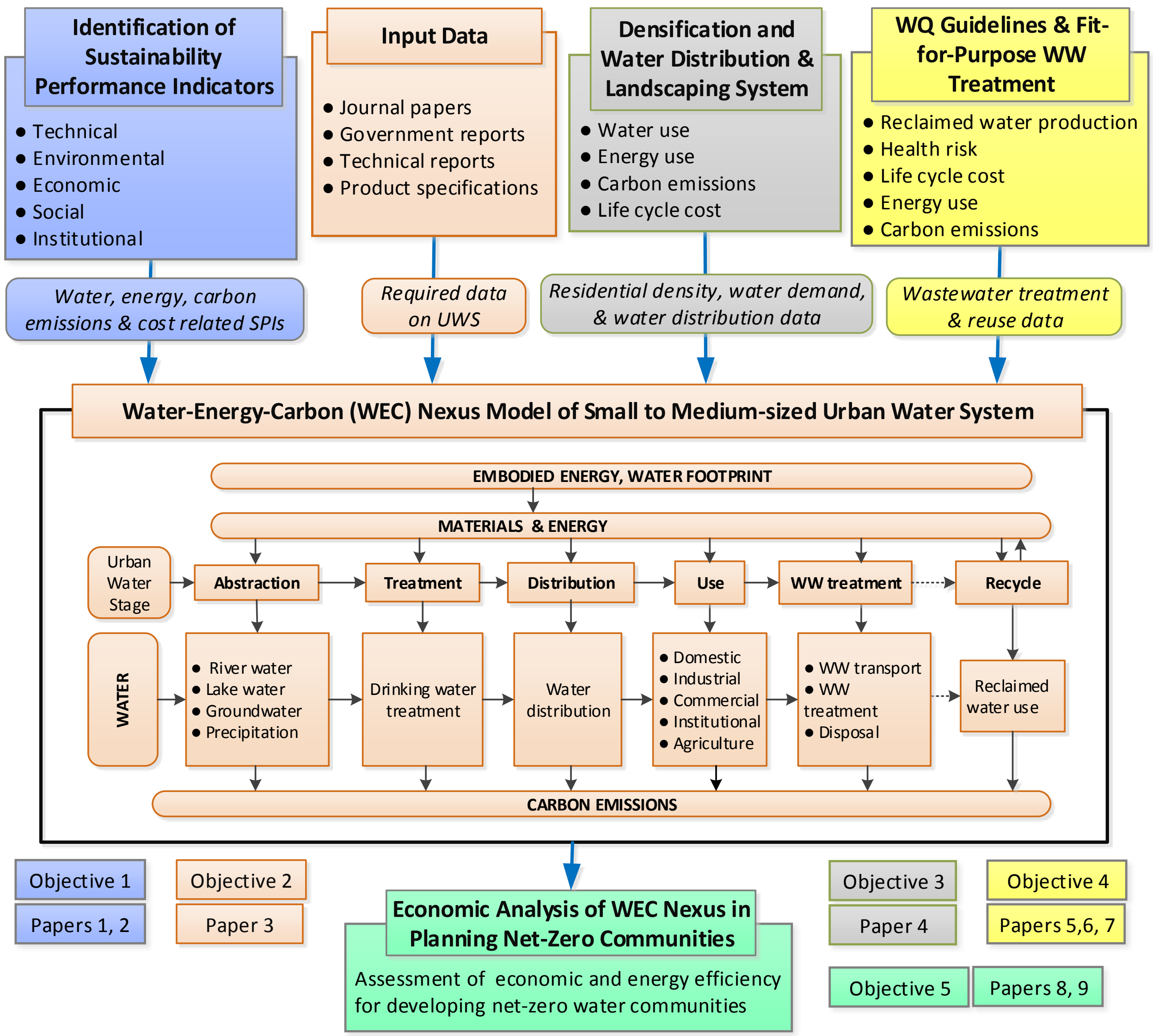
This thesis has used specific, technical vocabularies that have widely accepted definitions in the scientific and engineering community. However, certain principles and terminologies used in the thesis can have broad meanings. Such terms are specifically defined for the purpose of this thesis to ensure consistent understanding of the work as follows.

- *Decision support system (DSS)*: Represents a system of model, tool, and/or framework to aid decision making (e.g., WEC decision support system). It may comprise models, modules, sub-models, calculation tools (e.g., FitWater), and frameworks and is executable.
- *Framework*: Refers to holistic methods (e.g., framework for estimating WEC impact of neighbourhood densification). A framework can be a part of a DSS for a certain component of an urban water system.
- *Tool* : It is an executable model, framework, or a set of methods (e.g., FitWater tool) for a component/s of an UWS. A tool can be a part of a DSS for a certain component of an UWS.
- *Model*: It is a mathematical representation of a system or its component(s) (e.g., WEC model).
- *Technique*, *method*, and *methodology*: These terms have been used interchangeably for applying mathematical and statistical procedures.
- *Reclaimed water*, *recycled water,* and *reused water*: These terms have been used interchangeably referring to treated wastewater, rainwater, and/or storm water, which meet specific water quality criteria for beneficial uses.
- *Wastewater reuse*: This term has been used specifically to refer the use of reclaimed water produced by the treatment of wastewater.
- *Community*: This term has been used to describe the spatial scale served by small to medium-sized urban water systems.
- *Dollar values ($)*: Refer to Canadian dollars.

Chapter 2 Research Methodology

This chapter contains a brief description of research methodology, which was used to achieve the research objectives. An overview of methodology to achieve each objective is given in the following sections and the detailed methodology is presented in individual chapters.

A research framework (Figure 2.1) shows various methods and their interconnections under different objectives of this research. Several sustainability performance indicators (SPIs) have been identified to assess the sustainability of SMUWSs (Objective 1) and then a system dynamics-based WEC model for SMUWSs has been developed (Objective 2). The SPIs related to WEC nexus, cost, and health risk have been used in the WEC model. As important inputs to the WEC model, optimal residential density, energy use by a water distribution system, and water consumption by an entire community, including residential landscaping, were estimated (Objective 3). The microbial quality guidelines for reclaimed water in urban reuses have been proposed and a tool, called FitWater, has been developed for evaluating fit-for-purpose wastewater treatment and reuse potential (Objective 4). FitWater provides specific input data: reclaimed water quantity, life cycle cost, energy use, and carbon emissions of wastewater treatment and reclaimed water distribution for the WEC model. Finally, an economic analysis of the WEC nexus has been performed by adding a cost module in the developed WEC model. The cost-embedded WEC model has been used for assessing the economic and energy efficiency of developing net-zero water (Objective 5) to enhance the sustainability of SMUWSs.



Note: Objective # and paper # refer to the part of the framework with the similar background colour. Paper # corresponds to the list provided in preface.

Figure 2.1 Research methodology framework

2.1 Objective 1

The SPIs for assessing the sustainability of SMUWSs were initially identified from the critical review of indicators (SPIs). The SPIs belong to one of the sustainability dimensions: technical, environmental, economic, social, and institutional dimensions. These SPIs were further evaluated based on four criteria: relevance (importance) to sustainability, measurability, data availability, and comparability. Each SPI was rated using the Likert type scale. The relevance criteria was rated using a 5-point linguistic scale (very high, high, medium, low, and very low), whereas the measurability, data availability, and comparability criteria were measured using a 3-point linguistic scale (high, medium, and low). The relevance criteria was classified into five categories to capture the wide variability of rating provided by experts, whereas other three criteria were assssed more objectively having less variability in rating for which three categories were used. A multi-criteria decision analysis (MCDA) technique, called Fuzzy- ELECTRE I (Elimination and Choice Translating Reality I) was applied based on the four criteria in order to rank and select key SPIs. The identified SPIs are specific to the technical, environmental, economic, social, and institutional dimensions of sustainability.

The identified key SIPs can be used to develop an urban water sustainability index. However, the SPIs related to water consumption, energy use, carbon emissions, and life cycle cost of SMUWSs have been used in modelling the WEC model. The SPIs are variables, which are used in the consuctrion of the WEC model under Objective 2.

2.2 Objective 2

A WEC model for SMUWSs has been developed using system dynamics to assist municipalities, urban developers, and policy makers for neighbourhood water planning and management. The WEC model used several parameters and variables in modelling, including the SPIs related to water consumption, energy use, and carbon emissions indentified under Objective 1. The model comprises urban water stages: water abstraction, treatment, distribution, use, wastewater treatment, and water recycling if any. Water can be abstracted from river, lake, groundwater, and/or harvested from rainfall. The water consumption, energy use, and carbon emissions from each urban water stage have been included in the model. The water users comprise residential, commercial, institutional, and industrial sectors in the model. The dynamic WEC model and decision support system is for an operational phase of an UWS.

The required input data were collected from journal papers, government reports, technical reports, and product specifications. The model was validated and applied to an existing SMUWS. The necessary data can be classified into three levels: regional (R) containing municipal (site-specific) and regional data; national (N), and global (G). For example, required data (R, N, and

G) for the application of the developed DSS to the City of Penticton is given in in Appendix B.2 (Table B.1).

The developed WEC model is a broad framework consisting many variables affecting the WEC nexus of urban water systems. For the WEC model, an optimal residential density of a neighbourhood can be estimated using the framework developed under Objective 3. Also, the reclaimed water quality guidelines proposed under Objective 4 can be used to guide users in finding an appropriate level of wastewater treatment required for various reuse applications. Specifically, the developed tool under Objective 4, called FitWater, can be used to estimate the values of variables, such as quantity of reclaimed water, energy use of wastewater treatment, and the realted carbon emissions. These data can be used as inputs for the WEC model developed under Objective 2.

2.3 Objective 3

A framework has been proposed to study the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system. For this study, water demand was estimated for each neighbourhood design and a water distribution system (WDS) was designed for the corresponding neighbourhood design. The energy use by WDSs was estimated using the prepared WDS design. The related carbon emissions of energy use were estimated using the carbon emission factor of energy source (electricity). The carbon sequestration by residential landscaping, especially by trees, shrubs, and soil was estimated based on their annual carbon sequestration rates. The net carbon emissions were estimated as the balance of the carbon emissions and carbon sequestration. The water consumption, energy use, and net carbon emissions were aggregated by converting them into a common measurement unit – ecological footprint. In addition to the WEC nexus, the study has included the life cycle cost (LCC) of WDSs. The LCC was estimated as the sum of capital cost, operation cost, and repair and replacement cost of WDSs. The WEC nexus-based results were compared with the LCC.

A framework has been proposed to estimate the WEC-based optimal density. The framework was applied to a planned neighbourhood to demonstrate the effectiveness of the methodology. The framework can be used to estimate optimal residential density to be used as an input to the

WEC model of Objective 2. In addition, the estimated water demand, energy use, and net carbon emissions of water distribution can be used as inputs to the WEC model of Objective 2.

2.4 Objective 4

Probabilistic risk-based guidelines have been proposed for microbial quality of reclaimed water in various non-potable urban reuses. Health risk was estimated using quantitative microbial risk assessment. The estimation used two-dimensional Monte Carlo simulation to characterize variability and uncertainty in input data. The proposed guidelines were successfully applied to existing wastewater treatment effluents in the Okanagan Valley (BC, Canada). The guidelines help to identify an appropriate level of wastewater treatment required for various reuse applications. The level of treatment determines energy use and related carbon emissions from wastewater treatment plants.

In addition, a tool, called FitWater, has been developed for the evaluation of fit-for-purpose wastewater treatment and reuse potential in communities. The evaluation is based on the criteria: life cycle cost, health risk of reclaimed water, reclaimed water quantity, energy use, and the related carbon emissions. Uncertainty analysis was performed using probabilistic and fuzzy-based methods. The proposed FitWater tool was tested with the existing wastewater treatment plants and then implemented to an actual community. The tool can be used to develop and rank alternative wastewater treatment train and reuses for planning reclaimed water use in a community.

The proposed reclaimed water quality guidelines guide users to find an appropriate level of wastewater treatment for various reuse applications. In particular, FitWater provides the quantity of reclaimed water production, energy use, carbon emissions, and LCC of wastewater treatment and reuse. These data can be used as inputs to the WEC model of Objective 2.

2.5 Objective 5

The economic analysis of the WEC nexus has been conducted by adding a cost module in the WEC model developed under Objective 2 using system dynamics. The cost embedded WEC model also incorporated uncertainty anlysis using Monte Carlo simulations. The cost module includes the LCC of water conveyance, water treatment, distribution, wastewater collection, treatment, and reclaimed water distribution. The LCC was estimated as the sum of capital cost, operation cost, and repair and replacement cost of drinking water system, wastewater treatment system, and reclaimed water use system. The cost embedded WEC model was applied to a community to assess the economic and energy efficiency of net-zero water in different climatic and topographic regions. Also, the impacts of annual precipitation amount (climate), conveyance length (source water proximity), and net elevation head (topography) on energy use and cost of NZW development were assessed in detail in this study.

The cost embedded WEC model, i.e., the extended WEC model than that under Objective 2 can be used to estimate water consumption, energy use, carbon emissions, and LCC of SMUWSs. This objective included an extended WEC nexus analysis incorporating LCC and advanced WEC nexus analysis including uncertainty (Monte Carlo simulations).

Chapter 3 Literature Review

A version of this chapter has been published in *Clean Technologies and Environmental Policy* journal with the title "'Socializing' sustainability: A critical review on current development status of social life cycle impact assessment method" (Chhipi-Shrestha et al., 2015a).

This chapter contains the state-of-the-art literature related to this research and identifies drawbacks and limitations of current practices in SMUWSs. The chapter includes sustainability assessment approaches, namely life cycle sustainability assessment and sustainability performance indicators; water-energy-carbon nexus; system dynamics modelling; and net-zero water comprising impact of neighbourhood densification, reclaimed water use, and fit-for-purpose wastewater treatment.

3.1 Life cycle sustainability assessment (LCSA)

Sustainable development and sustainability are ideas that have been widely used since the 1980s in response to the negative impacts of development, policies, and strategies on the environment and society (UNEP/SETAC 2011; Turcu 2013; Fiksel et al. 2014). Sustainability has three main pillars, namely environment, economic, and social (Valdivia et al. 2011), which are referred to as the triple-bottom-line (TBL) (Sikdar 2007; Vinodh et al. 2012). Integrating life cycle thinking in product or process development with the TBL approach challenges the conventional waste management and pollution prevention mindset that mainly focuses on the factory site (UNEP/SETAC 2011). This new perspective avoids shifting the problem from one phase to another and from one geography to another (UNEP/SETAC 2009). This integrated approach is referred to as life cycle sustainability assessment (LCSA) (UNEP/SETAC 2011). LCSA is "the evaluation of all environmental, social, and economic negative impacts and benefits in decision-making processes towards more sustainable products throughout their life cycle" (UNEP/SETAC 2011). LCSA has three components: life cycle assessment (LCA), life cycle cost analysis (LCCA), and social life cycle assessment (S-LCA) (Klopffer 2003; UNEP/SETAC 2011).

3.1.1 Life cycle assessment (LCA)

Life cycle assessment is a technique that assesses potential environmental impacts of a product or service over its life cycle. This technique identifies opportunities to improve the environmental performance of products at different points in the life cycle, from raw material extraction to use, end-of-life treatment, recycling, and final disposal. LCA is performed in four stages: 1) Goal and scope definition, 2) Life cycle inventory (LCI) analysis, 3) Life cycle impact assessment (LCIA), and 4) Interpretation. Stage 1 includes the definition of purpose, functional unit (reference unit to which inputs and outputs are related), and system boundary. Stage 2 consists of data collection and calculation to quantify related inputs and outputs. Stage 3 involves the evaluation of the significance of potential environmental impacts based on the LCI results. Stage 4 includes the interpretation of findings from the LCI analysis and impact assessment. LCA is an environmental management technique that considers the entire life cycle of a product (ISO 2006a) (ISO 2006b).

3.1.2 Life cycle cost analysis (LCCA)

Life cycle cost analysis is an economic assessment method in which all costs arising from owning, operating, maintaining, and ultimately disposing of a project or product are considered (US DOE 1996). LCCA is defined "traditionally as the estimation of the total cost associated to an asset over time, including investment, operation, maintenance and major repairs and disposal" (Termes-Rifé et al. 2013). LCCA is a powerful economic tool and is particularly suitable for the evaluation of alternatives. LCCA assesses the long-term cost effectiveness of a project better than other economic methods that focus only on initial costs or on short-term operating costs (US DOE 1996).

UNEP/SETAC (2008) identified three types of life cycle cost (LCC): conventional, environmental, and societal LCC based on the 2004 survey of 33 LCC studies from 1984 to 2003. Conventional LCC is "the assessment of all costs associated with the life cycle of a product that are directly covered by the main producer or user in the product life cycle. The assessment is focused on real, internal costs, sometimes even without end of life or use costs if these are borne by others" (UNEP/SETAC 2008). It is a quasi-dynamic method and neglects external costs. The reference flow is mostly one unit of product (e.g., a building), which is easier for computation but may not be the most appropriate for sustainability assessment.

Environmental LCC is "an assessment of all costs associated with the life cycle of a product that are directly covered by one or more of the actors in the product life cycle (supplier, manufacturer, user or consumer, and/or end of life actor), with the inclusion of externalities that are anticipated to be internalized in the decision relevant future" (UNEP/SETAC 2008). It is LCA driven and steady-state in nature. Taxes and subsidies are included in environmental LCC if relevant (UNEP/SETAC 2008), whereas, societal LCC is "an assessment of all costs associated with the life cycle of a product that are covered by anyone in the society, whether today or in the long-term future. Societal LCC includes all of environmental LCC plus additional assessment of further external costs, usually in monetary terms (e.g., based on willingness-to-pay methods)" (UNEP/SETAC 2008). This method uses an expanded system boundary unlike other LCC types and comprises more costs including damage costs that will or could occur in the long term. It is quasi-dynamic in nature (UNEP/SETAC 2008).

The conventional LCC has been in practice throughout history for many government offices, public organizations, and firms (UNEP/SETAC 2008). The LCCA method is applicable to urban water systems. The stages of UWSs from water abstraction to wastewater disposal have several alternatives. These alternatives usually vary in water and energy performance, life span, initial investment cost, and operation and maintenance cost. LCCA can be applied to identify a cost effective decision for long-term scenarios.

3.1.3 Social life cycle assessment (S-LCA)

Social life cycle assessment (S-LCA) is a technique that assesses the potential social impacts of a product or service caused throughout its life cycle. S-LCA refers to the assessment of the real and potential social and socio-economic impacts of products or services including positive and negative impacts along their life cycle (Dreyer et al. 2010; Feschet et al. 2012; UNEP/SETAC 2009). Primarily, social impacts are related to human capital, human wellbeing, cultural heritage, socio-economy, and social behaviour (UNEP/SETAC 2009). S-LCA complements both LCA and LCCA in terms of sustainability assessment (UNEP/SETAC 2009). S-LCA has similar applications to LCA, such as sustainability labelling, sustainability management, and assessment of technology alternatives considering social aspects. In S-LCA, the area of protection is human dignity and wellbeing (Hauschild et al. 2008). More specifically, the area of protection is autonomy, well-being-freedom, and fairness based on a capability approach (Reitinger et al. 2011). The ultimate goal of S-LCA is the wellbeing of stakeholders over a product's life cycle

(UNEP/SETAC 2009). Similar to LCA, the general framework for S-LCA consists of the same four stages (ISO 2006a; UNEP/SETAC 2009).

3.1.4 Sustainability Performance Indicators for UWSs

Urban water utilities provide water to city dwellers through the following processes: water abstraction, treatment, distribution, use, wastewater treatment, and disposal. These human regulated urban water processes constitute a human hydrologic cycle or an urban water system (UWS) (Bagley et al., 2005). UWSs need to resolve emerging water issues related to population growth, urbanization and climate change (Inman and Jeffrey 2006; Lim et al. 2010; IPCC 2014). On the other hand, although UWSs achieve the fundamental requirements of providing clean drinking water and removal of wastewater to a higher degree, this sector is criticized from the sustainability perspective (Hellstro 2000). Sustainability, or sustainable development, in the 21st century is guided by Agenda 21, established by the UN Conference on Environment and Development in 1992 (UN 1992). Based on the Agenda 21 (UN 1992) and other literature such as Hellstro (2000) and Engel-yan et al. (2005), the major objectives of a sustainable UWS are to: a) provide clean and safe drinking water; b) reduce environmental impacts; c) develop an economically efficient system; and d) optimize water and other natural resource uses. These sustainability objectives can be viewed through five interrelated sustainability dimensions and are briefly described below (Daigger 2009; Harmancioglu et al. 2013; VanLeeuwen and Marques 2013; World Bank 2003).

- **i.** ***Technical***: The technical dimension refers to the reliable and proper functioning of UWS technologies and neighbourhood design. This dimension includes sustainability evaluation criteria, such as “neighbourhood location and design” and “water infrastructure and fixtures”.

- **ii.** ***Environmental***: Water resources face many threats, such as pollution and resource depletion due to climate change. These threats ultimately affect the reliability of the resource, i.e., quality and quantity of drinking water supply. On the other hand, water supplies and wastewater facilities themselves threaten the environment through the unsafe disposal of wastewater, emissions of pollutants, and over-consumption of resources. This dimension includes sustainability evaluation criteria such as “resource utilization”, “environmental impacts”, and “resource recovery”.

iii. **Econ*omic***: UWSs can only function if financial resources are available to meet the operation and maintenance costs, at a minimum. The economic dimension includes evaluation criteria, such as “water economics” and “wastewater economics”.

iv. ***Social***: UWS services need to satisfy consumers’ needs and expectations and also promote their health. This dimension includes sustainability evaluation criteria such as “service provision” and “public health”.

v. ***Institutional***: A community needs an institution in order to keep its UWS operational in order to serve consumers. This dimension includes the “governance and progress criteria”.

The performance or achievement of SMUWSs can be assessed using indicators (Murray et al. 2009). A sustainability performance indicator (SPI) is a parameter, or a value derived from parameters, which provides information about the sustainability achievement of an activity, a process or an organization (CWWA 2009). In particular, the literature outlined in Table 3.1 was found to be more relevant for the sustainability assessment of SMUWSs and was reviewed in detail.

Table 3.1 Major literature used for the screening of the SPIs for SMUWSs

SN	UWS and Water Utility Services Sustainability	SN	Neighbourhood and City Sustainability
1	City Blueprints (Van Leeuwen et al. 2012) [24]	10	LEED-ND (USGBC 2013) [19]
2	SI-UWS (Popawala and Shah 2011) [20]	11	BREEAM Communities (BREGL 2012) [11]
3	SCDS (Foxon et al. 2002)[62]	12	CASBEE-UD (IBEC 2008) [13]
4	ESI (Lundin and Morrison 2002) [15]	13	ECC (EarthCraft 2014) [22]
5	UWOT (Makropoulos et al. 2008) [22]	14	SCR (SCR 2009) [6]
6	UWCSS (Van Leeuwen and Marques 2013) [35]	15	Asian Green City Index (Siemens AG 2011) [7]
7	SI (Water UK 2011) [21]	16	Global City Indicators (World Bank 2008) [10]
8	PI (CWWA 2009) [30]	17	European Green City Index (Siemens AG 2009) [4]
9	BSS (Sydney Water 2013) [22]	18	IoS (SCI 2012) [4]

Note: Number of indicators used is mentioned in square brackets [no. of indicators].

3.2 Water-energy-carbon (WEC) nexus

The criteria and indicators related to the WEC nexus for SMUWSs are discussed in the sections below:

3.2.1 Energy *for* water

Energy is required for anthropogenic water use. The energy requirement for each urban water process significantly differs based on topography, technology (Tuladhar et al. 2014; Venkatesh et al. 2014), source water quality (Santana et al. 2014; Tuladhar et al. 2014), social factors (Venkatesh et al. 2014), and operational conditions, especially in water distribution (Cabrera et al. 2010; Giustolisi et al. 2016; Iglesias-Rey et al. 2016; Nardo et al. 2014). For example, in California (US), water supply and conveyance has a very wide range of energy intensity, ranging from zero to 3646 kWh/ML, indicating a higher dependency of energy use on supply methods (like ocean water desalination, and surface water and groundwater withdrawal) and conveyance distance. Similarly, wastewater collection and treatment also has a higher energy intensity, ranging from 291 to 542 kWh/ML, with variation mainly attributed to a variety of treatment methods (CEC and NC 2006). From the life cycle perspective of an UWS, the operational phase has been identified as the most energy intensive phase (Friedrich 2002; Nair et al. 2014). The operational phase of water treatment consumes 94% of total energy use and is responsible for 90% of total GHG emissions (Racoviceanu et al. 2007).

3.2.2 Energy *from* water

Energy can be generated from water. Water and wastewater flow contain kinetic energy, potential energy (Fontana et al. 2012), thermal energy, and chemically bound energy, especially in wastewater, all of which can be harnessed. The amount of kinetic and potential energy depends on topographical conditions. For instance, even at a height of 50 m, the potential energy content of water or wastewater is only 6 kWh per capita per year (Meda et al. 2012). Thermal energy in wastewater is mainly stored due to warm wastewater generation, from hot water showers, laundry, and dishwashing, and contains more energy than potential energy (Meda et al. 2012). Therefore, the greatest potential for heat recovery is from greywater, i.e., wastewater from baths, showers, laundry machines, and possibly kitchen sinks. For example, in Germany, the typical household’s greywater generation rate is 40L/p/day with a temperature difference of

$30^{0}C$, which can generate 509 kWh/capita/yr (Meda et al. 2012). This energy is higher than potential energy (6 kWh per capita per year, even at a height of 50 m height).

Chemically bound energy can be estimated from the carbon content, i.e., chemical oxygen demand (COD). On the basis of a daily COD load of 110 to 120 g/capita, the maximum theoretical energy content is approximately 146 kWh/capita/yr, under the assumption that all COD could be transferred to methane and be utilized (Meda et al. 2012). For comparison purposes, energy consumption of wastewater treatment plants ranges from 27 to 37 kWh/capita/yr in Canada (AECOM 2012), indicating the potential that wastewater treatment plants could be energy self-sufficient. However, the practically recoverable energy will be lower than the theoretical recoverable energy.

3.2.3 Water *for* energy

Water is required directly and indirectly for energy generation, which is the water footprint of energy. The water footprint of a product refers to the volume of freshwater used to produce the product, measured over the full supply chain (Hoekstra et al. 2011). Water is used directly for electricity generation by hydropower; however, a large amount of indirect water is used for the exploration, extraction, and beneficiation of fossil fuels, depending on the specific method used (Meda et al. 2012). Indirect water is also used for renewable energy production, such as energy crop cultivation (for biofuel production). Similarly, thermal power plants also use steam (water) directly for driving turbines and indirectly for cooling purposes, i.e., heat dissipation.

The quality and quantity of used water, i.e., wastewater from energy generation is an important factor (Meda et al. 2012). Some processes, such as hydraulic fracturing of oil and gas wells and power plant operation (e.g., ash handling) pollute water more than others (e.g., only the temperature increases in the water used in cooling towers). Additionally, the water used for crop cultivation is not directly available to further reuse, whereas the water used in cooling towers is available after its primary use. These processes indicate that the selection of energy types for urban water activities have different water implications.

3.2.4 GHGs *from* energy and water

Energy use and wastewater treatment processes release greenhouse gases (GHG). The GHG emissions or simply carbon emissions of energy uses differ by energy source and its generation method, such as fossil fuel, hydropower, thermal power, etc. Energy (e.g., grid electricity) generation methods are location specific and their carbon emissions vary. For instance, carbon emissions from grid electricity generation is 57 times higher for Alberta electricity (824.4 kg CO_2e/MWh) than for BC Hydro (14.4 kg CO_2e/MWh) (Ministry of Environment 2013). This is because 83% of the grid electricity is produced from fossil fuels (coal and natural gas) in Alberta (Alberta Energy 2014), whereas BC hydro produces 95% of its electricity from hydropower (BC Hydro 2015). In addition, GHGs are also released from wastewater treatment processes due to the microbial degradation of organic matter present in wastewater (IPCC 2006).

The WEC nexus is complex. Energy is needed for water production and energy can also be harnessed from wastewater. Water is needed for energy generation. Both energy use and wastewater processes emit GHGs. Because of this tight inter-linkage in the WEC nexus, decisions in one area could have inadvertent consequences in another (Rothausen and Conway 2011). Thus, only a holistic and generic model can capture the variability and dynamics of UWSs (Nair et al. 2014).

3.2.5 WEC models

The WEC model can broadly be categorized into static and dynamic models as follows:

3.2.5.1 Static models

Researchers have been working on WEC nexus and some have proposed static models and tools. GIZ/MENCBNS/IWA (2015) has developed an Excel-based Energy performance and Carbon emissions Assessment and Monitoring (ECAM) Tool for evaluating the energy performance and carbon emissions of water and wastewater utilities. However, the tool is only for utilities, and excludes energy performance in indoor water use. Similarly, Venkatesh et al. (2014) extensively studied the WEC nexus of four water utilities of Nantes (France), Toronto (Canada), Turin (Italy) and Oslo (Norway). The study lacks the inclusion of water consumption, which is a very important component of a UWS. Gu et al. (2016) investigated the WEC nexus of nine wastewater treatment plants in China, but they are limited only to a specific component of an

UWS, i.e., wastewater utility. Similarly, Stillwell et al. (2010) examined the WEC nexus of Texas and estimated potential water and energy savings by implementing water conservation and reuse practices. Arora et al. (2013) researched the life cycle energy use and GHG emissions of alternative urban water supply strategies for the Melbourne Metropolitan region in Australia using static models. Furthermore, the WEC nexus of UWSs including ten water and wastewater utilities in seven cities in Australia and New Zealand was studied by Kenway et al. (2008). Similarly, the carbon cost model was proposed by Reffold et al. (2008) for estimating GHG emissions and their cost for different water supply and demand options in the UK.

Some researchers only studied the water-energy nexus component of the WEC nexus for several community elements. Cutter et al. (2014) examined the water-energy nexus of the UWS of California as a means to evaluate the cost effectiveness of strategies for water utilities. Cheng (2002) investigated the water-energy nexus of residential buildings in Taiwan to evaluate the energy savings due to water conservation measures. Similarly, Abdallah and Rosenberg (2014) studied the water-energy linkages of residential indoor water use to determine the implications for water and energy conservation and management in the United States. Malinowski et al. (2015) investigated the energy-water nexus of integrated water management (IWM) measures, i.e., rainwater harvesting and greywater reuse to estimate the energy and cost saving at the national and local scale in the United States. Likewise, Pacetti et al. (2015) explored the water-energy nexus of biogas production from three energy crops: maize, sorghum, and wheat in Italy to estimate the water requirements for biogas from these energy crops. UDWR (2012) researched the water-energy nexus in Utah to determine the interconnection of these two resources at the state level.

EPRI (2002) and Gu et al. (2014) broadly explored the water-energy nexus at the national scale, respectively for the United States and China in order to determine the interdependency and sufficiency of these two resources in the future. In addition, Stillwell (2015) researched three energy-water nexus bills of the U.S. Congress to assess the sustainability of these public policies. None of the above mentioned frameworks and models can perform dynamic analysis and all lack system feedbacks. Also, they have limited flexibility for practical application and many of them do not include all the components of UWSs.

3.2.5.2 Dynamic models

An integrated urban water system was dynamically modelled by Fagan et al. (2010) to develop sustainability assessment framework. However, the model was developed by using rigorous mathematical equations. Other dynamic approaches used for urban water modelling are agent-based modelling (ABM) and system dynamics. Agent-based modelling has some disadvantages compared to system dynamics, e.g., higher data requirement for calibration (Gebetsroither-geringer 2014). In addition, the results of agent-based modelling are more difficult for evaluation, resulting in greater efforts required for model validation (Gebetsroither-geringer 2014) as revealed by researchers, such as Fagiolo (2006) and Werker and Brenner (2004). System dynamics has been used in only a few urban water studies (Zarghami and Akbariyeh 2012).

3.3 System dynamics modelling (SDM)

System dynamics is a well-established methodology to quantify complex feedbacks in system interactions (Forrester, 1961; Forrester, 1968). The methodology was initially developed by Forrester (1961). A system refers to "a collection of elements that continually interact over time to form a unified whole". Dynamics means change over time, where the values of variables and parameters change over time. Therefore, system dynamics is a methodology used to understand how a system changes over time (Martin 1997a). The system dynamics model (SDM) is often used to quantify system behaviors with feedback loops for more accurate projections (Qi and Chang 2011). The model allows for the effective trade-off analysis of multi-scenarios and the multi-attributes of the WEC nexus over time (Sehlke and Jacobson 2005). System dynamics involves the construction of "stock and flow diagrams" to mimic a dynamic system.

3.3.1 System dynamics model construction

3.3.1.1 Stock and flow diagram

System dynamics computer simulation programs such as STELLA provide a framework and easy-to-understand graphical interface to study the quantitative interaction of variables within a system. The interaction can be modelled using four building blocks as given in Figure 3.1.

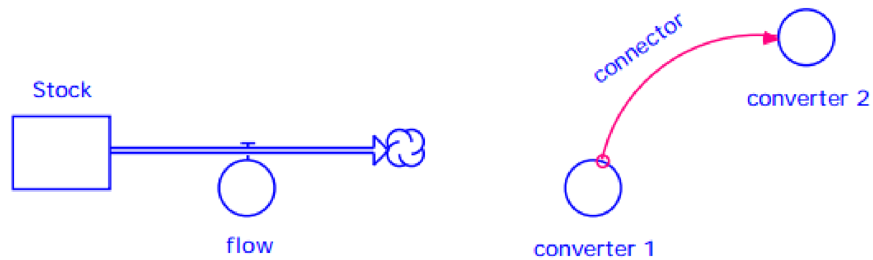


Figure 3.1 Representation of a stock, flow, converter, and connector

A stock is used to "represent anything that accumulates or drains over time" (e.g., water accumulating in a bucket). A flow is "the rate of change of a stock". A converter is used to "take input data and manipulate or convert input into some output signal(s)". "A connector is an arrow that allows information to pass between two converters, stocks and converters, stocks and flows, and converters and flows" (Martin 1997a).

Mathematical equations are developed by combining two fundamental ideas (Roberts 2001). First, a stock at present time equals the stock at a certain previous time, plus the change in the stock (net flow) that occurred over the specified time interval. Second, the change during a certain time interval equals the length of the interval, multiplied by the rate of change per time interval. The combination of these two ideas produces the following equation of the present stock at time 't' (Roberts, 2001; Porwal, 2013):

*Stock (at present time, t) = Stock (at certain previous time) + (Length of the time interval) * Rate of stock change*

$$= \int_0^t (\textbf{\textit{Stock variable}})\, \boldsymbol{dt} \qquad \textbf{Equation 3.1}$$

3.3.1.2 Feedback

"Feedback is a process whereby an initial cause ripples through a chain of causation ultimately to re-affect itself" (Martin 1997b). Feedback occurs when an output of a system is fed back into the system as an input (Martin 1997a). Feedback can occur in an open-loop system or closed-loop system; however, the latter one is more common. Feedback systems can be classified as positive or negative. Positive feedback system moves in the same direction to produce compounding or reinforcing behaviour. These systems drive growth and change (Martin 1997b). For instance, as shown in Figure 3.2, reproduction increases the rabbit population. The growth occurs by the birth of rabbit. The number of births per time depends directly on how many rabbits are already in the area considered and increases with the growth of the population size. The shaded arrows show the causal links and not material links or information links.

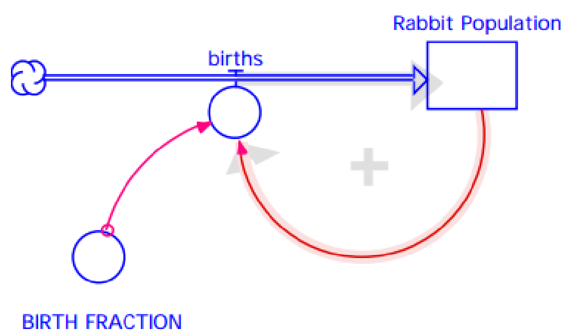


Figure 3.2 Positive feedback in growing rabbit population

On the other hand, a negative feedback system "moves in opposite directions to produce balancing or stabilizing behaviour". These systems negate change and stabilize systems (Martin 1997b). In positive feedback, a variable is eventually increased as a result of the increase in that variable. Whereas, in a negative feedback system, an increase in a variable eventually result in a decrease in that variable. For instance, Figure 3.3 shows a declining skunk population. Every year, a fraction of the total skunk population declines. The number of deaths per time (year) depends directly on the initial skunk population and decreases gradually with the decline of the population.

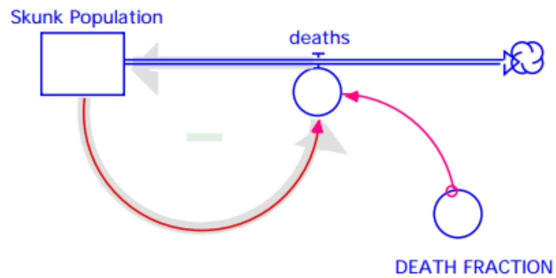


Figure 3.3 Negative feedback in declining skunk population

A causal loop diagram (CLD) is developed before developing a complete SDM. A CLD is the foundation of a SDM, and is used to identify relationships between individual system components and show feedback loops that affect system regulation (Nasiri et al. 2013). In the CLD, a ‘‘+’’ sign indicates a positive (reinforcing) relationship, whereas a ‘‘-’’ sign indicates a negative (balancing) relationship between two variables (Nasiri et al. 2013).

3.3.2 SDM for urban water management

A critical review of system dynamics-based urban water models was performed, and the strengths and weaknesses of the developed models are given in Table 3.2.

This critical review shows system dynamics has been applied to different aspects of urban water from county to city level but not applied to the WEC nexus for neighbourhoods or a community. Also, urban water processes perform differently in various geographic regions with respect to energy and carbon emissions. This requires a holistic and generic model to capture the variability and dynamics of UWSs (Nair et al., 2014). The WEC nexus model comprising interacting problems can be developed using system dynamics (Nasiri et al., 2013; Nair et al., 2014). The systems dynamics model can assist decision makers in understanding the implications of investment decisions and actions on a SMUWS (Kenway et al., 2011).

Table 3.2 Strengths and weaknesses of system dynamics based urban water models

Reference	Scale	Objectives/Strengths	Weaknesses
Zarghami and Akbariyeh (2012)	City	Determine the most effective policy to manage water demand & supply; considers different water sources, population, and cost	Not considered carbon emissions & energy analysis
Willuweit and O'Sullivan (2013)	City	Model effects of urban development and climate change on urban water cycle; considers social, environmental, economic, & functional indicators	Lacks a feedback loop between land use and water balance models
Zhang et al. (2008); Zhang et al. (2009a)	City	Identify an optimal plan; considers population, industry, agriculture, and water resource	Not considered energy analysis, carbon emissions, and water reuse
Nasiri et al. (2013)	County	Plan and manage reclaimed water use; considers cost, technology, and environmental factors	Focused on water, not considered energy, carbon emissions, & other benefits of water reuse, e.g., reduced energy use
Karamouz et al. (2012)	City	Assess reliability of water quantity and quality; considers daily time step	Not considered energy and carbon emissions and energy analysis; not calculated model accuracy
Qi and Chang (2011)	County	Estimate domestic water demand under changing macro-economy; considers socio-economy, population, and water	Not considered energy, carbon emission, and water reuse
Wang (2014)	Province	Effect of water price and wastewater purification ratio on the water demand; considers population, water demand and supply; based on World Water Model	Not mentioned accuracy, not considered energy and carbon emission
Zhang et al. (2009b)	City	Predict the WRCC*; considers water resource; industrial, agricultural and residential water; and WWT & reuse	Not considered energy analysis and carbon emissions; not validated
Tong and Dong (2008)	City	Analyze structure and functions of socio-economic-environmental system; considers domestic, industrial, agricultural, ecological water, and pollution control; used PSR** framework for SDM	Not considered energy, carbon emissions and water reuse
Nawarathna et al. (2009)	Catch-ment	Predict future water supply and demand under changing land use and climate; considers irrigation, urban and environmental water demand	Not considered energy and carbon emissions; model not validated
Wang (2013)	Country	Water and energy nexus; implications of biofuel development on water and energy	Model at national scale; preliminary model; focus on biofuel

*Water Resource Carrying Capacity;** Pressure, State, and Response Framework

3.4 Net-zero water (NZW)

Historically, the net-zero concept evolved from the building energy budget and its popularity has extended the concept to waste generation, carbon emissions, and water consumption (Joustra and Yeh 2014a). The concept of net-zero water (NZW) is similar to the carrying capacity of a system (Holtzhower et al. 2014). NZW is the balance of water demand and supply within a given areal boundary (Holtzhower et al., 2014). The US Army states “net-zero water limits the consumption of freshwater resources and returns water back to the same watershed so not to deplete the groundwater and surface water resources of that region in quantity or quality over the course of a year” (US Army 2011). The central theme of NZW emphasizes a balance so that the sum of all input water is offset by comparable output water (Joustra and Yeh 2014a). NZW presupposes that a community system can secure an adequate water supply within its boundaries, typically from surface water, groundwater, reclaimed water, and rainfall (Holtzhower et al. 2014). Achieving net-zero water similar to the natural cycle requires both the conservation of water and the creation of balanced water feedback loops (Joustra and Yeh 2014a). The nuances of NZW are given in Table 3.3. This study has considered the widely accepted definition proposed by the US Army (2011) for NZW.

NZW can also be interpreted as a sustainable tolerance of comfortable minimum use. The sustainable minimum use of water is approximated as 70 L/person/day, of which 20 L/p/d is for potable use and 50 L/p/d is for human sanitation and disease prevention according to the World Water Council and the United Nations Development Program (Ma 2014). A minimum use of water would lead to the development of NZW conveniently for communities. Moreover, Net-Positive Water (NPW) can also be developed with the generation of positive water balance. NZW or NPW explores the capabilities of communities for sustainable planning and use of water.

Table 3.3 Definitions of net-zero water and its nuances

Definitions	Source
Net-Zero water	
"Net-zero water limits the consumption of freshwater resources and returns water back to the same watershed so not to deplete the groundwater and surface water resources of that region in quantity or quality over the course of a year."	US Army (2011)
"Annual potable water use is no greater than annual rainfall"	Olmos and Loge (2013)
"A sustainable tolerance to comfortable use of minimum water. The sustainable minimum use of water is approximated as 70 L/person/day."	Ma (2014)
Zero water	
Zero-water compliance requires that the building water cycle operates independently from water and wastewater municipal systems. On-site wastewater recycling is crucial to zero-water success, and alternative water supplies are limited by the regional climate. However, the preservation of the local ecosystem must be considered when collecting precipitation for on-site use.	Joustra and Yeh (2014a)
Life-cycle zero water	
It requires that the embodied water required for the manufacture and transport of materials be considered over the building lifetime. Achievement of net-zero water over the building lifetime may be an unachievable objective without innovative techniques for on-site renewable water generation.	Joustra and Yeh (2014a)
Net-Positive water	
"One hundred percent of the project's water needs [except for regulated potable uses] must be supplied by captured precipitation or other natural closed -loop water systems and/or by recycling used project water, and must be purified as needed without the use of chemicals."	International Living Future Institute (2016)
Net-positive water balance in buildings as a result of restorative impacts.	Joustra and Yeh (2015)

A report published by the US National Research Council stated that "The use of reclaimed water to augment potable water supplies has significant potential for helping to meet future needs, …." and also recommended potable reuse with or without an environmental buffer as an alternative water management approach (National Research Council 2012). Similarly, water recycling for the augmentation of drinking water supplies has been promoted by the Australian government, who has been extensively applying reclaimed water and published guidelines for reclaimed water quality management (EPHC/NHMRC/NRMMC 2008). Also, in Canada, the provincial government of British Columbia (BC) has planned for the mandatory construction of dual water-plumbing (additional purple pipes for reclaimed water flow) in new buildings (BC Ministry of Environment 2008). Moreover, the BC government has endorsed a BC Wastewater Regulation that allows reclaimed water use in non-potable and potable reuses after treatment with the approval of local health authorities (BC Ministry of Environment 2013; MWR 2012). These initiatives show an increasing aspiration for reclaimed water use.

Actual NZW has been practised in several places in the world. For instances, the commercial building “The Bullitt Center” in Seattle, Washington is a NZW building that has been operational since 2013 (Crosson 2016; Killough 2016). Another pilot scale NZW building is a four-bedroom university residence hall unit that was built in 2012 (Englehardt et al. 2013; Gassie et al. 2016). At a larger scale, Namibia (Asano et al. 2007; Crook et al. 2005; WABAG 2016) and Singapore (Angelakis and Gikas 2014; Asano et al. 2007) have been utilizing reclaimed water for drinking. Namibia has been applying such practice since 1968 (Asano et al. 2007). Similarly, Cyprus reuses 100% of their wastewater (EU 2015, 2016), whereas Israel (Angelakis and Gikas 2014; Crook et al. 2005) and Malta (Crook et al. 2005; EU 2016) reuse approximately 80% of their wastewater.

NZW or NPW could be achieved by using wastewater recycling, rainwater harvesting, and storm water harvesting (Englehardt et al. 2013). However, NZW development may have higher associated costs (Englehardt et al. 2013; Gassie et al. 2016; Wang and Zimmerman 2015) and energy (Vieira et al. 2014; Wang and Zimmerman 2015). Therefore, economic viability and environmental sustainability of NZW is of high concern. In fact, these features are highly location specific, and the national Australian water reuse study recommended to evaluate water reuse project individually using a DSS (DSEWPaC 2012).

There are a number of studies that have proposed methodologies or DSSs for evaluating NZW potential. The studies are summarized in Table 3.4.

Table 3.4 DSSs for NZW analysis

DSSs	Applications	Limitations	Reference
ZeroNet decision support system (DSS) for the San Juan River Basin	Focus on drought planning and economic analysis at a watershed level; analyze critical water supply and demand information and assist water utilities in planning for water management in shortages	Not included energy consumption by water use; not included carbon emission & features such as water recycling and rainwater harvesting	Rich et al. (2005)
NZW alternative analysis methodology	Assist Department of the Army (US) in evaluating alternatives to achieve NZW in six pilot installations; multi-criteria-based; included criteria cost, environmental variables, and water quality, etc.	Methodology only at a conceptual stage; criteria are broadly presented	Payosova and Deason (2012)
Urban/suburban NZW treatment process for buildings	Based on laboratory experiments, mass balance, & kinetic model; experiments on four-bedroom university residence hall unit; NZW development with 10-20% rainwater make-up; capital cost of $ 6.20/m^3 (20 years of life); operational and maintenance cost of $1.83/m^3	Initial design than a decision support tool; only at a building level	Englehardt et al. (2013)
Offsetting of water conservation costs to achieve NZW	Reveal potential cost savings in both utility energy conservation and energy reductions in decreased hot water use; selling of carbon offset credits is feasible; NZW at single-family building and community level is feasible.	Conceptual case study; lacks dynamic analysis of water; not included wastewater treatment and its cost, energy use, & carbon emissions	Olmos and Loge (2013)
NPW matrix	Proposed guidelines for clean and dirty water management in buildings to develop NPW quality and quantity; NPW matrix helps to identify the means to develop domestic architecture for NPW in buildings	Only at a building level and does not include cost, energy use, and carbon emissions	Ma (2014)
Integrated Building Water Management (IBWM) Model	NZW decision support tool using system dynamics; flexible; and dynamically tracks potential water flows into, within, and from buildings	Only at a building level and do not include cost, energy use, and carbon emissions.	Joustra and Yeh (2014a); Joustra and Yeh (2014b)
NZW potential assessment	Analyze and map the NZW potential of the US; based on urban area clusters (UAC)	Preliminary analysis; not included surface and groundwater sources, temporal variation, cost, & carbon emissions; not an executable tool	Holtzhower et al. (2014)
Conceptual framework for NPW	Develop NPW buildings	Only at preliminary stage	Joustra and Yeh (2015)

Urban/suburban NZW treatment process for buildings	Present results of two years of operation of the system showing the effective NZW system with 85% recycling rate; system was projected to be capable of energy-positive operation	A "first design" rather than a decision support tool; only at a building level	Gassie et al. (2016)
NZW at a building level in severe drought prone areas	Achieve NZW at a building level in severe drought prone areas of Los Angeles, California; based on an office building of 250 Full Time Equivalent employees	Although in-depth analysis, not included cost, energy use, and related carbon emissions; only at a building level	Crosson (2016)
Model for economic feasibility of large-scale NZW management	Minimize cost and energy use for NZW systems; minimize the cost for recycling wastewater at the rate of $2.95/m^3	Not included carbon emissions and dynamic interaction of urban water components	Guo et al. (2016)

Almost all of the listed studies were conducted at the building scale. Among the existing methodologies, some are conceptual frameworks, whereas others lack one or more components such as energy, cost, or carbon emissions. For instance, Ma (2014) proposed a NWP matrix to identify the means to develop domestic architecture for NPW in buildings and Joustra and Yeh (2014a) developed a NZW decision support tool, Integrated Building Water Management (IBWM) Model. Both of the proposed methods are only at a building scale and do not include cost, energy use, and carbon emissions. Also, Joustra and Yeh (2015) proposed a framework for NPW buildings, which is a conceptual framework and at a preliminary stage. Englehardt et al. (2013) proposed an urban/suburban NZW treatment process for buildings based on laboratory experiments and Gassie et al. (2016) presented the results of two years of operation of the treatment process showing the effective NZW system. The proposed treatment process was a "first design" and at a building level rather than a decision support tool at a community scale.

ZeroNet Water-Energy Initiative developed the ZeroNet DSS for the San Juan River Basin with a focus on drought planning and economic analysis (Rich et al. 2005). The DSS does not include energy use by water consumption. Also, the DSS neither estimates carbon emissions nor includes the features such as water recycling and rainwater harvesting (reuses) that are important for developing NZW. Guo et al. (2016) developed a model for the analysis of economic feasibility of large-scale NZW management; however, it does not include carbon emissions and dynamic interaction of urban water components.

The cost, energy, and health risk of NZW are affected by various components of SMUWSs, including neighbourhood density. These components are reviewed as given below.

3.4.1 Impact of neighbourhood densification on WEC nexus

A Water Distribution System (WDS) comprises transmission mains, distribution pipelines, and pumping stations. WDSs are affected by residential landscaping practices that consume a significant amount of water. For instance, the water demand of residential landscaping ranges from 30% of domestic demand in coastal areas to 60% in hot inland areas in California, US (Gleick et al. 2003). This value has been shown to be as high as 77% in the Okanagan Valley, BC, Canada (OBWB 2016). WDSs have significant effects on energy use and related greenhouse gas emissions (Hellstro 2000). Energy use is the primary cost factor in the operation of water supply systems consuming approximately 80% of municipal water processing and distribution costs (EPRI 2002). For groundwater systems, almost all of the energy cost is associated with pumping except where ion exchange and physical or chemical treatment is required (Energy Center of Wiscosin 2003). In particular, water conveyance and distribution only consumed 1.6% of the total energy use, i.e., 386 GWh/year in the City of Toronto in 1998, indicating a high energy use for pumping (Cuddihy et al. 2005). Moreover, trees, shrubs, and soil of urban residential landscaping have significant carbon sequestration potential (Lal and Augustin 2012; Zirkle et al. 2011). These features show WDS and residential landscaping are connected in terms of water consumption, energy use, and net carbon emissions forming Water Distribution and Residential Landscaping System (WDRLS).

Energy requirements for WDSs depend on several factors, including service area topography (Bolognesi et al. 2014; Teixeira et al. 2016; Tuladhar et al. 2014), source water (Energy Center of Wiscosin 2003), urban form (Filion 2008; Speir and Stephenson 2002), population density (Filion 2008; Speir and Stephenson 2002), and adopted management strategies (Bolognesi et al. 2014; Teixeira et al. 2016). Numerous researchers have investigated the reduction in energy use in WDSs and many of them focussed on the optimization of distribution system operations. For example, energy cost optimization (Alighalehbabakhani et al. 2013); energy metrics of WDSs

(Dziedzic and Karney 2015); effects of different management strategies on energy consumption (Cherchi et al. 2015); life cycle energy use of water distribution pipes (Filion, Maclean, & Karney, 2004); and effects of raw water source on energy use by water conveyance (EPRI 2002).

Some researchers studied the influence of topography on energy use (Guo and Englehardt 2014), while others investigated the effects of housing patterns (i.e., single-family) on public water and sewer costs (Speir and Stephenson 2002) and effects of urban form (i.e., configuration) on water distribution energy (Filion 2008). All these studies assumed a constant rate of water use. The constant rate of water use in various residential densities is very different than reality, where landscaping water demand is high and differs much with residential densities. Residential density is affected by the mix of single-family (SF) and multi-family (MF) buildings. A maximum lot coverage (meaning the lot area covered by buildings) is typically lower in low density SF buildings than high density MF buildings. For instance, a maximum lot coverage in SF buildings is typically around 23% in the US (Zirkle et al. 2012; Zirkle 2010), around 30% in the most areas in Toronto (City of Toronto 2016); 40% in Penticton (City of Penticton 2015a), Kelowna (City of Kelowna 2007), and District of Peachland (District of Peachland 2014a), and 60% in Saskatoon (City of Saskatoon 2016). Similarly, a maximum lot coverage in high density MF buildings is 70% in Calgary (City of Calgary 2012) and Kelowna (City of Kelowna 2007), and 100% in Peachland (District of Peachland 2014a) and Penticton (City of Penticton 2015a) with landscaping on unbuilt land. These regulatory requirements indicate that per capita water use varies highly with residential densities.

3.4.2 Reclaimed water use

The challenges in water supply, e.g., growing population, variability in source water, etc. result in increasing water demands and competition among water utilities even across the provincial and national boundaries (Schaefer et al. 2004). In recent times even in Canada, seasonal water shortages have been experienced in various regions. Several cities in BC and Alberta, such as Vancouver, the Cowichan Valley, Penticton, and Calgary experience water restrictions in summer (Gulerian 2015; Water Conservation Company 2015). Water restriction is a municipal regulation to restrict water use in relatively less important activities. Some cities may even reach the severe

Stage 3 restriction prohibiting certain water uses, such as lawn irrigation, park irrigation, residential vehicle washing, street cleaning, and outdoor decorative water features.

Water resource management requires careful planning to address urban water shortages and the associated uncertainties. Supply-side and demand-side management are core strategies for water resources management (Kanta and Zechman 2014; Schaefer et al. 2004). Supply-side management includes water availability augmentation, water infrastructure expansion related to water, and new water source development, whereas demand-side management incudes water conservation activities, leakage control, and price setting (Kanta and Zechman 2014). Under supply-side management, reclaimed water use is an option. Reclaimed water refers to the municipal wastewater that is treated to meet specific water quality criteria, especially intended for beneficial uses. The term recycled water is also synonymously used for reclaimed water (Asano et al. 2007). Reclaimed water is an on-site water resource that can be generated at or near the vicinity of urban water consumption. Reclaimed water can be used for various purposes after treatment.

3.4.2.1 Global water reuse status and trend

Reclaimed water is the treated municipal wastewater that meets specific water quality criteria, primarily intended for beneficial uses. The major drivers triggering water reuse are lack of water, management of drought impacts, freshwater saving for first-use that demands high water quality, use of cheaper water sources, water reuse as low cost disposal option for wastewater, and water restoration to the environment (EU 2016; Jiménez Cisneros 2014).

Globally, 7000 Mm^3/year of reclaimed water was used after treatment in 2011, which comprised 0.59% of the total water use (EU 2016). More than 60 countries have applied reclaimed water for different uses (Angelakis and Gikas 2014). The amount of reclaimed water use in different regions and countries was reviewed and outlined in Table 3.5. Based on the total annual volume, China, Mexico and the United States (primarily, California, Florida, Texas, and Arizona) use the highest amount of reclaimed water in the world (Angelakis and Gikas 2014). However, China and Mexico reuse wastewater with little or no treatment similar to Pakistan (Table 3.5). The intensity of water reuse per capita was highest in Cyprus, Qatar, Israel, and Kuwait (EU 2016; Jiménez Cisneros 2014). Kuwait, Israel, and Singapore ranked first in terms of proportion of reuse with respect to total freshwater use volume (Jiménez Cisneros 2014). Furthermore, California, Singapore, and Japan are probably pioneers with respect to technological advancement in water reuse (Angelakis and Gikas 2014).

Also, Table 3.5 shows major types of water reuse applications comprising non-potable and potable uses in different countries. Non-potable reuse is common although potable reuse has been in practice in Namibia and Singapore. In particular, agricultural irrigation is a primary application reusing 32% of reclaimed water in the world. Other possible water reuses are landscape irrigation (20%), industrial uses (19%), urban uses (8%), environmental enhancement (8%), recreational uses (7%), groundwater recharge (2%), indirect potable use (2%), and others (2%) (EU 2016; Lautze et al. 2014). However, groundwater recharge and indirect potable reuse have high potential for future reuse (EU 2016).

People have a historical practice of reclaimed water use across the world. The trend of reclaimed water use has been increasing worldwide and Global Water Intelligence estimated that the world market of water reuse is expected to surpass desalination in the future. The estimation shows that water reuse will represent 1.66% (26,000 Mm^3/year) of the total global water use by 2030 (EU 2016). For example, the trend of reclaimed water use in some countries is given in Figure 3.4. Generally, the quantity of reclaimed water use has been increasing in the United States (California), Australia, and Europe since the 2000s or before.

Table 3.5 Status of global water reuse

Country	Reuse# (Mm^3/yr)	% of WW reused	Major applications	Reference
World	26,000 in 2030	1.66 in 2030*		EU (2016)
North & Latin America				
United States	3850	-	-	Angelakis and Gikas (2014)
California (US)	1271	-	AI, LI, GWR, IU	Asano et al. (2007)
Florida (US)	834	54	RI, WH	Asano et al. (2007)
Canada (BC)	-	3	AI	Schaefer et al. (2004)
Mexico	350,000 ha		AI@	Crook et al. (2005)
EU	1100	2.4		EU (2016); EU (2015)
Spain	347	~10	AI, IU, TF, LI, PIPU, RI	Crook et al. (2005); EU (2015, 2016)
Italy	233	~8		Crook et al. (2005); EU (2015, 2016)
Cyprus	20	100		EU (2015, 2016)
Germany	42	~1		Crook et al. (2005); EU (2015, 2016)
Malta	~4	~78		Crook et al. (2005); EU (2016)
Australia	300	16.8		DSEWPaC (2012)
New South Wales	63	9.8	LI, TF, AI, SI, IU, VW, CU, ENV	DSEWPaC (2012)
Victoria	100	24.1		”
Queensland	71	23.7		”
South Australia	22	28.1		”
Western Australia	19	12.0		”
Tasmania	3	6.2		”
Northern Territory	1.5	6.0		”
Austr. Capital Territory	3.5	13.3		”
Middle East				
Israel	300	~80	AI, GWR	Crook et al. (2005); Angelakis and Gikas (2014)
Qatar	760	-	AI, LI	MDPS (2016)
Iran	70	5	AI	Crook et al. (2005)
Kuwait	52	-	AI, LI	Crook et al. (2005)
United Arab Emirates	500	20	AI, LI	Crook et al. (2005)
Saudi Arabia	657	10	AI, LI, IU	Drewes et al. (2012); WHO (2005); Crook et al. (2005)
Asia				
China	7373	9.2	IU, LI, AI, TF @	Zhou et al. (2011)
Japan	187	-	TF, IU, ENV, AI,	Crook et al. (2005)
Korea	157	4**	IU, TF,CL	Crook et al. (2005)
Singapore	27	-	DW (2.5%) & NPW	Angelakis and Gikas (2014)
Pakistan	-	80	AI@	Crook et al. (2005)
Southern Africa				
South Africa	>45	3		Crook et al. (2005)
Namibia	7.67	4**	DW blending	WABAG (2016); Crook et al. (2005)

#Mm^3/year unless stated *% of total water use **% of water supply @ Little or no treatment

AI: Agricultural irrigation, LI: Landscape irrigation, GWR: Groundwater recharge, IU: Industrial use, RI: Recreational impoundment, WH: wildlife habitat, TF: Toilet flushing, PIPU: Planned indirect potable use, SI: Silviculture, VW: Vehicle washing, CU: Constructional use; ENV: Environmental applications (streamflow augmentation, dune stabilization, etc.), CL: Cleaning, DW: Drinking water, NPW: Non-potable water

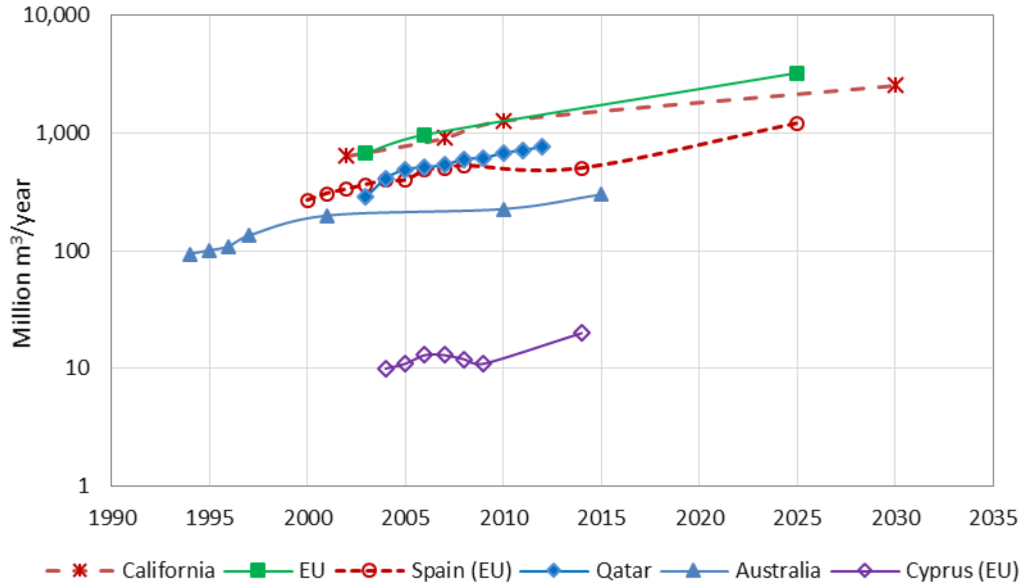


Figure 3.4 Trend of water reuse in different regions of the world

3.4.2.2 Microbial quality of reclaimed water

Reclaimed water use poses human health risks, primarily associated with pathogenic microorganisms, disinfection byproducts (DBPs), and pharmaceutical and personal care products (PPCPs). Pathogenic microorganisms in water primarily originate from sewage (feces) contamination (EPHC/NHMRC/NRMMC 2008) and also from natural freshwater bodies containing pathogens, such as *Lagionella* and *Aeromonas* (Health Canada 2013a). DBPs are created during water disinfection, primarily by the reaction of natural organic matter contained in water and chemical disinfectants (Tian et al. 2013). PPCPs may be present in treated water due to their presence in wastewater, which may not have been effectively removed during wastewater treatment (Kosma et al. 2014). This research is focused only on the human health risk associated with pathogenic microorganisms. Several groups of wastewater microorganisms have been identified as being pathogenic (EPHC/NHMRC/NRMMC 2008): a) Bacteria, e.g., *Campylobacter*, pathogenic *Escherichia coli*, *Shigella*, *Lagionella*, *Salmonella*, and *Vibrio cholera*; b) Viruses, e.g., adenovirus, rotavirus, norovirus, enterovirus, and Hepatitis A; c) Protozoa, e.g., *Cryptosporidium* and *Giardia;* and d) Helminths, e.g., *Taenia* (tapeworm), *Ascaris* (roundworm), *Trichuris* (whipworm), and *Ancylostoma* (hookworm). The human health risks posed by wastewater microorganisms have been estimated by quantitative microbial risk assessment (QMRA) since the 1980s (Haas et al. 2014).

Globally, standard guidelines do not exist for reclaimed water use. Indeed, the development of a practical guideline is complex. The complexity can be understood from the historical development of the reclaimed water use guidelines by the leading health organization – World Health Organization (WHO). The WHO published *Health guidelines for the use of wastewater in agriculture and aquaculture* in 1989 as a 76-page report and prescribed microbiological quality guideline values for wastewater reuse in agriculture (WHO 1989). The same organization published *WHO guidelines for the safe use of wastewater, excreta and greywater* in 2006 in four volumes, with some of them above 200 pages in length for agriculture and aquaculture (WHO 2006a). However, the risk-based four-volume guidelines have not prescribed any guideline value, rather procedures for developing guideline values suitable to local circumstances (WHO 2006a), indicating the practical complexity involved.

Reclaimed water has been used in various urban purposes across the world. The reclaimed water quality guidelines prescribed in different regions of the world could be a practical reference for developing new guideline values. The existing reclaimed water quality guidelines in different regions of the world were critically reviewed and are presented in Table 3.6. The review reveals that different countries and even provinces or states within a country, environmental organizations (e.g., US EPA), and health organizations (e.g., WHO) have proposed their own guidelines. Unlike drinking water, no internationally accepted standard guideline values exist for reclaimed water.

Table 3.6 Reclaimed water quality guidelines for urban reuses in various regions of the world

Country	Unrestricted urban reuse	Restricted urban reuse	Urban agriculture: food crops	Source
North America				
Canada (federal)	Toilet and urinal flushing: *E. coli* or thermotolerant: ND (med), max <=200	-	-	Health Canada (2010)
Canada (BC)	Fecal coliforms: med < 1 or < 2.2 MPN; max 14 (Greater exposure potential)	Fecal coliforms: Moderate expo- median 100, max 400; Low expo- med 100, max 1000	*E. coli* (for crops eaten raw): < 1 or < 2.2 MPN	MWR (2012)
Canada (Alberta)	-	-	Tot col. < 1000 (geom of wk samples (if storage provided) or daily samples (if storage not provided); Fecal coliforms <200	Alberta Environment (2000)
US	Fecal coliforms: not detectable (Med); max 14	Fecal coliforms: med<=200; max <=800	Fecal coliforms: not detectable (med); max 14	US EPA (2012a)
US (California)	Tot. coli: 2.2 (7-day med); 23 (not more than 1 sample exceeds this value in 30 d); max 240	Total coliforms: 23 (7-d med); 240 (not more than one sample exceeds this value in 30 d)	Tot. coli.: 2.2 (7-day med); 23 (not more than 1 sample exceeds it in 30 d); 240 (max)	US EPA (2012a)
US (Florida)	Fecal coliforms: 75% of samples ND; max 25; *Giardia* and *Cryptosporidium*: sampling once each 2-yr period for plants ≥1 mgd; once each 5-yr period for plants ≤ 1 mgd	Not specified	Fecal coliforms: 75% of samples ND; max 25; *Giardia*, *Cryptosporidium*: sampling once per 2-yr period for plants ≥ 1 mgd; once per 5-yr period for plants ≤ 1 mgd	US EPA (2012a)
US (Hawaii)	Fecal coliforms: 2.2 (7-day med); 23 (not more than one sample exceeds this value in 30 d); 200 (max) (R1)	Fecal coliforms: 23 (7-day med); 200 (not more than one sample exceeds this value in 30 d) (R2)	Fecal coliforms: 2.2 (7-day med); 23 (not more than one sample exceeds this value in 30 d); 200 (max) (R1)	US EPA (2012a)
US (Nevada)	Total coliforms: 2.2 (30-d geom); 23 (max) (Category A)	Fecal coliforms: 2.2 (30-d geom); 23 (max) (Category B)	Total coliforms: 2.2 (30-d geom); 23 (max) (Category A)	US EPA (2012a)
US (New Jersey)	Fecal coliforms: 2.2 (wk med); 14 (max) (Type 1 RWBR)	Fecal coliforms: 200 (mon geom); 400 (wk geom) (Type 2 RWBR)	Fecal coliforms: 2.2 (wk med); 14 (max) (Type 1 RWBR)	US EPA (2012a)
US (North Carolina)	Fecal coliforms or *E. coli*: 14 (mon mean); 25 (max) (Type 1)	Fecal coliforms or *E. coli*: 14 (mon mean); 25 (daily max) (Type 1)	Processed: Type 1; Non-processed: Type 2: Fecal coli. or *E. coli:* 3 (mon mean); 25 (daily max); Coliphage (virus): 5 (mon mean); 25 (daily max); Clostridium: 5 (mon mean); 25 (daily max)	US EPA (2012a)

US (Texas)	Fecal coliforms or *E. coli*: 20 (30-d geom); 75 (max); *Enterococci*: 4 (30-d geom); 9 (max) (Type 1)	Fecal coli. or *E. coli*: 200 (30-d geom); 800 (max); *Enterococci*: 35 (30-day geom); 89 (max) (Type 2)	Fecal coli. or *E. coli*: 20 (30-d geom); 75 (max); *Enterococci*: 4 (30-d geom); 9 (max) (Type 1)	US EPA (2012a)
US (Virginia)	Fecal coliforms: 14 (mon geom), CAT > 49; *E. coli*: 11 (mon geom), CAT > 35; *Enterococci*: 11 (mon geom), CAT > 24m (Level 1)	Fecal coliforms: 200 (mon geom), CAT > 800; *E. coli*: 126 (mon geom), CAT > 235; *Enterococci*: 35 (mon geom), CAT > 104 (Level 2)	Fecal coliforms: 14 (mon geom), CAT > 49; *E. coli*: 11 (mon geom), CA> 35; *Enterococci*: 11 (mon geom), CAT > 24 (Level 1)	US EPA (2012a)
US (Washington)	Tot. coli: 2.2 (7-d med); 23 (max) (Class A)	Tot. coli.: 23 (7-d med); 240 (max) (Class C)	Total coli.: 2.2 (7-d med); 23 (max) (class A)	US EPA (2012a)
Arizona	-	Fecal coli: < 200 in last 4 of 7 samples; 800 (max) (Class B)	Fecal coliforms: ND in last 4 of 7 samples (Class A)	US EPA (2012a)
Australia				
National	National level guidelines (2006-2009) implemented but lacks specific water quality value recommendation			(EPHC/NHMRC/NRMMC (2006, 2008)
Western Australia	High exposure: indoor, irrigation (lawn), toilet flushing, cold tap washing machines: < 1	Medium expo: Urban irrigation (restricted access), firefighting, water features, dust suppression: < 10	Irrigation (unprocessed foods):< 1	WA DoH (2011)
Queensland	Class A (toilet flushing, lawn & golf course irrigation, water features): <10 (med)	Class B (washdown of hard surfaces in agri industries): <100 (med)	Class A (food crops with raw consumed foods irrigation: <10 (med)	QS EPA (2005)
Southern Australia	Class A (dual recirculation): <10 (med); unrestricted municipal irrigation: < 10 (med); landscape irrigation: < 1000	Class B (municipal with restricted access): <100 (med)	-	SA DHA (2012)
Victoria	Class A (non-potable urban use: toilet flushing, lawn & golf course irrigation, fountains & water features): <10 (med); viruses: 7-log reduction, Protozoa: 6-log reduction	Class C: Urban (non-potable,, controlled public access): < 1000	Class A (food crops with raw consumed foods irrigation : <10; Class C (Processed/cooked food): <1000 (med)	EPA Victoria (2003, 2015)
Europe				
European Union	Lack of coherent and comprehensive legislative although some countries have own standards			European Commission (2016)
Spain	Garden irrigation: 0; landscape irrigation, street cleaning, fire hydrants and car washing: 200	-	Food crops eaten raw: 100; crops not eaten raw: 1000	Royal Decree 1620/2007 (DoET and SERI 2014)

France	Public green spaces (parks, golf courses): <= 250 (sanitary level A)	Crops, vegetables processed by industrial heat; sold cut flowers: <= 10,000 (sanitary level B)	Crops, vegetables not processed by industrial heat: <= 250 (sanitary level A)	Decree of France (2016)
Greece	-	Restricted irrigation (no public access) & crops (processed before consumption): <= 200 (med)	Unrestricted irrigation for all crops such as vegetables (raw eaten), vines: <= 5 (for 80% of samples) and <=50 (for 95% of samples)	Ilias et al. (2014)
Italy	-	-	Vegetable crops: 10	Lonigro et al. (2015)
Middle East and Asia				
UAE (Abu Dhabi)	Unrestricted: 100	Restricted: 1000	Food crops: 100	UAE Guideline 2010 (DoET and SERI 2014)
Jordan	Public parks and road sides: 100; ground water recharge: <2.2	Landscape irrigation: 1000	Cooked food crops: 100	Jordanian Standards (JS:893/2002) (WHO 2005)
Kuwait	-	-	Crop irrigation with raw eaten (100 coliforms); Crops not eaten raw (10,000 coliforms)	WHO (2005)
Saudi Arabia	-	-	Unrestricted irrigation: coliforms 2.2	WHO (2005)
Mediterranean regions	Residential (toilet flushing, vehicle washing, gardening) & urban reuses (parks, golf courses, firefighting & recreation impoundments (pond & stream except bathing)): <=200	-	Irrigation of vegetables, fruit trees, landscape impoundments without public contact: <=1000	Mediterranean guidelines (proposed) EMWater (2001)
Japan	Toilet flushing: ND; sprinkling water: ND; recreational water: ND	Landscape irrigation: <=1000 as coliforms groups	-	Tajima (2007)
WHO (Global)	Safe Use of Wastewater 2006 for agriculture and aquaculture; complex for practical application, not recommended specific water quality values			WHO (2006b)

Unit: cfu/100 mL and is for E. *coli* unless stated; mon is monthly, wk: weekly, med: median, max: maximum ND: Not detectable, geom: geometric mean; mgd: million gallons daily, RWBR: Reclaimed Water for Beneficial Reuse, expo: exposure, Tot.: Total, coli.: coliforms, CAT (Corrective Action Threshold) = A bacterial, turbidity or total residual chlorine standard for reclaimed water at which measures shall be implemented to correct operational problems of the reclamation system within a specified period.

3.4.3 Fit-for-purpose wastewater treatment

Different water reuse applications require various grades of water quality, resulting in a number of required treatment levels. The production of higher quality water than required can result in overtreatment, leading to unnecessary cost and over use of resources such as energy. DSEWPaC (2012) suggests that a water reuse project cost must be determined on a case by case basis. A wastewater treatment train for a water reuse project can be selected based on the end use of reclaimed water for achieving economic efficiency and environmental sustainability (US EPA 2012a). Such treatment is referred to as fit-for-purpose wastewater treatment. It aims to avoid overtreatment, and obviously under-treatment as it is legally prohibited. Water quality depends on the level of water and wastewater treatment, which is dictated by the end use of reclaimed water.

Wastewater treatment technologies differ mainly in terms of cost (Guo et al. 2014), treatment efficiency (Health Canada 2010), energy consumption (Chang et al. 2008), and the related carbon emissions. Wastewater treatment technologies also affect the efficiency of water recycling, i.e., the volume of reclaimed water produced, especially when influent wastewater is highly polluted. All above factors determine the required level of fit-for-purpose wastewater treatment, which requires a decision support tool (DST) for the evaluation of treatment trains for a community. The DST helps in ranking and identifying a cost-effective, risk-acceptable, and energy efficient treatment train to meet the water quality for an intended use.

Several DSTs are available and are in practice for the planning and operation of wastewater treatment plants. The DSTs related to fit-for-purpose wastewater treatment were reviewed and summarized in Table 3.7. The review reveals that various DSTs have their own objectives and applications. For example, an Excel-based ECAM tool was developed for evaluating the energy performance and carbon emissions of water and wastewater utilities (GIZ/MENCBNS/IWA 2015) and the WEST tool was developed to assess the environmental effects including water use, energy use, and carbon emissions of water and wastewater infrastructure (Stokes et al. 2011). Both of these tools lack the capability to estimate health risk associated with the treated water used for a specific purpose and rank the corresponding wastewater treatment chains. Some researchers developed QMRA tools, such as QMRAspot (Schijven et al. 2011) and QMRAcatch

(Schijven et al. 2015). The tools were developed only to assess the health risks associated with water use. However, these tools have included either drinking water or recreational water only and they cannot be used to screen various treatment processes. Therefore, no DST is flexible and capable enough to evaluate the potential of wastewater treatment and reuse for different purposes simultaneously based on cost, health risk, energy use, carbon emissions, and amount of reclaimed water production.

The decisions with regards to the planning of reclaimed water use projects for a specific reuse application should consider these major factors: quantity, quality, cost, energy, and carbon emissions (NASEM 2016; Nasiri et al. 2013; Zarghami and Akbariyeh 2012). This requires a DST to evaluate alternative wastewater treatment trains and reuses. However, such a tool is not available in the publically accessible literature. The national water reuse assessment report of Australia also revealed the need of a similar DST for a high-level evaluation of reclaimed water reuse projects, called hotspot analysis, across the country (DSEWPaC 2012).

Table 3.7 Existing DSTs related to fit-for-purpose wastewater treatment

DSTs	Applications	Limitations	Reference
Energy performance and Carbon emissions Assessment and Monitoring (ECAM)	Evaluate energy performance and carbon emissions of water and wastewater utilities	Lacks health risk estimation for specific reuse; not capable to rank alternative wastewater treatment chains	GIZ/MENCB NS/IWA (2015)
Water-Energy Sustainability Tool (WEST)	Evaluate environmental impacts of life cycle of water and/or wastewater infrastructure	”	Stokes et al. (2011)
water-energy nexus of UWS	Evaluate cost effectiveness strategies for water utilities in California	”	Cutter et al. (2014)
Carbon cost model	Estimate GHG emissions and associated cost in water supply and demand options in UK	”	Reffold et al. (2008)
Online design tool	Automatically designs a preliminary WWTP (activated sludge or food chain reactor) using the provided input: country (region); hydraulic capacity or population equivalent, and generic effluent criteria; design provides an overall energy intensity and an equipment list	Limited effluent criteria options; not specific to a particular water reuse; does not estimate cost	Organica Water Inc. (2016)
QMRAspot	Assess microbial risk of a drinking water production chain from surface water to potable water; determine drinking water treatment efficiency related to the legislative health-based target	Does not rank specific treatment processes to overcome the target risk	Schijven et al. (2011)
QMRAcatch	Catchment model to assess the health risks associated with *E. coli*, enterovirus, norovirus, *Campylobacter* and *Cryptosporidium* in water resources in a catchment	Includes only recreational and drinking water; does not rank treatment processes	Schijven et al. (2015)
Recycled water Irrigation Risk Analysis (RIRA)	Evaluate human health risks of reclaimed water use in irrigation; users can choose pathogens from the given list in the tool and input their concentrations in water or foods; deterministic model	Only for irrigation water; not able to recommend corresponding treatment units; does not estimate energy use and cost	Hamilton et al. (2007)
QMRA Wiki	Provides fundamental information, steps, and online calculators to conduct QMRA; intended to be a reference source for the QMRA community	Only for risk assessment; users should know background knowledge on risk assessment; calculators cannot recommend treatment units and cost for a specific water reuse	CAMRA (2016)
Energy Use Assessment Tool	Evaluate energy and cost of small and medium sized water and wastewater utilities	Not applicable to select an optimum treatment chain for safe water reuse in a specific reuse application	US EPA (2012b)

Chapter 4 Identification of Sustainability Performance Indicators

A version of this chapter has been published in *Water Environment Research* journal with a title "Sustainability performance indicators for small to medium sized urban water systems: A selection process using Fuzzy-ELECTRE method" (Chhipi-Shrestha et al., 2017a).

4.1 Background

Urban Water Systems (UWSs) can be viewed at different spatial scales: building, neighbourhood, community, city, and metropolis. Sustainability issues of UWSs can vary with these scales. In particular, the recovery of resources, such as energy, water, and nutrients, from wastewater is important from the urban water sustainability perspective. The recovered resources are required to be distributed efficiently to households to meet their needs. Therefore, the distance between a recovery station and potential users, especially for the reclaimed water use is a critical factor (Wang et al. 2008). However, such a distance is longer in centralized UWSs. In addition, conventional centralized UWSs, specifically wastewater systems, have poor flexibility in associated facilities, a high and long-term capital investment (Bieker et al. 2010). On the other hand, household level wastewater treatment may be appropriate in low-density households, but not in densely populated urban areas due to operation risk and limited space availability (Bieker et al. 2010). The limitations of the centralized level and that of decentralized level (household level) wastewater treatments could be overcome using an intermediate scale of UWS. CCME (2002), Asano et al. (2007), Bieker et al. (2010), and Zarski and Ancel (2012) have also indicated the suitability of UWSs at such scale. This level is referred to as the "*urban community*" scale or small to medium-sized urban water systems (SMUWSs).

4.1.1 Small to medium-sized urban water systems (SMUWSs)

The population size in small to medium-sized urban water systems (SMUWSs) depends on its location and should be guided by the principle "as small as possible, as big as necessary" (Bieker et al. 2010). For example, Bieker et al. (2010) proposed a community size of 50,000 to 100,000 or even lower population, whereas Böhm et al. (2011) proposed a population of 20,000 for the community in China. Since, drinking water is a major water input to an UWS and also primarily determines the magnitude of wastewater amount, the drinking water system can be considered as a basis of size classification of UWSs as given in Table 4.1.

Table 4.1 Classification of UWSs based on US EPA (2009a)

System size	Population served
Small	<3,300
Medium	3,300 – 100,000
Large	>100,000

Broadly, the population size of a SMUWS can be up to 100,000; however, the community should be compact as a lower dwelling density in a neighbourhood and a higher dispersion of neighbourhoods increase the cost of providing water and wastewater services (Speir and Stephenson 2002). A SMUWS serves a group of neighbourhoods that can be a town, a small city, a municipality, or a part of any of these. In Canada, the proportion of SMUWs is very high. For example, the municipalities with the population of 5,000 or less are above 80% (FCM and NRC 2005). The SMUWSs are different from large UWSs and have the following characteristics:

- Small service area and population
- Smaller infrastructure
- Limited data availability
- Institutional limitations because a SMUWS may cover only a small part of a municipality
- Low technical capability in terms of staff and equipment in smaller urban communities
- Limited financial resources in smaller urban communities

The sustainability of UWSs can be assessed using sustainability performance indicators (SPIs). Available SPIs are established mainly for large UWSs (Foxon et al. 2002; Van Leeuwen et al. 2012; Van Leeuwen and Marques 2013) and cannot be adopted as is for a sustainability assessment of a SMUWS due to different characteristics as listed above. This chapter addresses the gap and aims to develop a set of applied indicators to assess the holistic sustainability of small to medium-sized UWSs. The UWSs can be existing or new.

4.1.2 Fuzzy sets and fuzzy numbers

Zadeh (1965) introduced fuzzy sets to analyze uncertainty caused by imprecision and vagueness in decision making. Fuzzy sets are a useful tool for modelling language to approximate a system having fuzzy phenomena (Chhipi-Shrestha et al. 2016; Chu 2011).

The fuzzy set A can be represented as:

$$A = \{(x, f_A(x)) \,/\, x \,\varepsilon\, U\} \qquad \textbf{Equation 4.1}$$

where U is the universal set, x is an element in U, A is a fuzzy set in U,

$f_A(x)$ is the membership function of A at x. The larger $f_A(x)$, the stronger the grade of membership for x in A (Chhipi-Shrestha et al. 2016; Chu 2011).

Similarly, a real fuzzy number A is described as any fuzzy subset of the real line R with membership function f_A.

The membership function f_A of the fuzzy number A can be expressed as:

$$f_A(x) = \begin{Bmatrix} f_A^L(x), a \le x \le b \\ 1, \quad b \le x \le c \\ f_A^R\ (x), c \le x \le d \\ 0, \quad otherwise \end{Bmatrix} \qquad \textbf{Equation 4.2}$$

where $f_A{}^L(x)$ and $f_A{}^R(x)$ are left and right membership functions of A respectively,

$a \le b \le c \le d$, and A can be represented by (a, b, c, d).

Various fuzzy numbers can be used depending on the condition, but triangular fuzzy numbers (TFNs) are commonly used due to computational simplicity (Sevkli 2010). In this study, TFNs were used. TFNs can be defined as a triplet (p, q, r), where the parameters p, q, and r indicate the smallest possible value, the most promising value, and the largest possible value, respectively, which describe a fuzzy event (Sevkli 2010). A triangular fuzzy number $\tilde{A}$ = (p, q, r) is given in Figure 4.1 (Chhipi-Shrestha et al. 2016).

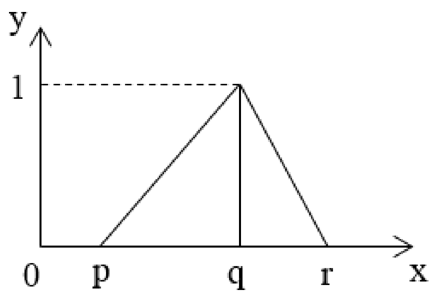


Figure 4.1 Membership function of $\tilde{A}$

The mathematical operations of TFNs for the two positive triangular fuzzy numbers (a_1,b_1,c_1) and (a_2,b_2,c_2) are given below: A TFN (a,b,c) is said to be a positive TFN if and only if $a \geq 0$.

$(a_1, b_1, c_1) + (a_2, b_2, c_2) = (a_1 + a_2, b_1 + b_2, c_1 + c_2)$ **Equation 4.3**

$(a_1, b_1, c_1) * (a_2, b_2, c_2) = (a_1 * a_2, b_1 * b_2, c_1 * c_2)$ **Equation 4.4**

$(a_1, b_1, c_1) * k = (a_1 * k, b_1 * k, c_1 * k)$, where $k \geq 0$ **Equation 4.5**

4.2 Methodology

A comprehensive literature review was performed using keywords search, through web-based scientific search engines and online databases. Several researchers have used specific keywords for searching literature for conducting comprehensive reviews (Jørgensen, 2013; Yi and Chan, 2014). The keywords include: urban water sustainability, sustainability indicators of urban water, city water sustainability, water sustainability assessment, neighbourhood sustainability assessment, city sustainability assessment, water sustainability, sustainability performance of water, and community water. These keywords were searched in databases such as Compendex Engineering Village, Web of Knowledge databases, electronic library of the University of British Columbia, Canada, and web-based search engine http://scholar.google.ca/. Since, the sustainability of an UWS at the community scale or SMUWS is to be assessed, three categories of literature have been reviewed: a) sustainability assessment of UWSs, b) performance assessment of water and wastewater services that focus on sustainability, and c) neighbourhood and city sustainability assessment.

The SPIs were selected based on the methodological framework (Figure 4.2) consisting the initial screening, development of selection criteria, Delphi method, and multi-criteria decision analysis using Fuzzy-ELECTRE I.

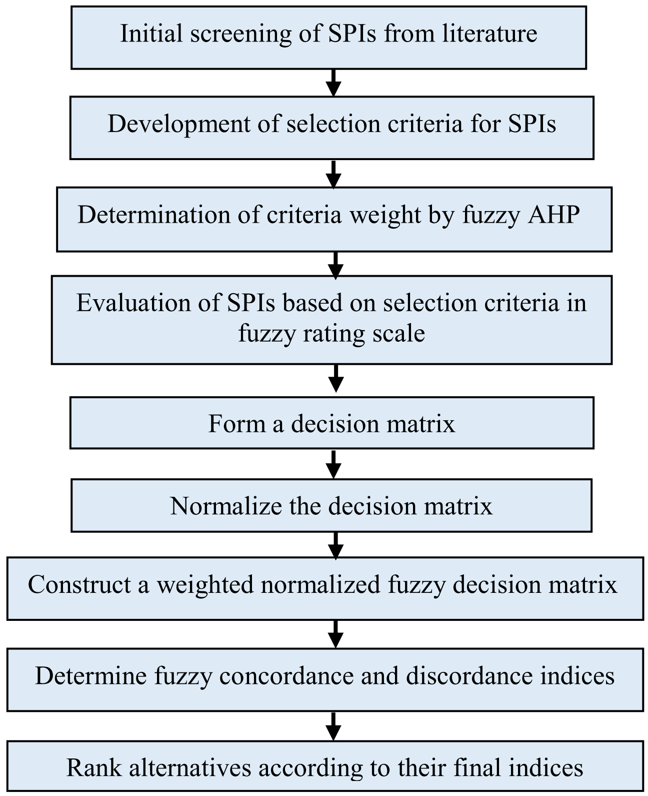


Figure 4.2 Methodological framework used for the selection of SPIs

4.2.1 Initial screening of SPIs

An initial screening of SPIs for SMUWSs was performed by a simple *checklist method* considering smaller infrastructures, data limitations (Haider et al. 2014a), small population, small service area, and institutional limitations of urban communities. The screened SPIs were categorized into five sustainability dimensions: technical, environmental, economic, social, and institutional dimensions (World Bank 2003; Van Leeuwen and Marques 2013).

4.2.2 Development of selection criteria

Four selection criteria developed for the selection of initially screened SPIs are *relevance (importance)* to sustainability, *measurability*, *data availability,* and *comparability* (adapted from

Lundin 2002 and Haider et al. 2014a) and their ratings are based on Likert type scale (Tveit 2009) as given in Table 4.2.

a. *Relevance*: How much an indicator is relevant and comprehensive to the sustainability of the small to medium-sized urban water systems? It is related to the technical, environmental, social, or economic relevance and the comprehensiveness of many features of the sustainability dimension.

b. Measurability: How much a variable is measurable accurately and requires the extent of observations for the calculation of indicators?

c. Data availability: How is the availability of the data for indicator calculation?

d. Comparability: How much the value of SPI is comparable with the available reference value? It is related to whether the indicator is used for urban water sustainability assessment in the region and/or at the international level.

The relevance criteria of SPIs was rated by using a 5-point linguistic scale (very high, high, medium, low, and very low), whereas the measurability, data availability, and comparability criteria were measured by using a 3-point linguistic scale (high, medium, and low). The relevance criteria was categorized into five categories in order to capture the wide variability of rating provided by experts, whereas other three criteria were evaluated more objectively having less variability in rating for which three categories were used. Similar approach was also used by Haider et al. (2014b), Hung et al. (2010), and Tveit (2009). Since these ratings are linguistic and imprecise, their calculation was performed using fuzzy sets. Zadeh (1965) introduced fuzzy sets to analyze uncertainty in decision making caused by imprecision and vagueness. The fundamentals of fuzzy sets, fuzzy number, and their mathematical operations (Chhipi-Shrestha et al. 2016) are provided in Section 4.1.2.

Table 4.2 Criteria for the selection of SPIs

Criteria (score[1])	Description
Relevance	
Very high (0.7,1,1)	Must be included, SPI is highly relevant and more comprehensive indicator of sustainability of SMUWSs.
High (0.5,0.7,0.9)	SPI is highly relevant and of average comprehensiveness.
Medium (0.3,0.5,0.7)	SPI is of average relevance and average comprehensiveness.
Low (0.1,0.3,0.5)	SPI has low relevance to the sustainability of SMUWSs.
Very low (0, 0, 0.3)	SPI seems to be irrelevant for sustainable development for SMUWSs.
Measurability	
High (0.5,1,1)	Variables have absolute values or one annual observation provides the data (e.g., proximity to drinking water system) for the variables.
Medium (0,0.5,1)	Variables have highly varying values that require a large number of observations in a year.
Low (0,0,1)	Variables have qualitative data or have estimated values.
Data availability	
High (0.5,1,1)	Data are available in public annual municipal reports (water, wastewater, and financial reports) and official water master plan.
Medium (0,0.5,1)	Data are available in raw form in internal official records.
Low (0,0,1)	Data are only available in occasional study reports or rarely available.
Comparability	
High (0.5,1,1)	SPI has been used for urban water sustainability assessment in the region (country).
Medium (0,0.5,1)	SPI has been used for urban water sustainability assessment outside the region.
Low (0,0,1)	SPI has rarely been used for urban water sustainability assessment.

(Adapted from: Lundin (2002), Tveit (2009) and Haider et al. (2014a))

The relevance criteria was evaluated based on a group decision of experts using the Delphi method. The other three criteria were rated based on the reported literature. For the rating of the data availability criteria, five small to medium-sized municipalities were randomly selected from a list of small to medium-sized municipalities by coding and then using a statistical random table (Gibbons et al. 1999) in each of three large provinces of Canada: Ontario, Alberta, and British Columbia. The selected municipalities are the Cities of Belleville, Brantford, North Bay, St. Thomas, and Stouffville of Ontario; the Cities of Airdrie, Leduc, Lethbridge, Red Deer, and Spruce Grove of Alberta; and the Cities of Parkville, Prince George, Penticton, Vernon, and District of Kitimat of British Columbia. Their public annual municipal reports (water,

[1] Triangular Fuzzy Number (TFN) represent by lowest possible(l), middle (m), and highest possible (u) values for all scales

wastewater, and financial reports) and official water master plan as far as available were referred to perform the ratings of the data availability criteria.

For the rating of the comparability criteria, national reports were used. Comparability was considered to be high if a SPI is available in the national reports, i.e., National Water and Wastewater Benchmarking Initiative (NWWBI) (AECOM 2012) or Municipal Water Use Report (Environment Canada 2011). The NWWBI report was prepared based on the assessment of 44 wastewater utilities, 41 water utilities, and 17 storm water management programs, whereas the municipal water use report was prepared based on the data of 2,779 Canadian municipalities. Similarly, the comparability criteria was considered to be medium if a SPI is used for urban water sustainability assessment by any international literature identified as the major literature in Table 3.1 (Chapter 3). The comparability criteria was considered low if a SPI has rarely been used for urban water sustainability assessment. Furthermore, the weights of the four selection criteria were determined based on a group decision of experts using the Delphi method as explained in detailed in Appendix A.1.

The weights of these selection criteria were determined by the fuzzy-Analytical Hierarchical Process (F-AHP) (Kaya and Kahraman 2011) using a group decision method. The calculated weights of the relevance, measurability, data availability, and comparability criteria in terms of TFNs are (0.19, 0.32, 0.49), (0.20, 0.27, 0.35), (0.18, 0.24, 0.35), and (0.11, 0.17, 0.27) respectively. The consistency ratio (CR) of the comparison matrix is 0.0076 that indicates a consistent matrix as the ratio is lower than 0.10 (Alonso and Lamata 2006). In addition, CRs were less than 0.1 for the comparison matrices of all participants.

4.2.3 Fuzzy-ELECTRE I

The rating of each SPI for the four criteria was converted to fuzzy scores using TFNs (Hung et al. 2010; Chen et al. 2008) as given in Table 4.2. A decision matrix was formed using these scores and then normalized. An outranking method called fuzzy- ELECTRE I (ELimination Et Choix Traduisant la REalite´, i.e., Elimination and Choice Translating Reality I) was applied for ranking and selecting SPIs. The ELECTRE method was developed by Bernard Roy in the late 1960s. The method uses concordance and discordance indices to determine outranking relations among the alternatives. Concordance and discordance indices can be viewed as satisfaction and dissatisfaction measurements that a decision maker chooses one alternative over the other (Rouyendegh and Erkan 2013). The fuzzy ELECTRE I method is the application of the usual ELECTRE I method to fuzzy data. The main benefits of the fuzzy- ELECTRE technique are as follows: it is highly applicable when criteria are measured in an ordinal scale, has small differences in evaluations, and is non-compensatory (Mousseau and Roy 2014). The method consists of the following steps (Sevkli 2010; Rouyendegh and Erkan 2013).

Step 1. A group of 30 decision makers knowledgeable in the field of urban water management with an experience more than three years was formed. The group was responsible for the evaluation of the relevance criteria and the weights of the four selection criteria. By using the Delphi method, consensus was reached to the rating of all SPIs and the criteria weights. The fuzzy importance weight for each criterion can be described as TFNs $\widetilde{w}_j = (l_j, m_j, u_j)$ for j =1, 2, 3, and 4, where a tilde (~) represents a fuzzy number. The other three criteria measurability, data availability, and comparability were rated based on the literature.

Step 2. Normalized decision matrix

A fuzzy decision matrix was formed for each sustainability dimension and normalized to obtain a normalized decision matrix $\tilde{X}$ as given below.

$$\widetilde{\boldsymbol{X}} = \begin{bmatrix} \tilde{\boldsymbol{r}}_{11} & \tilde{\boldsymbol{r}}_{12} \dots \tilde{\boldsymbol{r}}_{14} \\ \tilde{\boldsymbol{r}}_{21} & \tilde{\boldsymbol{r}}_{11} \dots \tilde{\boldsymbol{r}}_{24} \\ \dots \quad \dots & \dots. \quad \dots \\ \tilde{\boldsymbol{r}}_{m1} & \tilde{\boldsymbol{r}}_{m2} \dots \tilde{\boldsymbol{r}}_{m4} \end{bmatrix}$$ **Equation 4.6**

where $\tilde{r}_{ij} = (r^{l}_{ij}, r^{m}_{ij}, r^{u}_{ij})$ and **Equation 4.7**

$$r^{l}_{ij} = \frac{x^{l}_{ij}}{\sqrt{\sum_{i=1}^{m} (x^{u})^{2}_{ij}}} \ , \ r^{l}_{ij} = \frac{x^{m}_{ij}}{\sqrt{\sum_{i=1}^{m} (x^{m})^{2}_{ij}}} \ , \ r^{l}_{ij} = \frac{x^{u}_{ij}}{\sqrt{\sum_{i=1}^{m} (x^{l})^{2}_{ij}}}$$ **Equation 4.8**

where $\tilde{x}_{ij} = (x^{l}_{ij}, x^{m}_{ij}, x^{u}_{ij})$ is an actual rating score and its normalized score is *r* with i = 1, 2, ……, m (m= 17, 24, 8, 10, and 9 for technical, environmental, economic, social, and institutional dimensions respectively); j = 1, 2,……4, and the superscripts l, m, and u respectively refer to lower, middle, and upper values of TFNs.

Step 3. Normalized weighted matrix

A weighted fuzzy decision matrix was computed by multiplying the normalized decision matrix $\tilde{X}$ with the criteria weights ($\widetilde{w}_j$), and then normalized according to Equation 4.8. The normalized weighted matrix $\tilde{V}$ is shown in Equation 4.9.

$$\tilde{V} = [\tilde{v}_{ij}]_{m\times n} \qquad \textbf{Equation 4.9}$$

where i and j are same as previously defined; $\tilde{v}_{ij}= \tilde{r}_{ij} \times \widetilde{w}_j$; $\widetilde{w}_j = (w_{j1}, w_{j2}, w_{j3})$, i.e., the relative weight of the jth criterion and

$$V^l = \begin{bmatrix} v_{11}^l & v_{12}^l & \dots & v_{14}^l \\ v_{21}^l & v_{21}^l & \dots & v_{24}^l \\ \dots & \dots & \dots & \dots \\ v_{m1}^l & v_{m2}^l & \dots & v_{m4}^l \end{bmatrix}, \quad V^m = \begin{bmatrix} v_{11}^m & v_{12}^m & \dots & v_{14}^m \\ v_{21}^m & v_{21}^m & \dots & v_{24}^m \\ \dots & \dots & \dots & \dots \\ v_{m1}^m & v_{m2}^m & \dots & v_{m4}^m \end{bmatrix}, \text{ and } V^u = \begin{bmatrix} v_{11}^u & v_{12}^u & \dots & v_{14}^u \\ v_{21}^u & v_{21}^u & \dots & v_{24}^u \\ \dots & \dots & \dots & \dots \\ v_{m1}^u & v_{m2}^u & \dots & v_{m4}^u \end{bmatrix}$$

Equation 4.10

where $\tilde{v}_{ij}$ is a positive TFN.

Step 4. Concordance and discordance sets

The concordance and discordance sets were developed for each matrix $V^l, V^m,$ and V^u representing lower (l), middle (m), and upper (u) values of TFNs respectively. For each pair of alternative A_p and A_q (p, q = 1, 2, ..., m and p ≠q), the set of criteria was classified into two distinct subsets. If the alternative A_p was preferred over alternative A_q for all the criteria, then the concordance set was composed and expressed as:

$$C(p,q) = \{j \mid v_{pj} \geq v_{qj}\} \qquad \textbf{Equation 4.11}$$

where v_{pj} is the normalized weighted rating of the alternative A_p with respect to the jth criterion. In other words, C(p, q) is the collection of attributes where A_p is better than or equal to A_q. The complement of C(p, q) known as the discordance set, contains all the criteria for which A_p is worse than A_q and can be expressed as

$$D(p, q) = \{j \mid v_{pj} < v_{qj}\} \quad \textbf{Equation 4.12}$$

Step 5. Concordance and discordance indices

The concordance and discordance indices were computed for l, m, and u values of each criterion having the weights w_{j1}, w_{j2}, and w_{j3} respectively. The concordance index C_{pq} indicates the degree of confidence in pairwise - judgments ($A_p \rightarrow A_q$). The concordance index C_{pq} is defined as

$$C_{pq}^{l} = \sum_{j*} w_{j1},\; C_{pq}^{m} = \sum_{j*} w_{j2},\, C_{pq}^{u} = \sum_{j*} w_{j3} \quad \textbf{Equation 4.13}$$

where j* are attributes contained in the concordance set C (p, q).

Similarly, the discordance index measures the power of a discordance set, i.e., the degree of disagreement in ($A_p \rightarrow A_q$), which can be expressed as:

$$D_{pq}^{l} = \frac{\sum_{j+} |v_{pj+}^{l} - v_{qj+}^{l}|}{\sum_{j} |v_{pj}^{l} - v_{qj}^{l}|} \quad , \quad D_{pq}^{m} = \frac{\sum_{j+} |v_{pj+}^{m} - v_{qj+}^{m}|}{\sum_{j} |v_{pj}^{m} - v_{qj}^{m}|} \text{, and } D_{pq}^{u} = \frac{\sum_{j+} |v_{pj+}^{u} - v_{qj+}^{u}|}{\sum_{j} |v_{pj}^{u} - v_{qj}^{u}|} \quad \textbf{Equation 4.14}$$

where J^+ are the criterion contained in the discordance set D (p,q) and v_{ij} is the normalized weighted evaluation of the alternative i on the criterion j.

Step 6. Final indices calculation

The final concordance (C^*_{pq}) and discordance (D^*_{pq}) indices are geometric means of l, m, and u values separately of C_{pq} and D_{pq}. These indices can be considered as the defuzzification and were computed as:

$$C_{pq}^{*} = \sqrt[z]{\prod_{z=1}^{z} C_{pq}^{z}} \quad \text{and} \quad D_{pq}^{*} = \sqrt[z]{\prod_{z=1}^{z} D_{pq}^{z}} \quad \textbf{Equation 4.15}$$

where Z=3 denoting three values l, m, and u.

A larger final concordance index C_{pq} and a smaller final discordance index D_{pq} resulted in a stronger dominance relationship of the alternative A_p over the alternative A_q. The outranking relation was obtained by using Equation 4.16 and Equation 4.17.

If $C_{pq}^{*} \geq \bar{C}$ and **Equation 4.16**

$D_{pq}^{*} < \bar{D}$ **Equation 4.17**

where $\bar{C}$ and $\bar{D}$ are averages of C_{pq} and D_{pq} respectively.

In this method, A_p outranks (better than) A_q when Equation 4.16 and Equation 4.17 hold true, whereas, alternative A_p is indifferent to A_q when both hold false, and the alternative A_p is incomparable to A_q when one holds true with another false. Based on these relationships, an outranking diagram of SPIs were developed for each sustainability dimension (Yoon and Hwang 1995).

Step 7. Ranking SPIs

The net outranking relationships can be established using a net concordance index (C_p) and the net discordance index (D_p) for each SPI (alternative). C_p measures the degree to which the dominance of an alternative A_p over competing alternatives exceeds the dominance of competing alternatives over the alternative A_p and can be defined as follows (Yoon and Hwang 1995; Haider et al. 2014b):

$$C_p = \sum_{\substack{k=1 \\ k \neq p}}^{m} C_{pk} - \sum_{\substack{k=1 \\ k \neq p}}^{m} C_{kp}$$ **Equation 4.18**

Similarly, the D_p measures the relative weakness of alternative A_p with respect to other alternatives and can be defined as follows (Yoon and Hwang 1995; Haider et al. 2014b):

$$D_p = \sum_{\substack{k=1 \\ k \neq p}}^{m} D_{pk} - \sum_{\substack{k=1 \\ k \neq p}}^{m} D_{kp}$$ **Equation 4.19**

For making an overall preference (ranks), a higher C_p and lower D_p will receive a higher rank. The final ranking was performed based on the values of C_p and D_p of SPIs and their outranking relationships.

4.3 Results

A total of 68 potential SPIs were initially screened from the literature for the sustainability assessment of SMUWSs. The screened list is comprised of 17 SPIs in the technical, 24 in the environmental, 8 in the economic, 10 in the social, and 9 in the institutional dimension. These SPIs with their description and measurement method are given in Appendix A.2. Also, the methodological steps involved, including the application of F-AHP, is elaborated in detailed in Appendix A.3 for the ranking of SPIs in the economic dimension as an example. The application of the fuzzy-ELECTRE I method to the initially screened SPIs resulted in outranking relationships. The outranking relationships for each sustainability dimension are given in Figure 4.3, Figure 4.4, Figure 4.5, Figure 4.6, and Figure 4.7. These relationships can be used to identify important SPIs for sustainability assessment of SMUWSs.

The most important SPIs are those positioned at the upper level of outranking diagrams. The decision maker's boundary (DMB) was used as a cut-off boundary to select the final SPIs (Haider et al. 2014b). A DMB is selected by a decision maker and is based on the relative distance between the indices (concordance and discordance) of SPIs. The final SPI list consists of 38 SPIs including 8 SPIs in the technical, 13 SPIs in the environmental, 4 SPIs in the economic, 7 SPIs in the social, and 6 SPIs in the institutional dimensions as follows:

4.3.1 Technical

The selected eight technical SPIs belong to the "neighbourhood location and design" and "water infrastructure and fixtures" criteria (Figure 4.3). The SPIs within the neighbourhood location and design criteria are as follows: proximity to drinking water system/source, proximity to wastewater system, separation of wastewater and storm water, and dwelling density. The SPI proximity to drinking water system/source is one of two top level indicators. Another top level indicator, water leakage, is discussed below. Similarly, the selected SPIs of the water infrastructure and fixtures criteria are as follows: water leakage, water supply reliability, metered connection, and treated water storage capacity.

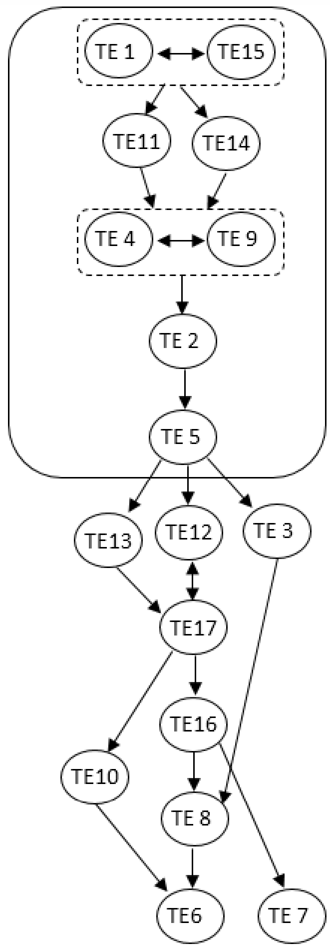


Figure 4.3 Outranking relations of the technical SPIs with DMB

Legend

TE1 → TE2 TE1 outranks TE2

TE1 ↔ TE2 TE1 and TE2 outranks each other (equally important)

SPIs grouped at a level

Decision maker's boundary (DMB) for final preferences

Descriptions of TE1, TE2, etc. are given in Appendix A.2.

4.3.2 Environmental

The relationship among environmental SPIs is presented in the outranking diagram in Figure 4.4. Altogether, 13 SPIs were selected in this dimension. The SPIs are grouped into the following criteria: resource utilization, environmental impacts, and resource recovery. The selected SPIs of the resource utilization criteria are as follows: water self-sufficiency, domestic water consumption, non-domestic water consumption, groundwater quality, surface water quality, energy use in water service, energy use in wastewater service, chemical use in water treatment, and chemical use in wastewater treatment. Water self-sufficiency and domestic water consumption are placed at the top level along with water reuse indicator. Similarly, the three selected SPIs of the environmental impacts criteria are as follows: discharged wastewater quality, biosolids quality, and disposal of backwash water. In the resource recovery criteria, the SPI water reuse was selected. In the resource recovery criteria, the SPI water reuse was selected. Water reuse is the third of the top level indicators along with the proximity to drinking water system and surface water quality indicators.

4.3.3 Economic

The economic SPIs and their relationships are depicted in Figure 4.5. Four SPIs have been selected in this dimension and were categorized into two criteria "water economics" and "wastewater economics". In the water economics criteria, the selected SPIs are operating cost coverage ratio for water service, average water fee rate, and non-revenue water. Similarly, in the wastewater economics criteria, the selected SPI is operating cost coverage ratio for wastewater service. The top level indicators were found to be operating cost coverage ratio for water service and that for wastewater service. Similarly, at the second level, two SPIs average water fee rate and non-revenue water are placed. The top level indicators identified are operating cost coverage ratio for water service and that for wastewater service. At the second level, two SPIs average water fee rate and non-revenue water are placed.

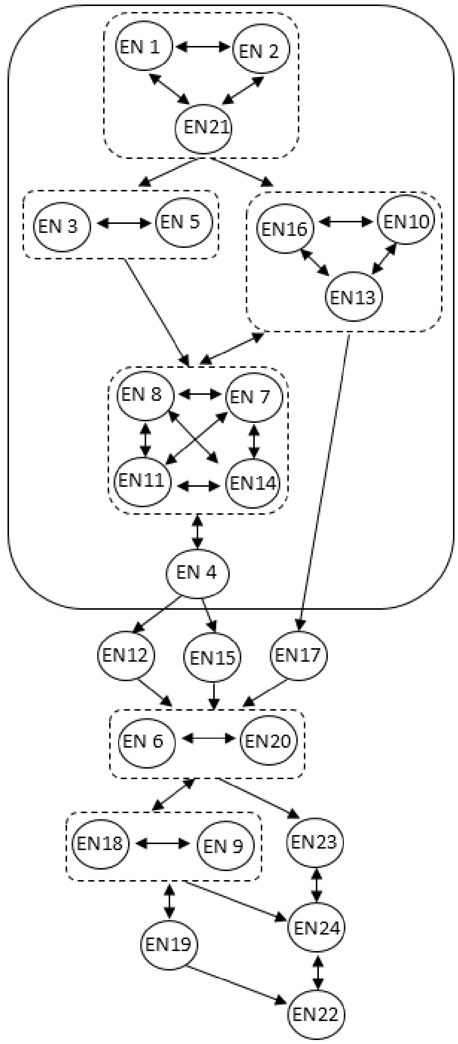


Figure 4.4 Outranking relations of the environmental SPIs with DMB

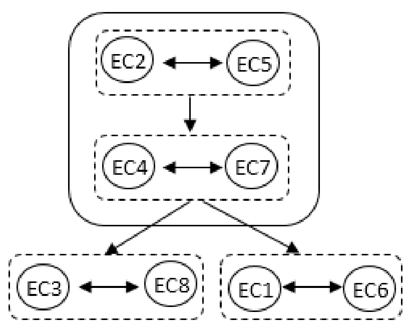


Figure 4.5 Outranking relations of the economic SPIs with DMB

4.3.4 Social

The relationships among social SPIs are presented in Figure 4.6. Altogether, seven SPIs have been selected. These SPIs belong to the "service provision" and "public health" criteria. The SPIs: access to water service, access to wastewater service, and drinking water quality outranked all other indicators. In the service provision criteria, the selected SPIs are as follows: access to water service, access to wastewater service, water restrictions, and public acceptability. Similarly, in the public health criteria, the selected SPIs are drinking water quality, boil water advisories, and safety (from flooding and drought). The SPI drinking water quality was found to be at the top level along with the access to water and wastewater service indicators.

4.3.5 Institutional

The institutional SPIs and their relationships are depicted in Figure 4.7. Six SPIs have been selected, and they belong to the "governance and progress" criteria. The top level SPI is urban water policies that is indifferent with the SPI achievement of water demand reduction target. The SPIs institutional capacity and personnel training are placed at the next level.

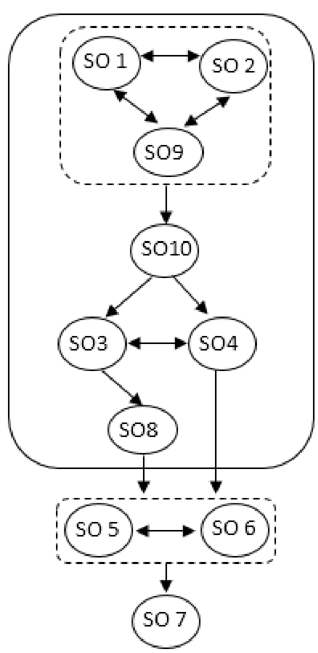


Figure 4.6 Outranking relations of the social SPIs with DMB

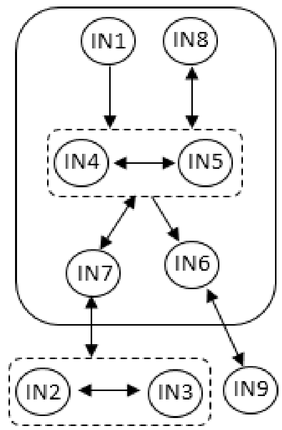


Figure 4.7 Outranking relations of the institutional SPIs with DMB

The list of selected SPIs are given in Table 4.3.

4.4 Discussion

The applicability of SPIs in urban communities is described under each sustainability dimension below and is confined to the final SPIs.

4.4.1 Technical

In the neighbourhood location and design criteria, the SPI proximity to drinking water system/source is an important indicator. The further the community is from existing water and wastewater systems, the less sustainable it is as it consumes more resources for the construction, operation, and maintenance of its water and wastewater systems. However, a community should be built at a certain buffer distance from a water source (USGBC 2013). Similarly, the proximity to wastewater system is an essential indicator. Speir and Stephenson (2002) showed that a longer distance between neighbourhoods (development tracts) and existing water and wastewater services increases the cost of providing these services.

Table 4.3 Final Ranks of the Selected SPIs of UWSs

Technical		Environmental		Economic		Social		Institutional	
R	SPI*	R	SPI	R	SPI	R	SPI	R	SPI
1	Proximity to water (TE1)	1	Self-sufficiency (W) (EN1)	1	Oper. cost (W) (EC2)	1	Access to water (SO1)	1	Demand reduc. (IN8)
2	Water leakage (TE15)	2	Domestic consumption (EN2)	2	Non-revenue water (EC5)	1	Access to WW (SO2)	2	Policies (IN1)
3	Water storage (TE11)	3	Water reuse (EN21)	3	Avg. water fee (EC4)	3	Drinking WQ (SO9)	3	Insti.capacit y (IN4)
4	Supply reliability (TE14)	4	Non-domestic consumption (EN3)	3	Oper. cost (WW) (EC7)	4	BWA (SO10)	3	Personnel training (IN5)
5	Separation of WW and SW (TE4)	4	Surface WQ (EN5)	-	-	5	Water restrictions (SO3)	5	Consvn. Programs (IN6)
5	Metered connection (TE9)	6	Chemical use WT (EN10)	-	-	6	Acceptabilit y (SO4)	6	Public participation (IN7)
7	Dwelling density (TE5)	7	Backwash water (EN16)	-	-	7	Safety from hazards	-	-
8	Proximity to WW (TE2)	8	Chemical use in WWT (EN11)	-	-	-	-	-	-
-	-	9	Bio-solids quality (EN14)	-	-	-	-	-	-
-	-	10	Energy use (W) (EN7)	-	-	-	-	-	-
-	-	10	Energy use (WW) (EN8)	-	-	-	-	-	-
-	-	12	Discharged WW quality (EN13)	-	-	-	-	-	-
-	-	13	Ground WQ (EN4)	-	-	-	-	-	-

Note: * Full description of SPIs with their measurement method is provided in Appendix A.2.
R: rank, WW: wastewater, W: water, WQ: water quality, WT: water treatment, WWT: wastewater treatment, Oper.: operating, Avg.: average, BWA: Boil water advisory, Insti.: institutional, consvn.: conservation

The next SPI dwelling density (residential building) represents a measure of compact development. A higher value of the SPI indicates a more dense urban community, and represents an efficient use of water infrastructures. Speir and Stephenson (2002) demonstrated that a larger lot size and higher dispersion of neighbourhoods significantly increases the cost of providing water and wastewater services. For a sustainable neighbourhood, the recommended value is a minimum of 8 dwelling units (DU) per acre for residential buildings (USGBC 2013; EarthCraft 2014). Finally in the neighbourhood location and design criteria, the SPI separation of wastewater and storm water occurs. This indicator is a measure of optimum resource use (Van

Leeuwen et al. 2012) because the degree of treatment required for storm water is usually less than that required for wastewater.

In the water infrastructure and fixtures criteria, water leakage shows the distribution efficiency of finished water. Its value ranges from 7.6% – 14.9%, with an average of 13.3% in Canada (Environment Canada 2011). The SPI water supply reliability can be expressed as the number of main breaks per 100 km length. Its value ranges from 1 to 19.7 main breaks per 100 km with a median of 5.9 main breaks per 100 km pipe length (AECOM 2012). However, the ratio may be higher in small and medium sized communities because of a shorter pipe length. So, care should be taken when comparing with these values.

The SPI metered connection is a measure of the efficient use of resources. The installment of water meters in homes encourages consumers to conserve water because higher water consumption increases their water bills. Water metering can reduce water consumption by 15 to 25% (DoP 2007). For instance, water consumption was reduced by 16% in the District of Peachland, British Columbia in 2007 when water metering began (DoP 2015). The next SPI, treated water storage capacity, measures the capacity of the water system to meet water demand even during treatment failures. This indicator shows the sustainability of the system (NRC 2009; CSA 2010). In Canada, storage capacity ranges from less than 1 to 96 hours with a median of 29 hours (AECOM 2012).

4.4.2 Environmental

In the resource utilization criteria, the SPI water self-sufficiency is a measure of the availability of the required water in a community territory. Water self-sufficiency can be measured in terms of licensed water or annual renewable water available in the community territory. However, they have different meanings. Water self-sufficiency in terms of licensed water represents the legally available water; however, the licensed amount of water may not necessarily be available in the water source. Water self-sufficiency in terms of renewable water represents the natural water availability in the source water. Usually, licensed water is based on the renewable water availability of the water bodies.

Water self-sufficiency, in terms of licensed water in Canadian municipalities, is higher than 130% (i.e., less than 1 to 76% of licensed water withdrawal) (AECOM 2012). The SPI domestic water consumption measures the extent of domestic water use indicating whether an urban community has an over-consumption or under-consumption of water with respect to a benchmark. Canadians have a higher domestic water consumption rate (Renzetti 1999; Ma 2014) with an average of 343 L/capita/day (Environment Canada 2014b), while non-domestic water consumption is 236 L/capita/day in their country (Environment Canada 2011). However, non-domestic water consumption depends on the type and extent of industrial establishments in an urban community. Domestic and non-domestic water may be supplied by groundwater or surface water or both. The SPI groundwater quality is measured in terms of Faecal Coliform, Nitrogen (N), and Phosphorus (P). Another SPI surface water quality is measured in terms of Faecal Coliform, N, P and Biochemical Oxygen Demand (BOD) of the major water body of the urban community or city (Van Leeuwen et al. 2012). Good quality of groundwater and surface water is required for environmental and human health (Van Leeuwen et al. 2012). Since, groundwater and/or surface water are the sources of public water supply, their preservation enhances the water sustainability of an urban community.

The SPI energy use is an important aspect of urban water sustainability (Chang et al. 2014). Energy is required for operating UWSs. Energy use significantly differs based on site-specific conditions, including distance to the water source, depth in case of groundwater, topography, water quality, and the technology used (Tuladhar et al. 2014). For example, in California, US, energy use varies from 0.211 to 8.243 kWh m^{-3} of water supplied and 0.291 to 1.321 kWh m^{-3} of wastewater treated (CEC and NC 2006). The next SPI, chemical use, in terms of major chemicals such as chlorine and coagulants in water and wastewater treatment is measured in order to determine the use of resources for water treatment (Lundin and Morrison 2002; Makropoulos et al. 2008; Popawala and Shah 2011; Water UK 2011; Van Leeuwen et al. 2012; Van Leeuwen and Marques 2013). Clean raw water and less polluted wastewater require less chemicals for their treatment, which enhances urban water sustainability.

In the environmental impacts criteria, discharged wastewater quality measures the impact of wastewater disposal on the environment. Discharge wastewater quality is measured as the number of days (or times) out of compliance for BOD, N, P, and heavy metals (Cd, Pb, Hg and

Cu). This non-compliance should be less than 5% (18 days in year) (World Bank 2008). Next, the SPI biosolids quality, measured in terms of heavy metal content, indicates the impacts of the biosolids disposal on the environment. The biosolids can have restricted or unrestricted use based on their heavy metal content (CCME 2010). A higher heavy metal content than a recommended value (for unrestricted use) results in a restricted use of biosolids, such as soil amendments. Furthermore, backwash water is generated by the cleaning (backwash) of the treatment plant equipment, for example filters. Backwash water may contain toxic chemicals, such as aluminum and manganese due to the use of coagulants in water treatment (Haider et al. 2014b). Because of the chemical content, the backwash water should be treated and the discharge of untreated backwash water to natural water bodies should be monitored.

In the resource recovery criteria, the SPI water reuse was selected. The wastewater of SMUWSs is a resource and can be recycled to obtain water of usable quality. The water recycling enhances urban water sustainability (Chang et al. 2015) because water reuse saves water that would otherwise be lost from the system. For example, in Canada, water reuse is very low with a value of approximately 3% in British Columbia - one of the water reusing provinces (CCME 2002). A very high use of recycled water, up to 88%, was achieved in Melbourne, Australia, where recycled water of different classes is produced and used for various activities such as toilet flushing, industrial wash down, and municipal watering (Western Water 2013). Similarly, 100% wastewater has been recycled and reused in Cyprus (EU 2015; 2016). Water reuse should be increased in an urban community for its water sustainability.

4.4.3 Economic

The top level indicators identified are operating cost coverage ratio for water service and that for wastewater service. This result is not surprising as any water utility is financially sustainable when water and wastewater revenues equal or exceed expenses for, at least, the operational and maintenance costs (World Bank 2003). In this case, the ratio is 1.0 or higher. For an economically sustainable SMUWS, a lower operating cost of water and wastewater is desired. A dense urban community can reduce water infrastructure requirements and operating costs thereby increasing economic sustainability.

At the second level, two SPIs average water fee rate and non-revenue water are present. The average water fee rate is a measure of affordability of consumers to pay for water and wastewater services delivered. This rate varies from \$152 to \$489 with a median of \$ 366 per 250 m^3 water in Canada (AECOM 2012). According to Renzetti (1999), user fees meet only 37% of operational and 66% of capital expenditures. This statistics indicates lower water user fees in Canada. The next indicator, non-revenue water (NRW), measures the water supplied with no revenues collected. NRW includes real losses, apparent losses (customer meter inaccuracies and unauthorized consumption), and unbilled authorized consumption (Kanakoudis and Tsitsifli 2010). NRW is calculated in terms of "liter/connection/day" based on the view that a major water loss occurs at service connections (Hamilton et al. 2006). NRW can be considered as a useful financial indicator (Kanakoudis and Tsitsifli 2010). For an economically sustainable SMUWS, an affordable water fee rate and a lower value of NRW are desired (Zhou et al. 2013).

4.4.4 Social

In the service provision criteria, the SPIs access to water and wastewater services are measured by the percentage of population served by public water supply and wastewater service (with a secondary level or higher treatment) respectively. Water service is required for the development of an individual human being (Van Leeuwen et al. 2012). Wastewater service is required for the safe disposal of wastewater in order to protect human and environmental health (Van Leeuwen et al. 2012). Therefore, access to these services is crucial for assessing the sustainability of SMUWSs. In Canada, the access to public water service varies from 50% to 98% with an average of 88.9% and the access to wastewater service varies from 37 to 76% with an average of 68% (Environment Canada 2011).

The SPI water restrictions, measured as the number of days per year, are the days that are restricted for water use for specific purposes, such as lawn irrigation. These regulatory measures are imposed by local water utilities to conserve water especially during peak demand. However, this measure restricts consumers from water use. In Canada, this measure is frequently practised. Water restrictions range from zero to 365 days per year with a median of 121 days per year

(AECOM 2012). Furthermore, public acceptability is an important aspect of social sustainability. This indicator can be expressed in terms of the number of complaints on water and wastewater services per 1000 population. A low number or no complaint indicates public acceptability to water and wastewater services. In Canada, the complaints on water and wastewater services range from 0.02 to 24.84 complaints per 1000 people with a median of 4.49 complaints per 1000 people per year (AECOM 2012).

In the public health criteria, the SPI drinking water quality was found to be at the top level along with the access to water and wastewater service indicators. The reason behind this may be – clean and safe water supply is one of the prime objectives of a sustainable UWS (Hellstro 2000; Engel-yan et al. 2005). This SPI is measured in terms of non-compliance of turbidity, total coliforms, residual chlorine, and nitrates of drinking water. The next SPI, boil water advisories (BWA), is a measure of public health risk due to the contamination of water supply. BWA is calculated as the number of household (HH)-days per year that boil water advisories are in effect as a % of total HH-days. This is an important indicator. For example, a majority of provinces have gone through many BWA and British Columbia has gone through the highest BWA in Canada (Water Chronicles 2014). The median BWA in Canada is 0.89 days per year with a range of zero to 12 days per year (measured as the number of BWA days *x* capita affected/total population served). Furthermore, the safety indicator qualitatively assesses plans, measures, and their implementation status in order to protect citizens against flooding and drought. This indicator is also considered by City Blueprints (Van Leeuwen et al. 2012).

4.4.5 Institutional

The top level SPI is urban water policies. The urban water policies indicator qualitatively assesses a local government's policies, action plans, and commitments for an integrated urban water management. A similar indicator has also been used by City Blueprints (Van Leeuwen et al. 2012). Moreover, water demand reduction in an existing community is an effective SPI to measure a community's progress toward the sustainability practice. The reduced demand is primarily achieved by institutional initiatives. Particularly, in high water consuming communities such as in Canada, the reduction of residential water consumption is an important step for achieving water sustainability. The average water demand reduction is 18 litre/capita/day, i.e., 5.5% per year from 2006 to 2009 (Environment Canada 2011).

At the next level, the SPI institutional capacity, in terms of fulltime equivalent (FTE) personnel, measures the strength of a municipality or water purveyor. This indicator is also used by IWA (2006), Government of Canada (2007), CSA (2010), and Sydney Water (2013). Similarly, the SPI personnel training measures the extent of organizational development and is also used by IWA (2006), AWWA (2008), and World Bank (2011). Furthermore, the SPI public participation measures a local community involvement for achieving healthy community activities (Brown and Farrelly 2009; Siemens AG 2009; Van Leeuwen et al. 2012). These SPIs are also used for assessing the sustainability of UWSs by Van Leeuwen et al. (2012) in City Blueprints. In addition, the SPI conservation program is a measure of conservation efforts of an institution and can be measured by annual expenses for running the program. A reduction in domestic water consumption can be achieved by effective conservation programs. In Canada, the expenses of conservation programs vary from less than $1 to $5.64 per person per year with a median value of $0.47 per person per year (AECOM 2012). However, the type of conservation programs required may differ from community to community.

4.5 Summary

UWSs are challenged by the sustainability perspective. Certain limitations of the sustainability of centralized UWSs and decentralized household level wastewater treatments can be overcome by managing UWSs at an intermediate scale, referred to as small to medium sized UWSs (SMUWSs). SMUWSs are different from large UWSs, mainly in terms of smaller infrastructure, data limitation, smaller service area, and institutional limitations. Moreover, sustainability assessment systems to evaluate the sustainability of an entire UWS are very limited and confined only to large UWSs. This research addressed the gap and has developed a set of 38 applied sustainability performance indicators (SPIs) by using fuzzy-ELECTRE I outranking method to assess the sustainability of SMUWSs. The developed set of SPIs can be applied to existing and new SMUWSs and also provides a flexibility to include additional SPIs in the future based on the same selection criteria. The SPIs related to water, energy, carbon emissions, cost, and health risk has been used for developing water-energy-carbon (WEC) model in the next chapter.

Chapter 5 System Dynamics Modelling of Water-Energy-Carbon (WEC) Nexus

A version of this chapter has been published in the *ASCE Journal of Water Resources Planning and Management* entitled "Water-Energy-Carbon nexus modelling for an urban water system: A system dynamics approach" (Chhipi-Shrestha et al., 2017b).

5.1 Background

A comprehensive framework and decision support system to quantify the WEC nexus and its dynamic behavior at the neighbourhoods or community level is required (Nair et al., 2014; Rothausen & Conway, 2011; Arora et al., 2013; Kenway, 2013). The extensive review of the water-energy nexus studies by Kenway et al. (2011) also concluded the lack of a unifying framework and consistent methodology for analyzing the WEC nexus. The WEC model comprising interacting problems can be developed by using system dynamics (Nasiri et al., 2013; Nair et al., 2014).

System dynamics has been used in only a few urban water studies (Zarghami and Akbariyeh 2012). The researchers, such as Zarghami and Akbariyeh (2012), Zhang et al., (2008), Zhang et al. (2009), Karamouz et al. (2012), Nasiri et al. (2013), and Zhang et al. (2009) applied system dynamics to identify effective and reliable water resources plan, policy or estimate water resource carrying capacity. Qi and Chang (2011), Wang (2014), Tong and Dong (2008), and Nawarathna et al. (2009) separately studied the dynamic effects, such as of macro-economy, water price, socio-economic-environmental system, or changing land use and climate on water demand and supply. All these system dynamics-based studies lack energy use and carbon emissions.

At the country level, Tidwell et al. (2012) estimated potential impact of water availability on future expansion of thermoelectric power generation, whereas Wang (2013) studied the implications of biofuel development on water and energy. The former lacks carbon emissions and the latter is a preliminary model. Specifically, Willuweit and O'Sullivan (2013) modelled the effects of urban development and climate change on urban water cycle. This critical review shows system dynamics has been applied to different aspects of urban water from county to country level but not applied to the WEC nexus for neighbourhoods, a community, i.e., SMUWS. Also, urban water processes perform differently in various geographic regions with respect to energy and carbon emissions. This requires a holistic and generic model to capture the variability and dynamics of UWSs (Nair et al., 2014). This chapter aimed to develop a dynamic WEC nexus model that can assist municipalities, urban developers, and policy makers in making informed decision for reducing water consumption, energy use, and carbon emissions in UWSs.

5.2 Methodology

The operational phase of a SMUWS has been identified as the most energy intensive phase from the life cycle perspective (Friedrich, 2002; Nair et al., 2014). This study has focused only on the operational phase of a SMUWS. The WEC model was based on system dynamics using STELLA® 10.1.3 (ISEE Systems 2016; Karamouz et al. 2012; Qi and Chang 2011). The system dynamics model (SDM) includes water module, energy module, and carbon module as elaborated in the following sections.

5.2.1 Water module

Water module is comprised of water consumer growth sub-models and water and wastewater sub-models as follows.

5.2.1.1 Water consumer growth sub-models

The water consumers: population; commercial, institutional and industrial (CII) sector; and agriculture were included in the WEC model. The Standard Industrial Classification code numbers 2000 through 3999 (Gleick et al. 2003; US EPA 2009b) was followed to define CII sector. The major commercial sectors included in this study are offices, restaurants, supermarkets and retail, and hotels; major institutions: government institutions, hospitals, and schools as identified by AWWA (2000), and industries in average. The dynamics of water consumers were analyzed by using the growth equation (Nasiri et al. 2013) (Equation 5.1 in Table 5.1).

5.2.1.2 Water and wastewater sub-models

The water and wastewater sub-models include municipal water use model, wastewater generation model, and water footprint (WF) model for the operational phase of the SMUWS.

a) Municipal water model

The municipal water use model represents the flow and use of drinking water in neighbourhoods. The municipal water flow occurs through urban water stages: abstraction and conveyance, treatment, distribution, and use. The municipal water use dynamics was modelled in Equation 5.2 in Table 5.1. The equation includes the water consumed by different urban water components over time: residential, CII, public parks, golf courses, and agriculture. Each of these urban water components was modelled by including all their unit water use activities. As an example, residential water is modelled in Equations 5.3, 5.4, and 5.5 in Table 5.1, and the similar equations were used for all other urban water components of Equation 5.2.

b) Wastewater generation and water footprint models

The wastewater generation model includes wastewater (WW) collection from residential and CII sector as well as infiltration and inflow to sewer network as shown in Equation 5.6. The modelled wastewater includes the indoor water consumed by the respective urban water components except the leakage. The water footprint model is represented by Equation 5.7.

5.2.2 Energy module

The energy module includes the operational energy of a SMUWS and embodied energy of major chemicals, such as chlorine, poly aluminum chloride, and polymers. The dynamics of energy use was modelled in Equations 5.8 and 5.9. The hot water energy for residential sector was modelled by using Equation 5.10 (Aguilar et al. 2005) and the similar equation was used for modelling indoor hot water energy of CII sector.

5.2.3 Carbon module

The carbon module represents carbon emissions of the operational phase of a SMUWS. The module includes direct carbon emissions in terms of CO_2e from energy use in a SMUWS, wastewater processes, and carbon footprint of major chemicals. The dynamics of carbon emissions were modelled in Equations 5.11 and 5.12 in Table 5.1.

Table 5.1 Equations of the WEC model

WEC nexus	Aggregation Equations	Eqn #
	Water module (Water consumer growth sub-model)	
W,E,C	$N_t = N_0 e^{r\,t} * (P_{resi})_t$	**[5.1]**
	Water module (Water and wastewater sub-models)	
W	$(W_{direct})_t = (W_{resident})_t + (W_{comm})_t + (W_{insti})_t + (W_{industry})_t + (W_{parks})_t + (W_{golf})_t + (W_{agri})_t + (W_{distrib\ loss})_t$	**[5.2]**
W	$(W_{resident})_t = (W_{in})_t + (I_{out})_t$	**[5.3]**
W	$(W_{in})_t = (TW)_t + (SW)_t + (FW)_t + (LW)_t + (DW)_t +$ Indoor water leakage $= [(f_T*\eta_{TW})_t + (f_S*d_S*\eta_{SW}*\eta_{SU})_t + (f_F*\eta_{FW}*\eta_{FU})_t + (f_L*\eta_{LW})_t + (f_D*\eta_{DW})_t]*(1+In_{Le})*Ic_t*N_t$	**[5.4]**
W	$(I_{out})_t = \frac{N_t}{DO} * \left(U_S * L_S * C_S + U_D * L_D * C_D + \frac{U_R*C_R*RU}{L_R*FAR_R} + \frac{U_{As}*C_{As}*RU}{L_{As}*FAR_{AS}} + \frac{U_{Al}*CAl*RU}{L_{Al}*FAR_{Al}}\right)_t * Ir_t * Irc_t * \mathrm{Ip}$	**[5.5]**
W	$(WW)_t =$ (Residential WW)$_t$ + (Commercial WW)$_t$ + (Institutional WW)$_t$ + (Industrial WW)$_t$ + (Infiltration and inflow to sewer network)$_t$	**[5.6]**
W-E	(Total WF)$_t$ = $(W_{direct})_t$ + (WF of direct energy use)$_t$ + (WF of major chemicals)$_t$	**[5.7]**
	Energy module	
E-W	$(E_{direct})_t = (E_{convey})_t + (E_{WT})_t + (E_{Distri})_t + (E_{resi\ HW})_t + (E_{CII\ HW})_t + (E_{WW\ transport})_t + (E_{WWT})_t + (E_{Biosolids})_t$	**[5.8]**
E-W	$(E_{total})_t = (E_{direct})_t + (EE_{chemicals})_t$	**[5.9]**
E-W	$(E_{resi\ HW})_t = (SE)_t + (FE)_t + (LE)_t + (DE)_t + (SL)_t$ $= [(f_S*d_S*\eta_{SW}*\eta_{SU}*\eta_{SE}*H_S)_t + (f_F*\eta_{FW}*\eta_{FU}*\eta_{FE}*H_F)_t + (f_L*\eta_{LE}*WHR_L*H_L)_t + (f_D*\eta_{DE}*WHR_D*H_D)_t$ $+ (SL_R)_t*(f_S*d_S*\eta_{SW}*\eta_{SU}*H_S + f_F*\eta_{FW}*\eta_{FU}*H_F + f_L*\eta_{LW}*H_L + f_D*\eta_{DW}*H_D)_t]*Ec_t*N_t$	**[5.10]**
	Carbon module	
C-E &W	$(C_{direct})_t = (C_{convey})_t + (C_{WT})_t + (C_{Distrib})_t + (C_{resiHW})_t + (C_{CIIHW})_t + (C_{WWtransport})_t + (C_{WWT})_t + (C_{biosolids})_t +$ $(C_{WWprocesses})_t$	**[5.11]**
C- E &W	$(C_{total})_t = (C_{direct})_t + (CF_{chemicals})_t$	**[5.12]**

Note:
Water module
Water consumer growth sub-model
Where for population, N_t is population in a month, N_0 is base population, r is population growth rate (monthly), t is time duration in months. Similarly for CII sector, N separately refers to the number of hotel rooms, hospital beds, and school students and N refers to floor area for other CII sectors (restaurants, offices, supermarkets, and industries); for irrigation water, N separately

refers to golf course area, neighbourhood and community park land, and agricultural land if present; r refers to their growth rate; P_{resi} refers to the proportion of population residing in that time and is 1 when seasonal migration is not considered.

Water and wastewater sub-models

a) In Equation 5.2, W_{direct} is direct water use (L), $W_{resident}$ is residential water use (L), W_{comm} is commercial water use (Offices including governmental offices; restaurants and supermarkets) (L), W_{insti} is institutional water use (Hotels, hospitals, and schools) (L), $W_{industry}$ is industrial water use (L), W_{parks} is parks water use (L), W_{golf} is golf courses water use (L), W_{agri} is agricultural water use (L), $W_{distrib\ loss}$ is water loss in distribution (L), and "t" refers to a month.
b) In Equations 5.3 and 5.4, W_{in} is indoor water use (L); I_{out} is outdoor irrigation water (L); f is frequency of use (per capita/month), η is efficiency (L/min for shower and L/use of others), d is duration of use (min/shower) and their subscripts T is toilet, TW is toilet water, S is shower, SW is shower water, SU is shower use, F is faucet, FW is faucet water, FU is faucet use, L is laundry, LW is laundry water, D is dishwasher, and DW is dishwasher water; In_{Le} is % of indoor water leakage; Ic_t is indoor water conservation rate (monthly) and for exponential change of this rate, $Ic_t = Ic_0e^{-rt}$ with r as the rate of change of indoor water conservation rate; N_t is population at time t (month).
c) In Equation 5.5, DO is dwelling occupancy (persons/residential unit); U is % of dwelling units, L is average lot size (ha), C is average % of lawn coverage, i.e., (1 - Average % of lot coverage), FAR is floor area ratio, and their subscripts S is single detached home, D is duplex, R is row house, "As" is small apartments, and "Al" is large apartments; RU is residential unit size (ha); Ir is garden irrigation rate; Irc is irrigation conservation rate (monthly) and for exponential change of this rate, $Irc_t = (Irc)_0e^{-rt}$ with r as the rate of change of irrigation conservation rate; I_p is irrigated garden proportion. The value of r could be different for Ic_t and Irc_t; however, they are considered equal in this study due to the lack of data.
d) In Equation 5.7, WF of direct energy use is the sum of the products of WF of particular energy source (L/kWh) and total amount of that energy (kWh) for all energy uses, and WF of major chemicals is the sum of the products of WF of a particular chemical (L/kg), rate of chemical use in water or wastewater (kg/L), and total amount of water or wastewater (L) for all chemical types in time "t"

Energy module

e) In Equations 5.8 and 5.9, E_{direct} is direct energy use (kWh), E_{convey} is raw water abstraction and conveyance energy (kWh), E_{WT} is water treatment energy (kWh), E_{Distri} is water distribution energy (kWh), $E_{resi\ HW}$ and $E_{CII\ HW}$ are the energy for indoor hot water use (kWh) in residential and CII sector respectively, $E_{WW\ transport}$ is wastewater transport energy (kWh), E_{WWT} is wastewater treatment energy (kWh), $E_{Biosolids}$ is biosolids transportation energy (kWh), E_{total} is total energy use (kWh), $EE_{chemicals}$ is embodied energy of major chemicals (kWh), and "t" refers to a month.
f) In Equations 5.8 and 5.9, $(E_{convey})_t$, $(E_{WT})_t$, $(E_{Distri})_t$, $(E_{resi\ HW})_t$, $(E_{CII\ HW})_t$, $(E_{WW\ transport})_t$, and $(E_{WWT})_t$ are individually estimated as the product of energy intensity of each process (kWh/L) and total amount of water or wastewater flow in the process (L) in time "t", and $(E_{Biosolids})_t$ as the product of energy intesnsity of biosolids transportation (kWh/kg), rate of biosolids generation per unit of wastewater (kg/L) and total volume of wastewater (L) in time "t" and $(EE_{chemicals})_t$ as the product of unit embodied energy of a chemical (kWh/kg) estimated from LCA, rate of chemical use in water or wastewater (kg/L) and total amount of water or wastewater (L) for all chemical types in time "t"
g) In Equations 5.10, f, η, d, T, TW, S, SW, SU, F, FW, FU, L, LW, D, DW, N_t , and t have same meaning as of Equation 5.4; SE is shower energy; FE is faucet energy; LE is laundry energy; DE is dishwasher energy; H is hot water ratio; WHR is water heating ratio; SL is standby energy loss; R is rate; Ec_t is hot water-energy conservation rate (per month) and for exponential change of this rate, $Ec_t = Ec_0e^{-rt}$ with r as the rate of change of hot water-energy conservation rate (per month).

Carbon module

h) In Equations 5.11 and 5.12, C_{direct} is carbon emissions (CE) from direct energy use (kg CO_2e), C_{convey} is CE from raw water abstraction and conveyance energy (kg CO_2e), C_{WT} is CE from water treatment energy (kg CO_2e), $C_{Distrib}$ is CE from water distribution energy (kg CO_2e), $C_{resi\ HW}$ and $C_{CII\ HW}$ are CE (kg CO_2e) from indoor hot water use respectively in residential and CII sector, $C_{WWtransport}$ is CE (kg CO_2e) from wastewater transport energy, C_{WWT} is CE (kg CO_2e) from wastewater treatment energy, $C_{biosolids}$ is CE (kg CO_2e) from biosolids transportation energy, $C_{WWprocesses}$ is CE (kg CO_2e) from wastewater processes (wastewater treatment), C_{total} is CE (kg CO_2e) from total energy use, $CF_{chemicals}$ is carbon footprint (kg CO_2e) of major chemicals, and "t" refers to a month
i) In Equation 5.11, $(C_{convey})_t$, $(C_{WT})_t$, $(C_{Distrib})_t$, $(C_{resiHW})_t$, $(C_{CIIHW})_t$, $(C_{WWtransport})_t$, $(C_{WWT})_t$, and $(C_{biosolids})_t$ are individually estimated as the product of carbon emission factor of the energy source (kg CO_2e/kWh) and total amount of energy consumption (kWh) in time "t", $(C_{WWprocesses})_t$ as the product of organics (BOD) generation rate (kg BOD/person), carbon emission factor of organics (kg CO_2e/kg BOD) and total population (persons) in time "t" (IPCC 2006).
j) In Equation 5.12, $(CF_{chemicals})_t$ is the product of carbon footprint of a particular chemical (kg CO_2e/kg) estimated from LCA, rate of chemical use in water or wastewater (kg/L), and total amount of water or wastewater (L) for all chemical types in time "t"

A causal loop diagram (CLD) was developed before developing a complete SDM. A CLD is a graphical representation that enables the visualisation of causal relationships between variables in a causal model. The causal diagram shows how each factor affects others and in turn is affected by other factors. The CLD is given in Figure 5.1, in which “+” indicates a positive relationship and “-” indicates a negative relationship in the UWS. As shown in Figure 5.1, community people use indoor and outdoor water (residential water). Similary, commercial, institutional, and industrial (CII) sector, agriculture, golf courses, and parks consume water (direct water use). The water for residential sector, CII sector, agriculture, golf course irrigation, and park irrigation combinely give water use in a community. Water abstraction supplies water for use and it generates wastewater after use. Raw water collection, water distribution, wastewater transport, and wastewater treatment require energy (water and wastewater conveyance and treatment energy). Indoor hot water use (residential and CII sectors) consumes energy (hot water energy). Energy use has water footprint. Also, chemicals used in water and wastewater treatment have water footprint, embodied energy, and carbon footprint.

The sum of direct water use, water footprint of water and wastewater conveyance and treatment energy and indoor hot water energy use, and water footprint of chemicals gives total water footprint. Similarly, the sum of water and wastewater conveyance and treatment energy, indoor hot water energy, and embodied energy of chemiclas gives total energy for an UWS. The sum of GHG emissions from conveyance and treatment energy use, GHG emissions from indoor hot water energy use, and carbon footprint of chemiclas provides total GHG emissions from an UWS. The complete SDM in the form of stock and flow diagrams developed based on the CLD is given in Appendix B.1.

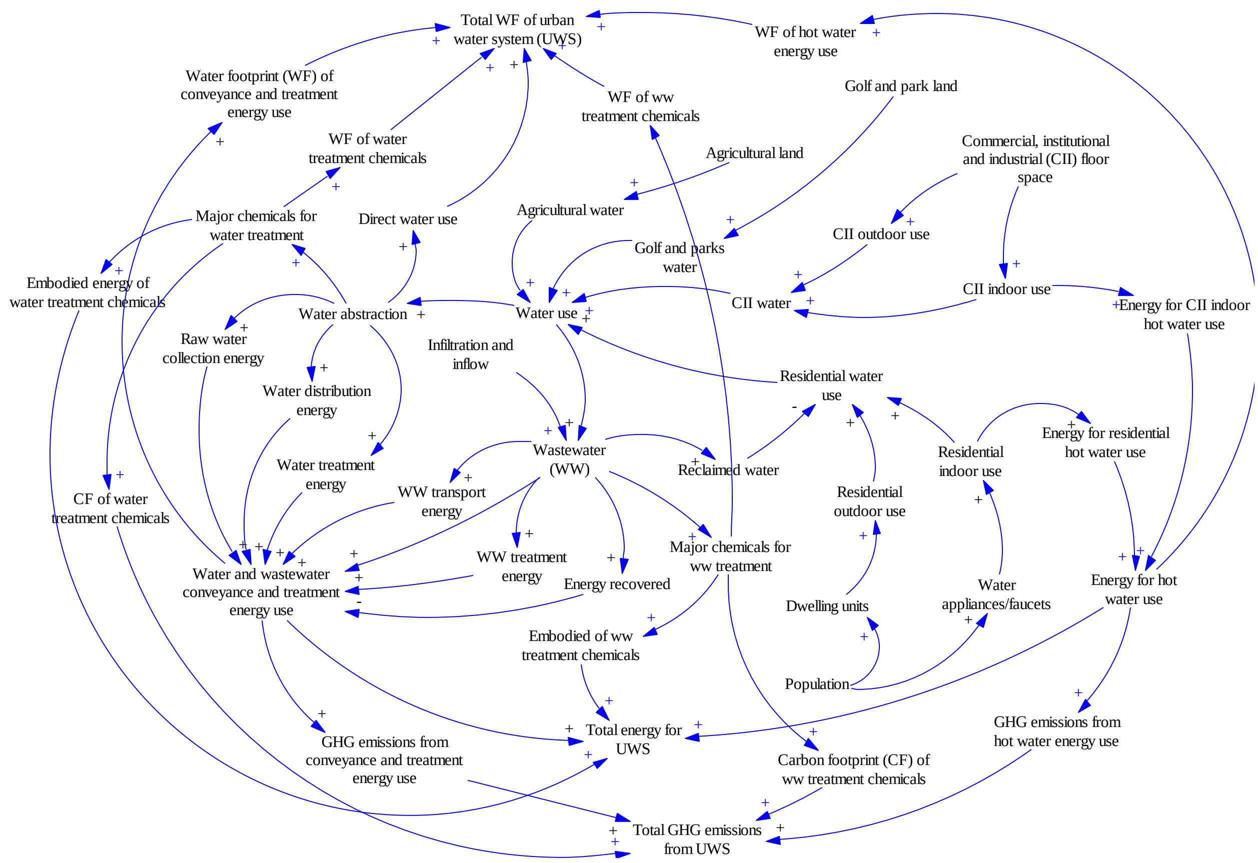


Figure 5.1 Causal loop diagram of the WEC nexus of a SMUWS

5.3 Results and Discussion

A WEC model and its DSS was developed in Stella 10.1.3. The model was calibrated and validated and then applied to the City of Penticton (CoP), Okanagan, British Columbia (BC), Canada as follows.

5.3.1 Model calibration and validation

Calibration refers to the estimation of parameter values, e.g., rate of water treatment energy use. The model was calibrated using the historical data for the years other than the validation period (Wang 2014; Willuweit and O'Sullivan 2013). The calibrated model was validated by using the historical data (Qi and Chang 2011; Willuweit and O'Sullivan 2013). The validation data constituted the monthly data for raw water collection, water treatment, distribution, water consumption, wastewater generation, wastewater treatment, energy use in water and wastewater transport and treatment, major chemical uses in the water and wastewater treatment of the CoP from 2005 to 2014. The water treatment plant capacity was upgraded in 2008 to 2009 (City of Penticton 2015b), whereas the wastewater treatment plant was upgraded during 2009 to 2012 in CoP (City of Penticton 2014a). The energy consumption rates of both treatment plants after these upgrades are more applicable for energy use modelling and forecasting. Therefore, for the validation of energy use in water and wastewater conveyance and treatment, the simulated results from 2013 to 2014 were used. It is noteworthy that the supplied drinking water was not used for agricultural irrigation in the CoP. The WEC model was further validated with direct structural tests, structure-oriented behavior tests, and behavior pattern tests (Barlas 1996).

5.3.2 Data requirements

The required data were collected from various sources. Different periods of data were used for model calibration and validation.

a. Water consumers

The data on base population, growth rate, and dwelling occupancy for Penticton were obtained from the census database (Statistics Canada 2015a). The average lot size of residential houses and CII buildings were estimated from the municipal GIS database using ArcGIS. They were verified with the zoning bylaws. In absence of data, the growth rate of CII sector was assumed equal to population growth rate as the growth of CII sector follows population growth. The baseline data on school students, hospital beds, and hotel rooms were obtained from the respective authentic sources as mentioned in detail in Appendix B.2.

b. Water and wastewater

The rates and water efficiencies of uses of water fixtures and appliances in residential and commercial and institutional (CI) sectors; industrial water use rate; and irrigation rates were obtained from literature as mentioned in detail in Appendix B.2. Moreover, average lot coverages (%) for different residential houses and CII buildings were estimated by using Google Earth and were verified with the zoning bylaws. The rate of change of indoor water conservation rate, monthly average infiltration and inflow rate, and the average decreasing rate of change of monthly infiltration-flow rate specific to CoP were estimated based on the Penticton data and their detail methods are explained in Appendix B.2. Moreover, the water footprint of major chemicals used in water and wastewater treatments were obtained by conducting a life cycle assessment (LCA) by using SimaPro 8.0.5 (Risch et al. 2014).

c. Energy consumption

The rate of change of indoor hot water energy conservation rate; monthly average energy consumption rates separately for raw water collection to wastewater treatment; average dosages of major chemical uses; and average rate of increase of monthly energy consumption rate for water and wastewater treatment in Penticton were estimated and explained in detail in Appendix B.2. The embodied energy of the major chemicals was obtained from the same LCA conducted for estimating the water footprint of these chemicals in the earlier section (Risch et al. 2014). As far as data was available, the data of Penticton was used for the estimation of the parameters. The average Canadian values were used for other parameters, for instance, hot water ratios for different water uses were obtained from the national Residential End Use Model (REUM) (Aguilar et al. 2005). The energy efficiency of water fixtures and appliances was obtained from the REUM model (Aguilar et al. 2005) for conventional fixtures and from ENERGY STAR (2014a; b) for efficient fixtures in Canada.

d. Carbon emissions

The carbon emissions from energy use were estimated using the carbon emission factors of the respective energy sources (Ministry of Environment 2013). The carbon emissions from wastewater processes were estimated based on the IPCC methodology (IPCC 2006). Moreover, the carbon footprint of major chemicals was obtained from the same LCA conducted for estimating the water footprint of these chemicals in the earlier section (Risch et al. 2014).

The data for all parameters are categorically shown as: regional containing regional and site-specific data (R); national (N), and global (G) in Appendix B.2. In the lack of full data set, the model can be run only by changing the regional data for other cities or communities.

5.3.3 WEC model for Penticton

In the developed WEC DSS, the major data can be input from the interface; however, if a community has detailed data in addition to those mentioned in the interface, all the data can be imported from a spreadsheet. The results can be exported to a spreadsheet and the major ones will be displayed on the interface. The interface of the WEC DSS is shown in Figure 5.2. The slider, button, and dropdown list are used for data input. After model simulation, the major outputs are graphically and numerically displayed.

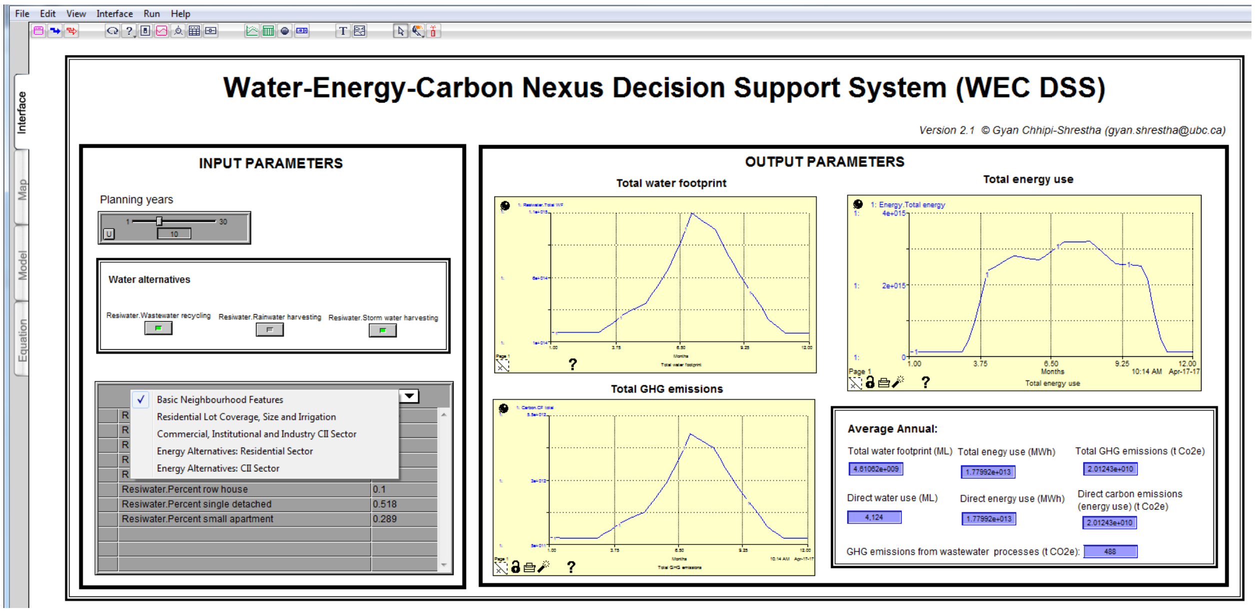


Figure 5.2 Screenshot of the WEC DSS interface

The developed WEC model was applied to the SMUWS of the City of Penticton. The city, with an area of 42.1 sq. km, had a population of 32,877 in 2011 with a growth rate of 0.6% per year (Statistics Canada 2015a). The CoP supplies drinking water through 197 km of water mains having three pump stations, two booster stations, and a water treatment plant (City of Penticton 2015b). The generated wastewater is collected by gravity system and then pump to the wastewater treatment plant (Biological Nutrient Removal) by using 10 lift stations (City of Penticton 2014a).

The WEC model was simulated for Penticton from 2005 to 2014. Based on the data availability, the developed model was validated by using the historical data of monthly water consumption and wastewater collection from 2005-2014, energy use for water and wastewater conveyance and treatment from 2013 and major chemical use for the treatments from 2010 to 2014.

5.3.3.1 Water use and wastewater generation

The simulated monthly water consumption from 2005-2014 has a coefficient of determination (r^2) of 0.89 and is compared with its actual data in Appendix B.3. For the 10-year period, the WEC model resulted in an average water consumption of 622 L/capita/day, which is 1 % higher than the actual value. In particular, the WEC model estimated the indoor water consumption rate to be 220 L/capita/day for 2005 to 2006, which is comparable with the metered data of 222 L/cap/day for indoor water in 2006 in Okanagan, including Penticton and Kelowna (Maurer 2010). The WEC model's simulation resulted in an average of 48.5% of residential water for outdoor irrigation for 2005-2006, which is comparable with the value of ~ 50% for Penticton given by previous studies (Maurer 2010; Neale 2005). From 2008-2014, the r^2 of the wastewater sub-model is 0.85 and the predicted average wastewater collection was 370 L/cap/day, a value 1.2% higher than the actual value.

5.3.3.2 Energy use

For the predicted energy use in raw water abstraction and conveyance, water treatment, and wastewater treatment, the values of r^2 were 0.84, 0.85, and 0.76 respectively. The differences in mean values of energy use by utilities in various urban water stages for actual and modelled data are insignificant ranging from -1.9% to -0.3%. The energy consumed by hot water use (excluding energy for water use in space heating and mechanical work in laundry and dish washing machines) is also an energy use component of a SMUWS. The baseline energy consumption estimated by the WEC model at the start of 2005 is 1906 kWh/capita/year, which is comparable with 1913 kWh/capita/year (based on the national dwelling size of 2.55) reported by the REUM model (Aguilar et al. 2005). From 2005 to 2011, the average hot water energy use at the national level was 2437 kWh/capita/year (Natural Resources Canada 2014), whereas the WEC model estimated 1814 kWh/capita/year for CoP. The value estimated by the WEC model was about 25% lower than the national value because the WEC model was based on the REUM model and the REUM model's estimation itself was 25% lower than the national value of 2,546 kWh/capita/year for 2005 (Natural Resources Canada 2014).

The WEC model estimated an average energy use of 26 kWh/m^2/year for indoor hot water use in the CI sector from 2005 to 2011. For the same period, the survey-based national historical database reported an average of 38 kWh/m^2/year (Natural Resources Canada 2014). The value estimated by the WEC model is 30% lower than the national average because the WEC model considered the same pattern of indoor hot water use in the CII sector as of the residential sector and the residential sector estimates a 25% lower value than the national database. However, in spite of different estimation methods of the WEC model (REUM model-based) and the national database (survey-based), the order of magnitude is similar, indicating that the results of the WEC model are reasonable.

5.3.3.3 Carbon emissions

The WEC model resulted in an average of 244 kgCO_2e/capita/year for hot water energy use in the residential sector in the CoP from 2005-2011, whereas the historical data reported 447 kgCO_2e/capita/year in Canada during the same period (Natural Resources Canada 2014). The WEC model estimated 45% lower than the national database. Similarly, the WEC model estimated an average of 3.6 kgCO_2e/m^2/year in the CI sector in the CoP from 2005-2011, whereas the national database reported 7 kgCO_2e/m^2/year in Canada during the same period. The estimated value is 51% lower than the national level value. The lower carbon emissions by hot water use in the residential and CI sector in the CoP has two major reasons. First, the energy use estimation based on the REUM model itself gives a 25% lower value than the national value. Secondly, the grid electricity in the CoP is provided by FortisBC, which has more than 95% of electricity generated from hydropower and renewable energy, such as wood waste (BC Hydro 2015; Ministry of Environment 2013), whereas the national grid electricity contains only 58% hydroelectricity with 28% thermal power-based electricity that has a higher carbon footprint

(Canadian Electricity Association 2006). However, some variations may also be resulted by a difference in theoretical estimation and actual data of energy use.

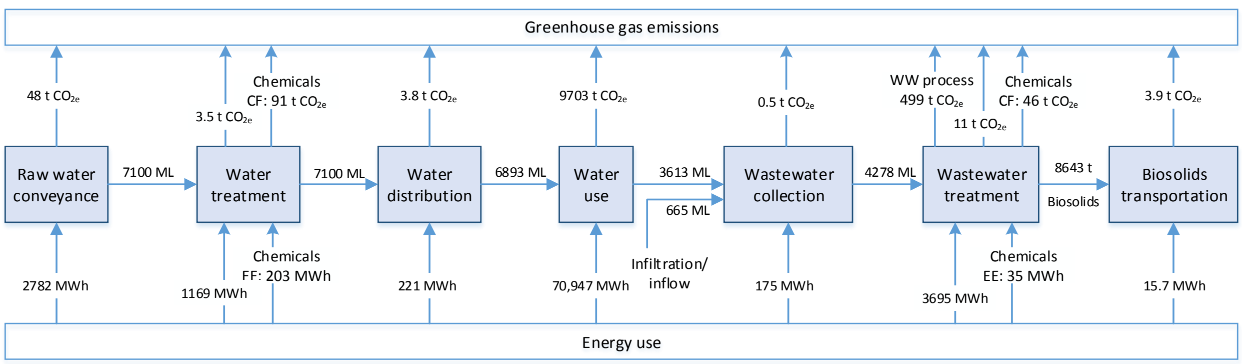
GHGs, i.e., methane and nitrous oxide emissions from wastewater treatment processes were estimated based on the IPCC methodology for a centralized aerobic treatment system (IPCC 2006), which is valid. For the validation of the carbon footprint (also for the water footprint and embodied energy) of the major chemicals, the simulated results for the amount of chemicals consumed are compared with their actual values. The r^2 value is 0.83 for chlorine and that for polymer is 0.88. Moreover, the mean difference in the actual and simulated values in all major chemicals are insignificant ranging from -1.6% to 0.2%. In addition, the difference in average monthly biosolids generation between the actual and modelled data is -1.6% from 2013-2014, indicating a valid estimation. Since, GHG emissions are globally estimated indirectly from energy consumption (Ministry of Environment 2013), the validated energy use, chemicals use, biosolids transportation, and valid emission factors result in a valid carbon module.

Overall, from 2005-2011 the WEC model estimated a reduction of carbon emissions by 1.3%/capita/year for the residential sector, whereas Natural Resources Canada (2014) reported a value of 1.8%/capita/year at the national level. Similarly, the WEC model estimated a reduction of 1.3%/m^2/year for CII sector, whereas Natural Resources Canada (2014) reported a national reduction of 1.4%/m^2/year in the same period. A slight variation in these trends was mostly due to the fact that the WEC model considered the static proportion of energy sources used for water heating as the model is primarily developed for the planning of water in new neighbourhood developments. Also, location-specific features can result in different emission trends in the CoP compared to the national trend; for example, the emission factor of grid electricity differs regionally.

5.3.3.4 Quantitative WEC nexus

All the modules of the WEC model have a complete dataset for the years 2013 and 2014. Therefore, the WEC model provides a completely validated result for all interconnected entities in that period. The annual direct water use, total energy use, and total carbon emissions in absolute values in various stages of the operational phase of the Penticton UWS are given in

Figure 5.3. The average annual water footprint of the UWS was 7,625 ML with 92.7% direct water use, 7.2% water footprint of energy use, and ~ 0.03% water footprint of major chemicals used in treatments. The UWS consumed an annual total energy of 83,625 MWh with 89.5% by indoor hot water use (residential 66% and CII sector 23.5%). In the total energy, embodied energy of chemicals is insignificant (0.3%). A similar result of approximately 90% of operational energy consumption by hot water was also obtained by other studies (Graaff and Klaversma 2012; Reffold et al. 2008).



Note: EE: Embodied energy, CF: carbon footprint

Figure 5.3 Annual direct water use, energy use and GHG emissions in various stages of Penticton UWS

The average annual carbon emissions for the operational phase of the Penticton UWS was 11,047 tCO_2e with the highest share of 93.5% from the residential (69.2%) and CII indoor (24.3%) hot water use. Surprisingly, the proportion of carbon emissions by indoor hot water use was 99.3% of carbon emissions by direct energy use. The very high proportion of carbon emissions by indoor hot water use is due to the highest energy use and primary use of natural gas for water heating (67%), which has high carbon footprint. Therefore, the energy use and carbon emissions of the UWS can be reduced significantly by using energy efficient hot water systems, behavioural change for reduced hot water use, and clean energy for indoor water heating. A saving of approximately 10% in hot water energy can provide energy for all other operational energy demand in the UWS.

The WEC nexus of the Penticton UWS for 2013-2014 is quantitatively shown in Figure 5.4. The overall correlations among water, energy, and carbon in an UWS are nonlinear. Spearman's rank correlation coefficient (ρ) was used to estimate their interrelationship. The Spearman's ρ between water and energy, water and carbon, and energy and carbon were 0.94 (p= 0.000), 0.89 (p = 0.000), and 0.83 (p = 0.000), respectively (Figure 5.4), indicating highly significant interconnections.

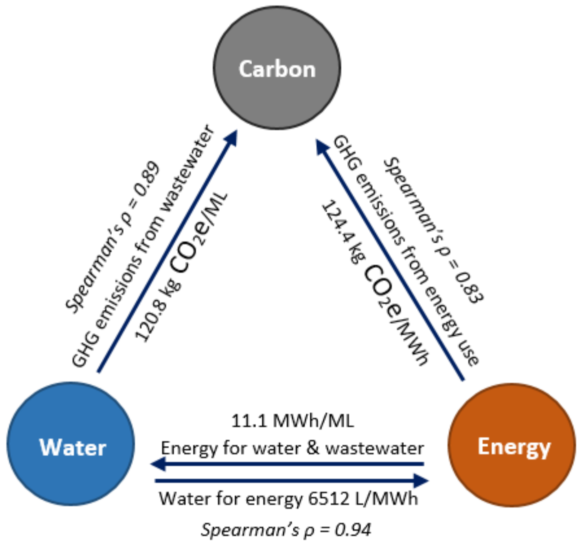


Figure 5.4 WEC nexus of Penticton UWS

5.3.3.5 Sensitivity analysis

Sensitivity analysis was performed to investigate the sensitivity of input parameters to the variations in the final outputs: total water footprint, total energy use and total carbon emissions. Since the WEC model has a large number of input parameters (more than 200) for sensitivity analysis, the input parameters were first screened and 102 parameters were selected based on the relatively important parameters identified by Venkatesh et al. (2014) and Kenway et al. (2008). A most commonly used sensitivity analysis method was used in which an approximate relative contribution of each parameter to the variance of the final outputs was estimated by squaring the rank correlation coefficients between input parameters and final output and then normalized to 100% (Hammonds et al. 1994; Sadiq et al. 2004b). The parameters with the highest relative contributions are considered to be the most sensitive input parameters, which would contribute to reduce the largest amount of overall uncertainty in the results (Hammonds et al. 1994).

The WEC model was simulated for the complete validation period 2013-2014 by using Monte Carlo simulations of 10,000 runs in Stella Professional® 1.0.3 by considering uniform distributions for the screened parameters (Sadiq et al. 2004b) as given in Appendix B.4. Since the contributions of input parameters to the final outputs were highly dispersed due to a large number of input parameters (102), the contributions were estimated for the aggregated inputs as given in the sensitivity analysis framework in Appendix B.4. Furthermore, the contributions of basic inputs to the variance of aggregated inputs can be estimated in the same way. The results of sensitivity analysis are shown in Table 5.2.

Table 5.2 Percent contribution of parameters to the variability of the WEC model

Parameters	% contribution	Parameters	% contribution
Total water footprint		***Total energy use***	
WF_chemicals	33	Hot shower energy (resi)	15
Residential outdoor irrigation	25	NR hot dishwasher energy	14
Shower water (resi)	7	Resi_standby energy loss	14
WF of energy use	7	Hot dishwasher energy (resi)	10
Faucet water (resi)	6	Hot faucet energy (resi)	10
Other municipal water	5	EE_chemicals	8
Toilet water	5	WW transport energy	6
Total carbon emissions		WW treatment energy	6
CF_residential hot water	46		
CF_NR hot water	27		
CF_chemicals	9		

Note: resi= residential, NR= non-residential or CII sector

As shown in Table 5.2, for the variance of the total water footprint of the Penticton UWS, the water footprint (WF) of chemicals (33%) and residential outdoor irrigation (25%) were the largest contributors. Although the WF of chemicals seems to have the highest contribution, in fact the major contributors to the variance of the WF of chemicals were water use (58%), residential wastewater (26%), and infiltration-inflow to sewer network (13%) rather than the unit WF of chemicals. Similarly, another parameter the WF of energy use also contributed over 5%; however, the major contributors to the variance of the WF of energy use were the amount of energy consumption (70%) and partly by unit WF of electricity (26%).

For the variance of the total energy use, the largest contributors were hot shower energy (residential) (15%) and residential standby energy loss (14%). All the parameters of energy use given in Table 5.2 are directly related to water use activities except the residential standby energy loss. However, the variance of the residential standby energy loss was also affected twothirds by indoor water use activities (shower, faucet, and laundry water), and one-third by the standby energy loss rate. Similarly, the variance of another parameter the embodied energy of chemicals was mainly affected by water use activities and partly by chlorine consumption and unit embodied energy of chlorine. For the variance of the total carbon emissions of the UWS, the highest contributors were the carbon footprint (CF) of residential hot water (46%) and carbon footprint of non-residential (NR) hot water (27%). The highest contribution by the carbon footprint of residential and NR hot water was obvious as they represent about 93% of the total carbon emissions of the UWS and variances in these inputs would have significant effects on the total carbon emissions. Although the carbon footprint of chemicals seem to affect significantly to the variance of the total carbon emissions, the major contributors to the variance of the carbon footprint of chemicals were water use activities rather than unit carbon footprint of chemicals. Monte Carlo-based based sensitivity analysis has widely been used, such as by Sadiq et al. (2004), Zio and Pedroni (2012), Veihe and Quinton (2000) and Veihe et al. (2000). The technique removes the difficulties with the traditional linear approach using single-valued inputs that in reality are random variables with the associated distributions (Veihe and Quinton 2000). However, it considers that input are independent (US EPA 1997; Veihe and Quinton 2000).

5.3.4 Scenario analysis

Various scenarios can be developed and analyzed by using the WEC model in order to identify an optimum WEC nexus or intervention in the SMUWS. In this study, 10 scenarios in five categories, namely business as usual (Category 1), indoor water demand management (Category 2), outdoor water demand management (Category 3), source water alternatives (Category 4), and water heating energy alternatives (Category 5), were developed for the CoP to improve the sustainability of the UWS (Table 5.3). In order to have a large possible improvement in the WEC nexus, the categories were developed as cumulative from Category 1 to 5; however, the letter suffices indicate alternative scenarios within a specific category.

Table 5.3 Scenarios developed for the UWS of Penticton

Scenarios	Features	Challenges/required actions
1	**Business as usual**	-
	Indoor water demand management	
2	Scenario 1 with efficient (best available) water fixtures: water efficient toilet, showerhead, faucet, waterless urinals (in CII sector) and energy and water efficient cloth washers and dishwashers	Awareness and rebate programs
	Outdoor water demand management alternatives	
3A	Scenario 2 with 50% irrigation demand reduction in lawn and parks	Xeriscaping and efficient irrigation
3B	Scenario 2 with lawn size reduction; lot coverage increased from 43% to 70% in single family houses and from 45% to 70% in duplex	Policy change to increase lot coverage of single family houses and duplex
3C	Scenario 2 with high density housing; residents moved from single family houses to small and large apartments equally; single family houses reduced from 51.3% to 5%	Awareness to prefer high density housing
	Source water alternatives	
4A	Scenario 3A with treated wastewater reuse for park and lawn irrigation and toilet flushing in residences	Secondary distribution pipes for reclaimed water is a challenge
4B	Scenario 3A with rainwater harvesting in residences; harvested water use in toilet, lawn, and laundry	Awareness and incentives for rainwater harvesting
	Water heating energy alternatives	
5A	Scenario 4A with natural gas increased from 66.8% to 95% and remaining 5% by electricity	Majority of residents switch to natural gas for water heating for hot water use
5B	Scenario 4A with electricity increased from 26.9% to 95% by replacing all natural gas and by reducing oil from 5.8% to 4.5%; 0.5% propane is as usual	Awareness and incentives to use electricity for water heating for hot water use
5C	Scenario 4A with 95% solar thermal energy and 5% electricity	Awareness and incentives to use solar thermal energy for water heating for hot water use

5.3.4.1 Business as usual scenario

The WEC model was simulated from 2015 to 2034. For Scenario 1, i.e., business as usual scenario, the monthly water footprint, energy use, and carbon emissions of the UWS is shown in Figure 5.5. Figure 5.5.shows a decreasing trend for all three elements primarily due to increasing water conservation and the use of efficient water fixtures and appliances.

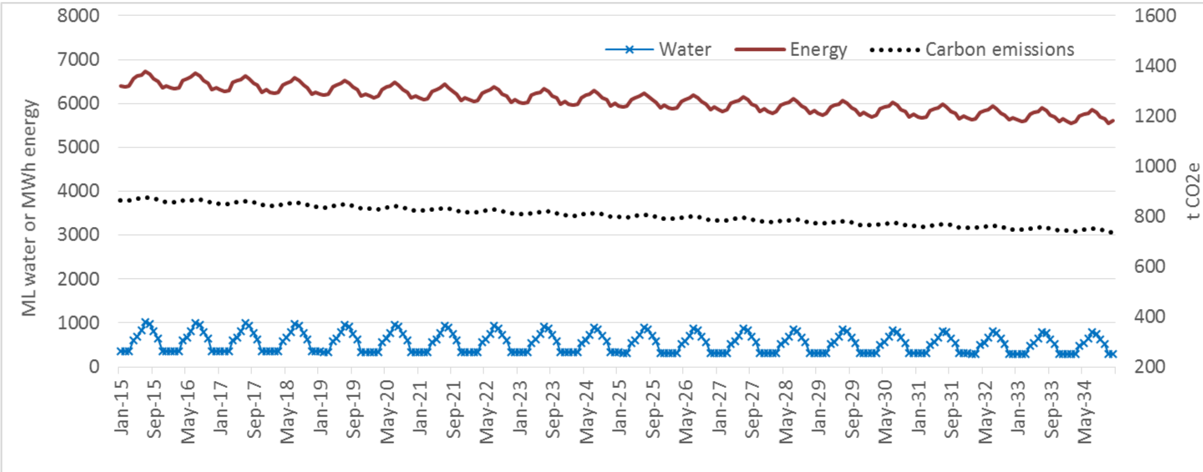


Figure 5.5 Monthly water footprint, energy use, and carbon emissions under Scenario 1 from 2015 – 2034

The average annual water footprint, energy use, and carbon emissions of the UWS will be 6,539 ML; 72,997 MWh, and 9,644 tCO_2e respectively. The results of all scenarios in terms of the change with respect to Scenario 1 are shown in Figure 5.6.

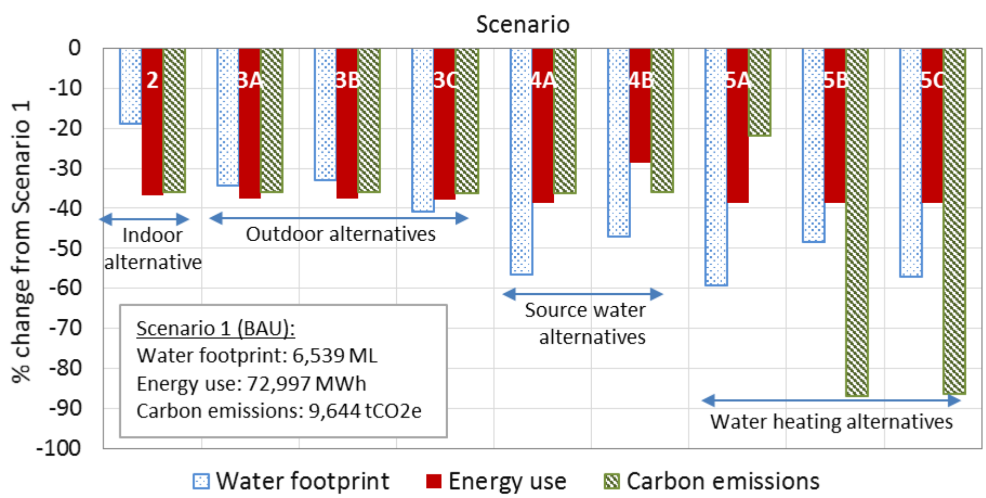


Figure 5.6 Change in average annual water footprint, energy use and carbon emissions compared to Scenario 1

5.3.4.2 Indoor water demand management

Indoor water demand management strategies in Scenario 2 can reduce the average annual water footprint, energy use, and carbon emissions by 19%, 37%, and 36% respectively compared to Scenario 1. Required actions for Penticton, as given in Table 5.3, include rebate programs for water fixtures and appliances (toilets, washing machines, and showers) although the toilet rebate program was once launched in 2006 (Maurer 2010).

5.3.4.3 Outdoor water demand management

Outdoor water demand management can be considered in addition to indoor water demand management as in Scenario 3C. The average annual water footprint, energy use, and carbon emissions can be reduced up to 41%, 38%, and 36% respectively in Scenario 3C compared to Scenario 1. However, it requires a larger behavioural change among the residents, specifically, 95% of them should prefer high density housing, such as apartments and row houses. Alternatively, municipal policy change towards a higher lot coverage as in Scenario 3B, can achieve similar results, especially for total energy use and carbon emissions, but it reduces landscaping area in the community. It is noteworthy that xeriscaping can reduce more than 50% of the water demand of lawn and park irrigation (Boot and Parchomchuk 2009). By xeriscaping as in Scenario 3A, the average annual water footprint, energy use, and carbon emissions can be reduced up to 34%, 38%, and 36% respectively. In this analysis, the carbon sequestration by landscaping was not considered.

5.3.4.4 Source water alternatives

Scenario 4A (wastewater reuse) can reduce the average annual water footprint, energy use, and carbon emissions up to 57%, 39%, and 36% respectively compared to Scenario 1. In Scenario 4A, the rate of energy use for the secondary distribution of reclaimed water was considered to be the same as that of drinking water distribution energy. Since, Penticton has been using reclaimed water for irrigating golf courses and public parks, additional treatment, except chlorination, of reclaimed water may not be required for further use. In Scenario 4B (rooftop rainwater harvesting) can reduce an average annual water footprint, energy use, and carbon emissions up to 47%, 29%, and 36%, respectively compared to Scenario 1, indicating increased energy use by

7% from Scenario 3C due to energy intensive rainwater harvesting in Penticton (semi-arid region).

5.3.4.5 Water heating energy alternatives

Scenario 5B (increased used of electricity in indoor water heating) reduces the carbon emissions of the UWS up to 87% as a result of a lower carbon footprint of hydro-based electricity than that of natural gas and oil (Ministry of Environment 2013). Alternatively, the increased use of solar thermal energy for water heating (Scenario 5C) can reduce carbon emissions by 86%. It is noteworthy that the trend of natural gas use for water heating has been increasing in Canada since the past 10 years (Natural Resources Canada 2014). However, Scenario 5A (increased use of natural gas) can reduce carbon emissions only by 22% compared to Scenario 1, indicating an increased carbon emissions of 14% compared to Scenario 4A due to a higher carbon footprint of natural gas (Ministry of Environment 2013). Scenarios 5B and 5C were better scenarios based on lower carbon emissions and energy use; however, Scenario 5C had the best performance with an additional decrease of annual water footprint, but a detailed feasibility study is recommended on the use of solar thermal energy to meet the energy demand of water heating throughout the year, especially in winter. The developed 10 scenarios include extreme cases in addition to the business as usual scenario. The extreme scenarios, such as Scenario 5C will assist a decision maker to approximate a range of extreme values or uncertainty in a particular intervention.

5.3.4.6 WEC nexus analysis of interventions

The WEC nexus for important individual interventions in UWSs identified in the previous sections were further analyzed for Penticton in 2015-2034. For this purpose, the individual water-based interventions considered were efficient water fixtures as in Scenario 2; xeriscaping as in Scenario 3A (without additional Scenario 2 features in Scenario 1); and wastewater reclamation as in Scenario 4A (without additional Scenario 3A features in Scenario 1). Similarly, the energy-based individual interventions considered are natural gas dominancy (as in Scenario 5A), electricity dominancy (as in Scenario 5B), and solar thermal energy dominancy (as in

Scenario 5C) without the additional Scenario 4A features in Scenario 1 for these interventions. The analysis results are shown in Figure 5.7.

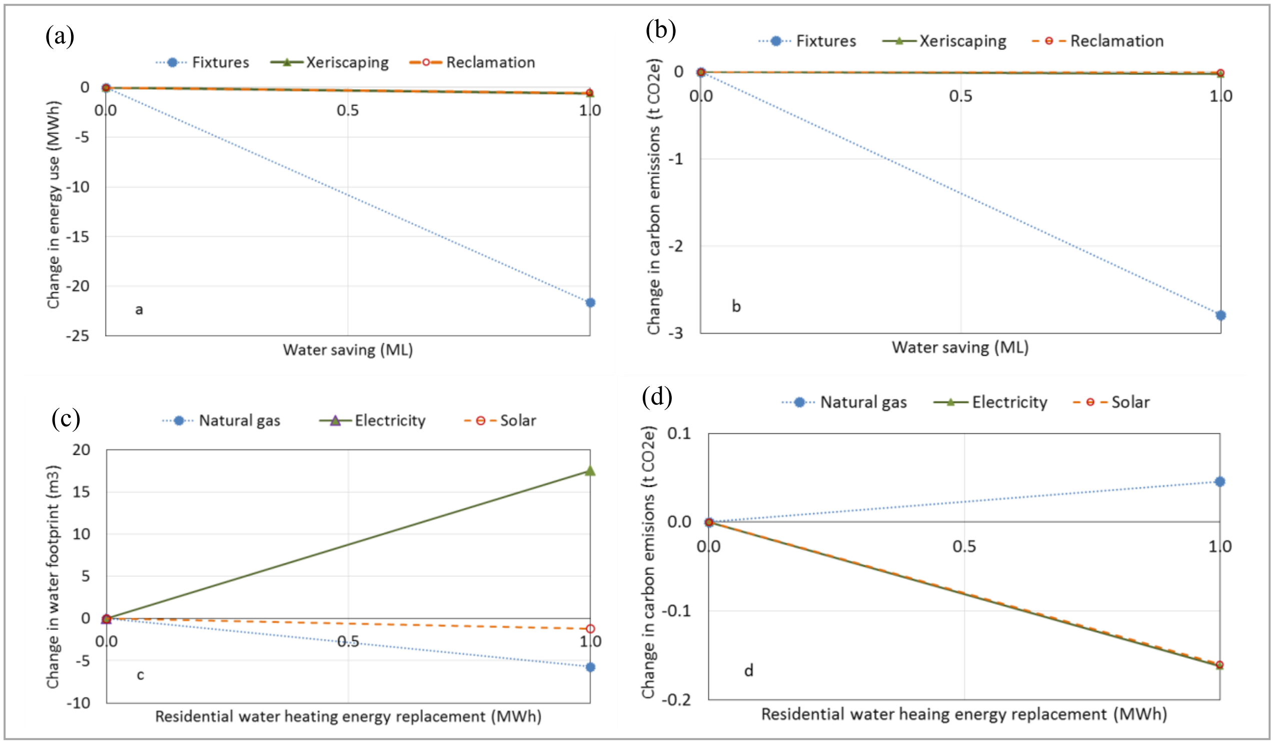


Figure 5.7 WEC nexus analysis of interventions: water-based (a & b) and energy-based (c & d)

The unit water saving would result in significantly different energy saving (p = 0.000) and carbon saving (p = 0.000) for various water-based interventions based on repeated measures ANOVA (Figure 5.7**)**.The very high energy saving by efficient water fixtures is due the fact that the reduced water eliminates its energy requirement from raw water collection to wastewater treatment and disposal, whereas the reduced water in irrigation (xeriscaping and water reclamation interventions) eliminates only upstream energy requirement (raw water collection to water distribution). Moreover, efficient water fixtures reduce hot water demand and improve energy efficiency (ENERGY STAR 2014a; c). The similar features as of energy saving were depicted by carbon saving per unit water saved as the carbon emissions are mainly from the energy use.

In the energy-based intervention analysis, the replacement of 1 MWh of indoor water heating energy by different interventions have significantly different mean water saving (p = 0.000) and mean carbon saving ($p = 0.000$). The natural gas and solar thermal would result positive water saving, whereas it would be negative for electricity because of a higher water footprint of hydro-based electricity (19.7 L/kWh) of Penticton (Okadera et al. 2014) than that of natural gas (0.4 L/kWh) (Okadera et al. 2014) and solar thermal (3.98 L/kWh) (Fulton and Cooley 2015). Furthermore, carbon emissions would be increased by 0.05 tCO_2e for the energy replacement by natural gas, whereas the carbon emissions would be decreased by about 0.16 tCO_2e for the replacement either by electricity or solar thermal energy. The carbon footprint of natural gas is higher (Ministry of Environment 2013) than that of electricity (Ministry of Environment 2013) and solar thermal (Menzies and Roderick 2010). This dynamic WEC nexus analysis of interventions would provide better results than a simple unit footprint-based calculation as the dynamic model incorporates the feedbacks in the UWS.

The estimated values may be associated with uncertainties, which can broadly be classified as aleatory uncertainty – due to natural variation resulting in uncertain data or parameter values (or parameter uncertainty) and epistemic uncertainty – due to imperfect understanding of the system (model uncertainty) (Dyck et al. 2014). In particular to the present application, exponential growth has been used for human population, CII sector, and other parameters, such as inflow-infiltration and energy consumption. The estimated exponential growth rates were found to be applicable to the historical data and the similar growth pattern with adjusted rates were used for future forecasting (20 years). However, the growth rates may be associated with high uncertainties when forecasted for a very long duration due to variations in dynamics among model variables. The parameter uncertainty can be approximated by Monte Carlo simulations as in sensitivity analysis and to some degree by scenario analysis. Furthermore, uncertainty can be reduced by better quality and region specific data with improved parameter relationships. Such improvement in sensitive parameters will highly reduce uncertainty in the WEC nexus.

5.4 Summary

A comprehensive water-energy-carbon (WEC) nexus model for an urban water system (UWS) by using system dynamics is proposed to assist municipalities, urban developers, and policy makers for neighbourhood water planning and management. The proposed model and decision support system was developed for the operational phase of SMUWSs by using Stella®. The model was validated using historical water and energy consumption data (2005-2014) of Penticton (British Columbia). Spearman's correlation coefficients between water and energy, water and carbon, and energy and carbon were 0.94, 0.89, and 0.83 respectively revealing highly significant interconnections. The energy for water was 11.1 MWh/ML, water for energy was 6512 L/MWh, and carbon emissions were 124.4 kg CO_2e/MWh from energy use and 120.8 kgCO_2e/ML from wastewater processes. The highest energy consumer was found to be the indoor hot water use in the residential and CII sector consuming approximately 90% of the operational energy demand and contributing about 93% to carbon emissions. Indoor hot water use should be prioritized for the reduction of energy use and carbon emissions. The contributions of residential outdoor irrigation, shower water, faucet water, and non-residential (CII) and residential hot dishwasher energy to the model variability were higher than other parameters.

A Monte Carlo-based sensitivity analysis showed residential outdoor irrigation and water heating energy for shower and dishwasher have higher contribution to model variability. The intervention analysis reveals significant differences in savings in water, energy, and carbon for various water and energy-based interventions in SMUWSs and the developed DSS is well capable for analyzing these dynamic savings. The developed decision support system is capable of dynamic analysis of different WEC-based interventions to improve the sustainability of SMUWSs. The decision support system can be used by utilities, urban developers, and policy makers for sustainable urban water planning to reduce water consumption, energy use, and carbon emissions in neighbourhoods. Furthermore, the tool can also be used for operational neighbourhoods to forecast future WEC nexus.

Chapter 6 Investigating Impacts of Residential Density on WEC Nexus

A version of this chapter has been published by the *Journal of Cleaner Production* entitled "Impacts of neighbourhood densification on water-energy-carbon nexus: Investigating water distribution and residential landscaping system" (Chhipi-Shrestha et al. 2017c).

6.1 Background

Water Distribution Systems (WDS) can significantly consume energy and release greenhouse gases (Hellstro 2000; Wu et al. 2015). Per capita energy requirements for WDSs can be reduced by developing high residential density in communities (Filion 2008). Densification is one of the major reasons to reduce per capita infrastructure and land requirement including water infrastructure (Duncan 1989; Frank 1989; Gleick et al. 2003). Dense residences comprising multi-family (MF) buildings can also have a lesser irrigation demand for landscaping since landscaping requirements for MF buildings are lower (City of Kelowna 2007; City of Penticton 2015a; District of Peachland 2014a). Reduced water demand also has a decreased upstream energy use. These interlinkages suggest that higher density can have reduced water use, energy requirement, and carbon emissions. However, a lesser amount of landscaping due to dense residences results in reduced carbon sequestration. Atmospheric CO_2 is photosynthesized and stored as plant biomass by herbs, shrubs, and trees. Shrubs and trees store carbon for a long term in their biomass as carbon stock, whereas herbs (e.g., grasses) decay and some of their biomass-carbon is humified and stored for a long term in soil as soil organic carbon (SOC) (Zirkle et al. 2011, 2012). Furthermore, the life cycle cost of WDSs may be lower in dense residences (Speir and Stephenson 2002).

Energy is required for water abstraction, conveyance, treatment, and distribution. The required energy varies with several factors, such as topography, source water quality, urban form, population density, and adopted management strategies. Energy can also be harvested from the hydraulic energy of WDSs (Ye and Soga 2011), as well as from the thermal and chemically bound energy of wastewater (Meda et al. 2012; Nowak et al. 2011). In addition, the change of residential density directly affect landscaping that in turn affect the amount of carbon sequestration. Water is required directly and indirectly for energy generation. Direct water is required for hydroelectricity generation, whereas a large amount of indirect water is necessary for the exploration, extraction, and beneficiation of fossil fuels (Meda et al. 2012) and also for renewable energy crop cultivation, such as biofuel. This interconnection shows a complex WEC nexus of Water Distribution and Residential Landscaping System (WDRLS). The impacts of neighbourhood densification on the WEC nexus of WDRLS are shown in Figure 6.1.

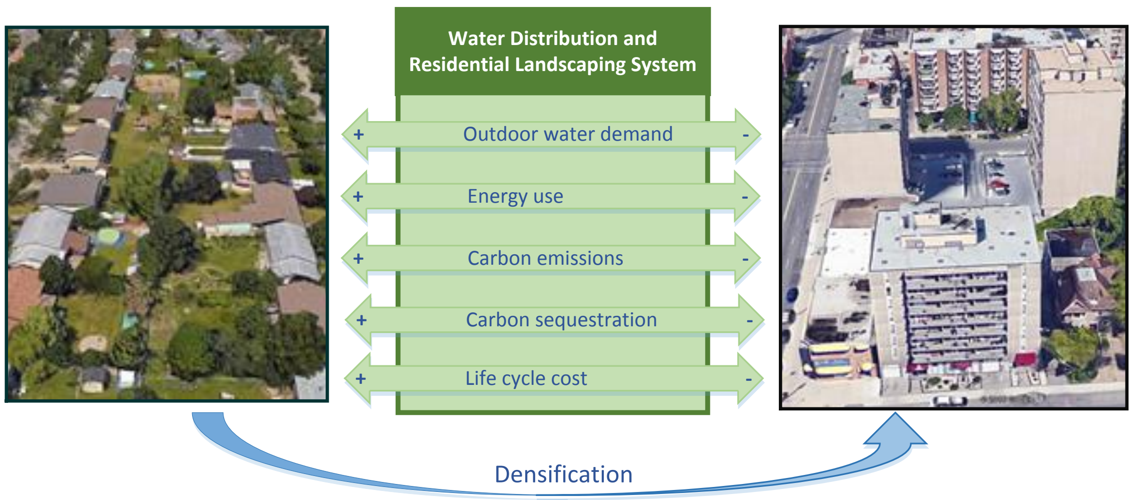


Figure 6.1 Impacts of neighbourhood densification on the WEC nexus of WDRLS (per capita)

This chapter aims to study the impacts of urban residential density on the WEC nexus of water distribution and residential landscaping system. The results will help municipalities and urban developers to identify water and energy efficient WDSs and landscaping with respect to different residential densities. In addition, the findings will help them to decrease the resulting carbon emissions and water distribution cost.

6.2 Methodology

A conceptual framework for studying the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system has been developed (Figure 6.2). The framework is explained in the following sections.

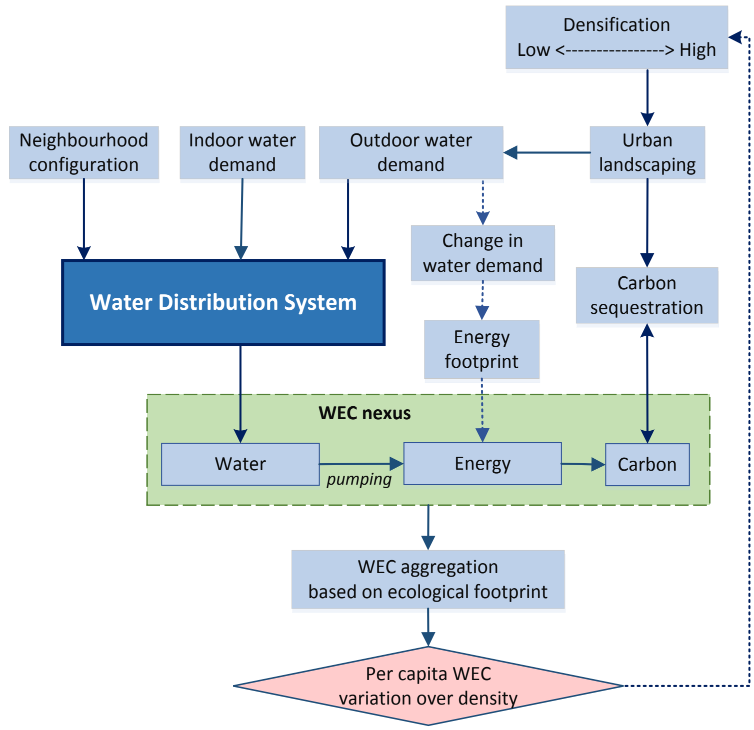


Figure 6.2 Conceptual framework to study the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system

6.2.1 Water demand

As the first step, a neighbourhood mainly consisting residential buildings is designed on a given land. The neighbourhood design should follow municipal bylaws for lot coverage, road size, building mix (SF and MF buildings), building height, and public park designation. Also, alternative designs are prepared by changing residential density. The residential density can be increased by increasing MF buildings and decreasing SF buildings, and vice-versa. The developed alternative designs have different total lot coverages, indicating varying per capita residential landscaping size. These designs alter their overall and spatial water demand. The average water demand of each alternative neighbourhood is estimated based on Equations 6.1 to 6.4. Then, WDSs including fire demand are designed as per the concerned municipal bylaw for each alternative by using EPANET 2 (Aydin et al. 2014).

$$W_{neighbour} = W_r + W_{ci} + W_{park} \qquad \textbf{Equation 6.1}$$

$$W_r = [\sum_i^n (\eta_f * F_f)_i] * P_t + L + (A_r)_l * I_l$$ **Equation 6.2**

$$W_{ci} = N_{ci} * C_{ci} + (A_{ci})_l * I_l$$ **Equation 6.3]**

$$W_{park} = A_p * I_p$$ **Equation 6.4**

where $W_{neighbour}$ is neighbourhood water demand, W_r is residential water demand, W_{ci} is commercial and institutional water use (Offices, retails, hotels, and schools), W_{park} is parks water demand; η is efficiency of fixtures (L/use), subscripts r, ci, f, l, and p refers to residential, commercial and institutional, fixture, landscaping, and parks respectively, i refers to fixtures toilet, shower, faucet, laundry, and dishwasher; F is frequency of fixture use (no. of uses/p/d), P_t is total population; L is leakage; A is area (ha); I is irrigation rate (L/ha/d); N_{ci} is number of rooms for hotels and floor space (sqft) for offices, retail, and schools, and C is indoor water consumption rate (L/room/d for hotels and L/sqft/d for offices, retails, and schools).

6.2.2 Energy use

The designed WDS is used to estimate the required energy use for water distribution based on the capacity of pump and its duration of use. The design provides utility energy (water mains energy). However, for mid-rise and high-rise residential buildings, i.e., for apartments, additional energy is required for booster pumps at apartments to supply water to elevated levels. The additional energy for a booster pump (house pumping energy) at each apartment can be estimated using Equations 6.5 and 6.6 for buildings under 15 stories (Cheng 2002).

$$P = \frac{\gamma Q H_p (1+\alpha)}{\eta * \eta_t * 1000}$$ **Equation 6.5**

$$E_h = P * (1 + f) * N$$ **Equation 6.6**

By combining Equations 6.5 and 6.6,

$$E_h = 2.23 * 10^{-3} * \gamma Q H_p * N$$ for α =0.2, f =0.3, η = 0.7 and η_t =1 **Equation 6.7**

where P is power of lift pump (kW), γ is specific weight of water (9806 N/m^3), Q is pumping capacity of lift pump (m^3/s) which can be estimated from an average water discharge considering a peak factor for hourly demand of 7.4 (Ontario Ministry of Environment 2008), H_p is height from the lift pump to the top of the building (m) and can be estimated as using Hp = 3.1 (F+1) with F as number of floor and 3.1 m as the floor-floor height for residential building (Council on Tall Buildings and Urban Habitat 2016), α is safety factor of pumping power (0.1 to 0.2), η is pump efficiency (65% to 85%), η_t is mechanical transmission efficiency (92% to 100%), E_h is Energy consumed (kWh) in a house pump, f is friction loss within pipes (30%), N is number of hours a pump is operated (Cheng 2002).

The energy required for WDS (E_{WDS}) is the sum of water mains energy (E_m) and house pumping energy (E_h) as in Equation 6.8.

$$E_{WDS} = E_m + E_h \quad \textbf{Equation 6.8}$$

The reduced water demand in high density residences also lowers upstream energy use of neighbourhood water demand, such as energy for conveyance and water treatment. The change in total energy requirements in different alternatives due to densification can be estimated by using Equation 6.9.

$$\Delta E_{total} = E_{WDS} + \Delta W * EI_{upstream} \quad \textbf{Equation 6.9}$$

where ΔE_{total} is change in total energy requirement (kWh), E_{WDS} is energy required for WDS of a particular neighbourhood design (kWh), ΔW is change in total water demand in the particular design compared to any design, and $EI_{upstream}$ is upstream energy intensity of water (kWh/m^3)

6.2.3 Carbon emissions and sequestration

The energy related carbon emissions of WDSs are estimated based on the energy use by using Equation 6.10.

$$C_E = E_{WDS} * E_f \quad \textbf{Equation 6.10}$$

where C_E is carbon emissions related to energy, E_{WDS} is energy required for WDS (kWh) and E_f is CO_2 emission factor for energy (kgCO_2e/kWh). If different sources of energy are used, estimate total carbon emissions for each energy source by multiplying energy use with its respective emission factor and then compute sum to estimate grand total carbon emissions.

The carbon sequestration of landscaping constitutes the sequestration of SOC, shrub biomass-carbon and tree biomass-carbon (Lal and Augustin 2012; Zirkle et al. 2011). For soil, the net SOC sequestration rate is the balance of gross carbon accumulation and carbon emission during lawn maintenance practices (mowing, irrigation, fertilizer, and pesticides use). The total carbon sequestration in a residential landscaping can be estimated by using Equation 6.11.

$$C_s = SOC_s * \Delta A_l + C_{st} * N_t + C_{ss} * N_s \qquad \textbf{Equation 6.11}$$

where Cs is total carbon sequestration (kg CO_2/yard/yr), SOC_S is net SOC sequestration (kg CO_2/m^2/yr), ΔA_l is net landscaping area after reducing the landscaping area by trees and shrubs canopy as the landscaping shaded by trees and shrubs is assumed not to be productive (Zirkle et al. 2012), C_{st} and C_{ss} are net carbon sequestration by trees and shrubs, N_t and Ns are number of trees and shrubs respectively.

The net carbon emissions by a WDS and residential landscaping in a neighbourhood can be estimated by using Equation 6.12.

$$C_n = C_E - \sum_{i=1}^{n} (C_s)_i \qquad \textbf{Equation 6.12}$$

where C_n is net carbon emissions (kg CO_2/yr), C_E is carbon emissions related to energy use in a WDS, Cs is total carbon sequestration by an individual landscaping (kg CO_2/m^2/yr), n is number of all residential landscaping with water supplied by the WDS.

6.2.4 WEC aggregation

Water and energy are natural resources, whereas emitted GHGs (carbon) are pollutants from a global warming perspective. Generally, freshwater (streamflow) is produced within a river basin (land area). Energy generation requires landmass too, e.g., hydroelectricity production needs land area for river catchment and reservoir site; a large land area is required for fossil fuel extraction. Furthermore, vegetated land is needed for the sequestration of carbon emitted to the atmosphere. All these three elements – water, energy, and carbon – have a common feature of the use of land resources. This common feature can be used to aggregate them together as WEC nexus.

Therefore, water consumption, energy use, and carbon emissions can be integrated by converting them into a common measurement unit – ecological footprint.

The ecological footprint is a well-known resource accounting tool for measuring biologically productive land and water area, an individual or a region requires to produce the resources it consumes and to absorb the waste it generates, using prevailing technology and resource management (Kitzes et al. 2013; Musikavong and Gheewala 2016; Wackernagel and Rees 1996). Global hectares are used as a common unit to express an ecological footprint. A global hectare refers to a hectare that is normalized to have the world average productivity of all biologically productive land and water in a given year (Kitzes et al. 2013). The ecological footprint of water, energy, and carbon emissions can be obtained from the related literature as given in the application section, which can be summed together to estimate aggregated WEC.

6.2.5 Life cycle cost analysis

Life cycle cost (LCC) analysis is an economic assessment method in which all costs arising from owning, operating, maintaining, and ultimately disposing of a project or product are considered (US DOE 1996). LCC of WDSs is estimated as the sum of capital cost, operation cost, and repair and replacement cost of water distribution infrastructure, namely pipes, pumps, and valves. The net present value (NPV) of the annualized LCC of WDSs is estimated by using Equation 6.13 (Davis et al. 2005).

$$NPV\ (i, N) = \sum_{t=0}^{N} \frac{R_t}{(1+i)^t} \qquad \textbf{Equation 6.13}$$

where i is discount rate, i.e., 4% (C-SHRP 2002; Umer 2015), t is number of years, N is total planning duration (years), R_t is net cash flow at time t.

6.3 Results

6.3.1 Application

The proposed conceptual framework was applied as a case study to a newly planned neighbourhood located in the Okanagan Valley, British Columbia (BC), Canada as described in the following sections.

6.3.1.1 Study area

The neighbourhood has an area of approximately 51 ha in a rugged topography with a maximum elevation difference of 80 m. The neighbourhood is planned for mixed use comprising residential, commercial, and institutional buildings. The neighbourhood is planned to have approximately 24% area covered by parks and trails. The planned residential population is approximately 4848 with a net residential density of 149 persons per hectare (persons/ha). Net residential density refers to the dwelling units or number of persons living in residential buildings divided by the land area covered by the buildings, private access ways, and local public roads (Landcom 2011). The neighbourhood has three zones: A – low density residential, B – medium density residential, and C – commercial area with high density residential buildings as per the information provided by the neighbourhood developer.

The initial neighbourhood plan with lot division and configuration was obtained from the developer. A typical SF building in the neighbourhood has an average lot coverage of 365 m^2 for building, garage, driveways, and sidewalk. The remaining area was assumed to be landscaped as per the municipal bylaw and provided neighbourhood plan. A lot coverage of 60% and 100 % was considered for medium and high density MF buildings respectively as per the municipal bylaw (District of Peachland 2014a). Without altering neighbourhood configuration, 11 neighbourhood designs were prepared by increasing residential density from D1 to D11 (Appendix C.1). The residential density was gradually increased by converting SF lots to MF lots by lot reorganization. In the series of design, Design D5 represented initially planned base design. The lot reorganization was performed by maintaining the same average unit residential area and number of stories in SF and MF buildings as of base design (Design D5).

A WDS was designed for the entire neighbourhood by following the development and servicing bylaw of the concerned municipality (District of Peachland 2004) and Ontario Ministry of Environment (2008). The maximum daily demand was estimated by multiplying the average daily demand (ADD) and maximum day factor, whereas peak hourly demand was estimated by multiplying ADD and peak hour factor. Also, the needed fire flow was estimated by using Fire Underwriters Survey (2007). Based on the municipal bylaw (District of Peachland 2004), the design flow is the greater value between peak hourly demand and the sum of the maximum daily demand and needed fire flow. The design flow was used to prepare WDS using EPANET 2 (Aydin et al. 2014). The energy intensity (kWh/m^3) of water mains was estimated based on the pump capacity used in the WDS. The water mains energy consumption for residential water only was calculated from the estimated energy intensity and total residential water demand. The same process was repeated for all alternatives. In addition, house pumping energy was estimated for multi-family buildings by using Equations 6.5 and 6.6. The number of stories in a MF building can vary from six to ten as per the bylaw and an average of eight stories was considered in this study. Then, the total energy for WDS was estimated by using Equation 6.8.

6.3.1.2 Data

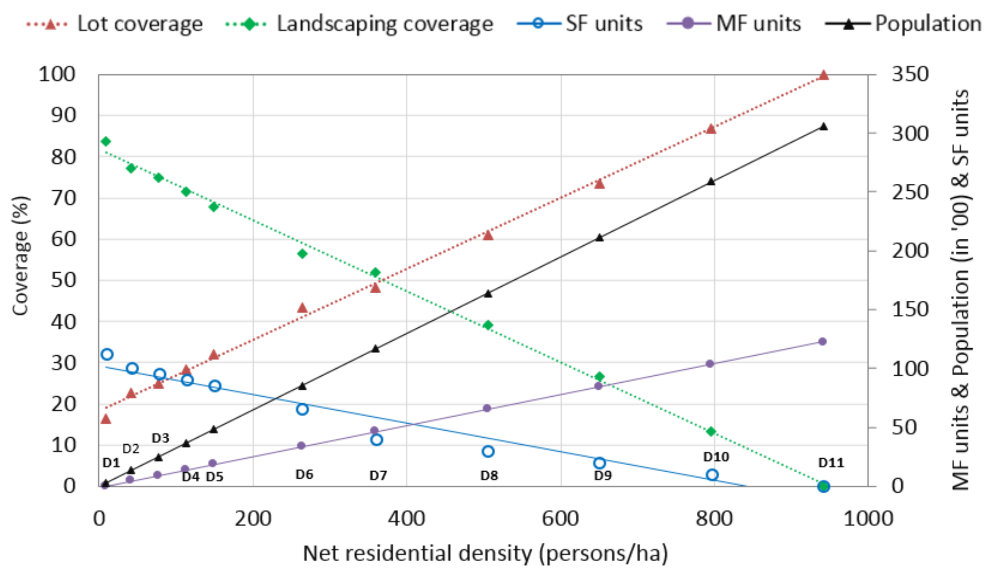
The neighbourhood was planned to be a sustainable community with the use of efficient water fixtures, e.g., efficient toilet, showers, cloth washers, etc. The efficiency and use frequency of various water appliances and fixtures as well as outdoor irrigation for residential and commercial and institutional (CI) buildings were estimated from the literature as given in detail in Appendix C.2. This study is focussed only on the residential water; however, CI water was also estimated to calculate total water demand. The total water demand is required to design a WDS to estimate energy use by residential water as in reality water is distributed by the same WDS to an entire neighbourhood comprising residential, commercial and institutional buildings, and public parks. The energy related carbon emissions were estimated by using the emission factor of the grid electricity. The cost data of water mains installment and repair/replacement, electricity, water pumps, and valves were obtained from the related literature as given in Appendix C.2 and carbon sequestration of residential landscaping was obtained from the related literature as given in Appendix C.3

The ecological footprint of Canadian freshwater is 1.08 x10^{-4} gha/m^3/yr that was estimated as the reciprocal of annual freshwater availability per unit river basin area (Kitzes et al. 2013; Meng et al. 2016; Statistics Canada 2003). Similarly, the ecological footprint of Canadian hydroelectricity is 3.9x10^{-6} gha/kWh/yr that was estimated as the ratio of per capita ecological footprint of hydroelectricity use (0.4 gha/p) (FCM 2005) and per capita hydroelectricity use (10,213 kWh/p/yr) (Environment Canada 2013). The ecological footprint of carbon emissions is 0.224 gha/tCO_2e (Kissinger et al., 2013). For the WDS, a 30-year planning period was considered (Speir and Stephenson 2002).

6.3.1.3 Residential density and WEC nexus

The SF residences were changed to MF residences to increase residential density. To show the impact of such changes on water distribution and residential landscaping system, the variation of characteristics: lot coverage, landscaping coverage, SF units, MF units, population, WEC nexus, and LCC with respect to net residential density is presented in the next sections.

i) **Lot coverage and landscaping**: The change in lot coverage and landscaping of residential lots with various net residential densities is presented in Figure 6.3. The number of SF and MF units considered and their population in various densities are also presented in the figure.

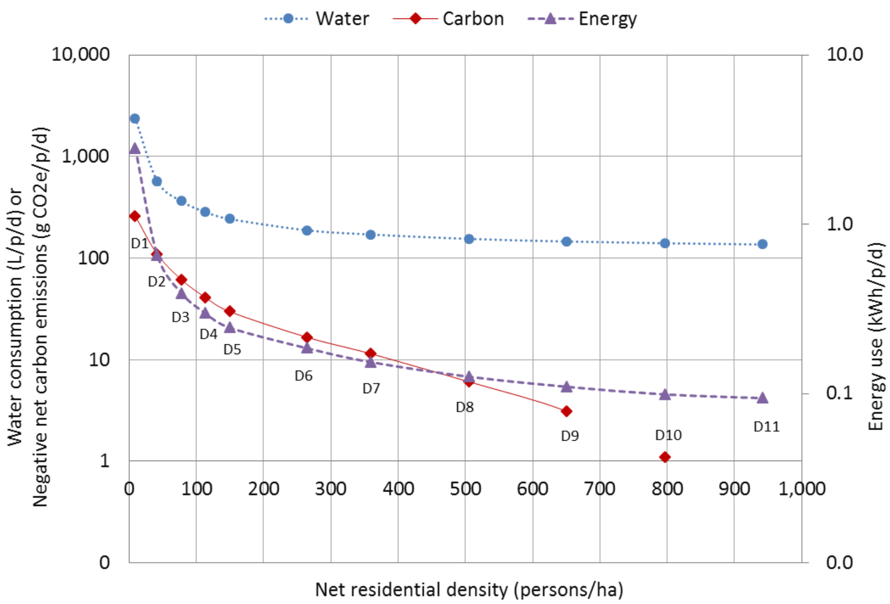


Note: D1 to D11 refers to the density range of the neighbourhood

Figure 6.3 Lot coverage, landscaping, and population variation over net residential density

Figure 6.3 shows that initially planned base Design D5 (149 persons/ha) had a lot coverage of approximately 32% and landscaping coverage of 68%. By changing all residential lots to SF lots, the net residential density will be very low with a value of 8.7 persons/ha (D1) with a lot coverage of about 16% and landscaping coverage of 84%. However, when all residential lots were used for MF residence, with a maximum of eight stories as per the bylaw, the net residential density would be very high with a value of 941.6 persons/ha (D11), with a lot coverage of 100%. This means that the neighbourhood can have a landscaping coverage from 0% (all MF residences) to 84% (all SF residences) as per the existing municipal bylaw and the information provided by the developer. Moreover, the change of SF and MF units resulted in the gradual increase of population from 283 (D1) to 30,631 people (D11). The variations of landscaping and population with net residential density have linear relationships with coefficient of determination (r^2) of approximately 1 for landscaping and population.

ii) WEC nexus of WDS and landscaping: The per capita water consumption, energy use, and net carbon emissions by WDS and residential landscaping in various densities are shown in Figure 6.4. In Figure 6.4, per capita water consumption, energy use, and net carbon emissions interact over net residential densities. The interaction indicates that all the three elements and respective net residential density should be taken into consideration in decision making.



Note: Negative net carbon emissions at 942 persons per ha (D11) is negative and not shown in the log scale

Figure 6.4 WEC nexus in various residential densities: Interaction plot

The three elements were integrated to WEC nexus by converting them to ecological footprint (gha/p/yr). The estimated WEC nexus is given in Figure 6.5. The WEC nexus or simply ecological footprint (EF) sharply decreases from 0.08 gha/p/yr (D1) to 0.008 gha/p/yr (D4) and then gradually decreases to about 0.006 gha/p/yr (D11). A total of 93% of the EF (WEC nexus) was reduced from Design D1 to D11 by increasing net residential density from 8.7 persons/ha (D1) to 941.6 persons/ha (D11) as shown by the characteristic curve in Figure 6.5. The increased residential density would also increase the lot coverage from 16% in D1 to 100% in D11. The decrease in per capita water demand was achieved due to the decrease in landscaping requirements. The curve has a power relationship with r^2 of approximately 84%. The power relationship can be used to identify an optimal density. The rate of change of the EF with respect to net residential density (dy/dx) is also shown in Figure 6.5. The value of dy/dx is very low and almost similar beyond D6, indicating the density around 264 persons/ha as an optimal density.

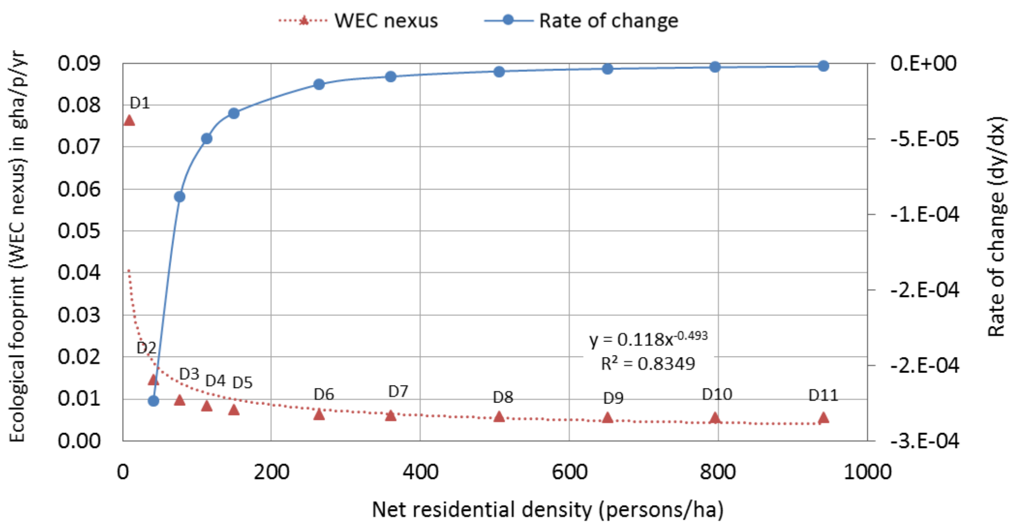


Figure 6.5 Ecological footprint of various residential densities D1 to D11

The per capita water demand decreased from approximately 2,373 L/p/d (D1) to 136 L/p/d (D11) with a total of 94% reduction. Similarly, the energy intensity of water use decreased gradually from 1.2 kWh/m^3 (D1) to 0.69 kWh/m^3 (D11) as shown in the characteristic curve of energy intensity. Also, the per capita energy use for the WDS decreased from 2.8 kWh/p/d to 0.09 kWh/p/d (D11) with a reduction of 97% (2.5 kWh/p/d). The energy use includes water mains energy (utilities) and house pumping energy (apartment). Moreover, the reduced water demand from Designs D1 to D11 leads to upstream energy saving of 1.3 kWh/p/d resulting in the total energy reduction of 3.8 kWh/p/d from D1 to D11. Energy use emits carbon, whereas residential landscaping sequester it. The net carbon emissions were negative in Design D1 to D10, but positive in D11. The negative net carbon emission or positive net carbon sequestration decreased from 260.9 g CO_2e/p/d (D1) to 1.1 g CO_2e/p/d in D10 and -0.3 g CO_2e/p/d in D11, indicating positive carbon emissions in D11. All these characteristic curves of water demand, energy intensity, energy use, and negative net carbon emissions have power relationships with r^2 ranging from 90% to 99%. Furthermore, the rate of change of per capita water demand, energy use, and net carbon emissions with respect to density (dy/dx) are relatively low and almost unchanged beyond D6 similar to that of the ecological footprint.

6.3.1.4 Life cycle cost and residential density

A characteristic curve of per capita LCC of the WDS and net residential density was prepared (Figure 6.6). The per capita LCC, in terms of net present value, decreased sharply from \$96.2/p/d (D1) to \$ 9.7/p/d (D4) and then gradually to \$ 2.6/p/d (D11), a 97% reduction from D1. The curve has r^2 of approximately 99%. The characteristic curve also has a power relationship with residential density similar to that of per capita EF. In addition, the value of dy/dx of the curve is very low beyond D6. Therefore, the density around D6 (264 persons/ha) also represents an optimal density with respect to the per capita LCC of the WDS.

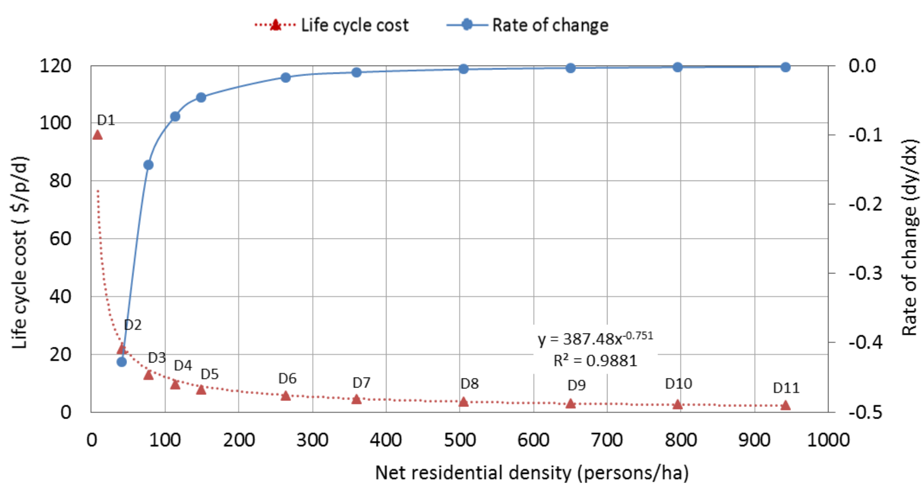


Figure 6.6 LCC of the WDS and its rate of change in different residential densities

6.3.1.5 Uncertainty and sensitivity analysis

The uncertainty analysis was performed by estimating probabilistic water demand to approximate the associated uncertainty. The sensitivity analysis was performed to identify sensitive parameters in water demand estimation.

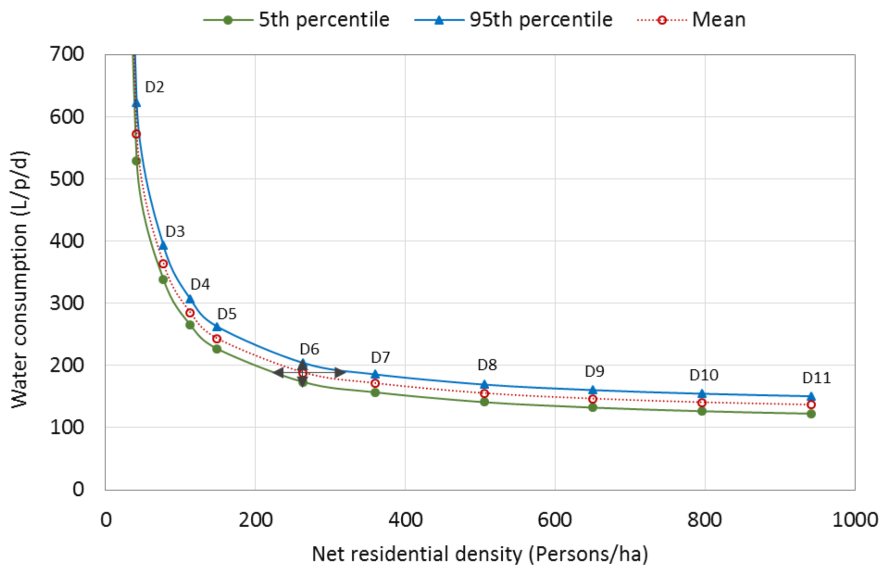
i) **Probabilistic water demand:** The estimated water demand is affected by several factors making the results uncertain. The uncertainty analysis was conducted based on the probabilistic technique using Monte Carlo simulation with 10,000 simulations in each WDS design by using the @Risk 7 (Lee et al. 2011). The simulations predict per capita probable water demand with randomly generated values for input parameters based on the given probability distribution (Lee et al. 2011). The parameters with their distributions are given in Table 6.1.

Table 6.1 Major factors affecting water demand and their distribution parameters

Factors	Distribution	Units	Remarks
Indoor water consumption rate	N~ (133, 8.5)	L/p/d	133 L/p/d is average for the neighbourhood[1] and 150 L/p/d of the Okanagan and North American[2] was considered maximum value
Lot coverage (MF)	T~(50, 60, 60)	%	60% is maximum lot coverage
Lawn irrigation	N~ (991, 53)	$L/m^2/yr$	991 $L/m^2/yr$ is average[3] & maximum value was considered as of golf irrigation (1005 $L/m^2/yr$)[3]
Dwelling occupancy	N~(2.5,0.15)	Persons/DU	2.5 is average for the neighbourhood[4] & 2.2 of a neighbouring city[5] was considered as a minimum value. This distribution includes the highest provincial value[6] of 2.6.

Note: Normal distribution: N ~ (μ, σ) & assumed a truncated distribution for the given min to max range with $\mu \pm 4\sigma$ representing 99.997% data; Triangular distribution: T~ (min, most probable, max values); DU= dwelling unit
1. From Appendix C.2 2. OBWB (2016) 3.OBWB (2010) 4. From developers' plan 5. Statistics Canada (2015)
6. Statistics Canada (2014b)

The results show that per capita water demand band (within 5th and 95th percentile boundary) varies with different net residential density (Figure 6.7). The previously identified optimal density at D6 is consistent with this band. For Design D6, the per capita water demand may vary from 172 L/p/d to 204 L/p/d with the mean value of 188 L/p/d. In another way, to achieve the same average water demand of 188.4 L/p/d, the net residential density can vary from 224 to 316 persons/ha. This range of 224 to 316 persons/ha can be considered as an optimal density range as the characteristic curves of per capita water demand and the ecological footprint are similar.



Note: D1 is not shown here as its water demand is very high than others

Figure 6.7 Water demand variability (5th and 95th percentiles) in different residential densities

ii) Sensitivity analysis: Sensitivity analysis was conducted to study the effects of the variation of input parameters on the final output residential water demand. The parameters with the highest relative effects are considered to be the most sensitive input parameters. A reduction in the level of uncertainty (i.e., reducing variance) of the most sensitive parameters would contribute to reduce the largest amount of overall uncertainty in the results (Hammonds et al. 1994). The Monte Carlo simulations for probabilistic water demand were used for sensitivity analysis. The results are almost similar in all 11 neighbourhood designs. The most sensitive input parameters in all designs are indoor water use rate and dwelling occupancy. The effects of indoor water use rate and dwelling occupancy on total residential water use vary from -11% to 11% and -9% to 11% respectively. The effects of input parameters are not high, which may be due to less variation considered in the input parameters. However, the result provides a relative sensitivity of various inputs.

6.3.1.6 Two-dimensional analysis for WEC nexus scenarios

The increase in residential density reduces per capita water demand to meet a given lot coverage requirement. Alternatively, increasing lot coverage requirement also reduces per capita water demand for a given density, indicating two-dimensional nature of the WEC nexus. The change in bylaws on lot coverage and/or application of xeriscaping would result in a change in water demand. A scenario analysis was performed by considering four scenarios for the same design series (D1-D11). Scenario S1 is with an existing lot coverage bylaw for all 11 designs. Scenarios S2 and S3 are with the modification of bylaw on lot coverage, whereas Scenario S4 is an application of xeriscaping in all 11 designs as shown in Table 6.2. Xeriscaping is low water-use landscaping in place of traditional turf (Sovocool et al. 2006). A xeriscaping of 15% of turf and 85% of water conserving species was designed in a typical SF building lawn of the neighbourhood and its detail is given in Appendix C.4

Table 6.2 Scenario features

Scenario	Lot coverage in residence		Remarks
	Single-family	Multi-family*	
S1	348 m^2 (10 - 40%)	60%	Existing neighbourhood plan and lot coverage bylaw
S2	40%	60%	Considered the existing maximum lot coverage guideline as an average coverage to be required
S3	70%	80%	Bylaw on lot coverage changed from maximum 40% to average 70% in SF and maximum 60% to average 80% in MF buildings
S4	348 m^2 (10 - 40%)	60%	Xeriscaping in residential landscaping in Scenario S1

* Medium density MF buildings

The results of two-dimensional analysis of the WEC nexus are shown in Figure 6.8. The results of scenario analysis show that the reduction in per capita EF (aggregated WEC nexus) of water distribution and landscaping was highest in Scenario S4 (xeriscaping) among four scenarios and Design D1 among all designs. In Scenario S4, the reduction in per capita EF would range from below 1% in Design D11 to 66% (0.02gha/p/yr) in Design D1 compared to Scenario S1. The range of reduction would be less than 1% to 34% in Scenario S3 and less than 1% to 13% in

Scenario S2. Moreover, the reduction of per capita EF of Design D5 would be about 1%, 5%, and 15% in Scenarios S2, S3, and S4 respectively.

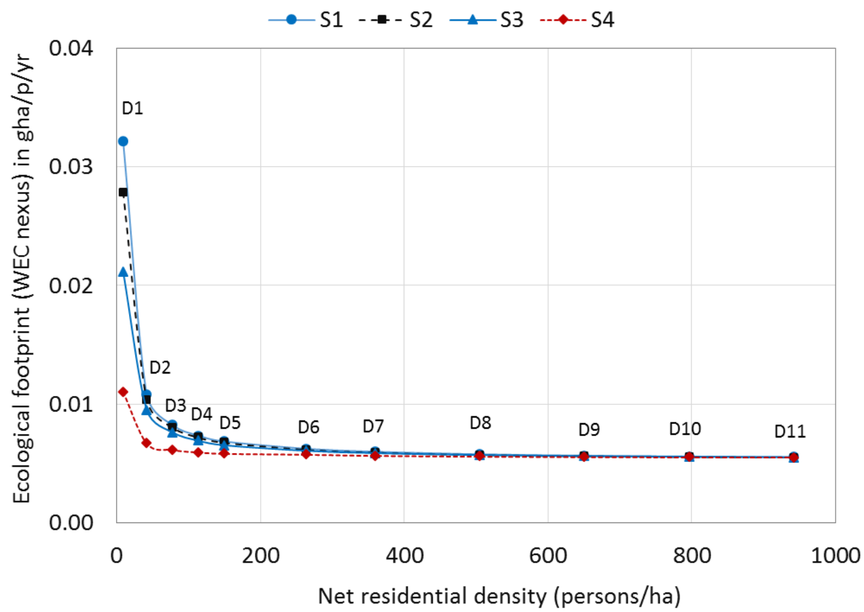


Figure 6.8 Two-dimensional WEC nexus: Varying scenario results in different densities

6.4 Discussion

Densification of neighbourhoods is generally preferred for sustainable communities, including sustainable water systems (EarthCraft 2014; USGBC 2013). This process will affect the WEC nexus of water distribution and residential landscaping. In this study, a planned neighbourhood with different alternative designs with varying residential densities was considered for the WEC nexus analysis. Although the variations of lot coverage, landscaping coverage, and population with net residential density are linear, the variation of per capita EF (i.e., WEC nexus) with net residential density has a power relationship. The power relationship is due to the decrease of per capita share of landscaping in MF residences coupled with the decrease of landscaping coverage requirements for high density MF buildings. The power relationships of all these parameters with density have a point or zone of inflection, which provides an optimal density.

The net residential density of around D6, i.e., 264 persons/ha or 106 units per ha (units/ha) or gross residential density of 170 persons/ha can be considered as an optimal density based on the per capita EF. It may vary from 224 to 316 persons/ha or 90 to 126 units/ha. LEED-ND and ECC also recommended a residential density above 25 units/ha for a compact and sustainable neighbourhood (EarthCraft 2014; USGBC 2013). The identified optimal density is lower than the highest densities of Canada and the US. Some of the highest densities of Canada are a gross residential density of 262 persons/ha in Blocks M4Y (Toronto), 212 persons/ha in V6E (Vancouver), and 204 persons/ha in M4X (Toronto) (Urban Toronto 2014) and that of the US are 386 persons/ha in New York, 363 persons/ha in Los Angeles and 297 persons/ha in Miami (Malouff 2013). The estimated optimal density in the present neighbourhood is lower than in other *highest densities* neighbourhoods, which may be due to lesser number of stories in the present neighbourhood buildings.

The reduction in per capita water demand and energy use from 8.7 persons/ha (D1) to 942 persons/ha (D11) (3.5 to 377 units/ha) was 94% and 97% respectively. However, Filion (2008) found a reduction of only 10% energy use in water distribution by increasing residential density from 4 to 110 units/ha. A high reduction in the present study is mainly due to the consideration of increasing lot coverage with increasing density as per the bylaw and also a wide range (108-fold) of density considered. This fact is also supported by the results of scenario analysis, in which the change of lot coverage only in Scenario S3 (lot coverage of 70% in SF and 80% in MF building), the per capita energy use reduction of 16% could be achieved in Design D1. On the other hand, Filion (2008) considered a constant per capita water demand, which deviates highly from an actual condition.

The reduction in per capita energy related carbon emissions is equivalent to the reduction in energy use, i.e., 97%. However, per capita carbon sequestration was reduced by 100% from Design D1 to D11 as D11 lacks landscaping. The per capita negative net carbon emissions or positive net carbon sequestration was reduced by about 99% from Designs D1 to D10 and became net carbon emitter in Design D11. This study has considered only the carbon sequestration benefit of landscaping besides its other benefits such as physical and mental health, economic benefits, and biodiversity (Kabisch et al. 2015). Similarly, the per capita EF was decreased by 93% from 8.7 persons/ha (D1) to 942 persons/ha (D11) and this high reduction is attributed to water component, which dominates the ecological footprint of WEC nexus. In addition, the characteristic curve of per capita LCC of WDSs is also similar with that of the WEC nexus. This means the WEC nexus-based optimal density is supported by the LCC of

WDSs. The findings show that residential density plays an important role in per capita water demand, energy use, net carbon emissions, and also LCC of WDSs. Higher the residential density, lower the per capita water demand, energy use, carbon emissions, and LCC. The reduction in per capita LCC of a WDS in dense neighbourhoods is also revealed by Speir and Stephenson (2002) although the study has mentioned that a major reduction would be in water treatment cost. The reduction of 97% of LCC from D1 to D11 is higher than that of 66% as mentioned by Duncan (1989), most probably due to the consideration of wide variation of density (108-fold increment) in this study.

The lot coverage requirements imposed by municipalities affect the landscaping size as the land areas not covered by buildings need to be landscaped (District of Peachland 2014a). This ultimately affects per capita EF. The two-dimensional WEC nexus scenario analysis shows that the reduction in per capita EF is more significant in low density housing as they are composed mainly of SF residences that have higher landscaping requirements. Specifically, the per capita EF was highly reduced in Scenario S4 (xeriscaping) than S3 (lot coverage of 70% for SF and 80% for MF buildings). The xeriscaping would reduce per capita water demand, energy use, and considerable amount of carbon sequestration (~30% reduction in soil organic carbon per unit of landscape area). Xeriscaping can save a high amount of water, such as up to 54% (Gleick et al. 2003) and 76% of irrigation demand (Sovocool et al. 2006). The estimated water saving of 51% of irrigation demand in xeriscaping in this study is comparable with Gleick et al. (2003) and Sovocool et al. (2006). The reduced water demand will also save the energy use in water distribution and upstream energy.

The present study included the impacts of neighbourhood densification on water distribution and residential landscaping system only in terms of water, energy, carbon emissions, carbon sequestration, and water distribution cost. Densification may also affect other neighbourhood elements, such as transportation, open space, etc., which are not considered in this research. Furthermore, this study was conducted in a medium-sized neighbourhood of approximately 51 ha with about 5,000 population. The developed characteristic curves may be site and size dependent, but the methodology is well applicable. The study can be extended by increasing the dimensions of the system, i.e., scale of economies in various neighbourhood configurations.

6.5 Summary

Neighbourhood densification is a strategy primarily applied to reduce per capita infrastructure and land requirement. In particular, densification alters residential landscaping that in turn affects water distribution systems. An integrated study of the water-energy-carbon (WEC) dynamics of water distribution and residential landscaping under neighbourhood densification is lacking in the published literature. A conceptual framework was developed and applied as a case study to a planned neighbourhood in the Okanagan Valley (BC, Canada). For this neighbourhood, 11 alternative designs with varying combinations of single-family and multi-family lots representing different residential densities were investigated. Water consumption, energy use, and net carbon emissions by water distribution and residential landscaping system were combined and represented by ecological footprint. The results show that per capita ecological footprint has a power relationship with net residential density despite of a linear relationship between population and net residential density. The power relationship reveals a high dependency of per capita ecological footprint on residential density, which helps to identify an optimal density. Two-dimensional analysis of the WEC nexus scenarios indicates that xeriscaping can reduce per capita ecological footprint ranging from roughly 1% reduction in high density to 66% in low density neighbourhood. Also, the effects of xeriscaping on the WEC nexus are highly density dependent. The results emphasize the importance of amending relevant policies for constructing medium to high-density buildings in urban neighbourhoods to achieve an optimal WEC nexus.

Chapter 7 Development of Microbial Water Quality Guidelines for Reclaimed Water

A version of this chapter has been published in the *Science of the Total Environment* journal with a title " Probabilistic risk-based investigation on microbial quality of reclaimed water for urban reuses" (Chhipi-Shrestha et al. 2017d).

7.1 Background

Canada has abundant freshwater supplies and is one of the water richest countries in the world based on per capita water availability (Asano et al. 2007; WRI 2001). However, there is a large regional disparity in water availability. The annual precipitation of Canada is approximately 600 mm, ranging from 100 mm in the high Arctic to over 3500 mm along the Pacific Coast. Many agricultural lands in the Prairies and British Columbia (BC) interior receive an average annual precipitation of 300 to 500 mm (Schaefer et al. 2004). In 1994-1999, about 26% of municipalities with water supply systems experienced water shortage due to droughts, deteriorating infrastructure, and increased consumption (Environment Canada 2004). In addition, the water and wastewater infrastructure conditions are anticipated to decline in the future due to inadequate reinvestment (Canadian Infrastructure Report Card 2016).

Reclaimed water use is an option to increase water supply. Public perception plays an important role in reclaimed water use. A Canada-wide survey on the public perception on reclaimed water use was conducted by Dupont (2013). The survey results show that at least 80% or more of people are willing to use reclaimed water for toilet flushing and irrigating garden grass and flowers, public parks, and golf courses. In addition, for the irrigation of agricultural crops and garden vegetables respectively 75% and 64% of people are willing to use reclaimed water. Moreover, they are willing to pay an additional annual amount of $142 to $155 per household for using reclaimed water to avoid water restrictions. The willingness to pay is approximately an additional 33% to their annual water bills. The results are consistent with another study on public attitudes on reclaimed water use in several cities in the Lake Simcoe Region in Ontario (LSRCA

2010). This public willingness shows that water reuse has a large potential in Canada in non-potable urban purposes.

At the federal level, Canada has the national plumbing code with an installation guide: *Design and installation of non-potable water systems/maintenance and field testing of non-potable Water Systems* (Canadian Standards Association 2011) and a treatment guide: *Performance of non-potable water reuse systems* (Canadian Standards Association 2012). The treatment guidelines have been prescribed for very small water use systems with a capacity of 10,000 L/d or less and does not cover custom-engineered systems (AEDA 2013; Canadian Standards Association 2012). In addition, Canada has reclaimed water quality guidelines at the federal level: *Canadian guidelines for domestic reclaimed water for use in toilet and urinal flushing* (Health Canada 2010). The federal guidelines are prescribed only for toilet and urinal flushing. The federal government has a long-term goal to develop reclaimed water use guidelines for many beneficial purposes besides toilet and urinal flushing (Health Canada 2010).

At the provincial level, BC promulgated Municipal Wastewater Regulation (MWR) in 2012, which is a holistic legislation for reclaimed water applications in non-potable and potable uses. The regulation has proposed guidelines for broad water reuse classes (MWR 2012): a) *Indirect potable reuse*, b) *High exposure potential* (e.g., agricultural and lawn irrigation, toilet flushing, etc.), c) *Moderate exposure potential* (e.g., commercially processed agricultural crop irrigation, pasture, nurseries, etc.), and d) *Low exposure potential* (e.g., industrial process water, dust control, concrete production, etc.). The provincial approach is different from the federal approach that has prescribed guidelines for specific water reuse applications, e.g., toilet and urinal flushing. Moreover, the trend of developing risk-based guidelines on reclaimed water use has increased in several countries, such as Australia (EPHC/NHMRC/NRMMC 2006, 2008), the US (US EPA 2012a), Canada (Health Canada 2010) including the province of Alberta (Canada) (WaterSMART Solutions 2015), and in the WHO (WHO 2006a).

Based on the long-term goal of the Canadian federal government, the recommendations of WHO, existing urban water shortage, and public willingness for water reuse, further research is required for investigating and developing reclaimed water use guidelines for specific reuses in non-potable purposes besides toilet and urinal flushing. Furthermore, a probabilistic approach can be applied to analyze uncertainty in risk estimate. This chapter investigates the risk-based guideline values for microbial quality of reclaimed water in non-potable urban reuses with a case study in the Okanagan Valley, BC.

7.2 Methodology

The microbial water quality of reclaimed water for urban reuses was investigated and guideline values were proposed by using the research framework given in Figure 7.1. The framework involve quantitative microbial risk assessment (QMRA) and the application of the guideline values as a case study.

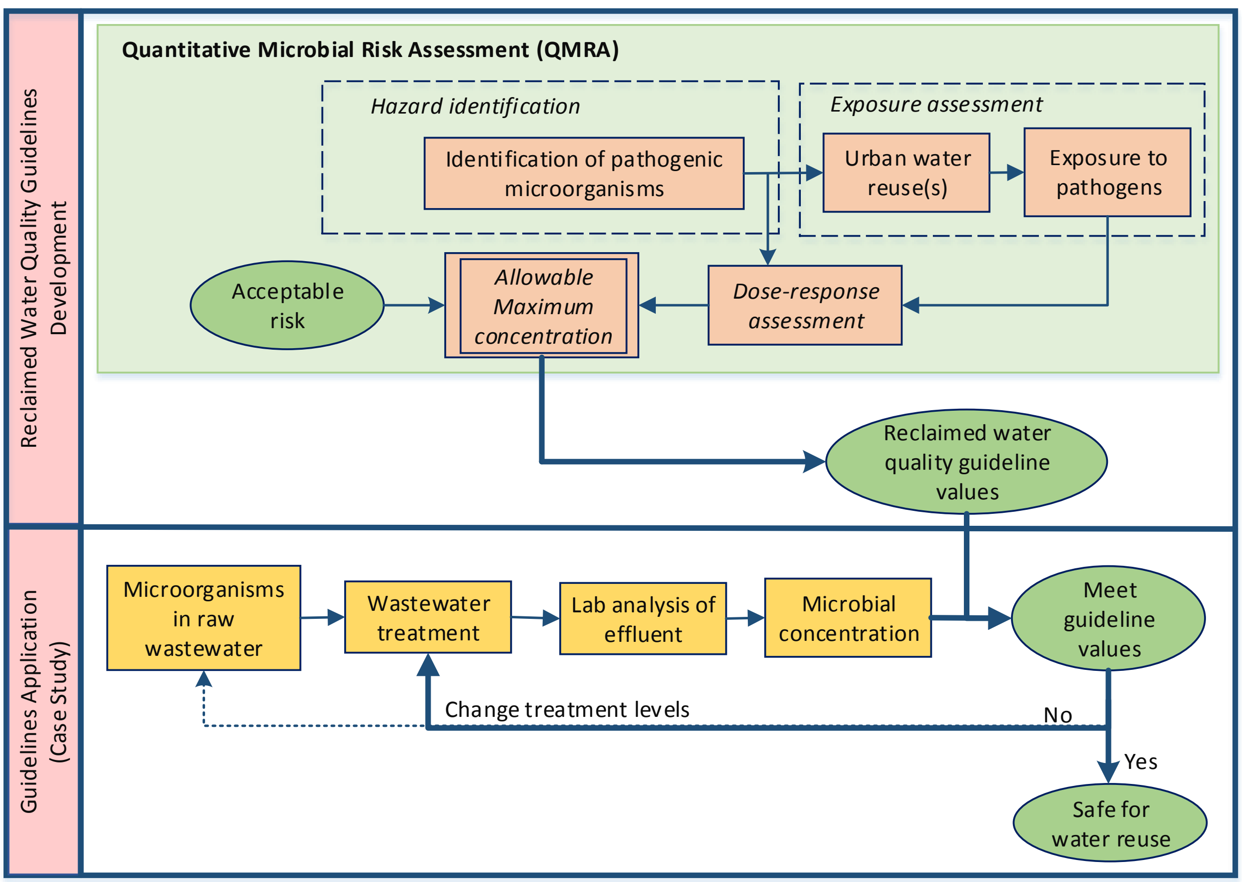


Figure 7.1 Research framework for developing and applying microbial water quality guidelines for reclaimed water

7.2.1 Quantitative microbial risk assessment

The health risk of reclaimed water due to pathogenic microorganisms was estimated by using QMRA (Haas et al. 2014). The QMRA includes four steps - hazard identification, exposure assessment, dose-response assessment, and risk characterization (Haas et al., 1999; Haas, 2002; Environment Canada/Health Canada, 2013). The propagation of variability and uncertainty in risk estimation was represented by using the two-dimensional Monte Carlo technique (Lim et al., 2015; Pouillot et al., 2016; US EPA, 2001). The QMRA steps are elaborated as follows:

7.2.2 Hazard identification

The major groups of wastewater pathogens are bacteria, viruses, and protozoans (Haas et al. 2014). In these groups, the pathogenic microorganisms were selected for risk assessment based on the indicator organism, adequacy of literature on the organism, the occurrence of water borne illness, and diseases as reported by health authorities (Katukiza et al. 2014). The selected pathogens were *Escherichia coli* O157:H7, *Salmonella* spp., and *Campylobacter jejuni* from bacteria; rotavirus, adenovirus, and norovirus from viruses, and *Cryptosporidium parvum* and *Giardia* spp. from protozoa. These microorganisms were the most relevant pathogens for risk assessment. All the selected pathogens cause gastrointestinal illness (US EPA 2010; WHO 2006b). Among these microorganisms, *E. coli* is the best available indicator because it does not usually multiply in the environment, is easily detectable even in high dilution due to its excretion in the faeces in large numbers (approximately 10^9 cells per gram), and has a life span on the same order of magnitude as those of other enteric bacterial pathogens (Health Canada 2013a). The indicator *E. coli* was used for the development of microbial water quality guideline values (Health Canada 2010, 2013a), whereas all the microorganisms were used for risk assessment in a case study.

7.2.3 Exposure assessment

Reclaimed water can be used for various urban purposes based on its quality and intended applications. The unit exposure volume of water in a reuse application and its annual application frequency are given in Table 7.1. Due to the lack of distribution data, a uniform distribution was assumed for most of the exposures similar to that of Mok et al. (2014) and Verbyla et al. (2016) with ±10% variation except for which a data range is available: “golf frequency” and “duration from crop irrigation to consumption", and a triangular distribution was assumed for "log reduction in natural die off".

Table 7.1 Exposure factors for different urban water uses

Reuse Applications	Exposure	Volume (mL)	Frequency/year	Source
Garden irrigation	Inhalation (aerosol)	Unif. (0.09, 0.11)	Unif. (81, 99)	1
Garden irrigation	Ingestion (Plant contact)	Unif. (0.9, 1.1)	Unif. (81, 99)	1
Garden irrigation	Ingestion (accidental)	Unif. (90, 110)	Unif. (0.9, 1.1	1
Public parks	Ingestion (Plant contact)	Unif. (0.9, 1.1)	Unif. (45, 55)	1
Golf courses	Ingestion (Plant contact)	Unif. (0.9, 1.1)	Unif. (26, 40)	1 & 2
Food crop lettuce	Ingestion	Unif. (4.5, 5.5)	Unif. (63, 77)	1
Other raw produce (commercial)	Ingestion	Unif. (0.9, 1.1)	Unif. (126, 154)	1
Fruits consumption (commercial)	Ingestion	Unif. (1.8, 2.2)	Unif. (243, 297)	1
Duration (field to consumption)	-	Unif. (1, 5) days for fish; Unif. (0.9, 1.1) day for lettuce, other raw produce & fruits		1
Fisheries	Ingestion	Unif. (2.7, 3.3)	Unif. (22.5, 27.5)	1
Toilet flushing	Inhalation (aerosol)	Unif. (0.009, 0.011)	Unif. (990, 1210)	1
Car washing	Ingestion, inhalation	Unif. (18, 22)	Unif. (45, 55)	1
Laundry machine use	Inhalation (aerosol)	Unif. (0.009, 0.011)	Unif. (90, 110)	1
Cross-connection by dual reticulation systems	Ingestion	Unif. (900, 1100)	Unif. (0.00028, 0.00034)*365	1
Fire fighting	Ingestion, inhalation	Unif. (18, 22)	Unif. (45, 55)	1
Log reduction in natural die off (D)*	-	Triang. (0.5, 0.5, 1) for cool weather		3
Log reduction in cleaning (D)*	-	Unif. (0.9, 1.1)		1
Log reduction in cooking (D)* (fisheries)	-	Unif. (4.95, 6.05)		1

Note: Unif. means a uniform distribution with minimum and maximum values in parenthesis; Triang. means a triangular distribution with minimum, most likely and maximum values in parenthesis; *Microbial reduction rate is calculated as 10^{-D}/day (WHO 2006b)

Source: 1: EPHC/NHMRC/NRMMC (2008) 2: NAGA (2012) 3: WHO (2006b)

Altogether 12 potential applications of reclaimed water were considered for non-potable urban purposes as follows:

a. Lawn (L) irrigation
b. Public park (P) irrigation
c. Golf course (G) irrigation
d. Agricultural (A) irrigation (raw eaten crops)*
e. P, L & G irrigation
f. P, L, G, & A irrigation

g. Toilet & urinal (T&U) flushing
h. Vehicle washing
i. Laundry machine
j. Firefighting
k. T&U flushing & laundry machine
l. Non-potable urban uses (all above reuses)

Note *agriculture in urban semi-urban areas

For the identified reuse types, annual exposure volumes were estimated by using the unit exposure and annual frequency (Table 7.1) employing Monte Carlo simulations.

7.2.4 Dose-response assessment

Dose-response models were used to estimate the probability of infection, which depend on incubation period (Haas et al. 1999; Katukiza et al. 2014). These models are specific to a microbial species. A Beta-Poisson model was used for *E. coli* (Health Canada 2010), *Campylobacter jejuni* (Haas et al. 1999; Katukiza et al. 2014), *Salmonella* spp. (Haas et al. 1999), and rotavirus (Health Canada 2010; Prez et al. 2015) with species-specific parameter values. Similarly, an exponential model was used for adenovirus (Katukiza et al. 2014; Lim et al. 2015; Vergara et al. 2016), norovirus (Messner et al. 2014; Schmidt 2015), and *Cryptosporidium parvum* and *Giardia* spp. (Robertson et al. 2005) with species-specific parameter values. The models and the annual risk estimation equation are as follows (Katukiza et al. 2014):

a) Beta-Poisson dose-response model

$$P_{inf}(d) = 1 - [1 + \left(\frac{d}{N_{50}}\right)\left(2^{\frac{1}{\alpha}} - 1\right)]^{-\alpha} \qquad \textbf{Equation 7.1}$$

b) Exponential dose-response model

$$P_{inf}(d) = 1 - \exp(-rd) \qquad \textbf{Equation 7.2}$$

c) Annual risk of infection

$$P_{inf(A)}(d) = 1 - [1 - P_{inf}(d)]^n \qquad \textbf{Equation 7.3}$$

where P_{inf} (d) refers to the probability or risk of infection to an individual exposed to a single pathogen dose "d"; d is the pathogen dose; $P_{inf(A)}$ (d) is estimated annual probability or risk of an infection from "n" exposures per year due to a single pathogen dose "d"; "α" and "r" are parameters referring to pathogen infectivity constant which characterize dose-response relationships; N_{50} is the median infective dose, i.e., the dose required to infect 50% of the exposed population. The parameter values of the dose-response models of different pathogens are given in Table 7.2.

Table 7.2 Parameter values of dose-response models

Pathogens	**Beta-Poisson**		**Exponential**	**Source**
	α	**N_{50}**	**r**	
E. coli O157:H7	0.2019	1120	-	Health Canada (2010)
Campylobacter jejuni	0.145	8.96E+02	-	Haas et al. (1999); Katukiza et al. (2014)
Salmonella spp.	0.3126	2.36E+04	-	Haas et al. (1999)
Adenovirus	-	-	0.4172	Katukiza et al. (2014); Lim et al. (2015); Vergara et al. (2016)
Norovirus	-	-	0.722	Messner et al. (2014); Schmidt (2015)
Rotavirus	0.27	5.6	-	Health Canada (2010)
Cryptosporidium parvum	-	-	0.004	Robertson et al. (2005)
Giardia spp.	-	-	0.0199	Robertson et al. (2005)

7.2.5 Risk characterization

Risk characterization was carried out by integrating hazard identification, exposure assessment, and dose-response assessment. Risk characterization results in the determination of a health outcome, such as the risk of infection, illness, and mortality. The final risk was expressed in disease burden, i.e., Disability-Adjusted Life-Years (DALYs) per year. The DALY is a common term to represent health impacts by death and unhealthy life periods. DALY was calculated by using Equations 7.4 and 7.5 (Howard et al. 2006).

$$Risk\ of\ disease\ (P_{ill}) = P_{inf(A)}(d) * P_{ill|inf} \qquad \textbf{Equation 7.4}$$

$$DALY = P_{ill} * DBPC * f_s \qquad \textbf{Equation 7.5}$$

where $P_{ill|inf}$ is risk of disease given infection, i.e., morbidity; DBPC is disease burden per case (DALY/year); and f_s is susceptibility fraction. The values of these parameters were obtained from literature as given in Table 7.3.

Table 7.3 Morbidity, disease burden per case and susceptibility fraction

Pathogens	Morbidity ($P_{ill\|inf}$)	Maximum disease burden (DALY/yr)	Susceptibility fraction (fs)
E Coli O157:H7	Unif. (0.2, 0.6) (US EPA 2010)	Unif. (0.0495, 0.0605)* (Health Canada 2010)	Unif. (0.8, 1) (Mok et al. 2014)
Campylobacter jejuni	Unif. (0.1, 0.6) (US EPA 2010)	Unif. (0.002, 0.0047) (Gibney et al. 2014)	Unif. (0.8, 1) (Mok et al. 2014)
Salmonella spp.	Unif. (0.18, 0.22)* (US EPA 2010)	Unif. (0.0318, 0.0574) (Gibney et al. 2014)	Unif. (0.8, 1) (Mok et al. 2014)
Adenovirus	Unif. (0.45, 0.55)* (Crabtree et al. 1997)	Unif. (0.0481, 0.0587)* (Health Canada 2010)	Unif. (0.8, 1) (Mok et al. 2014)
Norovirus	Unif. (0.3,0.8) (US EPA 2010)	Unif. (0.0004, 0.0008) (Gibney et al. 2014)	Unif. (0.8, 1) (Mok et al. 2014)
Rotavirus	Unif. (0.61, 0.73)* (US EPA 2010)	Unif. (0.0076, 0.0092)* (Health Canada 2011)	Unif. (0.05, 0.07) (Mok et al. 2014) (Health Canada 2010)
Cryptosporidium parvum	Unif. (0.2, 0.7) (US EPA 2010) (Health Canada 2010)	Unif. (0.0011, 0.0028) (Gibney et al. 2014)	Unif. (0.8, 1) (Mok et al. 2014)
Giardia spp.	Unif. (0.2, 0.7) (US EPA 2010)	Unif. (0.0015, 0.003) (Gibney et al. 2014)	Unif. (0.8, 1) (Mok et al. 2014)

Note: Unif. means a uniform distribution with minimum and maximum values in parenthesis; * considered ± 10% variation to estimate a range

For the development of guideline values in this research, reverse QMRA was applied. The target risk was considered to be 10^{-6} DALYs/year (WHO 2006b) and Equation 7.6 was used to estimate the equivalent concentration of *E. coli.*

$$Dose\ equivalent = \frac{Target\ risk}{Disease\ burden\ per\ case * P_{inf} * Morbidity * Susceptibiity\ fraction} \qquad \textbf{Equation 7.6}$$

7.2.6 Data variability and uncertainty

Quantitative risk assessment should reflect the variability in the risk and take into account the uncertainty associated with the risk estimate (Pouillot et al. 2016; US EPA 2001). The variability in QMRA, also called aleatoric uncertainty, represents the temporal and individual heterogeneity of the risk for a given population. The uncertainty in QMRA, also called epistemic uncertainty, stems from imperfect knowledge about the QMRA model structure and the associated parameters (Pouillot et al. 2016). A two-dimensional (or second-order) Monte Carlo Analysis (2-D MCA) was used to characterize variability and uncertainty in input variables. A 2-D MCA is a Monte Carlo analysis where the distributions reflecting variability and the distributions representing uncertainty are sampled separately in the simulation so that variability and uncertainty in the output may be assessed separately (Pouillot et al. 2016). In this analysis, the input parameters considered were: "exposure factors" as variable parameters, "pathogenic *E. coli* ratio" as an uncertain parameter, and "morbidity", "disease burden per case", and "susceptibility fraction" as variable and uncertain parameters. The 2-D MCA simulations with 10,000 iterations for the inner loop (variability) and 5000 iterations for the outer loop (uncertainty) were performed to make the risk estimates reliable (Ashbolt et al. 2010; Katukiza et al. 2014; Pavione et al. 2013; US EPA 2001) by using the R software (Pouillot et al. 2016). The 2-D MCA produced cumulative density functions (CDFs) of microbial concentration at different quantiles (e.g., 5%, 25%, median, 75%, and 95%). The measures of variability and uncertainty were estimated by using three ratios as shown in Equations 7.7 to 7.9 (Ozkaynak et al. 2009; Pouillot et al. 2016).

$$Variability\ ratio = \frac{B}{A} \qquad \textbf{Equation 7.7}$$

$$Uncertainty\ ratio = \frac{C}{A} \qquad \textbf{Equation 7.8}$$

$$Overall\ uncertainty\ ratio = \frac{D}{A} \qquad \textbf{Equation 7.9}$$

where A is the median of uncertainty (in 50% CDF) for the median of variability, B is the median of uncertainty (in 50% CDF) for the 97.5^{th} percentile of variability, C is the 97.5^{th}

percentile of uncertainty (in 97.5% CDF) for the median percentile of variability, and D is the 97.5^{th} percentile of uncertainty (in 97.5% CDF) for the 97.5^{th} percentile of variability.

7.3 Results

7.3.1 Microbial water quality investigation and guideline values

The microbial concentrations of reclaimed water for different reuses were estimated and their CDFs, for different intended uses, are shown in Figure 7.2. The figure shows a wide variation in uncertainty (along the x-axis) and variability (along the y-axis) in the concentration estimate. However, the variability, uncertainty, and overall uncertainty ratios in all water reuse types are similar as also seen from the identical shape of CDFs in Figure 7.2. In these water reuse types, variability ratios vary from 1.96 to 2.06, uncertainty ratios range from 2.31 to 2.34, and overall uncertainty ratios vary from 4.54 to 4.81. Moreover, in this study, the 95^{th} percentile is considered as the Reasonable Maximum Estimate (RME) as used by the US EPA (2001). For example, in lawn irrigation as shown in Figure 7.2 (Plot 1), the median of 0.06 cfu/100 mL at the 50^{th} percentile CDF (i.e., horizontal arrow) could range from 0.04 cfu/100 mL (5^{th} percentile CDF) to 0.13 cfu/100 mL (95^{th} percentile CDF) due to uncertainty. Similarly, the median of 0.06 cfu/100 mL (i.e., vertical arrows) could range from 0.04 cfu/100 mL (5% at median CDF) to 0.11 cfu/100 mL (95% at median CDF) due to variability. The uncertainty range of 0.04 to 0.13 cfu/100 mL is larger than variability range of 0.04 to 0.11 cfu/100 mL, which match with the higher values of uncertainty ratios compared to variability ratios.

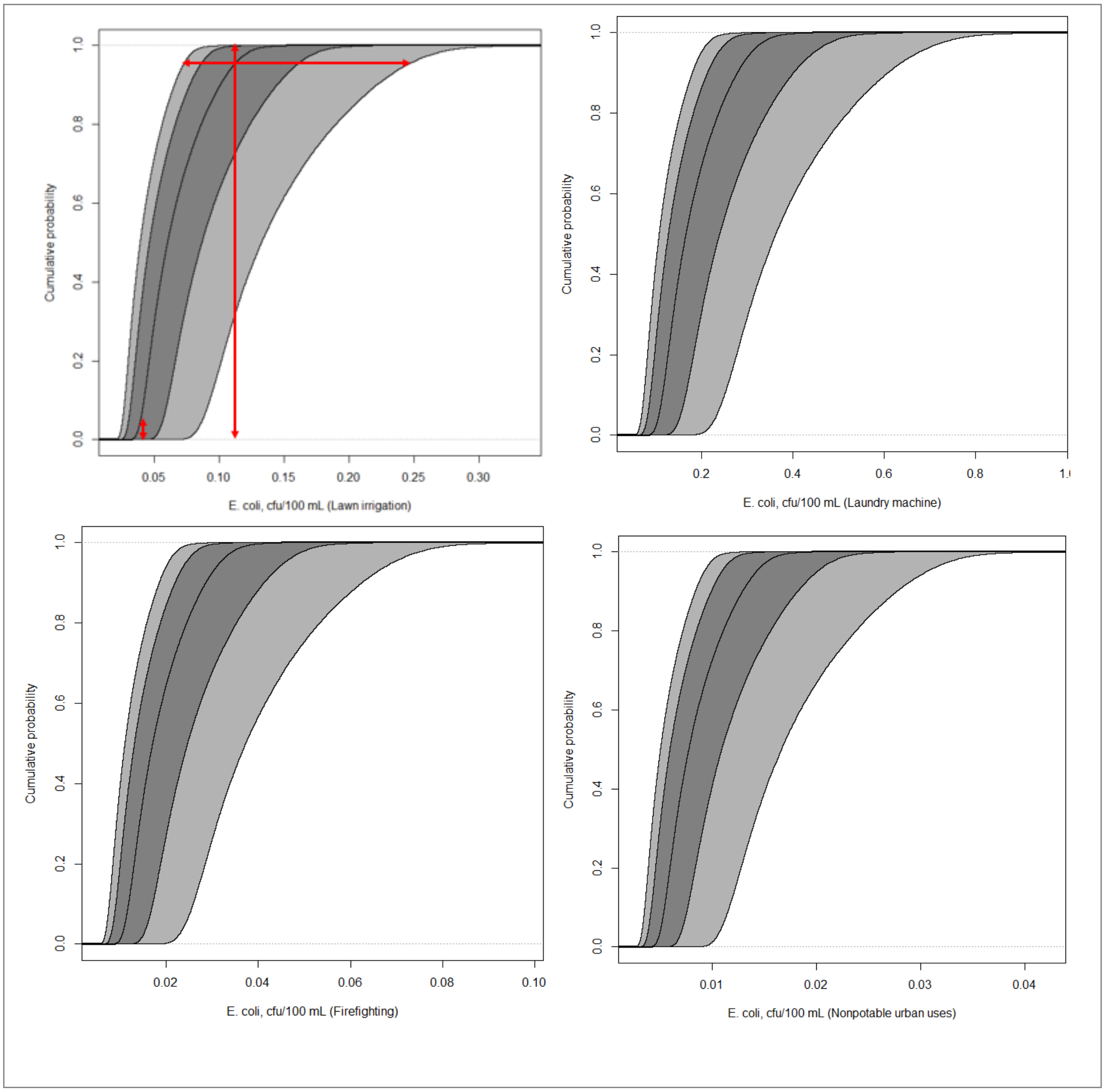


Figure 7.2 CDFs of *E. coli* concentrations with acceptable risks in different water reuse applications

Note: Five curves in each plot represent the 5th, 25th, median, 75th, and 95th percentile CDF from left to right respectively. The horizontal red arrow in the first plot represents a 90% confidence interval of the RME of 0.11 cfu/100 mL. Similar plots were drawn for all water reuses and not shown here due to space constraints

The RME of *E. coli* with a 90% confidence interval, i.e., the 5^{th} to 95^{th} percentiles for all water reuse types were extracted from their CDFs (Figure 7.2) and then plotted in Figure 7.3. The 90% confidence intervals are wider for all applications indicating a large uncertainty. The individual RME values of *E. coli* seem to be smaller; however, they provide a relative magnitude of potential risks in various urban reuse applications.

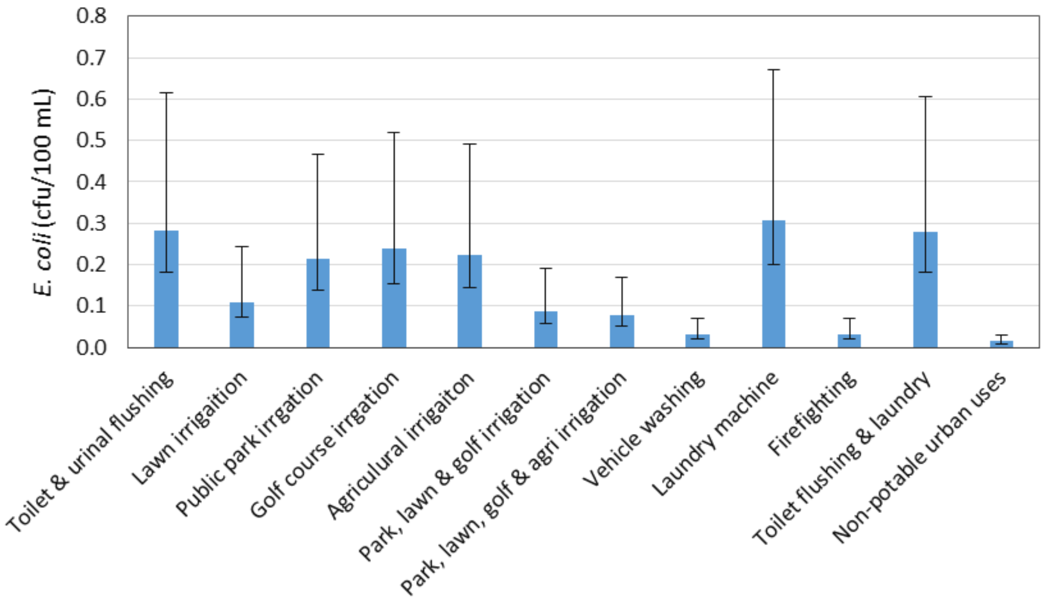


Figure 7.3 RME with 90% confidence interval of *E. coli* concentration for acceptable risk

The microbial water quality of reclaimed water use has been proposed in terms of the median and maximum value. The proposed median is based on the median estimate of the median CDF, whereas the proposed maximum value is based on the RME of the 95^{th} percentile CDF (Figure 7.2) and given in Table 7.4. The RMEs were initially expressed in terms of the presence (presence/absence) of *E. coli* and then their final equivalent concentrations were estimated in five samples. A minimum sample size of five with at least an hour apart was considered the same as that of the federal guidelines (Health Canada 2010) and BC regulation (MWR 2012).

Table 7.4 Microbial water quality guidelines for different reuses (*E. coli* in cfu/100 mL)

Water reuses	50% CDF-based		95% CDF-based		Guideline Values	
	Med value	RME	*E. coli* presence	For 5 samples	Med	Max
Lawn (L) irrigation	0.06	0.24	1/4	1.2/5	ND	≤ 1
Public park (P) irrigation	0.12	0.47	1/2	2.3/5	ND	≤ 2
Golf course (G) irrigation	0.13	0.52	1/2	2.6/5	ND	≤ 2
Agricultural (A) irrigation	0.12	0.49	1/2	2.5/5	ND	≤ 2
P, L & G irrigation	0.05	0.19	1/5	1.0/5	ND	≤ 1
P, L, G, & A irrigation	0.04	0.17	1/6	0.8/5	ND	< 1
Toilet & urinal flushing	0.15	0.62	1/2	3.1/5	#	#
Vehicle washing	0.02	0.07	1/15	0.3/5	ND	< 1
Laundry machine	0.17	0.67	1/2	3.4/5	ND	≤ 3
Firefighting	0.02	0.07	1/15	0.3/5	ND	< 1
T&U flushing & laundry*	0.15	0.61	1/2	3.0/5	ND	≤ 3
Non-potable urban uses	0.01	0.03	1/33	0.2/5	ND	< 1

*T&U: Toilet & urinal; Med: Median Max: Maximum; ND: Not detected

Not proposed as the federal government has already prescribed the guideline value

The required minimum water quality varies considerably for different reuse applications. Based on Table 7.4, the proposed median is Not Detected (ND), i.e., ~0 cfu/100 mL for all eleven reuse types (except toilet and urinal flushing), whereas the maximum value differs from below 1 to 3. However, the proposed maximum values would be larger if the RME was defined to be higher than the 95th percentile or the RME was estimated from a CDF higher than 95%, or both. All of these twelve reuse types fall under the reclaimed water class *Greater exposure potential* of the MWR (BC Ministry of Environment 2013). By combining the MWR class and the guideline values of this study (Table 7.4), the identified reuse types can be grouped under three categories:

a) *Greater exposure potential I*: Lawn irrigation, any irrigation in combination with lawn watering, vehicle washing, firefighting, non-potable urban uses (collective) – median: ND, maximum: 1 cfu/100 mL
b) *Greater exposure potential II*: Public park irrigation, golf course irrigation, agricultural irrigation – median: ND, maximum: 2 cfu/100 mL
c) *Greater exposure potential III*: Laundry machine and toilet and urinal flushing – median: ND, maximum: 3 cfu/100 mL

7.3.2 Sensitivity analysis

Sensitivity analysis was performed to study the effects of the variation of input parameters on the final output *E. coli* concentration. The parameters with the highest relative effects are considered to be the most sensitive input parameters. A reduction in the level of uncertainty (i.e., reducing variance) of the most sensitive parameters would contribute largely to reduce an overall uncertainty in the results (Hammonds et al. 1994). For sensitivity analysis, the QMRA model was run with Monte Carlo simulations of 50,000 iterations in @Risk®. The results of the analysis are similar for all reuse types and are shown in a tornado diagram for lawn irrigation (Figure 7.4) and in Appendix D.1 for other reuses.

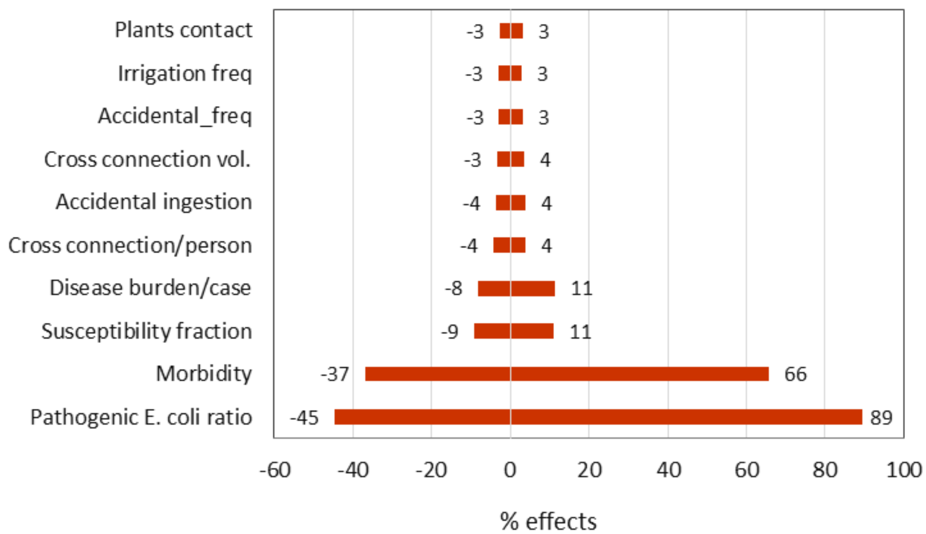


Figure 7.4 Effects of input parameters over their range on output mean in lawn irrigation

The tornado diagram (Figure 7.4) and Appendix D.1 show that the most sensitive input parameters in all water reuse types are the pathogenic *E. coli* ratio and morbidity. The effects of pathogenic *E. coli* ratio on the output are almost similar in all reuses and range from – 44% to 91%. The very high effect of the input parameter is due to the fact that the pathogenic *E. coli* ratio used in the simulations has a wide variation from 2% (Health Canada 2010) to 8 % (Haas et al. 1999; Howard et al. 2006), a four times difference. Similarly, the second most sensitive input parameter is morbidity, which affects the output mean by -38% to 68% in all water reuses. Such variation is a result of a large range of morbidity from 0.2 to 0.6 (US EPA 2010), a three times difference. For all water reuse applications, other common sensitive parameters are susceptibility fraction and disease burden per case. The range of effects on the output mean are -11% to 12% and -10% to 11% respectively for susceptibility fraction and disease burden per case. In fact, the values of these input parameters vary mildly from 0.8 to 1.0 for susceptibility fraction (Mok et al. 2014) and 0.0495 to 0.0605 DALY/year for disease burden per case of *E. coli* (Health Canada 2010).

7.3.3 Required treatment levels

The level of treatment required for the reuse of wastewater in different applications was estimated based on the 50th and 95th percentile CDFs. For this estimation, an initial concentration of *E. coli* of raw wastewater is required, which was obtained from the literature and is given in Appendix D.2. A lognormal distribution was considered for the concentration of *E. coli* (Pavione et al. 2013) with the given range truncated with $\mu \pm 4\sigma$ representing 99.997% of the distribution. The minimum levels of treatment required for treating the wastewater to be used in different reuse applications are given in Table 7.5.

Table 7.5 Log removal required for reclaimed water in different water reuses

Water reuses	Minimum log removal	
	Median	RME
Lawn (L) irrigation	7.1	7.8
Public park (P) irrigation	6.8	7.5
Golf course (G) irrigation	6.8	7.5
Agricultural (A) irrigation	6.8	7.5
P, L & G irrigation	7.2	7.9
P, L, G, & A irrigation	7.3	8.0
Toilet & urinal (T&U) flushing	6.7	7.4
Vehicle washing	7.7	8.3
Laundry machine	6.7	7.4
Firefighting	7.7	8.3
T&U* flushing & laundry	6.7	7.4
Non-potable urban uses	8.0	8.7

As shown in Table 7.5, the level of treatment required would be larger for non-potable urban reuses (combined reuses) with a log removal of 8.0 to 8.7. This reuse type has the highest log removal requirement as it is a collective use of all reuses considered, which has the largest exposure. However, the reuse types toilet and urinal flushing and laundry machine have the lowest log removal requirement of 6.7 to 7.4 as they have the lowest exposure. The estimated log removals are slightly higher than that prescribed by Western Australia and Victoria (Australia). For example, the minimum log removal required for toilet and urinal flushing and laundry is 6.7 to 7.4 and that for agricultural irrigation is 6.8 to7.5 proposed by this study, which is 6.0 by WA DoH (2011) for both reuses and that of non-potable urban reuses is 8.0 to 8.7 by this study and 7.0 by EPA Victoria (2003) and 6.5 by WA DoH (2011). The log removal requirements estimated by this study were found to be higher.

7.3.4 Uncertainty analysis

All the water reuse types considered in this study have a similar degree of variability and uncertainty as indicated by the estimated variability, uncertainty, and overall uncertainty ratios. The uncertainties associated with the input parameters and dose-response models are shown in terms of the credibility of data source, applicability to Canada, knowledge status, and nature of uncertainty in Table 7.6. For most of the input parameters and dose-response models, the knowledge status is low to medium, although the credibility of data source is mostly high and the applicability to Canada is medium to high (Table 7.6).

Table 7.6 Uncertainty in model and input parameters

Inputs/Model	Credibility of data source	Applicability to Canada	Knowledge status	Nature of uncertainty*	Uncertainty analysis#	Remarks
Exposure factors	H	M	M	A	Yes	Australian & Canadian data
Pathogenic *E. coli* ratio	M-H	M	L-M	E	Yes	Different ratios available
Morbidity	H	M-H	L-M	A & E	Yes	US & Canadian data
Disease burden/case	H	M-H	M	A & E	Yes	Australian & Canadian data
Susceptibility fraction	H	M-H	M	A & E	Yes	Australian & Canadian data
Dose-response models	M-H	M	L-M	A & E	No	> 1 dose-response model available for a pathogen
Model parameters	M-H	M	L	A & E	No	> 1 parameter value available for a model

Note: *A- Aleatoric and E- Epistemic # included in this study
High (H) – For credibility: data obtained from a relevant institutional study; for applicability: data produced in Canada or are eographically irrelevant; for knowledge status: universal (or highly developed) model/parameter/data; **Medium (M)** – For credibility: data obtained from a detailed individual study or have ≥ 3 data sources; for applicability: data produced in the North America/developed countries or are geographically less relevant; For knowledge status: moderately developed (or low variation) model/parameter/data; **Low (L)** – For credibility: data obtained from a relatively less detailed study; For applicability: data produced beyond the North America/developed countries & are geographically relevant; For knowledge status: less developed (high variation) model/parameter/data

7.3.5 Application

The proposed guideline values were applied as a case study to the wastewater treatment plants (WWTP) in the Okanagan Valley, BC (Canada) to study the effectiveness of the proposed values. Three WWTPs selected in the Okanagan Valley are Kelowna WWTP, Westside Regional WWTP, and Penticton WWTP. The features of the WWTPs given in Table 7.7.

Table 7.7 Features of wastewater treatment plants

Features	Kelowna WWTP	Westside Regional WWTP	Penticton WWTP
Treatment train	Screen and grit removal, primary sedimentation, BNR[1], cloth disk filtration, & UV[2] disinfection	Screen and grit removal, primary sedimentation, BNR, cloth disk filtration, UV disinfection, & chlorination[3]	Screen and grit removal, primary sedimentation, BNR, cloth disk filtration, ultrafiltration[3], UV disinfection, & chlorination[3]
Population served	117,312 (Statistics Canada 2016)	44,193 (RDCO 2014)	33,160 (City of Penticton 2014a)
Effluent discharge	To lake	Partial reuse for park irrigation	Partial reuse for landscaping irrigation

[1]Biological Nutrient Removal [2]Ultraviolet [3]Only for water reuse

The weekly data on the concentrations of *E. coli* in the treated wastewater in 2012–2014 were obtained from the WWTP annual reports. Specifically, the weekly *E. coli* data were available for the Westside Regional and Penticton WWTPs, whereas the Kelowna WWTP had the data only on fecal coliforms from which *E. coli* concentration was estimated as 95% of fecal coliforms (Howard et al. 2006). The median and maximum concentrations are given in Table 7.8.

Table 7.8 *E. coli* concentration in WWTP effluents in Okanagan from 2012-2014

WWTP (Without chlorination)	Median (cfu/100 mL)	Maximum (cfu/100 mL)	Within Guideline Values
Westside Regional*	1	93	No for both parameters
Kelowna	0	6.7	No for maximum value
Penticton	0	9.2	No for maximum value

*without chlorination and ultrafiltration

The proposed guideline values for public park irrigation is a median of ND *E. coli*/100 mL and maximum value of 2 cfu/100 mL (Table 7.4). Table 7.8 shows that the treated water from all three WWTPs has *E. coli* concentration above the proposed guideline values either for the maximum value or for both median and maximum value. The use of the WWTP effluent directly in public park irrigation would have significant health risks. This was also verified by the estimated human health risk (Table 7.9). The human health risk associated with reclaimed water use in public park irrigation was assessed by considering the lognormal distribution of *E. coli* (Howard et al. 2006). Moreover, the health risk was also estimated for water reuse with additional chlorination to the WWTP effluent and is shown in Table 7.9. For this, the microbial concentration after chlorination was estimated by using microbial removal efficiencies given in Appendix D.3.

Table 7.9 Human health risk of public park irrigation by WWTP effluents

WWTP effluent	Human health risk (RME) with		
	No chlorination	Chlorination (estimated)	Chlorination (actual)
Westside Regional	6.06E-05	1.08E-07	-
Penticton	3.13E-06	2.72E-07	< 1.0E-06
Kelowna	6.59E-06	8.69E-08	-

Table 7.9 shows that the health risk would be significant for the effluent of all three WWTPs without chlorination. However, additional disinfection such as, chlorination would make the WWTP effluent applicable for park irrigation as the estimated risk was lower than the risk benchmark. The actual data of the Penticton WWTP effluent after chlorination also verifies the acceptable risk level.

The *E. coli*-based indicators and pathogens in water have no direct quantitative relationship (O'Toole et al. 2012; Petterson et al. 2016; Sidhu et al. 2012). The health risk associated with other pathogenic microorganisms was also estimated so as to indicate the effectiveness of *E. coli*-based guideline values. Due to the lack of data, the concentrations of other pathogens in the WWTP effluent were estimated from the initial concentration of these pathogens in raw wastewater by log removal method (Lim et al. 2015). All the three WWTPs in the Okanagan Valley have the similar treatment units before the chlorination stage as indicated in Table 7.7. The given concentration range of different microorganisms in raw wastewater and the log removal credits of treatment technologies were obtained from the literature and are given in Appendix D.2 and Appendix D.3 respectively. Similar to *E. coli*, a lognormal distribution was considered for other pathogens (Pavione et al. 2013) with the given range truncated with $\mu \pm 4\sigma$ representing 99.997% of the distribution. A uniform distribution was used for the given range of log removal credits of treatment technologies (Mok et al. 2014). The health risks associated with different microorganisms estimated for effluents before and after chlorination are given in Table 7.10. Some pathogens, such as, *Campylobacter jejuni*, adenovirus, and rotavirus have significant health risk (RME) before chlorination, whereas the health risk will be insignificant after chlorination.

Table 7.10 Health risk of other pathogens by WWTP effluents

Pathogens	Without chlorination		With chlorination	
	Mean (DALY/yr)	RME (DALY/yr)	Mean (DALY/yr)	RME (DALY/yr)
Campylobacter jejuni	2.3E-05	1.0E-04	2.7E-08	9.2E-08
Salmonella spp.	1.1E-06	4.7E-06	1.2E-09	4.1E-09
Adenovirus	8.5E-05	4.0E-04	1.8E-07	8.0E-07
Norovirus	2.4E-08	1.1E-07	4.9E-11	2.0E-10
Rotavirus	8.8E-05	3.2E-04	2.5E-07	9.3E-07
Cryptosporidium parvum	8.3E-10	3.6E-09	2.4E-10	1.1E-09
Giardia spp.	1.2E-07	5.5E-07	3.7E-08	1.6E-07

7.4 Discussion

7.4.1 Microbial water quality guideline values

This study has proposed guideline values for microbial water quality for various specific and collective non-potable urban reuses. The guideline values were developed by applying the commonly used QMRA framework (Health Canada 2010, 2011, 2012, 2013b), although no definite risk assessment methodology has been prescribed for Canada (Dunn et al. 2014). The proposed median value of ~0 cfu/100 mL, i.e., ND for non-potable urban reuse is consistent with the guideline for the *Greater exposure potential class* (non-potable) of the BC Municipal Wastewater Regulation (MWR 2012). Similarly, a guideline value has not been proposed by this study for toilet and urinal flushing as the guideline value has already been prescribed by the federal government. The estimated median value of ~0 cfu/100 mL for toilet and urinal flushing matches with that of the prescribed guideline (Health Canada 2010). However, the proposed maximum limit of <1 cfu/100 mL by this study for non-potable urban reuse is lower than the generically prescribed guideline of 14 cfu/100 mL for the *Greater exposure potential class* (MWR 2012). On the other hand, the maximum limit for toilet and urinal flushing estimated by this study is 3 cfu/100 mL (not proposed as the guideline is already prescribed) that is lower than 200 cfu/100 mL prescribed by the federal government (Health Canada 2010).

The proposed guideline values or estimated maximum values by this study are lower because RME was defined to be the 95th percentile value at the 95th percentile CDF in this research. The RME of higher than the 95th percentile value and that estimated from a higher than the 95th percentile CDF will have a larger maximum value. On the other hand, the companion document to the MWR recommended the limit of *E. coli* of < 1 cfu/100 mL with daily monitoring for the *Greater exposure potential* class "when required due to the intended application, or if required by a health officer" (BC Ministry of Environment 2013). Therefore, the proposed guideline value (maximum) by this study for non-potable urban reuse is consistent with the additional guideline prescribed by the companion document. Similarly, the MWR prescribed the limit of *E. coli* of < 1 cfu/100 mL with daily monitoring for the agricultural irrigation (raw eaten crop), which is similar to the proposed guideline value of < 2 cfu/100 mL of *E. coli*.

In addition, the proposed three categories of non-potable urban reuses: *Greater exposure potential I, II* and *III* seem to have a minor difference in the proposed guideline values; however, a high water quality is required for Category I than Category III. For instance, non-potable urban uses under Category I have a median of 0.01 cfu/100 mL and maximum of 0.03 cfu/100 mL and the laundry machine use under Category III has a median of 0.17 cfu/100 mL and a maximum of 0.67 cfu/100 mL. Due to the discrete nature of *E. coli* data, the median becomes ~0 (ND) cfu/100 mL for Categories I and III. Also, the discrete data and the consideration of five minimum samples lead to maximum values of less than 1 cfu/100 mL for Category I and 3 cfu/100 mL or less for Category III.

The microbial quality of reclaimed water use prescribed in different countries and even in various provinces within a country are highly variable as presented in Table 3.6. For example, for unrestricted urban reuse: the median and maximum value of *E. coli* concentration are ND and 14 cfu/100 mL respectively as prescribed by the US EPA; 2.2 and 23 cfu/100 mL in Washington (US), and 2.2 and 240 cfu/100 mL in California (US) (US EPA 2012a) ; the median is < 1 cfu/100 mL in Western Australia (Australia) (WA DoH 2011), < 10 cfu/100 mL in Queensland (Australia) (QS EPA 2005), 100 cfu/100 mL in UAE (DoET and SERI 2014), <= 200 cfu/100 mL in Mediterranean regions (EMWater 2001) for the same reuse type, and the maximum value of 250 cfu/100 mL for public green space irrigation in Spain (DoET and SERI 2014). The prescribed guidelines depend on the water quality standard achievable by a country or region based on their economic status and tolerable risk (WHO 2006a).

The log removal requirements estimated by this study are generally higher, which may be due to the consideration of a wide range of input values or uncertainties in the estimate, including the lowest possible value (0) log removal credit of the BNR technology for virus, *Cryptosporidium*, and *Giardia*. Furthermore, the uncertainty associated with the estimation is discussed in the next section. A controversy exists in the presence of a quantitative relationship between *E. coli*-based indicator and other pathogens, e.g., the work by Maimon et al. (2014) in opposition to O'Toole et al. (2012), Petterson et al. (2016), and Sidhu et al. (2012). Therefore, the health risks associated with other pathogens were also estimated as a case study to indicate the effectiveness of *E. coli*-based method. *E. coli* is the best available indicator of recent faecal contamination; however, non-faecal pathogens, such as, *Legionella*, *Mycobacterium avium* complex, *Aeromonas*, and *Helicobacter pylori* are not transmitted by the faecal to oral route. Usually, these bacterial pathogens are naturally found in source waters. The detection of *E. coli*, i.e., a faecal indicator, does not provide any information on the potential presence of non-faecal pathogens, but indicators are not known yet for such pathogens (Health Canada 2013a).

7.4.2 Model uncertainty and limitations

Many factors can influence risk estimations and hence the proposed guideline values of *E. coli* concentrations in reclaimed water. Monte Carlo simulations allow the inclusion of uncertainty in inputs (Ashbolt et al. 2010; Pavione et al. 2013) and this study has included the uncertainties in input paramters by using 2-D MCA (Pouillot et al. 2016; US EPA 2001). In the microbial risk assessment literature, two risk benchmarks are available. One benchmark is the disease burden of $\leq 10^{-6}$ DALYs/person per year recommended by the WHO for safe drinking water (WHO 2006b). Another benchmark is the annual risk of infection of $\leq 10^{-4}$ recommended by the US EPA (Lim and Jiang 2013). Risk interpretations may be inconsistent with these two benchmarks.

However, this study has used the common risk benchmark ($\leq 10^{-6}$ DALYs/person/year) recommended by the WHO. In addition, disease burdens (DALYs) are widely used for cost-benefit analysis of microbial risks (Lim et al. 2015).

The knowledge status of the QMRA model is low to medium specially in dose-response models, model parameters, pathogenic *E. coli* ratio, and morbidity indicating the associated uncertainities. These uncertainities highly affect the estimated health risks and the recommeded values. This could be a reason in prescribing highly different values for reclaimed water quality across the world. In particular, the knowledge status of pathogenic *E. coli* ratio and morbidity is low to medium, which is supported by the highest effects of these inputs on the variation of the output mean as discussed in the sensitivity analysis section (Appendix D.1 and Figure 7.4). Similarly, other input parameters: disease burden per case, susceptibility fraction, and exposure factors have medium knowledge status as shown by their moderate effects on the output mean. Although a 10% variation was considered in most of the exposure factors due to the lack of data, only the inputs cross connection per person, annual unit cross connection volume, log removal by cleaning, days from crop irrigation to consumption, log reduction by transportation and storage, unit car washing volume, car washing frequency, firefighting frequency, and unit firefighting volume had the effects of 5% to 11% on the output mean in some reuse applications as shown in Appendix D.1 and Figure 7.4.

Uncertainties in microbial concentrations were considered by using Monte Carlo simulations for the entire range of data. Risk estimates also vary with the type of dose-response models used, such as, exponential and Beta-Poisson models and their parameter values (Gale and Lacey, 1998; Haas et al., 2014). This study has a limitation in that it could not include the variation in dose-response models and their parameters. However, the present study has used the updated models and parameter values for pathogens as far as these are available in the publically accessible literature, such as, norovirus (Messner et al. 2014; Schmidt 2015), *Campylobacter jejuni* (Haas et al. 1999; Katukiza et al. 2014), rotavirus (Health Canada 2010), and *Cryptosporidium parvum* and *Giardia* spp. (Robertson et al. 2005). Due to the lack of Canadian data, especially the morbidity, disease burden per case, and susceptibility fraction for some pathogens and exposure factors for many water reuses, the related data were obtained from the US and the studies based on other developed countries, which may affect the proposed guideline values. Furthermore, this work has not included the health risks to residents associated with the bioaerosol generated by on-site wastewater treatment, although the risks can be low (Benami et al. 2016). Moreover, this research does not claim the development of implementation ready guidelines, rather it has elaborated the direction and process of developing guidelines, showing the complexity involved. It has provided a discussion of and recommendations for the microbial quality of reclaimed water that would support the federal government in developing guidelines on reclaimed water quality for urban applications besides toilet and urinal flushing.

7.5 Summary

Canada has abundant freshwater resources; however, many cities still experience seasonal water shortage. Supply-side and demand-side management is a core strategy to address this water shortage. Under this strategy, reclaimed water, which the Canadian public is willing to use for non-potable purposes, is an option. However, no universal guidelines exist for reclaimed water use. Despite the federal government's long-term goal to develop guidelines for many water reuse applications, guidelines have only been prescribed for reclaimed water use in toilet and urinal flushing in Canada. At the provincial level, British Columbia (BC) has promulgated guidelines for wide applications of reclaimed water but only at broad class levels. This research has investigated and proposed probabilistic risk-based guideline values for microbial quality of reclaimed water in various non-potable urban reuses. The health risk was estimated by using quantitative microbial risk assessment. Two-dimensional Monte Carlo simulations were used in the analysis to characterize variability and uncertainty in input data. The proposed guideline values are based on the indicator organism *E. coli*. The required treatment levels for reuse were also estimated. In addition, the guideline values were successfully applied to three wastewater treatment effluents in the Okanagan Valley (BC, Canada). The health risks associated with other bacterial pathogens (*Campylobacter jejuni* and *Salmonella* spp.), virus (adenovirus, norovirus, and rotavirus), and protozoa (*Cryptosporidium parvum* and *Giardia* spp.), were also estimated. The estimated risks indicate the effectiveness of the *E. coli*-based water quality guideline values. Sensitivity analysis shows the pathogenic *E. coli* ratio and morbidity are the most sensitive input parameters for all water reuses. The proposed guideline values could further be improved by using national or regional data on water exposures, disease burden per case, and the susceptibility fraction of population.

Chapter 8 Development of Decision Support Tool for Fit-For-Purpose Wastewater Treatment and Reuse

As part of this chapter, two research papers have been submitted and are under review in the *Science of the Total Environment* for possible publication entitled:

- "Fit-for-purpose wastewater treatment: Conceptualization and development of decision support tool (I)" (Chhipi-Shrestha et al. 2017e) and
- "Fit-for-purpose wastewater treatment: Testing and implementation of decision support tool (II)" (Chhipi-Shrestha et al. 2017f).

8.1 Background

Water reuse has been increasing across the globe; however, certain challenges exist affecting its universal applications. The key barriers and challenges of water reuse identified by various studies are that its management is more complicated than conventional water resources (e.g., infrastructure requirements), it generally has higher costs than conventional water, public perceives that water reuse can pose high health risks, and possible trade barriers for food products grown using reused water (Crook et al. 2005; DSEWPaC 2012; EU 2016). The first two challenges are primarily faced by institutions and are related to the cost and the latter two are concerned with public perception, especially the stigma that the reused water can be a health risk. Reclaimed water of any quality can be produced, resulting in risk reduction, by using available technologies if financial resources are adequate. This indicates cost as the ultimate factor if public perception can be improved by increasing awareness.

Urban water supply systems, including water reuse applications are complex involving uncertainties (Nasiri et al. 2013; Roozbahani et al. 2013). Decision making in environmental applications including reclaimed water use, is associated with uncertainty that is unavoidable and inevitable (Sadiq and Tesfamariam 2009). The uncertainty in estimates related to water reuse should be incorporated in decision making. In particular, the variability and uncertainty in health risk can be computed by using a probabilistic approach (Ashbolt et al. 2010; Katukiza et al. 2014; Pavione et al. 2013). The probability analysis can be performed using Monte Carlo simulations. While certain attributes in water reuse decisions are not precisely assessed due to unquantifiable, incomplete, and non-obtainable information and partial ignorance (Sadiq et al. 2004a), for example, weights (importance) of selection criteria, parameter values given in a range, etc. Such types of uncertainty caused by imprecision and vagueness can be included in the analysis by applying fuzzy set theory (Agwa et al. 2013; Zadeh 1965). A fuzzy number can be expressed in a Triangular Fuzzy Number (TFN) for simplicity.

The cost, human health risk of water reuse, and energy consumption are important factors affecting the design of fit-for-purpose wastewater treatment plants (Chang et al. 2008; Guo et al. 2014; Health Canada 2010; NASEM 2016). The cost of wastewater treatment can be estimated by using life cycle cost analysis (LCCA). LCCA is an economic assessment method which takes into account all costs arising from owning, operating, maintaining, and ultimately disposing of a project or product (US DOE 1996). Moreover, reclaimed water use can pose various health risks, mainly associated with pathogenic microorganisms, disinfection by-products (DBPs), and pharmaceutical and personal care products (PPCPs) as elaborated in detail in Chapter 7. However, this study only includes the health risks associated with pathogenic microorganisms.

The major groups of wastewater pathogens are bacteria, viruses, and protozoans (Haas et al., 1999). In these groups, the pathogenic microorganisms are selected for risk assessment based on the indicator organisms, adequacy of literature on the organisms, and the occurrence of water borne illness and diseases as reported by health authorities (Katukiza et al. 2014). The most relevant pathogens are *Escherichia coli* O157:H7, *Salmonella spp.*, and *Campylobacter jejuni* in bacteria; Rotavirus, Adenovirus and Norovirus in viruses, and *Cryptosporidium parvum* and *Giardia spp.* in protozoa. Human health risk can be estimated by using quantitative microbial risk assessment (QMRA) (Haas et al. 2014). Also, energy consumption by a treatment train can be estimated as the sum of energy use by individual treatment processes of the train. This chapter aims to develop a decision support tool for evaluating the potential of fit-for-purpose wastewater treatment and specific reuse for a community. The tool will be able to assess health risks of reclaimed water use in one or more urban applications simultaneously, estimate the LCC of wastewater treatment, and estimate energy consumption and associated carbon emissions.

8.2 Methodology

A wastewater treatment train comprises of an array of treatment technologies (units) in different treatment stages to meet the criteria for a specific reuse application. Alternative treatment technologies in different treatment stages: primary, secondary, tertiary, and advanced treatment have various performance levels in terms of different evaluation criteria (e.g., cost, treatment efficiency, etc.). A conceptual model of a tool has been proposed for evaluating fit-for-purpose wastewater treatment and reuse potential. The conceptual model includes the steps: estimation of microbial concentration, quantitative microbial risk assessment (QMRA), development of alternative treatment trains, estimation of reclaimed water quantity and its distribution, LCC analysis, estimation of energy use, estimation of carbon emissions as well as multi-criteria decision analysis as shown in Figure 8.1.

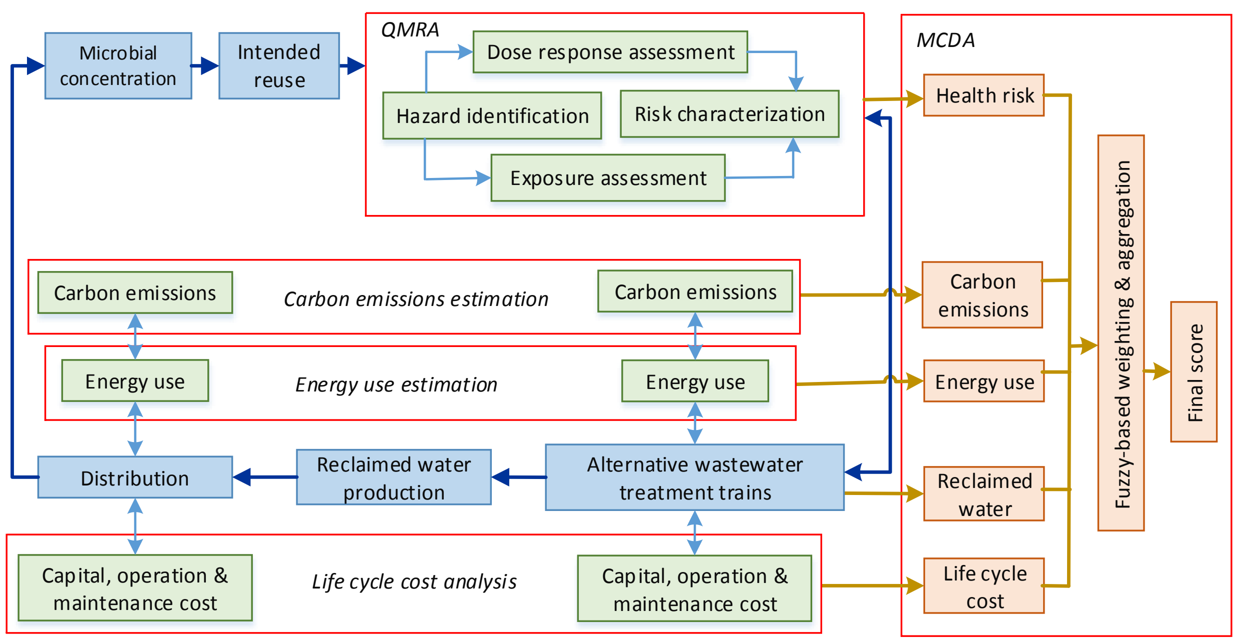


Figure 8.1 Conceptual model for evaluating fit-for-purpose wastewater treatment and reuse potential

8.2.1 Estimation of microbial concentration in reclaimed water

The concentration of microorganisms in raw wastewater and greywater was determined based on laboratory analysis or obtained from literature in absence of such data. The estimated microbial concentration is expressed as a TFN represented by the lowermost possible (l), most probable (m), and uppermost possible (u) values. The fuzzy microbial concentrations of raw wastewater and greywater obtained from the literature were used as default values in the proposed tool and is expressed in terms of TFN (l, m, u) as given in Appendix E.1 and Appendix E.2 respectively. The “l”, “m”, and “u” values were considered to be the 5th, 50th, and 95th percentiles of the concentration, respectively (a wide range considered in order to capture most date), which were obtained from Monte Carlo simulations (MCS) of 10,000 iterations of raw wastewaster microbial concentrations considering their log normal distribution (Pavione et al. 2013). In addition, the concentration of microorganisms in the reclaimed (treated) water was estimated by using log removal method for all treatment units contained in a WWTP (Lim et al. 2015). The log removal credits of different technologies are expressed in terms of TFNs. The “l”, “m”, and “u” values were considered to be the 50th, 75th, and 95th percentiles, respectively (considered 50th percentiles and above assuming good performance of water reuse projects), which were obtained from MCSs of 10,000 iterations considering uniform distribution (Table 8.1).

Table 8.1 Alternative technologies and pathogen removal efficiency (log removal in TFNs)

Treatment unit	Bacteria	Virus	*Cryptosporidium*	*Giardia* spp.
Primary sedimentation	(0.50, 0.75, 0.95) (Health Canada 2010)	(0.50, 0.75, 0.95) (Health Canada 2010)	(0.75, 0.87, 0.97) (Health Canada 2010)	(0.25, 0.37, 0.47) (Health Canada 2010)
Trickling filter	(1.50, 1.75, 1.95) (Health Canada 2010)	(1.25, 1.63, 1.92) (Health Canada 2010)	(0.75, 0.87, 0.98) (Health Canada 2010)	(0.75, 0.87, 0.97) (Health Canada 2010)
Activated sludge	(1.50, 1.75, 1.95) (Health Canada, 2010)	(1.25,1.63, 1.92) (EPHC/NHMRC/NRMMC 2008)	(0.75, 0.87, 0.98) (Health Canada 2010)	(0.75, 0.87, 0.97) (Health Canada 2010)
Biological Nutrients Removal (BNR)	(1.50, 1.75, 1.95) (Health Canada 2010)	(1.25,1.63, 1.92) (Health Canada 2010)	(0.75, 0.87, 0.98) (Health Canada 2010)	(0.75, 0.87, 0.97) (Health Canada 2010)
Membrane Bioreactor	(6.10, 6.45, 6.73) (Hai et al. 2014)	(3.25, 4.22, 5.01) (Hai et al. 2014)	(7.00, 7.50, 7.90) (Hai et al. 2014)	(7.00, 7.50, 7.90) (Hai et al. 2014)
Sequencing Batch Reactor	(1.50, 1.75, 1.95) (considered same as Activated Sludge)	(1.25,1.63, 1.92) (considered same as Activated Sludge)	(0.75, 0.87, 0.98) (considered same as Activated Sludge)	(0.75, 0.87, 0.97) (considered same as Activated Sludge)
Coagulation and Flocculation	(2.20, 3.05, 3.73) (Health Canada 2013)	(2.25, 2.82, 3.28) (Health Canada 2011)	(2.05, 2.88, 3.53) (Health Canada 2012); (Hijnen et al., 2010)	(1.65, 2.47, 3.13) (Health Canada 2012)
Microfiltration	(5.00, 5.50, 5.90) (Health Canada 2010); (EPHC/NHMRC/NRMMC 2008)	(4.25, 5.12, 5.82) (if proceed after coagulation) (Health Canada 2010)	(6.20, 7.10, 7.82) (Health Canada 2010)	(6.35, 7.17, 7.83) (Health Canada 2010)
Depth filtration	(2.25, 2.72, 3.10) (Health Canada 2013b)	(1.95, 2.87, 3.61) (Health Canada 2011)	(6.10, 6.45, 6.73) (Health Canada 2012)	(6.10, 6.45, 6.73) (Health Canada 2012)
Surface filtration	(0.50, 0.75, 0.95) (Asano et al. 2007)	(0.25, 0.37, 0.47) (considered half effective to bacteria) (Asano et al. 2007)	(0.50, 0.75, 0.95) (Asano et al. 2007)	(0.50, 0.75, 0.95) (Asano et al. 2007)
Granular Activated Carbon	(0.60, 0.85, 1.05) (Hijnen et al. 2010)	(0.45, 0.57, 0.67) (Hijnen et al. 2010)	(2.00, 2.35, 2.63) (Hijnen et al. 2010)	(2.00, 2.35, 2.63) (Hijnen et al. 2010)
Ultrafiltration	(5.0, 5.5,5.9) (Health Canada 2013b)	(4.50, 5.25, 5.85) (Health Canada 2011)	(6.2, 7.10, 7.82) (Health Canada 2012)	(6.35, 7.17, 7.83) (Health Canada 2012)
Reverse osmosis	(5.50, 6.25, 6.85) (Kitis et al. 2003)	(4.85, 5.92, 6.78) (Hai et al. 2014)	(8.00, 8.50, 8.90) (Health Canada 2012)	(8.00, 8.50, 8.90) (Health Canada 2012)
Chlorination	(4.00, 5.00, 5.80) (Health Canada 2010); (EPHC/NHMRC/NRMMC 2008)	(3.0, 3.5, 3.9) (Health Canada 2010)	(0.75, 1.12, 1.43) (Health Canada 2010)	(0.75, 1.12, 1.43) (Health Canada 2010)
UV radiation	(3.0, 3.5, 3.9) (Health Canada 2010); (EPHC/NHMRC/NRMMC 2008)	(2.12, 3.06, 3.81) (Health Canada 2010); (EPHC/NHMRC/NRMMC 2008)	(3.5, 3.75, 3.95) (Health Canada 2010); (EPHC/NHMRC/NRMMC 2008)	(3.5, 3.75, 3.95) (Health Canada 2010);(EPHC/NHMRC/NRMMC 2008)
Ozone	(3.50, 3.90, 4.22) (Xu et al. 2002)	(3.00, 3.50, 3.90) (EPA-Ireland 2011)	(2.00, 2.50, 2.90) (EPA-Ireland 2011)	(2.00, 2.50, 2.90) (EPA-Ireland 2011)

Note: TFN: (l, m, u) refers to (50^{th}, 75^{th}, 95^{th} percentiles)

8.2.2 Quantitative microbial risk assessment (QMRA)

The QMRA includes four steps - hazard identification, exposure assessment, dose-response assessment, and risk characterization (Haas et al., 1999; Haas, 2002; Environment Canada/Health Canada, 2013). The same steps as explained in Chapter 7 were followed for QMRA. In particular, the volume of water exposed to an individual due to unit exposure of water reuse applications and their annual frequency of applications were consistent with Table 7.1 (Chapter 7). This chapter has included 13 possible categories of urban reuses including potable reuses as follows:

a. Toilet and urinal flushing
b. Lawn irrigation
c. Public park irrigation
d. Golf course irrigation
e. Agricultural irrigation (raw eaten crops)
f. Lawn, park & golf course irrigation (non-agri)
g. All irrigation (lawn, park, golf course & agriculture)
h. Vehicle washing
i. Laundry machine
j. Firefighting
k. Both toilet flushing & laundry machine
l. All above non-potable reuses
m. Potable use

In each urban water reuse, the total volume of water exposed annually was estimated by using MCSs of 10,000 iterations. For MCSs, a uniform distribution was considered for most of the exposure factors (Mok et al. 2014; Verbyla et al. 2016) and a triangular distribution was assumed for "log reduction in natural die off" due to the lack of distribution data,. The exposure volumes were expressed in terms of TFNs with "l", "m", and "u" values as the 50th, 75th, and 95th

percentiles respectively (considered a high exposure being risk averse). Moreover, the parameter values of the dose-response models of different pathogens are given in Table 7.2 (Chapter 7).

Finally, the health risk was estimated in DALY by using Equation 8.1 (Howard et al. 2006).

$$DALY = P_{inf(A)}(d) * P_{ill|inf} * DBPC * f_s \qquad \textbf{Equation 8.1}$$

where $P_{ill|inf}$ is risk of disease given infection or morbidity; DBPC is disease burden per case (DALY/year); and f_s is susceptibility fraction. The values of morbidity, disease burden per case, and susceptibility fraction were obtained from literature and are given in Table 7.3 (Chapter 7).

In Equation 8.1, $P_{inf(A)}(d)$ is a fuzzy value attributed to the fuzzy input of microbial concentration. For the other three components ($P_{ill|inf} * DBPC * f_s$), MCSs of 10,000 iterations considering a uniform distribution were performed (Chapter 7). The 50th, 75th, and 95th percentiles obtained from simulations were used as l, m, and u respectively. Finally, DALY was estimated in TFNs (50th, 75th, and 95th percentiles) by multiplication of $P_{inf(A)}(d)$ and ($P_{ill|inf} * DBPC * f_s$) as given in Equation 8.1. The estimated health risk was compared with the acceptable risk benchmark of 10^{-6} DALYs/year (WHO 2006b).

8.2.3 Life cycle cost analysis (LCCA)

Life cycle cost (LCC) includes capital cost and operation and maintenance cost of WWTPs, wastewater collection, and reclaimed water distribution sytems in this study. The LCC is expressed in terms of annualized net present value (NPV). NPV is estimated using Equation 6.13 in Chapter 6 (Davis et al. 2005).

The equations used in the cost estimation of each treatment technology are given in Table 8.2. These equations are already validated equations. All the costs were converted to constant 2015 Canadian dollars ($). The total cost of an entire treatment train was estimated as the sum of the LCC of each treatment unit. The total LCC was expressed in TFNs with the input of a fuzzy discount rate. Also, the fuzzy data were used for unit cost of treatment technologies that have no specific cost equations as given in Table 8.2.

The economic benefit-cost analysis of a planned water reuse scheme can also be performed. Water reuse reduces the demand of municipal drinking water as well as the volume of sewage to be treated, which can save the cost of drinking water supply and wastewater treatment. The economic benefits or cost savings were estimated by using Equations 8.2 to 8.4.

$$S_{dw} = C_{dw} * Q_{rw} * P \qquad \textbf{Equation 8.2}$$

$$S_{ww} = C_{ww} * Q_{ww} * P \qquad \textbf{Equation 8.3}$$

$$S_{total} = S_{dw} + S_{ww} \qquad \textbf{Equation 8.4}$$

where “S” is saving in cost ($/d), “C” is average unit price ($/$m^3$), “Q” is average discharge per capita (m^3/d/person), “P” is total population and subscript “dw” refers to drinking water supply, “rw” is reclaimed water consumed, and “ww” is wastewater generated.

8.2.4 Estimation of energy use and carbon emissions

The energy consumed by each treatment process was estimated based on its energy intensity obtained from the literature, such as journals and government and industry reports. Most of the energy equations were generated from the database that were prepared based on the survey of 15,617 WWTPs in the US (WEF/EPRI 2013). The equations for the estimation of energy intensities are given in Table 8.3.

Table 8.2 Equations used for life cycle costing of treatment technologies

Technologies	Capital cost (CC) ($)	Annual O&M cost($)	Life span (yrs)	Reference
Screen and Grit removal	$(0.00064Q^{0.625} + 0.000343Q^{0.7} + 0.0000056Q + 0.4)\ x10^{6}$	OM = (4%, 5%, 6%) of CC (assumed)	15	Ahmed et al. (2002)
Primary sedimentation	$(0.000028\ Q + 0.4)\ x10^{6}$	(4%, 5%, 6%) of CC (assumed)	15	Ahmed et al. (2002)
Secondary Clarifier (SC)	$6020\ Q^{0.58}$	(4%, 5%, 6%) of CC (assumed)	15	Ahmed et al. (2002)
Trickling filter	$117\ Q + 291{,}330 + SC\ cost$	$16{,}875.78\ Q + SC\ cost$	20	Capital cost (Ahmed et al., 2002); O&M cost (Zahid, 2007)
Activated sludge	$10^{0.256\ (log\ Q)^{1.556}+4.545} + SC\ cost$	$25{,}731\ Q + SC\ cost$	20	Guo et al. (2014)
Biological Nutrients Removal	$2299\ Q + 367293\ + SC\ cost$	$22.3\ Q + 0.083$	20	US EPA (2007)
Membrane Bioreactor	$10^{0.569\ (log\ Q)^{1.135}+4.605}$	$0.083\ x\ 10^{0.639\ (log\ Q)^{1.143}+2.63}$	20	Guo et al. (2014)
Sequencing Batch Reactor	$65.47\ Q\ + 85{,}978 + SC\ cost$	2 Q	20	US EPA (1999); Q in kilogallon/d for OM cost
Coagulation and Flocculation	$10^{0.222\ (log\ Q)^{1.516}+3.071}$	$10^{0.347\ (\log Q)^{1.448}+2.726}$	20	Guo et al. (2014)
Microfiltration	$1{,}628{,}571\ Q$	114,286 Q	8	Furrey et al. (2000); Q in milliongallon/d
Surface filtration	$323.88\ Q$	4.51 Q	20	City of Odessa (2015), Q in kilogallon/d
Granular Activated carbon	$10^{0.722\ (log\ Q)^{1.023}+3.443}$	$10^{1.669\ (log\ Q)^{0.559}+2.371}$	4	Guo et al. (2014)
Ultrafiltration	$10^{1.003\ (log\ Q)^{0.83}+3.832}$	$10^{1.828\ (log\ Q)^{0.598}+1.876}$	4	Guo et al. (2014)
Electrodialysis	$19{,}250\ Q^{0.6}$	$0.15\ x\ Capital\ cost$	15	Ahmed et al. 2002)
Reverse osmosis	$10^{0.966\ (log\ Q)^{0.929}+3.082}$	$10^{0.534\ (log\ Q)^{1.253}+2.786}$	15	Guo et al. (2014)
Chlorination	$A + BQ$ (for dose of 4 mg/L without hypochlorite storage)	$A' + B'Q$ (for dose of 4 mg/L without hypochlorite storage)	20	US EPA (2006) with different parameter (A, B, A', B') values ; Q in kilogallon/d
UV	$48{,}463\ Q +\ 93{,}748$	3,913.6 Q + 660.85	20	Q in million gallon daily US EPA 1996)
Ozone	$A + BQ$	$A' + B'Q$	20	US EPA (2006) with different parameter values of (A, B, A', B'); Q in kilogallon/d
Sludge thickening and dewatering	(25%, 30%, 35%)	(25%, 30%, 35%)	-	Molinos-Senante et al. (2013)

Note: Q is plant capacity in m^3/d unless stated; CC= capital cost; OM= Operational and maintenance cost

Table 8.3 **Energy consumption equations for treatment technologies**

WWT Components	Energy (E)	r^2	References
Screen and Grit removal	$0.0035\ Q + 128.14$	0.9963	WEF/EPRI (2013)
Primary sedimentation	$0.0082\ Q - 7.0234$	0.9999	WEF/EPRI (2013)
Trickling filter	$00.134\ Q + 26.891$	0.9999	WEF/EPRI (2013); Metcalf and Eddy (2003)
Activated sludge	$0.1611\ Q + 1345.1$*	0.9996	WEF/EPRI (2013)
Biological Nutrient Removal	$0.0249\ Q + 328.97$	0.9953	WEF/EPRI (2013)
Membrane Bioreactor	$0.7149\ Q - 2.2808$	0.9999	WEF/EPRI (2013)
Sequencing Batch Reactor	$0.2668\ Q + 465.96$	0.9998	WEF/EPRI (2013)
Coagulation and Flocculation	$0.0113\ Q + 28.222$	0.9999	PGEC/SBWC (2006)
Microfiltration	$0.0264\ Q$	-	Chang et al. (2008)
Surface filtration	$0.0077\ Q + 11.156$	0.9999	WEF/EPRI (2013)
Depth filtration	$0.0153\ Q + 22.312$	0.9999	WEF/EPRI (2013)
Ultrafiltration	$0.2114\ Q$	-	Chang et al. (2008)
Electrodialysis	1.32 Q (l); 1.85 Q (m); 2.38 Q (u)	-	Chang et al. (2008)
Reverse osmosis	0.53 Q (l); 1.06 Q (m); 1.59 Q (u)	-	Chang et al. (2008)
Chlorination	$0.022 + {}^{0.5234}/_{Q}$	0.9999	WEF/EPRI (2013)
UV	$0.0618 - {}^{1.996}/_{Q}$	0.9999	WEF/EPRI (2013)
Ozone	$0.0743 + {}^{449.93}/_{Q}$**	-	WEF/EPRI (2013)
Sludge thickening	0.007 Q (l); 0.02 Q (m); 0.04 Q (u)	-	WEF/EPRI (2013)
Sludge dewatering	0.011 Q (l); 0.054 Q (m); 0.106 Q (u)	-	WEF/EPRI (2013)
Plant utility water	$0.0105\ Q + 11.07$	-	WEF/EPRI (2013)
Non-process loads (buildings, lighting, computers, pneumatics, etc.)	$0.0473\ Q + 154.66$	-	WEF/EPRI (2013)

Note: E= energy in kWh/d and Q in average flow in m^3/d

The total energy intensity (kWh/m^3) of a WWTP was estimated as the sum of energy intensities of its treatment units and non-treatment processes such as plant utility water use and non-process loads, e.g., buildings, lighting, computers, pneumatics, etc. The total energy use (kWh) of a WWTP was computed as its total energy intensity multiplied by influent flow rate (m^3/d). In addition, the carbon emissions associated with the total energy use was estimated by multiplying energy use (kWh) and carbon emission factors of electricity (kg CO_2e/kWh) for the region. The provincial emission factors of Canada were obtained from the Canadian Geoexchange Coalition (2010). The total energy use was expressed in TFNs with the input of fuzzy energy intensity of unit processes as far as data is available to generate fuzzy numbers. Consequently, the fuzzy energy use and fuzzy carbon emission factor give fuzzy carbon emission estimates.

8.2.5 Estimation of reclaimed water quantity and its distribution

Users have to input an average and maximum wastewater flow rate of the community under study. In absence of such data, a user has to provide total population and then the tool automatically uses default values of average wastewater generation rate of 240 L/p/d (Briere 2010) and that of greywater generation rate of 50% of wastewater (Environment Canada 2014b). Similarly, if users only input either average flow rate or maximum flow rate, the tool considers an approximate peaking factor of 1.8 to estimate a maximum flow rate from an average flow rate (Briere 2010; Guo et al. 2016). Also, the quantity of reclaimed water produced was estimated based on the recycling efficiency of wastewater treatment as follows.

*Reclaimed water (m^3/day) = Influent flow (m^3/day) * Recycling efficiency* **Equation 8.5**

A recycling efficiency of 85% to 90%, i.e., a TFN (0.85, 0.875, 0.90) was used as a default value (City of Penticton 2014a); however, users can input their own value in the tool. It will give a fuzzy value of reclaimed water production.

In addition, a secondary distribution network of reclaimed water should be designed by using any network design software, such as EPANET. Also, a secondary wastewater collection network may also need to be designed, such as for greywater recycling and use. The total pipe length, unit pipe installment cost, unit operational energy, and unit operational and maintenance cost for wastewater collection and reclaimed water distribution should be estimated using the network design. These data should be input into the tool.

8.2.6 Multi-criteria decision analysis (MCDA)

Alternative wastewater treatment trains can be evaluated based on the following criteria: a) reclaimed water production, b) health risk of reclaimed water use, c) life cycle cost of treatment and conveyance system, d) energy use, and e) carbon emissions. The values of all these criteria are fuzzy, i.e., TFNs.

The fuzzy data of each criteria, i.e., $(x_{ij}^{l},\ x_{ij}^{m},\ x_{ij}^{u})$, was normalized to obtain a unitless value $\tilde{r}_{ij}\ i.e., (r_{ij}^{l},\ r_{ij}^{m},\ r_{ij}^{u})$ using Equations 8.6 and 8.7 (Yoon and Hwang 1995).

Benefit criteria:

$r_{ij}^{l} = \frac{x_{ij}^{l}}{x_{j^*}};\ r_{ij}^{m} = \frac{x_{ij}^{m}}{x_{j^*}}$; and $r_{ij}^{u} = \frac{x_{ij}^{u}}{x_{j^*}}$ **Equation 8.6**

Cost criteria

$r_{ij}^{l} = \frac{x_j^{-}}{x^{l}{}_{ij}}$; $r_{ij}^{m} = \frac{x_j^{-}}{x^{m}{}_{ij}}$; and $r_{ij}^{u} = \frac{x_j^{-}}{x^{u}{}_{ij}}$ **Equation 8.7**

where superscripts l, m, and u refer to lower, middle and upper values of respective variables; i = 1, 2, ……, m (m= no. of alternatives) and j = 1, 2,…5; x^*_j and x_j^{-} are the maximum and minimum values of the jth criteria, including l, m, and u.

Each evaluation criteria may have different weights, i.e., importance. The criteria were rated using an importance scale: very high, high, medium, low, and very low. These linguistic weights that primarily depend on a specific community, are provided by decision makers or are obtained from a survey. The linguistic ratings of the importance values very high, high, medium, low, and very low were transformed to fuzzy weights ($\tilde{w}_j$), i.e., TFNs (0.75, 1, 1), (0.5, 0.75, 1), (0.25, 0.5, 0.75), (0, 0.25, 0.5) and (0, 0, 0.25) respectively (Chen et al. 2008). The ratings of criteria and their weights were aggregated by a fuzzy weighted average technique as given in Equation 8.8. The technique is often used in risk and decision analysis to obtain a final score ($\tilde{y}$) (Guu 2002).

$$\tilde{y}_i = \frac{\tilde{r}_{ij} * \tilde{w}_j}{\sum_{j=1}^{5} \tilde{w}_j} \qquad \textbf{Equation 8.8}$$

The final crisp score (y_i) was computed by the defuzzification of $\tilde{y}_i$ using the centroid method as given in Equation 8.9 (Ross 2004).

$$y_i = \frac{y_i^l + y_i^m + y_i^u}{3} \qquad \textbf{Equation 8.9}$$

The final scores were expressed in the scale of 0 to 100 for each alternative, i.e., treatment train. Higher scores denote better performance and vice versa.

8.3 Results

A spreadsheet-based tool called "FitWater" has been developed for the evaluation of fit-for-purpose wastewater treatment and reuse potential. The data entry and flow in FitWater is given in Appendix E.3. The tool will help in evaluating treatment trains for various water reuses based on reclaimed water production, health risk, life cycle cost, energy use, and carbon emissions. The interface of FitWater is shown in Figure 8.2. Users have to enter the weights (importance) of evaluation criteria specific to their community in terms of the linguistic scale: very high, high, medium, low, and very low. Also, they can input a carbon emission factor for their region; however, a dropdown list is provided to select the default provincial carbon emission factor of grid electricity of Canada (Canadian Geoexchange Coalition 2010). The tool automatically calculates a final aggregated score by using the fuzzy weighted average method and then ranks the alternatives.

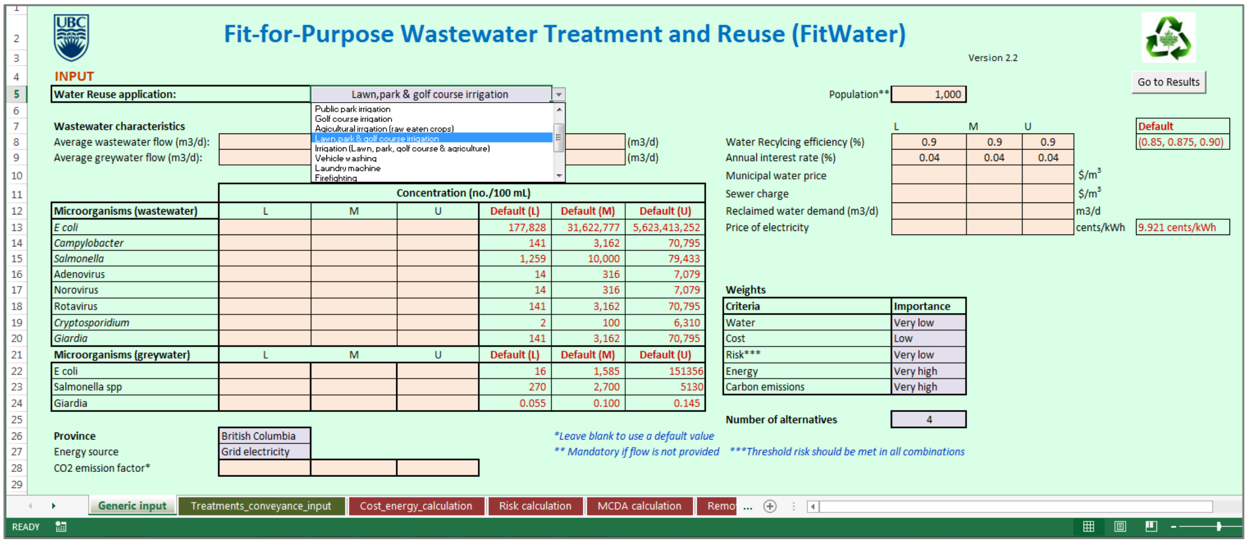


Figure 8.2 Sceenshot of the FitWater interface

The proposed FitWater is practical and flexible in developing and then evaluating the performance of wastewater treatment trains for one or more specific water reuses. The tool has provided a total of 13 water reuse options, including non-potable and potable water use. The tool is user-friendly and requires minimal input data for analysis. In the lack of such data, default values can be used, such as wastewater microbial concentration, per capita wastewater generation rate, carbon emission factor of electricity, and water recycling efficiency. However, location specific data is preferred for the analysis.

The uncertainty in data is incorporated in the analysis using probabilistic technique and fuzzy set theory in FitWater. The probabilistic technique is applied to include stochasticity and fuzzy set theory is used to analyze imprecise and vague information. Moreover, the tool included major wastewater microorganisms: bacteria, virus, and protozoa in health risk estimation. Also, all estimated costs are expressed in 2015 Canadian dollars for consistency. The proposed tool can be applied to any community with wastewater flow higher than 500 m^3/d. FitWater is applicable to both wastewater and greywater treatment.

8.3.1 Testing of FitWater

The proposed FitWater tool was tested with several existing and planned WWTPs to study the effectiveness of the tool in estimating cost, energy intensity, and health risk of reclaimed water use. In addition, the application of FitWater is illustrated with an example as given below.

8.3.1.1 Testing with existing data

Fitwater was applied to several existing WWTPs of the United States and Canada. The WWTPs are briefly described as follows:

1) *Big Spring Regional Reclamation Project*: This reclamation project is located in Texas, US. Its maximum capacity is 5,724 m^3/d. The reclaimed water is used in landscape and golf course irrigation (Freese and Nichols Inc., 2007).

2) *Groundwater Replenishment System*: The replenishment system is located in the Orange County, California, US. The maximum flow is 443,816 m^3/d. The reclaimed water is used to recharge groundwater as well as injected to reduce seawater intrusion (Woodside and Westropp 2015).

3) *East Mesa Water Reclamation Facility*: This facility is located in Las Cruces, New Mexico, US. The average flow is 1,308 m^3/d. The reclaimed water is used for irrigating city parks and golf courses (Corbin 2014).

4) *King County Reclaimed Water Plant*: The reclamation project is planned in Washington, US. The average flow is 1,893 m^3/d. The reclaimed water is planned to be used for landscape and agricultural irrigation (Interbay Strategy) (KCWTD 2012).

5) *Penticton WWTP*: The treatment plant is located in the Okanagan Valley, BC, Canada. The average flow is 11,438 m^3/d. A portion of the treated water is used in city park and golf course irrigation (City of Penticton 2014a, 2015c).

6) *Sechelt Water Resources Centre*: The Sechelt WWTP is established in BC, Canada. Its average flow is 2200 m^3/d and maximum flow is 4000 m^3/d. Its water is used for landscape irrigation (Maple Reinders 2016; Organica Water Inc. 2012).

7) *Robert O. Pickard Environmental Centre*: The Centre has a WWTP that is located in Ottawa, ON, Canada. Its average flow is 545,000 m^3/d and maximum flow is 1,362,500 m^3/d. The treated water is discharged to environment (City of Ottawa 2016).

8) *Four planned WWTPs*: Four WWTPs have been planned for installation in BC, Canada. Three plants use membrane bioreactor technology, which have an average flow of 1000, 2000, and 3000 m^3/d. Another WWTP uses activated sludge technology, and has an average flow of 1000 m^3/d. The treated water is planned to be discharged to the environment for all WWTPs (Gigliotti 2016).

For all of these WWTPs, capital cost, operational and maintenance cost, energy intensity, and health risk were estimated and compared with actual values, based on the available data (Table 8.4). Table 8.4 shows that the estimated capital cost, operational and maintenance cost, and energy intensity using FitWater are mostly similar to their respective actual values. In addition, the estimated health risk ("m" value, i.e., 75th percentile) is acceptable for the defined water reuse as the estimated risk is less than 1.0E-06 DALYs/year (WHO 2006b) except in the East Mesa Water Reclamation Facility, which has high risk associated with norovirus (9.0E-05 DALYs/yr). The reclaimed water use for irrigating city parks and golf courses may have significant health impacts. However, the estimation is based on the microbial quality of average raw wastewater obtained from the literature Health Canada (2010) and EPHC/NHMRC/NRMMC (2008). It is recommended to use location specific data on microbial quality of reclaimed water or raw wastewater. Moreover, the effluent quality of Robert O. Pickard Environmental Centre (28 cfu/100 mL) and the other four designed WWTPs for BC (<1 cfu/100 mL) discharge effluent to the environment, and the estimated *E. coli* concentrations in effluent are within the effluent guidelines of 50 cfu/100 mL (City of Ottawa 2013; MWR 2012).

Table 8.4 Applications of FitWater to existing wastewater treatment plants

Wastewater treatment plants	Flow (m^3/d)	Treatment train	Actual cost and EI		Estimated from FitWater		
			Cost ('000 $)	EI (kWh/m^3)	Cost ('000 $)	EI (kWh/m^3)	Health risk
1. Big Spring Regional Reclamation Project, Texas, US (Freese and Nichols Inc. 2007)	5,724 (max)	Microfiltration, RO, UV (Advanced oxidation)	C: 11,323 O: 798	1.07	C: 9,897 - 9,981 O: ~989	0.72 - 1.77	7.7E-13 DALYs/yr
2. Groundwater Replenishment System, Orange County, California, US (Woodside and Westropp 2015)	443,816 (max)	Microfiltration, RO, UV (Advanced oxidation)	C: 519,480 O: 43,200	0.67	C: 476,152 - 481,179 O: ~ 57,913	0.68 - 1.73	1.3E-09
3. East Mesa Water Reclamation Facility, Las Cruces, New Mexico, US (Corbin 2014)	1308 (avg)	Primary sedimentation (PS), BNR, disc filtration, UV	O: 260 (2012/13 yr)	1.59	C: 14,015 - 15,172 O: 173 - 270	1.01 - 1.13	9.0E-05 DALYs/yr
4. King County Reclaimed Water Plan, Washington, US (KCWTD 2012)	1893 (avg)	PS, Activated sludge (AS), Sand filtration, CF, MBR, Cl_2	C: 15,730 O: 127	1.88	C: 15,601 - 16,849 O: 158 - 231	0.99 - 1.12	1.3E-07 DALYs/yr
5. Penticton WWTP, BC, Canada (City of Penticton 2014a, 2015c)	11,438 (avg)	PS, BNR, Cloth filtration, UV, Sludge processing, Cl_2	O: 1166 to 1393	0.61 - 0.83	C: 91,906 - 99,573 O: 618 - 868	0.56 - 0.69	3.9E-08 Actual: << 1E-6 DALYs/yr
6. Sechelt Water Resources Centre (WWTP), BC, Canada (Maple Reinders 2016; Organica Water Inc. 2012)	2200 (avg) 4000 (max)	Primary, FBR, CF, Surface filtration, Sludge processing, UV, Cl_2	C: 21,296 O: 520	0.9 - 1.1	C: 21,868 - 23,954 O: 1,127 - 1,196	1.05 - 1.16	< 5.9E-11 DALYs/yr
7. Robert O. Pickard Environmental Centre (WWTP), Ottawa, ON, Canada (City of Ottawa 2016)	545,000 (avg) 1,362,500 (max)	Primary, BNR, Sludge processing, UV	LCC: $1.42/m^3	-	LCC: $1.41 - 1.47/m^3	0.45- 0.47	28 cfu /100 mL*
8. WWTP 1, BC, Canada (Gigliotti 2016)	1000 (avg)	Primary, MBR, UV, Sludge processing	C: 9,500 O: 133	1.25	C: 10,531 - 11,401 O: 119 - 182	1.15 - 1.27	<1 cfu /100 mL*
9. WWTP 2, BC, Canada (Gigliotti 2016)	2000 (avg)	Primary, MBR, UV, Sludge processing	C: 16,000 O: 173	1.2	C: 16,325 - 17,686 O: 149 - 222	1.00 - 1.13	<1 cfu /100 mL*
10. WWTP 3, , BC, Canada (Gigliotti 2016)	3000 (avg)	Primary, MBR, UV, Sludge processing	C: 22,500 O: 226	1.1	C: 21,500 - 23,303 O: 177 - 259	0.96 - 1.09	<1 cfu /100 mL*
11. WWTP 4, BC, Canada (Gigliotti 2016)	1000 (avg)	Primary, AS, UV, Sludge processing	C: 8,600 O: 127	-	C: 6,311 - 6,844 O: 159 - 254	0.69 - 0.81	28 cfu /100 mL*

Note: C: Capital cost, O: Operational and maintenance cost, EI: Energy intensity (kWh/m^3), LCC: Life cycle cost, FBR: Fed Batch Reactor (cost and energy intensity estimated based on MBR (Organica Water Inc. 2012)), RO: Reverse osmosis, UV: Ultraviolet disinfection, Cl2: Chlorination, MBR: Membrane Bioreactor, AS: Activated sludge, BNR: Biological nutrient removal, CF: Coagulation and flocculation, * Max limit of 50 cfu/100 mL (City of Ottawa 2013; MWR 2012).

8.3.1.2 Tool demonstration

The application of the developed FitWater has briefly been demonstrated by considering a community with a population of 10,000. The community evaluated two options for water reuse applications: toilet flushing and all non-potable urban uses. They identified three alternative wastewater treatment trains under each water reuse option. For evaluating these treatment trains, the assigned importance (weight) to the selection criteria: water, cost, health risk, energy, and carbon emissions are medium, very high, medium, very low, and very low respectively. The default values provided within FitWater were considered for microbial concentration, recycling efficiency, and interest rate. The results of the evaluation are shown in Table 8.5.

Table 8.5 Treatment train alternatives for various water reuse purposes and their ranking

Alt.	Treatment units	Water (ML)	Cost ($\$/m^3$)	Risk (DALYs/Yr)	Energy (kWh/m^3)	Carbon ($kgCO_2e/m^3$)	Score	Rank
	Purpose: Toilet flushing							
1A	PS+ BNR+ C&F+ SF+ STD+Cl_2	1022	1.37	2.4E-07	0.84	0.045	73.35	1
2	PS+ MBR+ UF+ STD+ UV	1022	3.27	1.3E-12	1.40	0.074	46.34	3
3	PS+AS+C&F+SF+UF+STD+Cl_2	1022	2.93	1.2E-13	0.83	0.044	59.25	2
	Purpose: Non-potable uses							
1B	PS+BNR+C&F+SF+STD+UV+Cl_2	1022	1.39	8.3E-10	0.91	0.049	72.89	1
2	P + MBR + UF + STD + UV	1022	3.27	2.5E-11	1.40	0.074	46.73	3
3	PS+AS+C&F+SF+UF+STD+Cl_2	1022	2.92	2.32E-12	0.83	0.044	59.71	2

Note: Alt.: Alternative, PS: Primary sedimentation, BNR: Biological nutrient removal, C&F: Coagulation and flocculation, SF: Surface filtration, STD: Sludge treatment and dewatering, Cl_2: Chlorination, MBR: Membrane bioreactor, UF: Ultrafiltration, UV: Ultraviolet radiation, AS: Activated sludge, MF: Microfiltration

Table 8.5 shows that health risk is approximately 10 times larger for Alternatives 2 and 3 in non-potable urban reuses than that in toilet flushing, although treatment trains were the same in both reuses. This is due to larger exposure volume in non-potable reuses (many reuses in addition to toilet flushing) than for toilet flushing. Also, in Alternative 1B for non-potable urban reuses, the use of the same treatment train as toilet flushing would lead to risk higher than the acceptable limit ($>10^{-6}$). Therefore, an UV radiation treatment unit was added to Alternative 1B, compared to Alternative 1A for toilet flushing reuse. Moreover, Alternatives 1A and 1B are the most preferred alternative for each toilet flushing and non-potable urban reuses respectively. The preferred alternative may change with the change of weights to the selection criteria.

8.3.2 Implementation of FitWater

FitWater was implemented to a newly planned neighbourhood for evaluating alternative wastewater treatment train and reuses. The implementation is explained in detail as follows.

8.3.2.1 Study area

The study area is a neighbourhood located in the Okanagan Valley, British Columbia. The neighbourhood has an area of approximately 51 ha with the planned residential population of 4848. The neighbourhood is planned for mixed use comprising of residential, commercial, and institutional buildings. The neighbourhood will have approximately 24% of its area covered by parks and trails as per the information provided by the neighbourhood developer. The neighbourhood has been planned to use reclaimed water in lawn and public parks irrigation.

8.3.2.2 Alternative treatment trains

Urban water reuse potential for various purposes can be evaluated using FitWater. Urban reuses can broadly be classified into three categories based on the literature such as Ahmed et al. (2002) and Asano et al. (2007). These categories may require different minimum water quality grades, namely Class A (Excellent), B (Good), and C (Moderate) as shown in Table 8.6.

Table 8.6 Minimum water quality required for urban reuses

Water reuse applications	Water quality grade
Direct and indirect potable use (groundwater recharge, augment water supply)	Class A (Excellent)
Toilet flushing, gardening, irrigating raw eaten crops, public parks, and golf course, vehicle washing, laundry, fire-fighting, etc.	Class B (Good)
Cooked or processed human food crops, livestock grazing and fodder, human food crops grown over a meter above the ground, non-food crops such as instant turfs, woodlots, and flowers	Class C (Moderate)

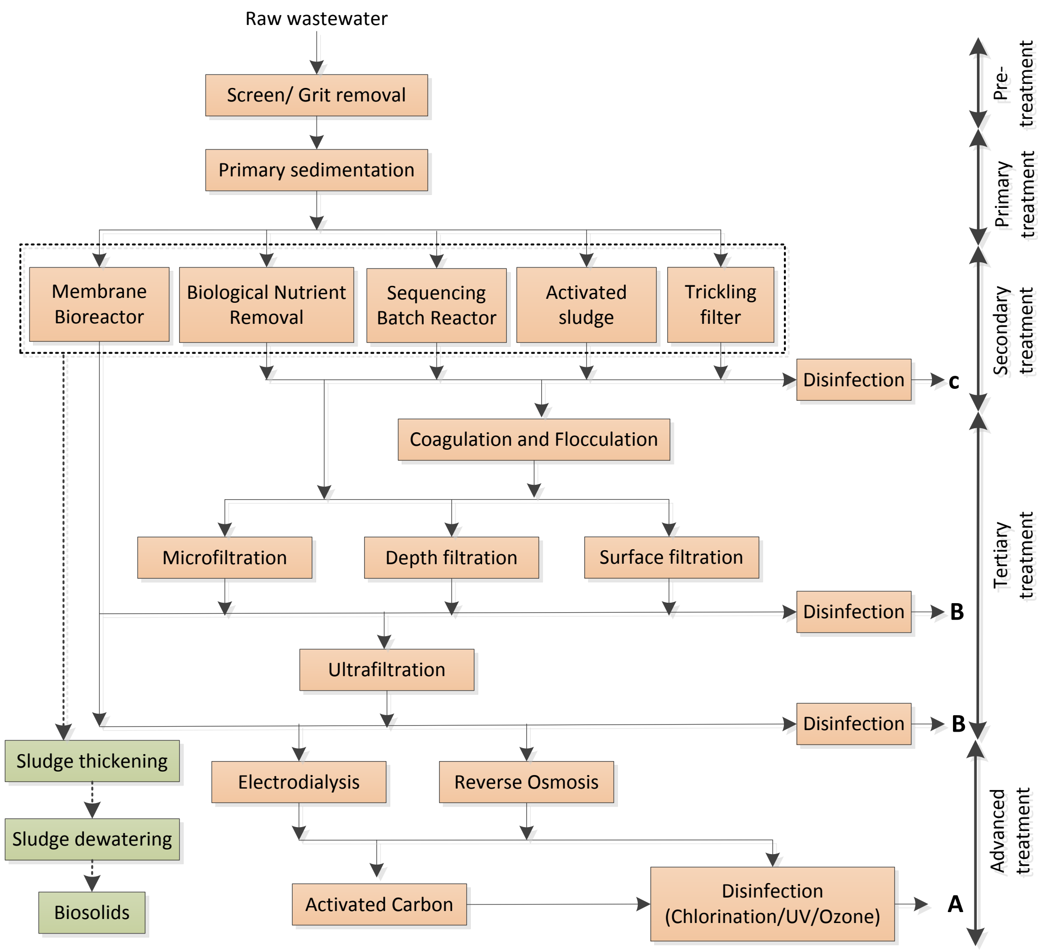


Figure 8.3 Flowchart for preparing alternative treatment trains

Different grades of microbial quality of reclaimed water can be produced by various combinations of wastewater treatment processes. An alternative treatment train can be prepared in FitWater using Figure 8.3. Each treatment train must include pre-treatment, primary treatment, secondary treatment, and sludge processing with disinfection at the end as minimum processes. Other processes of tertiary treatment, advanced treatment, and additional disinfection can be added for acquiring better quality of reclaimed water.

The neighbourhood can use either wastewater (entire) or greywater treatment for reclaimed water production. Altogether 12 alternatives have been developed with the treatment trains as given in Table 8.7. For this analysis, default values of microbial concentration in raw wastewater and water recycling efficiency of treatment plants provided in FitWater were used.

Table 8.7 Alternative treatment trains

Alt. #	Wastewater treatment	Alt. #	Greywater treatment
1	PS+ BNR + CF + SF + SP + UV + Cl_2	7	PS + SBR + Cl_2
2	PS + BNR + MF + SP + Cl_2	8	PS + SBR + SF + Cl_2
3	PS + AS + CF + SF + UF + SP + Cl_2	9	PS + SBR + SF + O_3
4	PS + SBR + CF + UF + SP + UV	10	PS + MBR + UV
5	PS + BNR + CF + SF + UF + SP + UV + Cl_2	11	PS + MBR + O_3
6	PS + MBR + SF + RO + SP + Cl_2	12	PS + MBR + Cl_2

Note: Alt.: Alternative; PS: Primary sedimentation; Coagulation and flocculation: CF; BNR: Biological nutrient removal; AS: Activated sludge; SF: Surface filtration; MF: Microfiltration; UF: Ultrafiltration; MBR: Membrane bioreactor; SBR: Sequencing batch reactor; SP: Sludge processing; UV: Ultraviolet radiation; Cl_2: Chlorination; O_3: Ozonation.

8.3.2.3 Design of wastewater collection and reclaimed water distribution system

The reclaimed water use requires the collection of wastewater and the distribution of reclaimed water for use in lawn and parks irrigation. For the implementation of FitWater in the planned neighbourhood in the Okanagan Valley, a gravity collection system was designed for a central collection of greywater in the neighbourhood using EPANET 2. Since, the flow was by gravity, pumping was not required. The collected greywater would be treated in a greywater treatment plant in the neighbourhood. After treatment, the reclaimed water would be distributed to the same community. So, a reclaimed water distribution system was also designed using EPANET 2 (Aydin et al. 2014). The total pipe length for each greywater collection and reclaimed water distribution was 4.4 km. The estimated EI of reclaimed water distribution is 0.33 kWh/m^3 based on the prepared network design.

The capital (installation) cost and operational and maintenance (repair) cost of pipes were estimated by using the unit cost obtained from Kabir (2016) and a local consulting firm (Focus Engineering 2014) and are also given in Appendix C.2. The unit cost of pipe installation was assumed to be the same for both greywater collection and reclaimed water distribution, whereas the unit cost of pipe operational and maintenance for collection pipes (gravity system) was assumed to be one-third of that of reclaimed water distribution pipes (pressurized system) as such cost would be lower in gravity flow pipes. Similarly, the cost of a water pump was obtained from a pump company (Franklin Electric 2012) and the cost of valves was obtained from a local consulting firm (Focus Engineering 2014). However, for a wastewater (sewage) treatment system alternative, additional collection infrastructure would not be required for a reclaimed water use project as a wastewater collection system would already be in place whether wastewater were reused or not. Furthermore, the EI and unit LCC of the treated wastewater distribution system would be the same as the network design for either treated greywater or wastewater because recycled greywater and wastewater would individually meet the demand of lawn and parks irrigation. The estimated unit LCC of pipe installation was \$1,563/km/yr and that of operation and maintenance of the pipes was \$1,514/km/yr.

Communities may have different importance weights for the evaluation criteria: water, cost, health risk, energy, and carbon emissions. Therefore, four scenarios with varying importance (weights) of these criteria were developed representing different possible communities. The importance ratings of the criteria used in the analysis are presented in Table 8.8.

Table 8.8 Scenarios with varying importance of selection criteria

Scenario	Community	Weights (Importance score)				
		Water	Cost	Risk	Energy	Carbon emissions
1	Economically constrained	L	VH	VL	VL	VL
2	Arid	VH	L	VL	VL	VL
3	Pro-social	VL	L	VH	VL	VL
4	Climate conscious	VL	L	VL	VH	VH

Note: VL: Very low, M: Medium, VH: Very high

8.3.2.4 Ranking of treatment trains

The tool performed a MCDA of alternatives with the input of weights and displayed results in a radar diagram and table for each simulation (scenario). The results obtained from FitWater show that annual reclaimed water production ranged from 306.6 to 324.6 million litres (ML) in Alternatives 1 to 6 (wastewater treatment), and from 232.4 to 246.0 ML in Alternatives 7 to 12 (greywater treatment). In addition, for the best alternative the amount of reclaimed water production, energy intensity, carbon intensity, and unit cost were presented in a water-energy-carbon-cost (WECCo) triangle as shown in Figure 8.4. The ranks of all alternatives in four scenarios are given in Table 8.9.

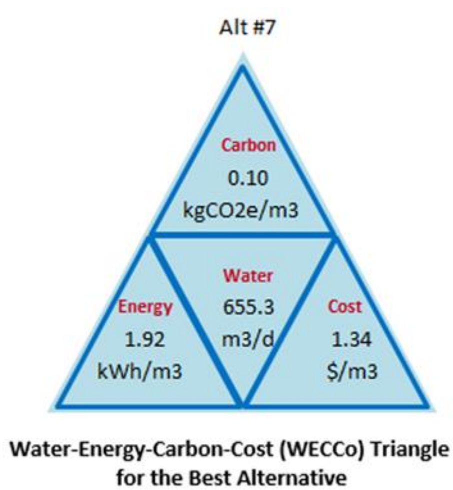


Figure 8.4 WECCo triangle for Alternative 7 in Scenario 1

Table 8.9 Ranks of alternative treatment trains in different scenarios

Rank	Scenario 1		Scenario 2		Scenario 3		Scenario 4	
	Score	Alt #	Score	Alt #	Score	Alt #	Score	Alt #
1	86.84	7	83.65	2	23.23	6	87.44	12
2	85.13	8	82.74	1	22.99	7	84.42	10
3	69.91	9	81.26	3	22.96	12	80.97	2
4	64.21	12	80.55	6	22.57	8	80.14	7
5	62.73	10	80.32	5	18.62	10	79.01	8
6	54.68	11	79.34	4	18.06	9	73.15	3
7	45.78	1	74.93	7	15.51	11	65.76	5
8	41.74	2	74.51	8	15.13	2	61.52	11
9	32.59	3	71.71	12	14.22	1	61.43	1
10	31.30	4	70.52	10	12.84	3	58.37	9
11	30.54	5	70.01	9	12.09	5	53.04	4
12	24.16	6	67.30	11	10.82	4	40.94	6

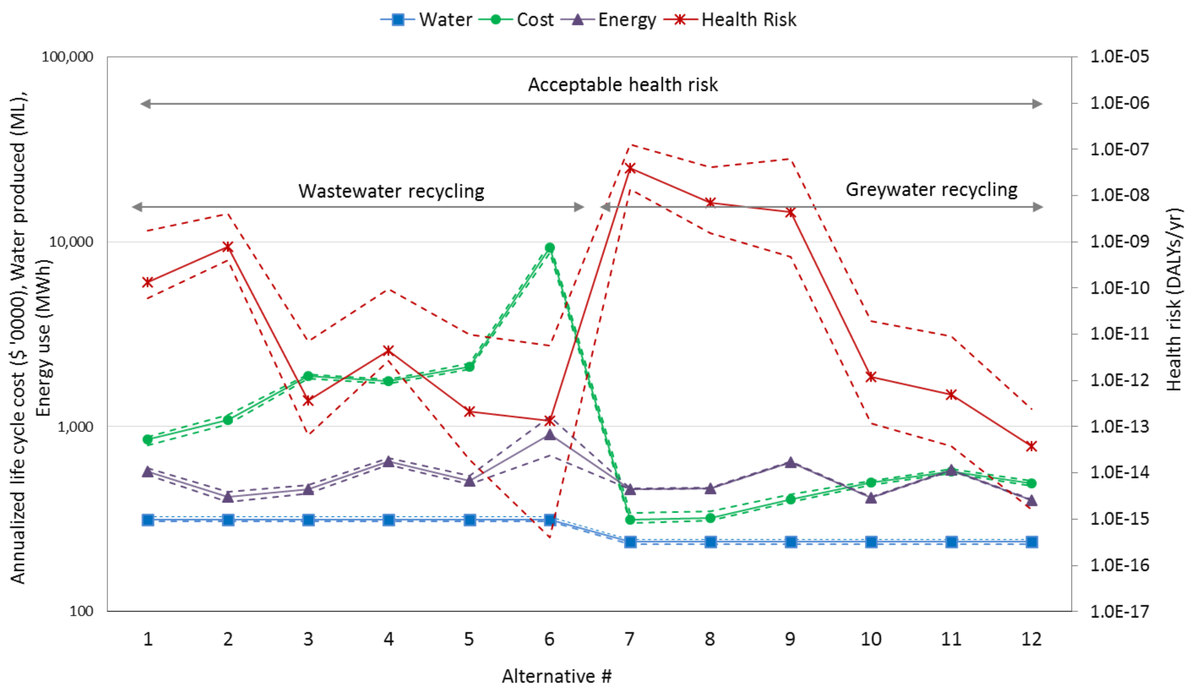
Broadly, Alternatives 7 to 12, i.e., greywater treatments were ranked higher than Alternatives 1 to 6, i.e., wastewater treatments in Scenario 1, an economically constrained community, as it assigned a higher weight to cost. Greywater treatment having a lower volume of wastewater resulted in comparatively lower cost. However, wastewater treatment trains (Alternatives 1-6) were ranked higher than greywater treatment trains in Scenario 2, an arid community, which assigns a higher weight to the volume of water. Obviously, wastewater treatment produces a larger quantity of reclaimed water than greywater treatment. Moreover, in Scenario 3 (Pro-social) and Scenario 4 (Climate conscious), the ranks of various greywater and wastewater treatments were intermixed as these scenarios depend on health risk and energy use respectively. The health risk and energy use by reclaimed water primarily depend on the types of treatment technologies used in a treatment train rather than the volume of wastewater treated.

In Scenario 1 (economically constrained community), Alternative 7 performing greywater treatment was ranked first. This is attributed to the fact that the cost had very high importance and Alternative 7 had the lowest LCC of \$302,000/yr to \$342,000/yr; while water had low importance and the other three criteria had very low importance. In Scenario 2 (arid community), Alternative 2 was the most preferred alternative as water had very high importance. In Alternative 2, entire wastewater would be treated producing one of the largest amount reclaimed water with the lowest energy use (393 to 445 MWh/yr) and carbon emissions (19 to 26 tCO_2e/yr) and the second lowest LCC (\$1,032,000 to \$1,155,000/yr) among the alternatives in entire wastewater treatment. For Scenario 3 (pro-social community), assigning a very high importance to health risk, Alternative 6 was most preferred. Alternative 6 had the least health risk, ranging from 4.1E-16 to 5.7E-12 DALYs/yr. Similarly, in Scenario 4 (climate conscious community), Alternative 12 was ranked first, as it had the lowest energy use of 400 to 404 MWh/yr and the lowest carbon emissions of 19 to 23 tCO_2/yr. Therefore, a community can rank and select an optimal alternative based on their need for water reuse planning using FitWater.

8.3.3 Trade-off analysis

A specific wastewater treatment train with a particular health risk has a definite LCC, reclaimed water production, energy use, and carbon emissions. In fact, these five elements are interconnected and their relationships vary with the types of wastewater treatment trains.

Such variation was analyzed using the data of the previous application of FitWater for the planned neighbourhood (Okanagan Valley) as a case study (Figure 8.5).



Note: Dotted lines show an uncertainty band with upper and lower bounds.

Figure 8.5 Variation of water, health risk, energy use, and life cycle cost in different treatment alternatives

In Figure 8.5, LCC and energy use generally increase with the decrease of health risk and vice-versa. The reduction of health risk of reclaimed water usually requires more number of treatment units in tertiary treatment, advanced treatment, and disinfection. Additional treatment units incur cost and energy use leading to increased LCC, energy use, and carbon emissions. However, it is noteworthy that health risk of all treatment trains and reuse are lower than the acceptable risk of 10^{-6} DALYs/year (WHO 2006b). The health risk, LCC, and energy use were found to interact over alternative treatment trains.

8.3.4 Cost and energy use of reducing health risk in varying plant capacities

The reduction of health risk associated with microorganisms, i.e., improvement of microbial quality of reclaimed water requires higher treatment cost and energy use. The health risk reduction can be measured in terms of log removal of microorganisms in wastewater. One log removal is equivalent to 10-fold health risk reduction. The required treatment cost and energy use per unit log removal of microorganisms in different technologies in secondary and tertiary treatments in various plant capacities (flow rates) were estimated using FitWater. The estimated unit LCC per unit log removal in secondary treatment: trickling filter (TF), sequencing batch reactor (SBR), membrane bioreactor (MBR), activated sludge (AS), and biological nutrient removal (BNR); tertiary treatment: depth filtration (DF), ultrafiltration (UF), and coagulation and flocculation (C&F); advanced treatment: granular activated carbon (GAC), electrodialysis, and reverse osmosis (RO); and disinfection: chlorination, ultra violet radiation (UV), and ozone technologies are shown in Figure 8.6. Similarly, the estimated energy intensity in secondary treatment: trickling filter, SBR, MBR, activated sludge, and BNR; tertiary treatment: surface filtration, depth filtration, and coagulation and flocculation; and disinfection: ozone technology are shown in Figure 8.7.

As shown in Figure 8.6 and Figure 8.7, the unit annualized LCC and energy intensity per unit log removal are highly variable with treatment technologies even within the same treatment stage. Obviously, they are very different for various treatment stages. For instance, unit LCC of BNR and activated sludge in secondary treatment; unit LCC of ultrafiltration and depth filtration in tertiary treatment; unit LCC of GAC and reverse osmosis in advanced treatment; and LCC of chlorination and UV in disinfection. Similarly, the EI of trickling filter and BNR in secondary treatment and EI of surface filtration and depth filtration in tertiary treatment are highly variable.

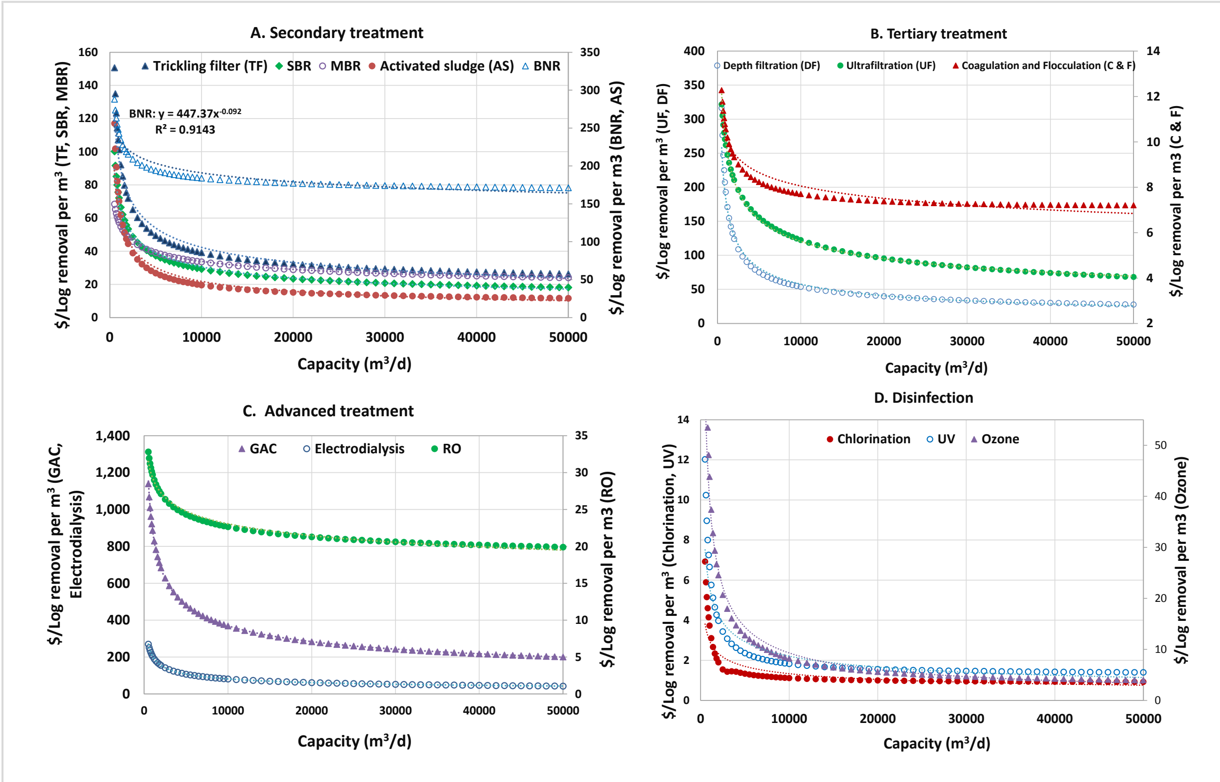


Figure 8.6 Unit annualized LCC per unit log removal for different treatment technologies

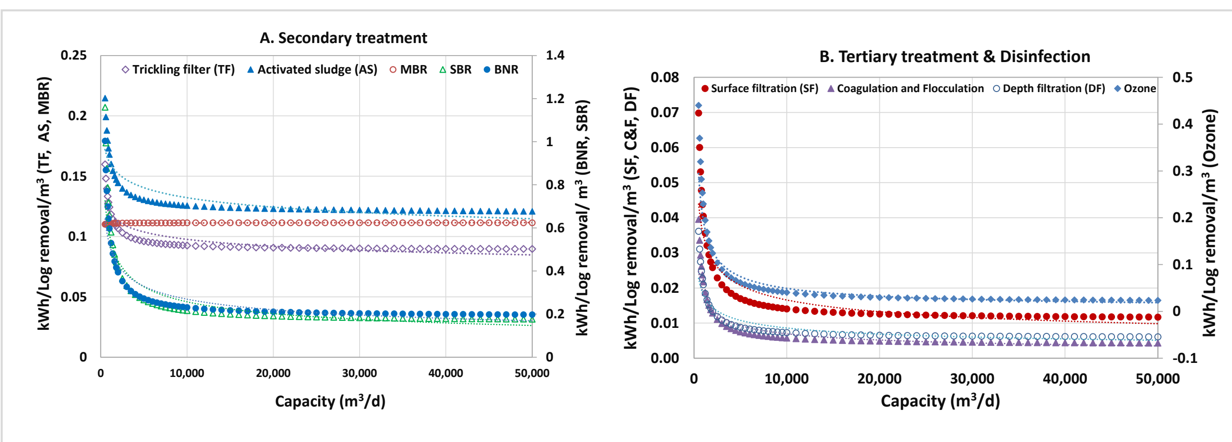


Figure 8.7 Energy intensity per unit log removal for different treatment technologies

The results show that the unit LCC per unit log removal and plant capacity have a power relationship. A power relation indicates a very high LCC when the plant capacity is very low and the LCC decreases sharply with increasing plant capacity for a certain range and then decreases gradually levelling off at a very high plant capacity. A similar power relationship has been observed in between energy intensity per unit log removal and plant capacity. Both power relations clearly show the economies of scale. The equations have been developed for such variation and are given in Table 8.10.

Table 8.10 Equations developed for unit annualized LCC and energy intensity per unit log removal

WW treatment technologies	Unit annualized LCC ($/Log removal/m^3)	r^2	Energy intensity (kWh/Log removal/m^3)	r^2
Trickling filter	$1071.6\,x^{-0.351}$	0.98	$0.2197\,x^{-0.088}$	0.79
Activated sludge	$3553\,x^{-0.468}$	0.98	$0.2972\,x^{-0.088}$	0.80
Biological Nutrients Removal	$447.37\,x^{-0.092}$	0.91	$3.8798\,x^{-0.29}$	0.88
Membrane Bioreactor	$266.98\,x^{-0.224}$	0.99	$0.1096\,x^{0.0016}$	0.74
Sequencing Batch Reactor	$823\,x^{-0.358}$	0.99	$6.2557\,x^{-0.347}$	0.90
Coagulation and Flocculation	$20.518\,x^{-0.101}$	0.92	$0.3435\,x^{-0.423}$	0.92
Depth filtration	$5767.8\,x^{-0.5}$	0.99	$0.17\,x^{-0.324}$	0.89
Surface filtration	-	-	$0.3308\,x^{-0.325}$	0.88
Ultrafiltration	$2830.6\,x^{-0.343}$	0.99	-	-
Granulated Activated Carbon	$12261\,x^{-0.38}$	0.99	-	-
Electrodialysis	$3247.8\,x^{-0.4}$	0.99	-	-
Reverse Osmosis	$60.14\,x^{-0.104}$	0.99	-	-
Chlorination	$33.523\,x^{-0.35}$	0.85	-	-
Ozone	$3036.4\,x^{-0.628}$	0.98	$12.077\,x^{-0.598}$	0.96

Note: y = \$/Log removal/m^3 in cost equations and kWh/Log removal/m^3 in energy equations
x = Treatment capacity in m^3/d

8.4 Discussion

The average and maximum wastewater flow rates are important factors for the planning of reclaimed water use. In the absence of such data, FitWater uses default values of per capita average wastewater generation rates, greywater generation rates, and peaking factors based on Briere (2010) and Environment Canada (2014b). Similarly, the users can input their own values of recycling efficiency or use default values. These features have made the tool user friendly.

FitWater has utilized validated cost equations that have a high r^2 with a value of 0.99 for many technologies (Guo et al. 2014). For consistency in the estimated LCCs, the cost estimated by various equations are expressed in 2015 Canadian dollars. Users can input their own interest rate in terms of TFNs. Moreover, human health risks associated with microorganisms were estimated by applying the widely used QMRA method and presented the results in DALYs/year. The updated dose-response models and parameter values of pathogens were used, as long as they were part of the publically accessible literature, such as for norovirus (Messner et al. 2014; Schmidt 2015), *Campylobacter jejuni* (Haas et al. 1999; Katukiza et al. 2014), rotavirus (Health Canada 2010), and *Cryptosporidium parvum* and *Giardia* spp. (Robertson et al. 2005).

The uncertainty in health risk was approximated by using Monte Carlo simulations. Monte Carlo simulations were performed separately for the estimation of microbial concentration in raw wastewater, exposed water volume, log removal of treatment technologies, and one for the combined computation containing morbidity, disease burden per case, and susceptibility fraction. The 50^{th}, 75^{th}, and 95^{th} percentiles were considered for the TFNs of the respective parameters except that in microbial concentration of raw sewage (5^{th}, 50^{th}, and 95^{th} percentiles). However, users are recommended to input their community specific microbial concentration in the tool. These fuzzy inputs produce the final health risks in terms of TFNs representing the 50^{th}, 75^{th}, and 95^{th} percentiles. By default, FitWater uses all values of TFNs and compute their average. However, FitWater has an option for a risk averse individual to use only the 95^{th} percentile risk and that for a risk taker to use only the median value in the evaluation. It would be better to use a higher percentile risk for the safety of a larger proportion of population.

The energy equations of different technologies were generated based on the energy intensity database given by WEF/EPRI (2013). The database was prepared based on the survey of a large number of WWTPs, i.e., 15,617 (WEF/EPRI 2013). The equations have higher r^2

values with 0.99 or above. However, their estimations may have uncertainties. The total energy use by WWTPs is expressed in TFNs providing a range of output values. Such a range includes the associated uncertainties.

The proposed FitWater tool was tested by applying it to several existing WWTPs. FitWater utilized cost and energy equations that were developed in separate studies. The compatibility of these two categories of equations were evaluated by applying the energy equations to the same case studies used in development of cost equations (Guo et al. 2014). The energy intensities estimated for the same WWTPs used by cost equations, i.e., Big Spring Regional Reclamation Project (US) and Groundwater Replenishment System (Orange County, US), are similar to actual energy intensities. These facts show that the two equation categories are compatible.

The application of FitWater to existing WWTPs shows that the estimated capital cost and operation and maintenance cost of WWTPs even in different countries are similar to their actual values. However, the estimation may still be associated with uncertainty. In fact, the construction cost of WWTPs are affected by several factors, such as project location (remote vs. local), weather, access to site, labor productivity, price fluctuation of materials, political situation, economic instability, fluctuation in currency exchange rate, government policy, and accuracy of planning (Knight and Fayek 2000; Mahamid 2014). Moreover, for consistency in the estimated LCCs, the costs computed by various equations are expressed in 2015 Canadian dollars. In the calculation of LCC, users can input their own interest rate in terms of TFNs.

Similarly, the energy intensities of the existing and planned WWTPs of different countries estimated by FitWater are similar to actual values, which also shows the effectiveness of the tool. However, their estimations may still have uncertainties. The total energy use by WWTPs is presented in TFNs providing a range of outputs. Such a range captures the associated uncertainties.

The improvement in reclaimed water quality for reducing health risk requires additional treatment processes that incur surplus cost and energy use. Therefore, trade-offs exist between health risk, cost, and energy use. Such trade-offs can be analyzed by developing and simulating several alternative treatment trains in FitWater. In addition, the unit annualized LCC and energy intensity of reducing health risk can also be estimated using the tool. The estimated unit annualized LCC and energy intensity per unit log removal of microorganisms or per 10-fold health risk reduction are highly variable with technologies in each treatment stage. Equations have been developed for unit annualized LCC and energy intensity per unit log removal of microorganisms in various plant capacities, which indicate the economies of scale. This is so because their source cost equations (Guo et al. 2014) and energy data (WEF/EPRI 2013) show economies of scale as reality.

Although the primary treatment is for the physical removal of suspended materials and secondary treatment is for the biological removal of dissolved and colloidal organic matter, both treatments also remove a certain level of microorganisms. In fact, the removal of microorganisms is primarily performed by disinfection and tertiary and advanced treatment. These treatment stages should be focused for the removal of microorganisms. Log removal is particularly high for bacteria and viruses in chlorination (Health Canada 2010), and is higher for protozoa (*Cyptosporidium parvum* and *Giardia* spp.) in depth filtration (Health Canada 2012), higher for bacteria and protozoa in GAC (Hijnen et al. 2010), and also higher for all considered microorganisms in MBR, ultrafiltration, RO, electrodialysis, UV, and ozone (EPA-Ireland 2011; Hai et al. 2014; Health Canada 2010, 2011, 2012, 2013b; Xu et al. 2002).

As noted in the introduction of the tool, primary treatment, secondary treatment, sludge processing, and disinfection are minimum treatment stages for wastewater treatment, but greywater treatment may not need sludge processing depending on the strength of the greywater (Li et al. 2009). Although this DST focuses only on microbial water quality, these four minimum stages will considerably reduce biological pollutants (BOD and COD) (Ahmed et al. 2002; Li et al. 2009). This shows that the alternative treatment trains developed for the safe microbial quality of reclaimed water would also produce physio-chemically high quality treated water. Furthermore, the FitWater tool can be extended to include the physio-chemical quality of wastewater, additional wastewater treatment technologies, other water reuse application types, energy use other than electricity, and carbon emissions from wastewater processes. FitWater can be used to approximate the economic impacts of developing microbiologically stringent effluent standards. The developed DST can be applied for the screening and planning of reclaimed water use in a community with wastewater flow higher than 500 m^3/d (Ahmed et al. 2002; Guo et al. 2014; WEF/EPRI 2013).

8.5 Summary

Water reuse has increasingly been practised around the world. Wastewater can be treated based on the intended end use of reclaimed water by considering the economic viability and environmental sustainability, which is referred to as fit-for-purpose wastewater treatment (WWT). WWT technologies differ mainly in terms of cost, treatment efficiency, energy use, and associated carbon emissions. The planning and screening of water reuse projects requires a decision support tool to evaluate alternative WWT train and reuses. However, such tool is not available in the publically accessible literature. A tool, FitWater, has been developed for the evaluation of wastewater treatment for various urban reuses. The evaluation is based on the criteria: amount of reclaimed water production, health risk of water reuse, cost, energy use, and carbon emissions. The cost is estimated as annualized life cycle cost and health risk is estimated using quantitative microbial risk assessment. Uncertainty analysis has been performed using probabilistic and fuzzy-based methods. Multi-criteria decision analysis using fuzzy weighted average is employed to aggregate different criteria and generate a final score. Fitwater ranks alternative wastewater treatment trains based on the final score.

The proposed FitWater has been tested and implemented. FitWater has been tested with several existing WWT plants in Canada and USA, which shows FitWater is useful in estimating life cycle cost (LCC), health risk, and energy use. FitWater is also implemented to a newly planned neighbourhood in the Okanagan Valley (BC, Canada) by developing 12 alternative WWT trains for water reuse in lawn and public parks irrigation. The results show that FitWater can effectively screen the alternatives based on the LCC, health risk, the amount of reclaimed water, energy use, and carbon emissions. Moreover, relationships have been developed for the variation of unit annualized LCC and energy intensity per unit log removal of microorganisms in different treatment technologies with varying plant capacities. The relationships have power relations, showing the economies of scale. FitWater can be applied to identify a cost-effective, risk-acceptable, and energy efficient wastewater treatment train with a plant capacity of 500 m^3/d or more. Furthermore, FitWater can be used to approximate the economic impacts of developing microbiologically stringent effluent standards. The capability of FitWater can be enhanced by including physio-chemical quality of wastewater, additional treatment technologies, and carbon emissions from wastewater decomposition processes.

Chapter 9 System Dynamics Analysis of Economic and Energy Efficiency of Net-Zero Water Communities

A conference paper has been published and a journal paper is under review as part of this chapter as follows:

- A conference paper has been published in the *Proceeding of Canadian Society for Civil Engineering (CSCE) conference* with a title "System dynamics modelling for an urban water system: net-zero water analysis for Peachland (BC)" (Chhipi-Shrestha et al., 2015b).
- A journal paper has been submitted and is under review in the *ASCE Journal of Sustainable Water in the Built Environment* with a title "Economic and energy efficiency of net-zero water communities: A system dynamics analysis" for possible publication (Chhipi-Shrestha et al. 2017g).

9.1 Background

Decision support frameworks and tools on net-zero water (NZW) are limited due to the relative newness of the NZW concept (Joustra and Yeh 2014a). The previously developed WEC model (Chapter 5) is unable to perform economic evaluation of UWSs. Economics is an important triple-bottom-line dimension for evaluating urban water sustainability (VanLeeuwen and Marques 2013; World Bank 2003). An UWS is economically sustainable when water and wastewater revenues equal or exceed expenses for, at least, the operational and maintenance costs (World Bank 2003). Therefore, the WEC model should be able to estimate the LCC of UWSs for assessing their sustainability. However, there is a lack of decision support tool to assess the location specific economic and environmental feasibility of NZW development. This research has added a cost module in the WEC model developed in Chapter 5 and performed the economic analysis of the WEC nexus incorporating uncertainty. Also, hydraulic equations have been added to the model to estimate the energy required for water conveyance (source to water treatment plant) and distribution of drinking water and reclaimed water. The cost embedded WEC model has been used to analyze the environmental factors affecting the economic and energy efficiency of NZW communities.

9.2 Methodology

The WEC model developed in Chapter 5 has been extended by adding a cost module and hydraulic equations to estimate the energy for water and reclaimed water flow in the energy module using Stella 10.1.3® (ISEE Systems 2016). The new cost module and added components in other modules are as follows:

9.2.1 Cost module

The life cycle cost (LCC) was modelled for an entire SMUWS: water conveyance, water treatment, distribution, wastewater collection, treatment, and reclaimed water distribution. It's stock and flow diagram is given in Appendix F.1. The LCC was estimated as the sum of capital cost, operation cost, and repair and replacement cost of water, wastewater, and storm water pipes; LCC of water and wastewater treatment; and energy cost of water conveyance, water and reclaimed water distribution. The net present value (NPV) of the annualized LCC of WDSs was estimated by using Equation 6.13 in Chapter 6 (Davis et al. 2005). Other cost equations are as follows:

a) Cost of drinking water system (freshwater)

$(C_{WC})_t = (UC_{WCI} + UC_{WCOM})_t * L_{WC} + (EI_{WC} * V_w * UC_E)_t$ **Equation 9.1**

$(C_{WT})_t = (UC_{WT} * V_W)_t$ **Equation 9.2**

$(C_{WD})_t = (UC_{WDI} + UC_{WDOM})_t * L_{WD} + (EI_{WD} * V_w * UC_E)_t$ **Equation 9.3**

$(C_{WS})_t = (C_{WC} + C_{WT} + C_{WD})_t$ **Equation 9.4**

b) Cost of wastewater treatment and reuse system

$(C_{WWC})_t = (UC_{WWCI} + UC_{OMWWC})_t * L_{WWC}$ **Equation 9.5**

$(C_{WWT})_t = (UC_{WWT} * V_{WW})_t$ **Equation 9.6**

$(C_{RWD})_t = (UC_{RWDI} + UC_{OMRWD})_t * L_{RWD} + (EI_{WD} * V_w * UC_E)_t$ **Equation 9.7**

$(C_{WWTR})_t = (C_{WWC} + C_{WWT} + C_{RWD})_t$ **Equation 9.8**

c) Cost of storm water harvesting

$$(C_{SWC})_t = (UC_{SWCI} + UC_{SWCOM})_t * L_{SWC}$$ **Equation 9.9**

where C is cost ($), UC is unit cost ($/m for WC, WWC, RWD, and SWC; $/L for WT, WWT; and $/kWh for E), L is length of pipe (m), V is volume (L), EI is energy intensity (kWh/m^3), E is electricity (kWh), and subscripts WC, WWC, SWC are water, wastewater, storm water collection; WCI, WWCI, and SWCI are installation of water, wastewater, and storm water collection infrastructure; WCOM, WWCOM, SWCOM are operational and maintenance of water, wastewater, and storm water collection infrastructure; WT and WWT are water and wastewater treatment; WDI and RWDI are installation of water (freshwater) and reclaimed water distribution infrastructure; WDOM and RWDOM are operational and maintenance of water and reclaimed water distribution, WD and RWD are water and reclaimed water distribution; WS and WWTR are water system and wastewater treatment and reuse; and "t" refers to time (month). It is assumed that the collected storm water will be treated to the same wastewater treatment plant as of sewage.

d) Cost of rainwater harvesting

$$UC_{RWH} = 14.347\, e^{(-0.015\, * {P_a}/_{10})}$$ **Equation 9.10**

where UC_{RWH} is unit cost of rainwater harvesting ($\$/m^3$) and P_a is annual precipitation (mm), and it has r^2 of 0.93. The equation has been developed from the data of Wang and Zimmerman (2015).

The energy and water modules of the WEC model have been updated for net-zero analysis. The added components in the modules are given below.

9.2.2 Energy module

In the energy module, basic energy equations were added for estimating the energy required for water conveyance (source to water treatment plant) and for distribution of drinking water and reclaimed water.

a) Energy for freshwater conveyance

$$D = \sqrt{4 * Q/(\Pi * v)}$$ **Equation 9.11**

$$h_f = 10.674\, L * C_{HW}{}^{-1.852} * D^{-4.871} * Q^{m1}$$ **Equation 9.12**

where D is diameter of pipe (m), Q is discharge (water demand) (m^3/s), v is velocity (1.3 m/s) (Briere 2010), h_f is head loss due to friction (m), L is length of pipe (m), C_{HW} is Hazen-William constant (150), and m_1 is 1.85 (Briere 2010).

$$h_e = L * Sin\,\theta$$ **Equation 9.13**

where h_e is net elevation head (m), L is length of pipe for water conveyance (m) and θ is average gradient of conveyance pipe (^{0})

The values of h_f and h_e are used to obtain the head at pump (H_p). The estimated H_p is used in Equation 9.15 to compute the energy for water conveyance (kWh).

b) Energy for water (finished and reclaimed) distribution

The pumping energy for drinking water and reclaimed water distribution can be estimated by using Equations 9.15 (Cheng 2002).

$$E_t = [\frac{\gamma\, Q\, H_p\,(1+\alpha)}{\eta * \eta_t} * (1+f) * T * W]_t$$ **Equation 9.14**

$$E_t = (2.23 * 10^{-3} * \gamma Q H_p * T)_t \text{ for } \alpha = 0.2,\ f = 0.3,\ \eta = 0.7 \text{ and } \eta_t = 1$$ **Equation 9.15**

where P is power of lift pump (kW), γ is specific weight of water (9806 N/m^3), Q is pumping capacity of lift pump (m^3/s) which can be estimated from an average water discharge, H_p is the sum of average pressure requirement at houses (m) and net elevation head between reclaimed water production site and maximum elevation point of service area (m); α is safety factor of pumping power (0.1 to 0.2), η is pump efficiency (65% to 85%), η_t is mechanical transmission efficiency (92% to 100%), E_h is energy consumed (kWh), f is friction loss within pipe and is assumed to be 40% as a conservative value, as 30% is used for 15 m lengths by Cheng (2002), T is time duration that a pump is operated (secs) (Cheng 2002), W is a conversion factor for Joule to Kilowatt-hour (kWh) (2.78E-07), and "t" refers to a month.

$$(EI)_t = (1000 * \frac{E_t}{V})_t \quad \textbf{Equation 9.16}$$

where EI is energy intensity (kWh/m^3), E_t is energy consumption (kWh) for specific urban water stage, V is volume of water (L), and t refers to a month.

c) Energy for RWH

$$EI_{RWH} = 8.74\, e^{(-0.015 * {P_a}/{10})} \quad \textbf{Equation 9.17}$$

where EI_{RWH} is energy intensity of rainwater harvesting (kWh/m^3) and P_a is annual precipitation (mm), and it has r^2 of 0.85. The equation has been developed from the data of Wang and Zimmerman (2015).

9.2.3 Water module

The equations added in the water module of the WEC model are as follows:

a) Rainwater and storm water harvesting

The amount of water collected by rainwater harvesting (RWH) has been modelled in the WEC model. The equation for the amount of water collected by storm water harvesting (SWH) was added in the WEC model. Both equations are as follows (Kim et al. 2016; RiverLink 2014).

$$(RWH_R)_t = {P_t}/{1000} * A_{Rr} * H_r * 1000 * \eta_f * K_r \quad \textbf{Equation 9.18}$$

$$SWH_t = {P_t}/{1000} * (A_B * 10{,}000 - A_{Rr}) * 1000\ K_u \quad \textbf{Equation 9.19}$$

where RWH and SWH are amount of rainwater harvesting (L) and storm water (L) harvested; P is average monthly precipitation in the form of rain (mm); A is area (m^2); H is harvestable area proportion (0.8 for roofs, assumed); η is rainwater filtration efficiency (0.9) (Wang and Zimmerman 2015); K is runoff coefficient (0.85 for roofs and 0.7 for urban built up area) (Kim et al. 2016; RiverLink 2014); subscripts R is residential, r is roof, p is proportion, B is built-up area (ha), u is urban, and t refers to time (month).

b) Water stress index

The water stress index of SMUWSs was added in the WEC model as follows (Jiménez Cisneros 2014).

$$Water\ stress\ index = \frac{Water\ abstraction}{Water\ availability} * 100 \quad \textbf{Equation 9.20}$$

where water availability is actual water available after meeting ecological flows, including creeks, lakes, and groundwater. It can be approximated as licensed water.

c) Net-zero water

The potential of a SMUWS to achieve net-zero water can be estimated using Equation 9.21.

$$(NZW)_t = (RW)_t + (RWH)_t + (SWH)_t - W_t \quad \textbf{Equation 9.21}$$

where NZW is net-zero water, W is water use, RW is reclaimed water use, RWH is rainwater harvested, SWH is storm water harvested, and "t" refers to a month (time).

9.2.4 Uncertainty analysis

The uncertainty in the results produced by the extended WEC model was approximated by using Monte Carlo simulations of 10,000 runs in Stella Professional® 1.0.3 (ISEE Systems 2017). For most input data that lack distribution, a uniform distribution was assumed (Sadiq et al. 2004b), whereas log-normal and triangular distributions were used for others. Altogether 121 parameters and variables, which are relatively important based on Venkatesh et al. (2014) and Kenway et al. (2008) were used for Monte Carlo simulations. The list of input parameters and variables used in the analysis are enlisted in Appendix F.2. These inputs are both time-variables and non-time variables.

9.2.5 Data requirements

The cost embedded WEC model has been applied to the City of Penticton. The data on water consumption by various sectors of the SMUWS was estimated using the use rate and duration of water fixtures and appliances based on Mayer et al. (1999), ENERGY STAR (2014a), ENERGY STAR (2014b); US EPA 2009b; Dziegielewski et al. (2000); and (Briere 2010) and irrigation rates for lawns, public parks, and agricultural land based on OBWB (2010) and Statistics Canada (2015b). These data were already validated using the historical data of City of Penticton in Chapter 5. Similarly, the data on energy use by utilities for raw water collection, water treatment, distribution, wastewater transport, and wastewater treatment were obtained from the City of Penticton (City of Penticton 2015c; d). The GHG emissions from energy use were estimated by using the GHG emission factors of the respective energy sources (Ministry of Environment 2013). The GHG emissions from wastewater processes were estimated based on the IPCC methodology (IPCC 2006). The detailed data on water consumption, energy use, and carbon emissions and their validation are provided in Chapter 5; however, the data used for Monte Carlo simulations are given in Appendix F.2. Moreover, the analysis in this Chapter has included only the operational energy use in utilities excluding the embodied energy of chemicals and energy consumption by indoor hot water use and the related carbon emissions. The exclusion of indoor water heating energy would prevent the masking of output energy use value as hot water use consumes ~90% of the energy used in the SMUWS (Chapter 5). Moreover, the historical rainfall data of Penticton was obtained from the governmental database for the years of 1980 to 2014 (Government of Canada 2016).

For the cost module, a total pipe length of water conveyance, water distribution, and wastewater collection were obtained from the City of Penticton (City of Penticton 2013, 2014a). The length of reclaimed water distribution and storm water collection was assumed to be the same as that of wastewater collection, as all three sewer, reclaimed water distribution, and storm water collection systems would primarily serve the built-up area. The unit costs of water mains and wastewater mains installment and their repair/replacement were estimated assuming the average diameter of pipes that would be required in the system (Appendix C.2). The price of electricity of 9.921 cents/kWh was used (Fortis BC 2016).

9.3 Results

The results obtained from the simulations of the WEC model are explained in the following sections.

9.3.1 Economics of WEC nexus of NZW communities

The economics of the WEC nexus was analyzed for NZW in the City of Penticton, British Columbia (BC), Canada. The city has an area of 42.1 sq. km with a population of 32,877 and a growth rate of 0.6% per year in 2011 (Statistics Canada 2015a). The city operates a drinking water treatment plant for water supply. It also manages wastewater generated in the city using an advanced wastewater treatment plant comprising of biological nutrient removal technology. Moreover, the impacts of environmental factors: precipitation amount, source water proximities, and topography on NZW were analyzed for a typical community with an UWS as of the City of Penticton. The potential for the city to achieve NZW was analysed using the extended WEC model. Five different scenarios were developed for the analysis as given in Table 9.1.

Table 9.1 Scenarios for net-zero water analysis

Scenarios	Features	Remarks
1	Business as usual	No reuse assumed although some public parks and golf courses irrigated by treated water; such areas excluded from analysis
2	Treated wastewater reuse	Potable & non-potable use
3	Scenario 2 with rainwater harvesting in residences	Potable & non-potable use after treatment
4	Scenario 3 with storm water harvesting and reuse	Potable & non-potable use after treatment
5	Scenario 2 with storm water harvesting and reuse	Potable & non-potable use after treatment

A scenario analysis was conducted for achieving NZW from 2016 to 2035. The monthly variation of net water, total energy use, life cycle cost, and total carbon emissions of the SMUWS in 10 years are given in Figure 9.1. These elements highly vary between months, i.e., seasons due to high irrigation demand ranging from 38% to 50% of the total water demand by outdoor lawn and public parks. The net water was lowest in July as it has the highest irrigation demand (OBWB 2010; Statistics Canada 2015b). As can be expected, the net water was higher in non-irrigating months, January to March, November, and December. The total energy use, LCC, and carbon emissions were highest in July in Scenarios 1 and 2, but they were highest in June in Scenarios 3 to 5. This is because Scenarios 3 to 5 have rainwater harvesting (RWH) and/or storm water harvesting (SWH) and monthly rainfall is highest in June in Penticton (Government of Canada 2016). In addition, the estimated energy intensity (EI), carbon intensity, and unit cost are higher for RWH compared to wastewater treatment and reuse in the city (Scenarios 3 and 4). Moreover, the treatment of all storm water was considered in Scenario 5 resulting in high EI, carbon intensity, and unit cost per unit of water use.

Interestingly, although the total energy use was higher in Scenario 5 than in Scenario 3, the total LCC was lower in Scenario 5 compared to Scenario 3, because Scenario 3 consists of reclaimed water use consisting of about 25% water supply by RWH, and Scenario 5 has reclaimed water use with SWH but without RWH. The energy intensity of RWH (5.6 kWh/m^3) is only slightly higher than the EI of a wastewater system with SWH excluding RWH (4.1 kWh/m^3), whereas the unit cost of RWH (\$9.1/m^3) is much higher than that of a wastewater system with SWH (\$3.4/m^3). Furthermore, the Monte Carlo simulations show a very high annual variability. For instance, in Scenario 1 average annual water use, energy use, LCC, and carbon emissions varied from 3,065 ML to 19,901 ML; 8.2 GWh to 23.3 GWh; \$6.7 million to \$20.7 million, and 23 to 76 tCO_2e respectively.

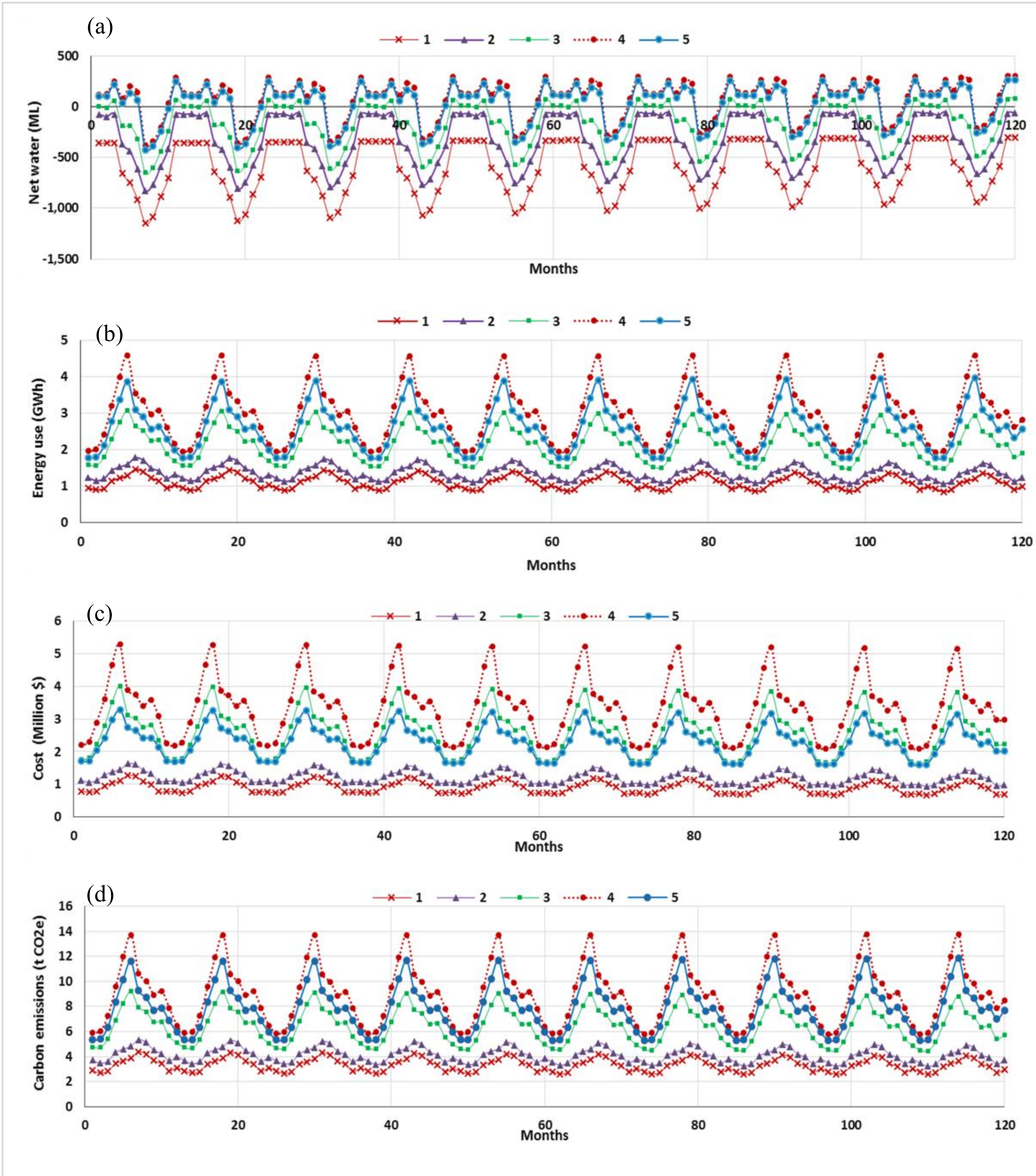


Figure 9.1 Median monthly net water and its energy use, cost, and carbon emissions

The average annual net water and water stress index from 2016 to 2035 are presented in Table 9.2. As shown in Table 9.2, the average annual net water of SMUWS is anticipated to gradually increase from – 7200 ML in Scenario 1 to 1300 ML in Scenario 5. Similarly, the water stress index is anticipated to gradually decrease from approximately 20% in Scenario 1 to approximately 0% in Scenario 5, indicating a large decrease of freshwater withdrawal. Compared to Scenario 1, net water can be increased by 39% using wastewater reuse and by 68% using wastewater reuse and RWH. By further including storm water harvesting, net water can be increased by 115%. However, using treated wastewater and storm water only, net water can be increased by 105%.

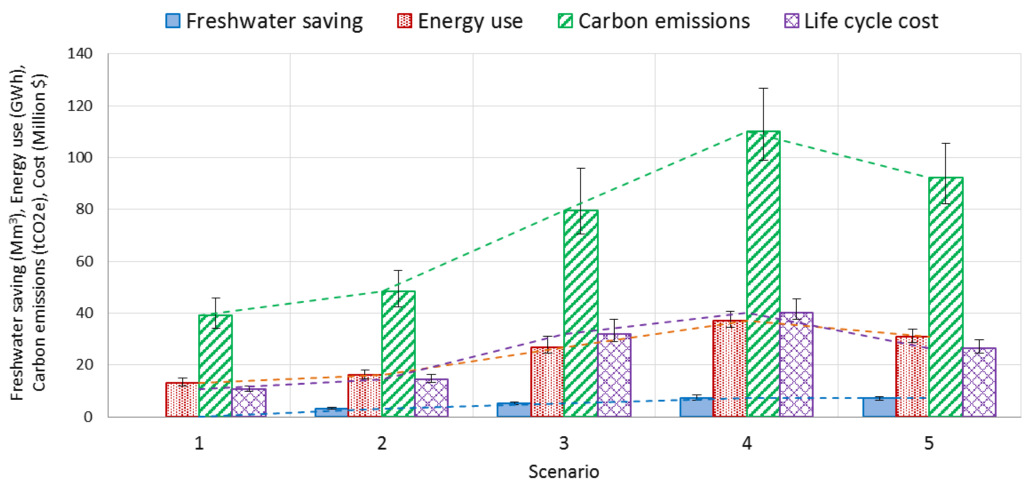
Table 9.2 Average annual net water in different scenarios for 2016 to 2035

Scenarios	Net water balance (Mm^3)	Average % change from Scenario 1	Water stress index (%)	Water reuses/ harvesting
1	(-8.6, -7.2, -6.4)*	NA	(17.6, 19.8, 23.4)	-
2	(-5.2, -4.0, -3.3)	39	(9.0, 10.9, 14.3)	WW
3	(-2.7, -1.6, -0.9)	68	(2.6, 4.4, 7.5)	WW, RWH
4	(1.3, 1.6, 1.8)	115	(0, 0, 0)*	WW, RWH, SWH
5	(0.4, 1.3, 1.5)	105	(0, 0, 0)*	WW, SWH

Note: (i) * Values in parenthesis includes 5th, Median, and 95th percentile respectively; (ii) RWH: Rain water harvesting: WW: Wastewater; SWH: Storm water harvesting; *mean water stress index is 1.9% and 2.3% for Scenarios 4 and 5 respectively.

Among the different scenarios, net water was lowest in Scenario 1 as it did not use reclaimed water at all. The net water was highest in Scenario 4, which was slightly larger than Scenario 5, which only considered SWH (without RWH). The total water recycling efficiency of SWH is 0.56, comprising an urban run-off coefficient of 0.7 (RiverLink 2014) and WWT recycling efficiency of 0.8 (City of Penticton 2014a), whereas RWH (harvesting a part of storm water in the catchment) considered a total recycling efficiency of 0.77, and is comprised of a roof runoff coefficient of 0.85 (CMHC 2012; Kim et al. 2016) and filtration efficiency of 0.9 (Wang and Zimmerman 2015). In reality, on-site RWH would be more water efficient than centralized storm water harvesting for the harvested portion of rainwater.

The annual freshwater saving, total energy use, total carbon emissions, and LCC of the SMUWS are shown in Figure 9.2. Interactions between freshwater saving, energy use, and LCC over scenarios exist as the rates of change of freshwater saving, energy use, and LCC differ with secarnios (Figure 9.2). Scenario 4 was found to have the highest energy use, LCC, and carbon emissions with the highest freshwater saving. The resource use and carbon intensities were approximately, 4.9 kWh/m^3, 14.8 tCO_2e/ Mm^3, and \$5.6/m^3 of water use in Scenario 4 compared to 1.8 kWh/m^3, 15.5 tCO_2e/ Mm^3, and \$1.2/m^3 of water use in Scenario 1. Although freshwater savings are higher in Scenarios 4 and 5, they could be energy intensive and costly.



Note: An error bar indicates an uncertainty range with 5th and 95th percentiles and dotted lines connect median values showing interactions

Figure 9.2 Median annual freshwater saving, energy use, cost and carbon emissions in different scenarios

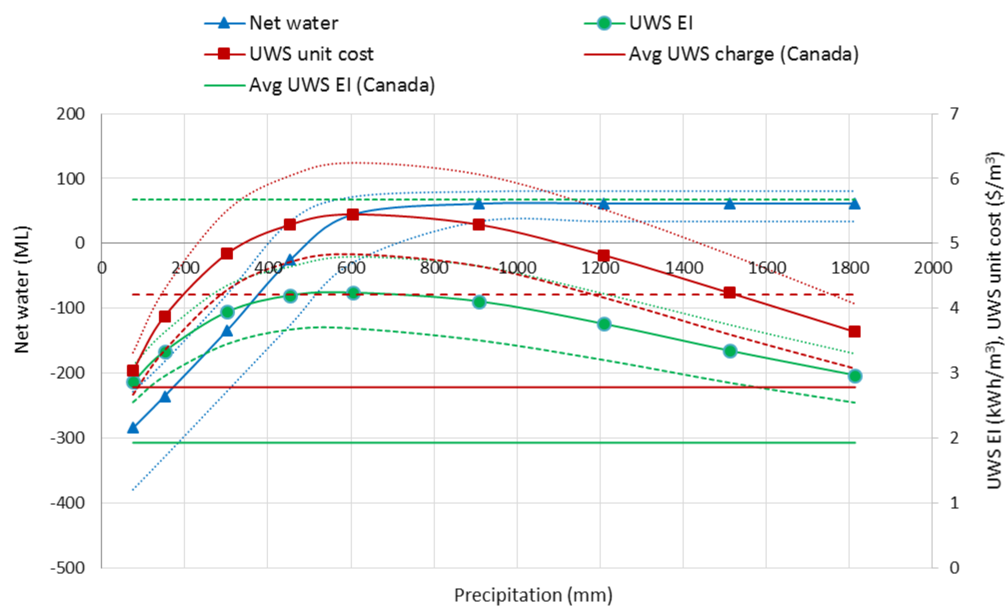
9.3.2 Economic and energy use impacts of environmental factors on NZW

The important components of NZW development are the amount of energy use and cost of RWH and wastewater reuse. The energy efficiency and economic viability of RWH primarily depend on climate, i.e., amount of precipitation in a community. However, the energy efficiency and economic viability of wastewater reuse can mainly be affected by the degree of advanced wastewater treatment applied and the infrastructure requirements for secondary distribution of treated wastewater. Secondary distribution is indeed affected by distance and net elevation head from a wastewater treatment plant to end users of reclaimed water. Assuming a wastewater treatment plant within a community, the primary factor affecting secondary distribution is topography, i.e., net elevation head between water reclamation station and the highest altitude of the reclaimed water service area. Moreover, reclaimed water use can save partial energy use and cost of freshwater abstraction, i.e., withdrawal and conveyance, from source to a water treatment plant. Usually for developing NZW in any community, wastewater reuse and RWH are important. Therefore, the impacts of precipitation amount, conveyance length, and net elevation head on energy use and cost of NZW development were assessed in detail based on Scenario 3 as follows.

9.3.2.1 Impact of precipitation amount on NZW

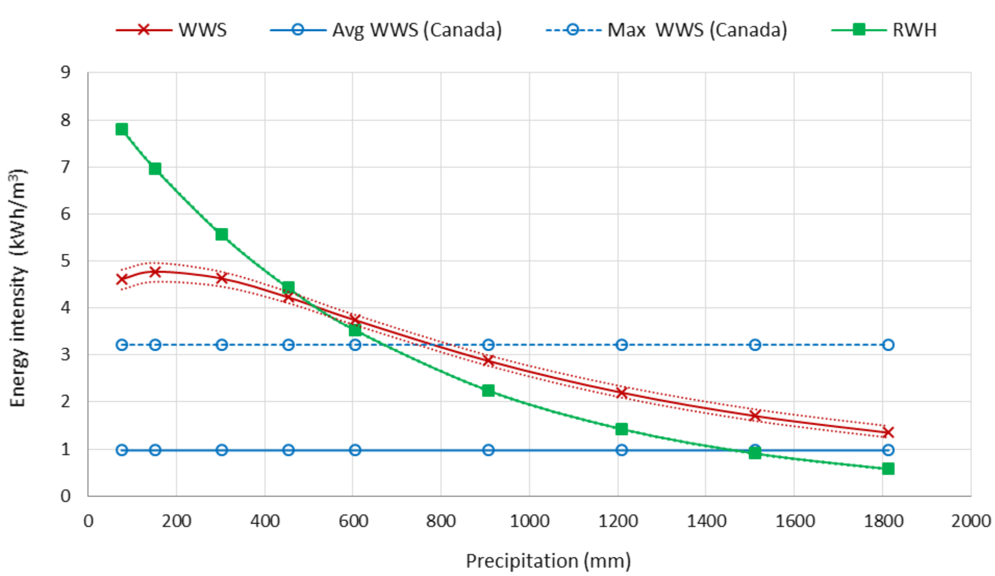
The amount of precipitation, mainly rainfall, highly affects the capacity of rainwater and storm water harvesting. The energy use and cost of NZW development in regions with varying annual precipitation were analyzed considering a typical community (Scenario 3) with an UWS similar to Penticton but located in different precipitation zones. The annual net water, overall EI, and unit cost of the entire SMUWS of the community in various precipitation zones are shown in Figure 9.3. Net water would almost be constant beyond the precipitation of approximately 1000 mm, because the household RWH would satisfy all the monthly residential demand with that precipitation and the excess harvested water would be discharged on-site as natural rain water. Interestingly, the monthly analysis shows that even an annual precipitation of 900 mm would be able to satisfy all the monthly residential water demands, but only after 2022 (Year 7).

The overall EI and unit cost of the SMUWS would be highest in the annual precipitation of about 600 mm (Figure 9.3). This is because EI and unit cost of RWH as part of reclaimed water use are higher in low precipitation regions as shown in Figure 9.4 and Figure 9.5. Also, the proportion of harvested rainwater compared to the total water demand would be lesser in low precipitation leading to a lower overall EI and unit cost. Both median EI and unit cost of the SMUWS were higher than average EI (1.9 kWh/m^3) (AECOM 2012) and average unit charge (water and wastewater) of the Canadian UWS (\$2.8/m^3) (AECOM 2012) even at the precipitation of 1800 mm. However, the median EI was lower than the maximum EI of the Canadian UWS (5.7 kWh/m^3) (AECOM 2012) in the entire precipitation range. Also, the median unit cost was lower than the maximum unit charge of the Canadian UWS (\$4.2/m^3) (AECOM 2012) (considering water and wastewater charge) beyond 1500 mm. In fact, the increased amount of RWH simultaneously decreases the energy use and cost of drinking water system (DWS) and wastewater reuse in NZW development.



Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

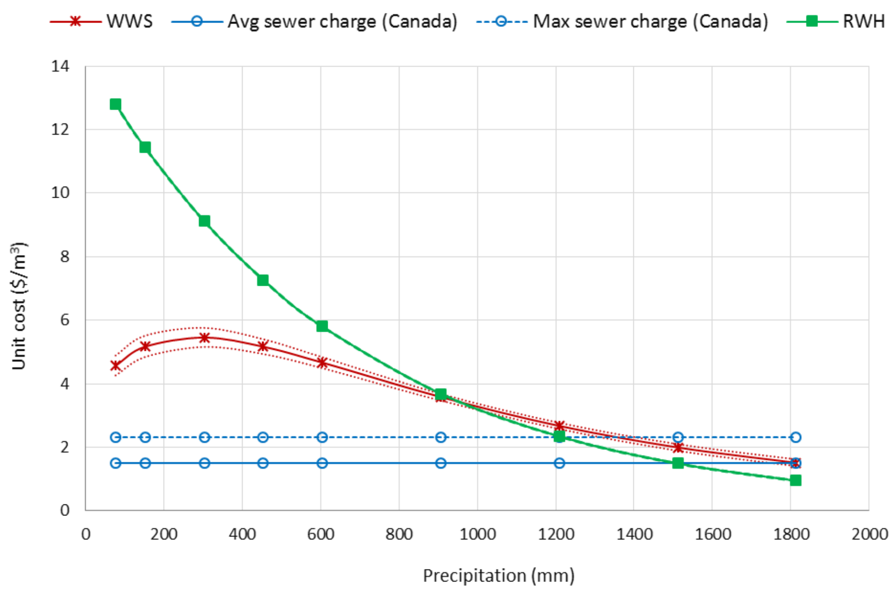
Figure 9.3 NZW and its energy intensity and unit cost in different precipitation



Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

Figure 9.4 Energy intensities of wastewater system and RWH in different precipitation

As shown in Figure 9.4, the EI of RWH would be lower than that of overall reclaimed water use (containing RWH) approximately beyond an annual precipitation of 500 mm, indicating energy efficient RWH compared to reclaimed water use beyond that level of precipitation. Moreover, the EI of reclaimed water use would be lower than the maximum EI (3.2 kWh/m^3) of Canadian wastewater treatment systems (AECOM 2012) beyond an annual precipitation of 800 mm and would further be lower than an average EI (1.0 kWh/m^3) of the Canadian wastewater system (WWS) (AECOM 2012) beyond approximately 2000 mm.



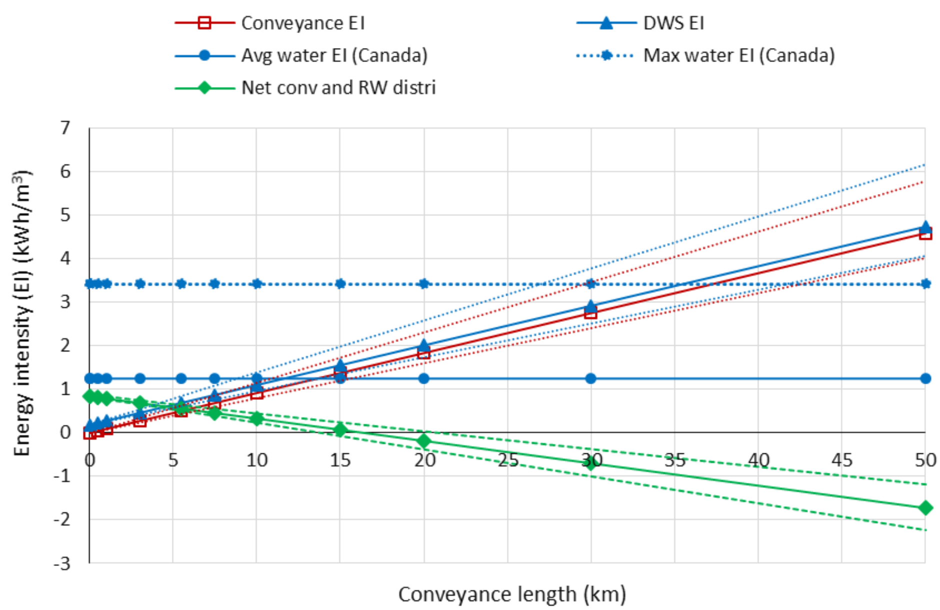
Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles; conv: conveyance; RW: Reclaimed water

Figure 9.5 Unit cost of wastewater system and RWH in different precipitation

As shown in Figure 9.5, the unit cost of RWH would be lower than that of reclaimed water use system beyond approximately 950 mm annual precipitation, indicating a cost-effective RWH compared to reclaimed production from sewage beyond that precipitation level. In addition, the unit cost of reclaimed water use would be lower than the maximum Canadian sewer charge for wastewater management ($2.3/m^3) (AECOM 2012) beyond the annual precipitation of 1350 mm and would further be lowered than an average Canadian sewer charge ($1.5/m^3) (AECOM 2012) beyond approximately 1800 mm. Therefore, RWH would be energy efficient in scenarios where annual precipitation surpassed 500 mm and be cost effective beyond 950 mm annual precipitation for NZW development compared to overall reclaimed water use.

9.3.2.2 Impact of source water proximity on reclaimed water use

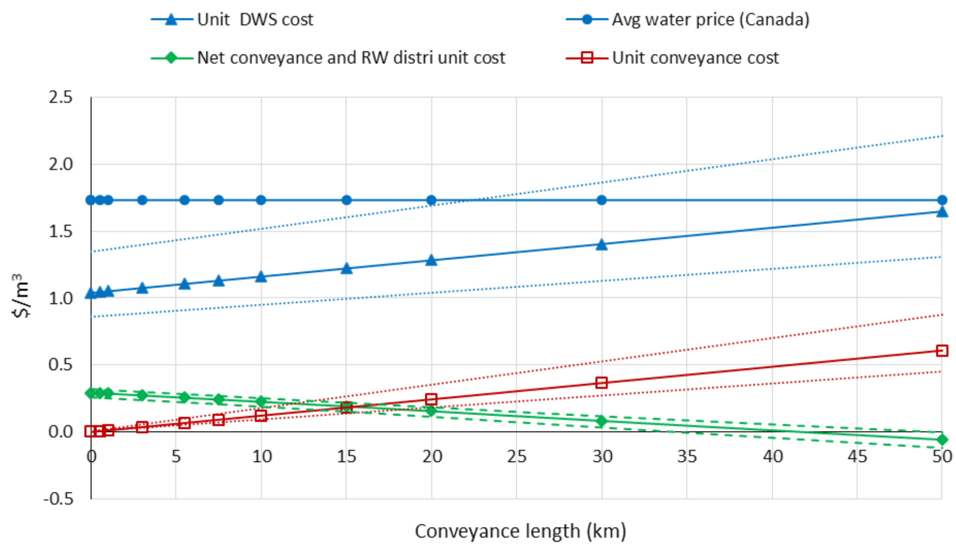
The energy use and cost of drinking water system (DWS) in various conveyance length were analyzed considering a typical community (Scenario 3) with an UWS similar to Penticton, but with different conveyance lengths. The results are shown in Figure 9.6 and Figure 9.7.



Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

Figure 9.6 Energy intensities of DWSs in varying conveyance length

The EI of the DWS increased from 0.2 kWh/m^3 with no conveyance length (0% conveyance energy) to 4.7 kWh/m^3 with 50 km conveyance length (97% conveyance energy) of DWS. The longer the freshwater conveyance length, the more energy can be saved by reducing freshwater withdrawal due to reclaimed water use. The EI of the DWS would be equal to the average EI of Canadian DWSs (1.2 kWh/m^3) (AECOM 2012) at a conveyance length between 9 km and 13 km. Also, the EI of the DWS would be higher than the Canadian maximum EI (3.4 kWh/m^3) (AECOM 2012) beyond the conveyance length of approximately 36 km. Overall, the energy use of reclaimed water distribution could be balanced by the energy saving in freshwater conveyance at a conveyance length between 15 km and 20 km with the same level of treated wastewater reuse (44% of water demand) as in Scenario 3.



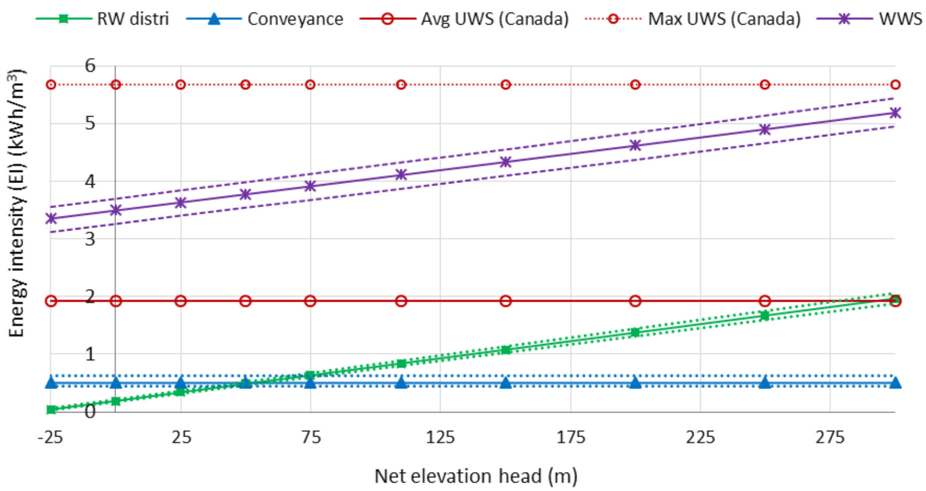
Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

Figure 9.7 Unit LCC of DWSs in varying conveyance lengths

As shown in Figure 9.7, the unit cost of the DWS increased from \$1.0/m^3 with no conveyance (0% conveyance cost) to \$1.7/m^3 with 50 km long conveyance (37% conveyance cost) of the DWS. If the freshwater conveyance length is longer, a higher cost would be saved by reducing freshwater use due to reclaimed water use. The unit cost of the DWS would still be lower than the average Canadian water price (\$1.7/m^3) (AECOM 2012) at a conveyance length of 50 km. This shows that conveyance length has lesser influence on the cost of DWSs compared to the conveyance energy use. Overall, the cost of reclaimed water distribution could be balanced by saving the cost of freshwater conveyance at a conveyance length between 34 km and 47 km. Therefore, reclaimed water use could be energy efficient if the freshwater conveyance length is longer than 15 km and cost-effective for conveyance lengths longer than 34 km with the same level of treated wastewater reuse (44% of water demand) as in Scenario 3.

9.3.2.3 Impact of elevation head on reclaimed water distribution

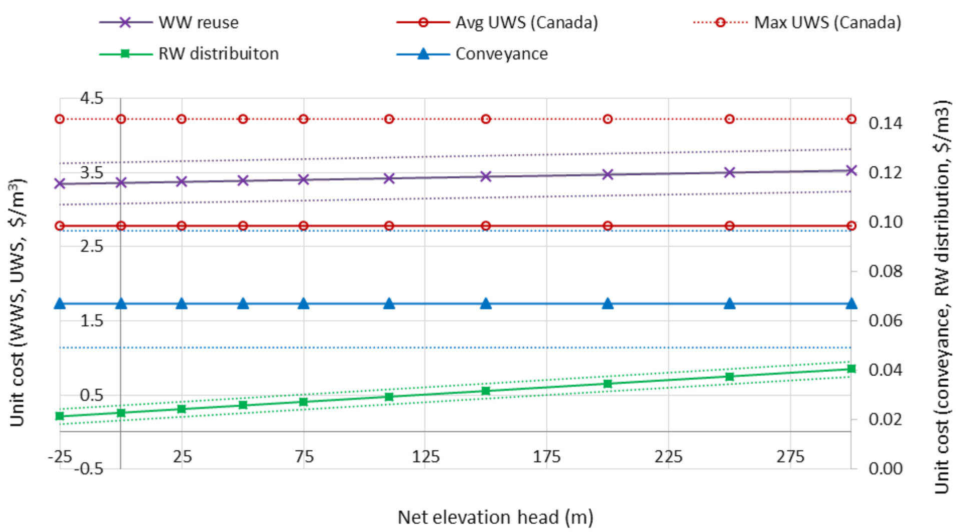
The energy use and cost of reclaimed water use in various net elevation heads were analyzed considering a typical community (Scenario 3) with an UWS similar to Penticton but with different net elevation heads. The results are shown in Figure 9.8 and Figure 9.9.



Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

Figure 9.8 Energy intensities in different net elevation heads

The EI of the reclaimed water (RW) distribution increased from 0.04 kWh/m^3 at -25 m (negative sign means a lower elevation than a reclamation station) net elevation head (1% of wastewater reuse energy excluding RWH) to 2.0 kWh/m^3 at 300 m net elevation head (38% of wastewater reuse energy) (Figure 9.8). Overall, the EI of wastewater reuse increased from 3.4 kWh/m^3 to 5.2 kWh/m^3 due to the increase in net elevation heads. Higher net elevation head would demand more energy use for RW distribution. However, the EI of the wastewater reuse was higher than the Canadian average EI (1.9 kWh/m^3) (AECOM 2012) and lower than the Canadian maximum EI (5.7 kWh/m^3) (AECOM 2012) of UWSs for all the considered net elevation heads. Compared to the country average, the higher EI of wastewater reuse in the typical community could be due to the advanced treatment of wastewater for reuse purposes, which is usually higher than the EI of treatment of drinking water sourced from freshwater. Moreover, the performance of wastewater reuse was compared with an entire UWS comprising water and wastewater services because wastewater reuse involves wastewater treatment and treated wastewater supply for reuse. Furthermore, around the net elevation head of 55 m, the EI of reclaimed water distribution would be equal to the EI of freshwater conveyance (0.44 kWh/m^3), indicating a comparable or lesser EI of reclaimed water distribution up to the net elevation head of 55 m.



Note: Dotted lines indicate an uncertainty band with 5th and 95th percentiles

Figure 9.9 Unit LCC of wastewater reuse with different elevation heads of secondary distribution

As shown in Figure 9.9, the unit cost of reclaimed water (RW) distribution increased from \$0.02/m^3 at -25 m net elevation head (0.6% of wastewater reuse cost) to \$ 0.4/m^3 at 300 m net elevation head (1.1% of wastewater reuse cost). Overall, the unit cost of wastewater reuse increased from \$3.3/m^3 to \$3.5/m^3 due to such increase in net elevation heads. This shows that the LCC of RW distribution is below 2% of the total LCC of the wastewater reuse system. The lower proportion of RW cost is primarily due to high wastewater treatment cost (\$2.3/m^3) for producing high quality water for reuse (Guo et al. 2016). The estimated unit cost of wastewater reuse was higher than average Canadian UWS charge (\$2.8/m^3) (AECOM 2012) and lower than the maximum Canadian UWS charge (\$4.2/m^3) (AECOM 2012) for all the considered net elevation heads. The estimated unit LCC of wastewater reuse was higher than the average UWS charge (water and wastewater charge) of the country, most likely due to government subsidies for water and sewer charges. However, wastewater reuse negates the expenditure on wastewater treament that has to be performed either way, perhaps to a lesser degree of treatment if wastewater is not reused. A sewer charge is a potential stream of revenue for wastewater reuse projects. Overall, the EI of reclaimed water distribution would be equal to that of freshwater conveyance around the net elevation head of 55 m, whereas the LCC of RW distribution was below 2% of the total LCC of wastewater reuse systems even at a net elevation head of 300m.

9.4 Discussion

The cost-embedded WEC model has been developed by adding a cost module and hydraulic equations to estimate the energy demand for water distribution in the previously developed WEC model. Cost and energy use are important factors in determining the economic viability and energy efficiency of SMUWSs. The proposed model is applicable in analyzing the dynamic NZW potential of communities in terms of energy use, LCC, and carbon emissions. The water, energy, and carbon modules were already validated in the WEC model developed in Chapter 5.

The newly added cost module included a cost function of RWH systems, which was developed from the country-wide extensive data of RWH at office buildings in different precipitation zones in the United States (Wang and Zimmerman 2015). The cost function has a high r^2 of 0.93. Based on the same research, the energy function of RWH was developed, with approximately 7% of the RWH cost attributed to energy use (Wang and Zimmerman 2015). The energy function also has r^2 of 0.85. Other cost data such as water and wastewater pipes and treatment were obtained from the published literature and government reports as mentioned in the data requirement section. Similarly, the newly added energy equations in the energy module are fundamental hydraulic equations obtained from a textbook (Briere 2010). Based on these equations, the estimated EI for water conveyance ranged from 0.406 kWh/m^3 to 0.465 kWh/m^3 in which the actual value 0.415 kWh/m^3 lies (City of Penticton 2015c) in the business as usual scenario. Similarly, the estimated EI of water distribution is 0.0325 kWh/m^3, which is similar to the actual value of 0.0330 kWh/m^3 (City of Penticton 2015c), indicating the effectiveness of the hydraulic equations.

Uncertainty is an unavoidable and inevitable element of environmental modelling (Sadiq and Tesfamariam 2009). The uncertainty of NZW analysis was approximated by using Monte

Carlo simulations by varying the relatively important 121 input variables and parameters based on Venkatesh et al. (2014) and Kenway et al. (2008). Based on these simulations, the uncertainty band with the 5^{th} and 95^{th} percentile was estimated for all the outputs considered. In fact, the band includes two dimensions of uncertainty: random variation at a particular time (month) and variation across time, i.e., monthly variation for 10 years. So, the uncertainty band is wider in some outputs such as NZW.

The proposed model has been applied to the City of Penticton. Five scenarios were analyzed for developing NZW in the city. Certainly, Scenario 4 with wastewater reuse, residential RWH, and SWH would result in the highest net positive water; however, it would be costly (\$5.6/m^3) and energy intensive (4.9 kWh/m^3) compared to the business as usual Scenario 1 with unit cost (\$1.2/m^3) and EI (1.8 kWh/m^3). This is due to high unit cost and high EI of RWH in Penticton being it is a semi-arid region and the use of advanced treatment for entire harvested storm water. Although expensive, this research has considered advanced treatment methods comprising of membrane bioreactor (MBR), iron-mediated aeration (IMA)/sand filtration, peroxone mineralization of organics, and residual chlorination to produce treated water with no detection of the analyzed 97 hormones and pharmaceuticals and personal care products (PPCPs) (Guo et al. 2016; Wu and Englehardt 2016). Instead, the unit LCC estimated by FitWater can also be used for wastewater treatment. Moreover, the estimated water stress index of Penticton is around 20% in Scenario 1. This level of water stress results in the recommendation for Penticton for wastewater reuse in Penticton (Jiménez Cisneros 2014).

The cost effectiveness and energy efficiency of NZW development are affected by many factors. This study has dynamically analyzed the three factors in detail: climate - precipitation; hydrological (source water) proximity- conveyance distance, and topography - net elevation head of reclaimed water service area. The dynamic analysis was conducted by considering a typical community with an urban water system similar to the City of Penticton, but with different climate, hydrological proximities, and topography in Scenario 3. Such an analysis will identify potential conditions for developing an economically viable and energy efficient NZW in different precipitation zones, hydrological proximities, and topography.

The NZW analysis for different precipitation zones shows that the residential RWH would satisfy all the domestic demand beyond the annual precipitation of 1000 mm. This precipitation range has a high potential for RWH. The commercial, institutional, and industrial (CII) sectors can also benefit from RWH in their buildings. This will help communities to assess the potential to supply water for all community demands by RWH at higher precipitation levels. Moreover, the significance of dynamic analysis was portrayed by RWH. The monthly analysis shows that an annual precipitation of 900 mm would not be able to meet all the monthly residential water demand (especially in July and August) until 2022 (Year 7); but such precipitation would be able to satisfy all monthly demand after 2022 due to the city's intensive water conservation programs that have progressively been reducing per capita water consumption in the recent past (City of Penticton 2006, 2013).

The economic and energy impacts of NZW development with respect to precipitation amount, hydrological proximities, and topography show that energy efficiency and cost effectiveness vary in these circumstances. For example, RWH would be energy efficient beyond an annual precipitation of 500 mm and cost effective beyond 950 mm; wastewater reuse can be energy efficient if the freshwater conveyance is longer than 15 km and be cost effective only for conveyance distances longer than 34 km. This is because energy cost is only a portion of the total LCC. The cost of other infrastructure also plays a significant role in determining the LCC of an UWS or its components, such as the cost of equipment in wastewater treatment. Therefore, the energy efficiency and cost effectiveness of NZW development are site-specific, but their effectiveness would increase with an increased pressure on source water due to decreasing freshwater availability, increasing population, etc.

The WEC model can further be improved by considering RWH in CII sectors that will help to assess and develop NZW communities by minimizing health risks, at least the perceived risk, of wastewater reuse in all sectors. The WEC model includes the cost and energy functions of RWH and energy functions of water, wastewater, and reclaimed water distribution varying with a spatial scale. The model can further be improved by including the cost functions of water and wastewater pipes with respect to scale as well as the cost of water fixtures (e.g., toilet, faucet), appliances (e.g., laundry machine, dishwasher), and indoor water heating equipment. Furthermore, the uncertainty analysis can also be extended by considering additional inputs with random Monte Carlo simulations. Such features will make the WEC model a more sophisticated tool for planning and analyzing entire urban water systems, such as for NZW potential.

9.5 Summary

A Net-zero water (NZW) community can be developed by combining various water supply sources, conservation measures, and reuse over time. However, there is a lack of decision support systems to assess the site-specific economic and environmental potential of NZW development. This chapter has developed a cost-embedded WEC model by adding a cost module and scale-dependent energy estimation of water and wastewater flow in the previously developed WEC model using STELLA®. The extended WEC model has been applied to analyze the economics of the WEC nexus in developing NZW in the City of Penticton by considering five different scenarios for 2016-2035. The uncertainty was approximated using Monte Carlo simulations.

Results show that Penticton can achieve net-zero and even net-plus water by wastewater reuse, storm water harvesting, and rainwater harvesting (RWH) (Scenario 4) or without RWH (Scenario 5). However, they would be energy intensive and costly with 4.9 kWh/m^3 and \$5.6/m^3 in Scenario 4 versus 1.8 kWh/m^3 and \$1.2/m^3 in the business as usual scenario. Moreover, a detailed environmental analysis shows that RWH would be energy and cost efficient beyond an annual precipitation of 500 mm and 950 mm respectively; and wastewater reuse would be energy and cost efficient for the system with freshwater conveyance longer than 15 km and 34 km respectively. Therefore, the energy and cost effectiveness of NZW development are site-specific but their effectiveness would increase with an increased pressure on source water. Factors such as decreasing freshwater availability due to climate change, increasing population growth, water security enhancement, etc. would increase the energy and cost effectiveness of NZW development.

Chapter 10 Conclusions and Recommendations

Urban Water Systems (UWSs) face increasing challenges due to increasing population, lower household occupancy, higher prices of water and energy, lifestyle changes, and climate change. These factors ultimately affect the sustainability of UWSs. The sustainability of UWSs is largely affected by water consumption, energy use, and carbon emissions, along with associated health risks and costs. The elements: water, energy, and carbon emissions are interconnected and have complex interactions that form a water-energy-carbon (WEC) nexus. These pervasive interactions require integrated solutions. Urban water sustainability can be assessed by quantifying the WEC nexus of UWSs; however, the WEC nexus should also take into account the associated human health risk of reclaimed water and life cycle costs.

10.1 Conclusions

The main objective of this research was to propose a decision support system (DSS) for assessing the WEC nexus of UWSs for planning and managing sustainable urban water systems. This objective was accomplished by critically reviewing state-of-the-art urban water sustainability assessment systems, WEC nexus studies, and health risk assessment of reclaimed water use. In this research, the WEC nexus modelling was demonstrated using small to medium-sized urban water systems (SMUWSs). A system dynamics-based WEC model, spreadsheet-based FitWater tool, and microbial water quality guidelines for non-potable urban reuses (accounting the associated health risks, life cycle cost, and WEC nexus) were proposed. The key findings of this research are summarized below.

Under Objective 1, Sustainability Performance Indicators (SPIs) for assessing the sustainability of SMUWSs have been proposed. Certain limitations of the sustainability of UWSs, both for centralized UWSs and decentralized wastewater treatments, can be overcome by managing UWSs at an intermediate scale, i.e., SMUWS. A set of 38 SPIs has been developed for assessing the sustainability of SMUWSs. The selected SPIs include 8 technical SPIs, 13 environmental SPIs, 4 economic SPIs, 7 social SPIs, and 6 institutional

SPIs. Water consumption, energy use, carbon emissions, health risk, and cost related SPIs were used in the WEC model and decision support system (DSS).

Under Objective 2, a dynamic WEC model and DSS for the operational phase of a SMUWS has been proposed. The system dynamics-based DSS is capable of dynamic analysis of different WEC-based interventions to improve the sustainability of SMUWSs. For the UWS of Penticton (BC), Spearman's correlation coefficients between water and energy, water and carbon, and energy and carbon were 0.94, 0.89, and 0.83 respectively showing very strong interconnections among water, energy and carbon. The highest energy consumer was found to be indoor hot water use in residential and commercial, institutional, and industrial sectors consuming approximately 90% of the operational energy demand and contributing about 93% to carbon emissions. The WEC DSS can be used by utilities, urban developers, and policy makers for sustainable urban water planning to reduce water consumption, energy use, and carbon emissions in neighbourhoods. Furthermore, the WEC DSS can also be used for operational neighbourhoods to forecast future WEC nexus.

Under Objective 3, a framework has been proposed to assess the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system. The proposed framework has been applied to a planned neighbourhood in the Okanagan Valley (BC). The findings show that a higher net residential density, will result in lower will be per capita water demand, energy use, negative net carbon emissions, and LCC of the water distribution systems. The characteristic curves of these parameters with net residential density have power relationships showing a distinct point or region of inflection, beyond which the change in parameters with respect to the net residential density is insignificant. This distinct point or region provides an estimation of an optimal density. The optimal residential density and the parameters related to water, energy, and carbon emissions of an optimal water distribution system can be used as inputs to the WEC model.

Under Objective 4, microbial water quality guidelines for reclaimed water use have been proposed for various non-potable urban purposes. Globally, standard guidelines do not exist for reclaimed water use. This research has developed microbial water quality guidelines for reclaimed water in various non-potable urban reuses. The required treatment levels were also estimated for all reuse applications. In addition, a FitWater tool has been proposed for the evaluation of fit-for-purpose wastewater treatment and reuse potential in communities. The evaluation is based on the LCC of treatment and wastewater conveyance, health risk, energy use, and carbon emissions. FitWater can be used to develop and screen alternative wastewater treatment trains and reuse plans. The tool can be applied to both wastewater and greywater treatment. The proposed FitWater tool has been tested with several existing wastewater treatment plants in Canada and the United States, showing the effectiveness of the tool in estimating capital, operational and maintenance costs, energy use, carbon emissions, and health risks. Moreover, equations have been developed for unit annualized LCC and energy intensity per unit log removal in various plant capacities, indicating economies of scale. Furthermore, FitWater can be used to assess the economic impacts of developing microbiologically stringent effluent standards.

Under Objective 5, the economics of the WEC nexus of SMUWSs has been analyzed using the WEC model with LCC. The cost embedded WEC model has been applied to the City of Penticton for analyzing NZW potential. Penticton can achieve net-zero and even net-plus water by wastewater reuse, storm water harvesting, and rainwater harvesting. It is also possible for Penticton to achieve its NZW without rainwater harvesting. However, the aforementioned interventions would be energy intensive and costly. A dynamic analysis of environmental factors: precipitation amount, source water proximity, and topography affecting NZW potential shows that rainwater harvesting would be energy efficient beyond an annual precipitation of 500 mm and cost effective beyond 950 mm annual precipitation. Wastewater reuse would be energy efficient for freshwater conveyance longer than 15 km in length and cost effective for systems longer than 34 km in length. Therefore, the energy and cost effectiveness of NZW development are site-specific, but their effectiveness can increase with increasing stress on source water.

10.2 Originality and contribution

The main contribution of this research work is the development of a decision support system (DSS) to simultaneously quantify dynamic water consumption, energy use, and carbon emissions of UWSs. The WEC nexus modelling has been demonstrated by applying to a

SMUWS; however, the developed WEC DSS can be applied to large UWSs, where there can be more than one water treatment plant, wastewater treatment plant, etc. Users can input average values of parameters, such as average energy intensity of water treatment plants, average energy intensity of wastewater treatment plants, etc. for large UWSs. The developed DSS will contribute to the improvement of the sustainability of UWSs. The WEC model at the community level is the first of its kind and can assist decision makers to identify optimal alternatives in different stages of UWSs to simultaneously minimize water consumption, energy use, and carbon emissions over time. The WEC model is capable of incorporating dynamic variables (e.g., population growth, toilet water efficiency improvements, etc.) in the analysis. In addition, this model is capable of forecasting the future water consumption, energy use, and carbon emissions of urban communities. Moreover, the proposed WEC model can be used to assess the potentiality of developing economic and energy efficient net-zero water communities in different climatic and topographic regions. The model is especially important for water-scarce communities as well as for water stewardship communities. Therefore, the developed model can be used to support decisions for municipalities, urban developers, and policy makers to improve urban water sustainability.

This research work has also developed microbial water quality guidelines for urban water reuses in non-potable purposes. In addition, this research has developed a FitWater tool to evaluate the potential of fit-for-purpose wastewater treatment and specific reuse for a community. The proposed tool is able to assess health risks of reclaimed water use simultaneously in one or more urban applications, estimate the LCC of wastewater treatment, and its WEC nexus and then rank alternative wastewater treatment and reuse plans. The output of FitWater can be used as inputs to the WEC model.

A framework has been proposed to assess the impacts of neighbourhood densification on the WEC nexus of water distribution and residential landscaping system. The characteristic curves of these parameters have power relationships with net residential density showing a distinct point or region of inflection. This distinct point or region provides an estimation of an optimal density. The optimal residential density and the parameters related to water, energy, and carbon emissions of an optimal water distribution system can be used as inputs to the WEC model. Furthermore, a set of sustainability performance indicators (SPIs) have been developed, which can be used to assess the overall sustainability of SMUWSs.

In addition, this research has been able to make community impact. There were several presentations of research findings, demonstration of the developed tools, discussions, and feedback collection from the District of Peachland (BC, Canada) as a community partner of this project. The WEC DSS and FitWater were successfully tested to the District of Peachland and the District appreciated these tools. Moreover, the discussions with the District provided the state-of-the-art knowledge on water and sustainability to the personnel of the District. These interactions increased their awareness level on water sustainability and motivated them to apply the recommended water conserve measures, such as source water protection, xeriscaping, monitoring and reduction of non-revenue water, and use of interpretive signage on water. The role of increased level of awareness and use of conservation measures were appreciated by the District of Peachland to win the water sustainability award of the Okanagan Valley (BC, Canada) in 2016.

10.3 Limitations

The present research work experienced data limitation to some extent. The proposed WEC DSS requires extensive data for small and medium-sized urban water systems. Site-specific data were not available for some parameters and variables. Due to this limitation, the WEC DSS was simulated by using provincial and national data for such parameters and variables. In the WEC model, the exponential growth was used for human population, CII sector, and other parameters, such as inflow-infiltration and energy consumption. The prediction may be associated with high uncertainties if predicted for a very long duration due to variations in dynamics among model variables. The cost module of the WEC DSS included a cost function of rainwater harvesting systems, which were derived from US-based rainwater harvesting data of office buildings in different precipitation zones (Wang and Zimmerman 2015). Also, based on the same research, the energy function of RWH was developed by approximating 7% of the RWH cost attributed to energy use (Wang and Zimmerman 2015).

This research only investigated the impact of neighbourhood densification on water distribution and residential landscaping system. Densification may also affect other elements, such as transportation, open space, aesthetic value, etc., which were not considered in this research. Also, this research did not include the variation in dose-response models and their parameters although it used the updated models and parameter values of pathogens as far as these are available in the publically accessible literature. Due to the lack of Canadian data, especially for morbidity, disease burden per case, and susceptibility fraction for some pathogens, and of exposure factors for many water reuses, the related data were obtained from the US and other developed countries, which may affect the proposed microbial water quality guideline values. Probabilistic risk analysis for developing microbial water quality guidelines considered a 10% variation with a uniform distribution in most of the exposure factors due to the lack of data. The developed FitWater focused only on microbial water quality for developing treatment trains although primary treatment, secondary treatment, sludge processing, and disinfection are mandatory treatment stages under FitWater, which considerably reduces organic pollutants (BOD and COD).

10.4 Recommendations for future work

The following recommendations are provided for the future research.

- The proposed WEC model can be enhanced by adding feedback systems by incorporating quantitative relationships between water demand variation and behavioural change.
- The proposed framework for assessing the impacts of neighbourhood densification on the WEC nexus of water distribution system and residential landscaping system can be extended by including the WEC nexus effects of neighbourhood configuration and the number of stories in buildings. The results of this study also recommend to amend relevant policies for constructing medium to high-density buildings in urban neighbourhoods to achieve an optimal WEC nexus.
- The proposed microbial quality guidelines of reclaimed water for non-potable urban reuses could further be improved by using national or regional data on water exposure

in different urban reuse applications, disease burden per case, and the susceptibility fraction of population to different diseases.

- The proposed FitWater tool can be enhanced by including the physio-chemical quality of wastewater, additional treatment technologies, and carbon emissions from treatment processes.
- The cost embedded WEC model can further be improved by including the cost functions of water and wastewater pipes with respect to spatial scale as well as the cost of water fixtures (e.g., toilets, and faucets), appliances (e.g., laundry machines, and dishwashers), and indoor water heating equipment.

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Appendices

Appendix A: Additional Information on SPI Identification

A.1 Delphi method

A group decision was used for the rating of the relevance criteria and the determination of criteria weights. The group decision was made based on the expert interviews using a Delphi method. A Delphi method is a technique for reaching a properly thought-through consensus among experts (Juwana et al. 2010). The urban water experts were from three categories: a) utilities: managers, treatment plant engineers, water engineers, financial managers, and urban planner; b) consultants: design engineers and water consultants; c) academics: professors and researchers working on water, were involved in the Delphi method. Emails were sent to 85 water experts and organizations for their participation in the study and among them, 30 experts participated. Twelve were from utilities, nine were consultants, and other nine were academics. The Delphi method was conducted by individual interviews (Keil et al. 2013) and some of the experts provided their views by questionnaire after they were explained about the research (Pirdashti et al. 2011; Juwana et al. 2010). The professional experience of the participant experts ranges from 4 to 28 years with an average of 13 years as given in Table A.1.

Table A.1 Expert participants and their professional experience

	% of participants with professional experience			
Experts	**4 - 10 years**	**10 - 19 years**	**≥ 20 years**	**Total**
Utilities	10.0	20.0	10.0	40.0
Consultants	10.0	16.7	3.3	30.0
Academics	10.0	10.0	10.0	30.0
Total	30.0	46.7	23.3	100.0

The consensus for all questions were reached after three rounds of interviews. Altogether, 30 experts participated in the first round, whereas 20 and 10 experts participated in the second and third round interviews respectively. Additional interviews after the third round were not required as coefficient of variations (CV) of responses for all questions were below 0.5 (Dajani et al. 1979; Kim et al. 2013). The calculated CV ranges from 0.13 to 0.47 for the rating of the relevance criteria for all SPIs and from 0.26 to 0.46 for the comparison of the selection criteria to derive weights. Similarly, the content validity ratio (CVR) measures an agreement among the participants as to how essential a particular item is. A CVR was calculated by using Equation A.1 (Wilson et al. 2012; Kim et al. 2013):

$$Content\ Validity\ Ratio\ (CVR) = \frac{N_e - {}^{N}/_{2}}{{}^{N}/_{2}} \qquad \textbf{Equation A.1}$$

where Ne is the number of expert participants indicating that an item is "essential" (i.e., high and very high relevance in this context) and N is the total number of expert participants. A CVR greater than 0.29 can be considered as an appropriate level of agreement (Wilson et al. 2012; Kim et al. 2013). The CVR ranging from 0.33 to 1.00 for the rating of the relevance criteria for all SPIs and from 0.38 to 0.62 for the comparison of the selection criteria indicate a consensus was reached. Furthermore, the respondents were asked to "base your answer on your professional knowledge rather than a specific location or situation" by mentioning it on the Delphi questionnaires.

A.2 Screening of SPIs for SMUWSs

Table A.2 Initially screened SPIs for SMUWSs

Code	Indicators	Measurement method	Description	Sources[2]
I. TECHNICAL				
Neighbourhood location and design				
TE1	Proximity to drinking water system or source water	Distance to nearby drinking water system (existing) or water source	Proximity to existing or planned water services or source	10,14
TE2	Proximity to wastewater system	Distance to nearby wastewater system	Proximity to existing or planned wastewater services	10, 13, 14
TE3	Proximity to electricity service	Distance to nearby electricity service	Proximity to existing or planned energy services	14
TE4	Separation of wastewater (WW) and storm water	% of population served with separate WW and storm water services	Measure of use of resources	1
TE5	Dwelling density (Residential)	No. of dwelling unit (DU)/ Total area of buildable land for residential uses	Measure of compact development (land conservation)	10, 13
TE6	Floor area ratio (Non-residential)	Total area of all floors of all non-residential buildings/Area of buildable land available for non-residential uses	Measure of compact development (land conservation)	10, 13
TE7	Flexibility and adaptability	Level of accommodation in design (adding or removing components from system) (Qualitative)	Ability to accommodate future changes to the system	12
TE8	Occupancy	Persons per DU	Measure of compact development (land conservation)	19,20
Water infrastructure and fixtures				
TE9	Metered connection	% of connections metered	Measure efficient use of resources	2
TE10	Water efficiency of appliances and faucets	Average % more efficient than the minimum (local) government standard for all buildings	Water efficiency of toilets, showers, faucets, dishwashers and clothes washers	10, 11
TE11	Treated water storage capacity	Storage capacity as % of average daily demand	Measure of sufficient capacity to meet demand even in treatment failures	21, 22
TE12	Storm water storage capacity	% of average daily flow	Measure of system capacity to control flash floods	13
TE13	Landscape water efficiency	% of water reduction in landscape irrigation with respect to calculated midsummer baseline case	Measure efficient use of resources	10, 11, 13
TE14	Water supply reliability	Number of main breaks per 100 km length	Reliability of water service	2,6,7, 9,16
TE15	Water leakage	% of water loss per year	Distribution efficiency	1,2,5,7, 8,12,15, 16,17
TE16	Incidents of sewer flooding	Counts of internal sewer flooding per year	Measure of reliability of wastewater service	12
TE17	Infrastructure maintenance	% of water and WW linear infrastructure less than 40 years	Measure of maintenance	1,6,9

[2] See the sources at the end of this table

II. ENVIRONMENTAL				
Resource utilization				
EN1	Water self-sufficiency	Ratio of annual licensed water (or renewable surface and groundwater) to water demand	Measure of the needed water availability within own territory	1,18
EN2	Domestic water consumption	L per capita per day	Current drinking domestic water consumption	1,7,8,14, 15,16,17
EN3	Non-domestic water consumption	L per capita per day	Current non-domestic water consumption	8,14
EN4	Groundwater quality	Faecal Coliform, N, P (% samples not complied)	Requirement for human and environmental health	4,5,6,13
EN5	Surface water quality	Faecal Coliform, N, P, BOD (% samples not complied)	Requirement for human and environmental health	1,10,11, 12,13, 16,18
EN6	Biodiversity of surface water	Diversity of benthic organisms	Assessment of ecological quality of main surface water(s), required for perpetuating functions of water bodies	1,6,10, 12,13
EN7	Energy use in water service	kWh per m^3 water supplied	Measure of the use of resources for water treatment	1,9,12
EN8	Energy use in wastewater service	kWh per m^3 water treated	Measure of the use of resources for wastewater treatment	2,6,12
EN9	Energy use for domestic hot water	kWh per m^3 domestic water consumed	Measure of the use of resources in domestic use	2,4,5, 6,7,14
EN10	Chemical use in water treatment	Tonnes per ML (major chemicals)	Measure of the use of resources for water treatment	1,2,4, 5,6,7
EN11	Chemical use in WW treatment	Tonnes per ML (major chemicals)p	Measure of the use of resources for wastewater treatment	4
Environmental impacts				
EN12	Wastewater generation rate	L per capita per day	Measure of pollutants generation	2,4,5, 6,7,8
EN13	Discharged wastewater quality (BOD, N, P, heavy metals: Cd, Pb, Hg and Cu)	Water quality compared to effluent standards (no. of days not complied per year)	Measure of impacts of wastewater disposal	2,3,4,9, 12,15, 16,17
EN14	Bio-solids quality	Heavy metals content	Measure of impacts of sludge disposal	3
EN15	Disposed storm water quality	% of reduction of water pollutants (TSS, Total N, P, litter) than untreated urban storm water	Measure of impacts of storm water pollution	13,14
EN16	Disposal of backwash water	% of residuals that reach natural water bodies	Measure of impacts of backwash water pollution	21
EN17	Effective impervious area	% of total basin area contributing to surface runoff and directly connected to drainage collection system	Measure of surface runoff	2,10,11, 12,13,14
EN18	GHG emissions from water service	Ton CO_2-e per (Million litres) ML water supplied	Measure of contribution towards global warming	3,6,7,9
EN19	GHG emissions from WW service	Ton CO_2-e per ML WW treated	Measure of contribution towards global warming	3,6,7,9
Resource recovery				
EN20	Rainwater harvesting (RWH)	% of HH with > 5% hard surface (roofs and free standing surface) used for RWH	Measure of the use of resources	4,5,6,11
EN21	Water reuse	% of reclaimed water use in total water consumption	Measure of the use and depletion of resources	2,6,10, 11,13
EN22	Energy recovery	kWh per m^3 of wastewater treated	Measure of the use and depletion of resources	1,4,6

EN23	Sludge reused	% of sludge reused per year	Measure of the use and depletion of resources	1,2,4, 7,9
EN24	Nutrients recovery	% of WW used for Phosphate recovery	Measure of the use and depletion of resources	1
III.ECONOMIC				
Water economics				
EC1	Total cost coverage ratio for water	Total actual costs/ Revenue for water service (yearly)	Measure of financial viability	6,9
EC2	Operating cost coverage ratio for water service	Operating costs/Revenue for water service	Measure of operational and management cost	2,3,5
EC3	5 yrs running average capital reinvestment/ replacement value	%	Measure of capital investment	2,3, 5,6
EC4	Average water fee rate	\$ per 250 m^3	Measure of affordability of householders to pay for water and wastewater services delivered	3,5
EC5	Non-revenue water	L/connection/day	Measure of financial loss in water supply	22, 23
Wastewater economics				
EC6	Total cost coverage ratio for WW	Total actual costs/ Revenue for WW service (yearly)	Measure of financial viability	6,9
EC7	Operating cost coverage ratio for WW service	Operating costs/Revenue for wastewater service	Measure of operational and management cost	2,3,5
EC8	5 yrs running average capital reinvestment/ replacement value	%	Measure of capital investment	2,3,5,6
IV. SOCIAL				
Service provision				
SO1	Access to water service	% of population served by public water supply system	Requirement for the development of an individual	1,2,3,5, 6,11, 16,18
SO2	Access to wastewater service	% of population served by public WW system	Requirement for human and environmental health	1,2,3,5, 6,11, 15,16
SO3	Water restrictions	Days per year	Measure of service quality	21
SO4	Public acceptability	Number of complaints per 1000 people	Measure of acceptability of water and wastewater systems to stakeholders	3,5, 6,9
SO5	Willingness to change behaviour	% of people willing to change behaviour for water conservation	Measure of public support towards sustainability direction	3
SO6	Water aesthetics	Water supporting the urban landscape aesthetics measured by residents' sentiment (Qualitative)	Requirement for quality of life for urban residents	1,18
SO7	Vandalism against agency	No. of acts of vandalism per 1000 people served	Measure of public security	21
Public health				
SO8	Safety (from flooding and drought)	Assessment of plans, measures and their implementations to protect citizens against flooding and drought (Qualitative)	Requirement for the development of people	1,2,10, 11,12, 13
SO9	Drinking water quality	Compliance with drinking water quality standard	Requirement for the development of an individual	1,3,6, 7,9,15, 16,18
SO10	Boil water advisories	No. of HH-days per year that boil water advisories are in effect as a % of total HH-days	Measure of risk to human health	3,5,16

V. INSTITUTIONAL				
Governance and progress				
IN1	Urban water policies	Local government's policies and commitments for integrated urban water management (Qualitative)	Measurement of participatory, adaptive, coordinated, and integrated management	1,6, 15,17
IN2	Institutional cooperation	Adequacy and status of water related agreements with other institutions (Qualitative)	Measure of institutional cooperation for managing shared water resources	25, 26
IN3	Consideration of water use impacts in decision making	Application of water centric decision making framework for large scale infrastructure development (Qualitative)	Measure of consideration of water use impacts in decision making (e.g., infrastructure development)	27,28
IN4	Institutional capacity	(FTE personnel * Work experience)/Volume of water supplied	Measure of institutional strengths	9, 29, 30, 22
IN5	Personnel training	No. of training hrs/ No. of employees in a year	Measure of organizational development	23,30, 31
IN6	Conservation programs	Average annual expenses per person	Measure of conservation efforts	6
IN7	Public participation	Proportion of individuals who volunteer for an organization or involved in planning process	Measure of local community strength	3,5,6, 7,8
IN8	Achievement of water demand reduction target	Current year's water usage as a % of target water use	Index to demonstrate continued progress in a multi-year conservation program	8,9,11
IN9	Research and development (R and D) initiatives	% of total budget invested for R and D per year	Initiatives for innovations	2

Notes: Sources 1-18 refer to SN 1 -18 of Table 3.1 (Chapter 3) respectively and sources 19-31 are: 19) Butler (1993); 20) Guerra Santin et al. (2009); 21) NRC (2009); 22) CSA (2010); 23) World Bank (2011); 24) AECOM (2012); 25) Mirumachi and Van Wyk (2010); 26) Herrfahrdt-Pahle (2014); 27) Adeoti (2010); 28) Bayart et al. (2014); 29) Government of Canada (2007); 30) IWA (2006); 31) AWWA (2008)

A.3 Application of Fuzzy-ELECTRE I method for ranking SPIs in economic dimension

The weights of selection criteria were determined using Fuzzy-AHP. The pairwise comparison matrix obtained through expert interviews using the Delphi method is given in Table A.3.

Table A.3 Pairwise comparison matrix for estimating the weights using F-AHP

	Relevance (C1)	Measurability (C2)	Data availability (C3)	Comparability (C4)
Relevance	(1, 1, 1)	(1/2, 1, 3/2)	(1, 3/2, 2)	(3/2, 2, 5/2)
Measurability	(2/3, 1, 2)	(1, 1, 1)	(1/2, 1, 3/2)	(1, 3/2, 2)
Data availability	(1/2, 2/3, 1)	(2/3, 1, 2)	(1, 1, 1)	(1, 3/2, 2)
Comparability	(2/5, 1/2, 2/3)	(1/2, 2/3, 1)	(1/2, 2/3, 1)	(1, 1, 1)

where (1, 1, 1) = Just equal; (0.5, 1, 1.5) = equally important; (1, 1.5, 2) = weakly important; (1.5, 2, 2.5) = strongly more important; (2, 2.5, 3) = very strongly more important; (2.5, 3, 3.5) = absolutely more important

In this study, the geometric mean method was used to derive fuzzy weights in the fuzzy-AHP (Kaya and Kahraman 2011). In the comparison matrix, a row wise geometric mean was calculated and then normalized by using Equation 4.8 (Chapter 4). The respective normalized values thus obtained are the weights of each criteria. The estimated weights in terms of TFNs are (0.19, 0.32, 0.49), (0.20, 0.27, 0.35), (0.18, 0.24, 0.35), and (0.11, 0.17, 0.27) for the relevance, measurability, data availability, and comparability criteria respectively. The consistency ratio (CR) of the comparison matrix is 0.0076, which is very less than 0.10 indicating a consistent matrix (Alonso and Lamata 2006).

In the economic dimension, eight SPIs were selected in the initial screening. The fuzzy scoring matrix is given in Table A.4, which was developed based on Table 4.2 (Chapter 4) through expert interviews using the Delphi method for Criteria C1 and using literature for Criteria C2, C3, and C4. These scores are higher the better. The normalized weighted fuzzy matrix calculated using Equation 4.6 to Equation 4.10 of Chapter 4 is given in Table A.5.

Table A.4 Scoring matrix for the economic dimension along with criteria weights

Code	SPIs / *Weights*	C1 (0.19, 0.32, 0.49)	C2 (0.20, 0.27, 0.35)	C3 (0.18, 0.24, 0.35)	C4 (0.11, 0.17, 0.27)
EC1	Total cost coverage ratio for water	(0.5, 0.7, 0.9)	(0.5, 1, 1)	(0, 0, 0.5)	(0, 0.5, 1)
EC2	Operating cost coverage ratio for water service	(0.7, 1, 1)	(0.5, 1, 1)	(0.5, 1, 1)	(0.5, 1, 1)
EC3	5 yrs running avg. capital reinvestment/ replacement value	(0.5, 0.7, 0.9)	(0, 0.5, 1)	(0, 0.5, 1)	(0.5, 1, 1)
EC4	Average water fee rate	(0.5, 0.7, 0.9)	(0.5, 1, 1)	(0.5, 1, 1)	(0.5, 1, 1)
EC5	Non-revenue water	(0.7, 1, 1)	(0.5, 1, 1)	(0, 0.5, 1)	(0.5, 1, 1)
EC6	Total cost coverage ratio for wastewater	(0.5, 0.7, 0.9)	(0.5, 1, 1)	(0, 0, 0.5)	(0, 0.5, 1)
EC7	Operating cost coverage ratio for wastewater service	(0.5, 0.7, 0.9)	(0.5, 1, 1)	(0.5, 1, 1)	(0.5, 1, 1)
EC8	5 yrs running average capital reinvestment/ replacement value	(0.5, 0.7, 0.9)	(0, 0.5, 1)	(0, 0.5, 1)	(0.5, 1, 1)

Table A.5 Normalized Weighted Fuzzy matrix

Code	Lower				Middle				Upper			
	C1	C2	C3	C4	C1	C2	C3	C4	C1	C2	C3	C4
EC1	0.04	0.04	0.00	0.00	0.31	0.39	0.00	0.20	2.46	3.26	3.25	4.40
EC2	0.06	0.04	0.03	0.03	0.45	0.39	0.52	0.39	2.73	3.26	6.50	4.40
EC3	0.04	0.00	0.00	0.03	0.31	0.20	0.26	0.39	2.46	3.26	6.50	4.40
EC4	0.04	0.04	0.03	0.03	0.31	0.39	0.52	0.39	2.46	3.26	6.50	4.40
EC5	0.06	0.04	0.00	0.03	0.45	0.39	0.26	0.39	2.73	3.26	6.50	4.40
EC6	0.04	0.04	0.00	0.00	0.31	0.39	0.00	0.20	2.46	3.26	3.25	4.40
EC7	0.04	0.04	0.03	0.03	0.31	0.39	0.52	0.39	2.46	3.26	6.50	4.40
EC8	0.04	0.00	0.00	0.03	0.31	0.20	0.26	0.39	2.46	3.26	6.50	4.40

The concordance and discordance sets calculated using Equation 4.11 and Equation 4.12 are given in Table A.6 and Table A.7.

Table A.6 Concordance sets for SPIs in the economic dimension

Lower value							
C (1,2) = {2}	C (2,1) = {1,2,3,4}	C (3,1) = {1,3,4}	C (4,1) = {1,2,3,4}	C (5,1) = {1,2,3,4}	C (6,1) = {1,2,3,4}	C (7,1) = {1,2,3,4}	C (8,1) = {1,3,4}
C (1,3) = {1,2,3}	C (2,3) = {1,2,3,4}	C (3,2) = {4}	C (4,2) = {2,3,4}	C (5,2) = {1,2,4}	C (6,2) = {2}	C (7,2) = {2,3,4}	C (8,2) = {4}
C (1,4) = {1,2}	C (2,4) = {1,2,3,4}	C (3,4) = {1,4}	C (4,3) = {1,2,3,4}	C (5,3) = {1,2,3,4}	C (6,3) = {1,2,3}	C (7,3) = {1,2,3,4}	C (8,3) = {1,2,3,4}
C (1,5) = {2,3}	C (2,5) = {1,2,3,4}	C (3,5) = {3,4}	C (4,5) = {2,3,4}	C (5,4) = {1,2,4}	C (6,4) = {1,2}	C (7,4) = {1,2,3,4}	C (8,4) = {1,4}
C (1 6) = {1,2,3,4}	C (2,6) = {1,2,3,4}	C (3,6) = {1,3,4}	C (4,6) = {1,2,3,4}	C (5,6) = {1,2,3,4}	C (6,5) = {2,3}	C (7,5) = {2,3,4}	C (8,5) = {3,4}
C (1,7) = {1,2}	C (2,7) = {1,2,3,4}	C (3,7) = {1,4}	C (4,7) = {1,2,3,4}	C (5,7) = {1,2,4}	C (6,7) = {1,2}	C (7,6) = {1,2,3,4}	C (8,6) = {1,3,4}
C (1,8) = {1,2,3}	C (2,8) = {1,2,3,4}	C (3,8) = {1,2,3,4}	C (4,8) = {1,2,3,4}	C (5,8) = {1,2,3,4}	C (6,8) = {1,2,3}	C (7,8) = {1,2,3,4}	C (8,7) = {1,4}
Middle value							
C (1,2) = {2}	C (2,1) = {1,2,3,4}	C (3,1) = {1,3,4}	C (4,1) = {1,2,3,4}	C (5,1) = {1,2,3,4}	C (6,1) = {1,2,3,4}	C (7,1) = {1,2,3,4}	C (8,1) = {1,3,4}
C (1,3) = {1,2}	C (2,3) = {1,2,3,4}	C (3,2) = {4}	C (4,2) = {2,3,4}	C (5,2) = {1,2,4}	C (6,2) = {2}	C (7,2) = {2,3,4}	C (8,2) = {4}
C (1,4) = {1,2}	C (2,4) = {1,2,3,4}	C (3,4) = {1,4}	C (4,3) = {1,2,3,4}	C (5,3) = {1,2,3,4}	C (6,3) = {1,2}	C (7,3) = {1,2,3,4}	C (8,3) = {1,2,3,4}
C (1,5) = {2}	C (2,5) = {1,2,3,4}	C (3,5) = {3,4}	C (4,5) = {2,3,4}	C (5,4) = {1,2,4}	C (6,4) = {1,2}	C (7,4) = {1,2,3,4}	C (8,4) = {1,4}
C (1,6) = {1,2,3,4}	C (2,6) = {1,2,3,4}	C (3,6) = {1,3,4}	C (4,6) = {1,2,3,4}	C (5,6) = {1,2,3,4}	C (6,5) = {2}	C (7,5) = {2,3,4}	C (8,5) = {3,4}
C (1,7) = {1,2}	C (2,7) = {1,2,3,4}	C (3,7) = {1,4}	C (4,7) = {1,2,3,4}	C (5,7) = {1,2,4}	C (6,7) = {1,2}	C (7,6) = {1,2,3,4}	C (8,6) = {1,3,4}
C (1,8) = {1,2}	C (2,8) = {1,2,3,4}	C (3,8) = {1,2,3,4}	C (4,8) = {1,2,3,4}	C (5,8) = {1,2,3,4}	C (6,8) = {1,2}	C (7,8) = {1,2,3,4}	C (8,7) = {1,4}
Upper value							
C (1,2) = {2,4}	C (2,1) = {1,2,3,4}	C (3,1) = {1,2,3,4}	C (4,1) = {1,2,3,4}	C (5,1) = {1,2,3,4}	C (6,1) = {1, 2, 3, 4}	C (7,1) = {1,2,3,4}	C (8,1) = {1,2,3,4}
C (1,3) = {1,2,4}	(2,3) = {1,2,3,4}	C (3,2) = {2,3,4}	C (4,2) = {2,3,4}	C (5,2) = {1,2,3,4}	C (6,2) = {2,4}	C (7,2) = {2,3,4}	C (8,2) = {2,3,4}
C (1,4) = {1,2,4}	C (2,4) = {1,2,3,4}	C (3,4) = {1,2,3,4}	C (4,3) = {1,2,3,4}	C (5,3) = {1,2,3,4}	C (6,3) = {1,2,4}	C (7,3) = {1,2,3,4}	C (8,3) = {1,2 3,4}
C (1,5) = {2,4}	C (2,5) = {1,2,3,4}	C (3,5) = {2,3,4}	C (4,5) = {2,3,4}	C (5,4) = {1,2,3,4}	C (6,4) = {1,2,4}	C (7,4) = {1,2,3,4}	C (8,4) = {1,2,3,4}
C (1,6) = {1,2,3,4}	C (2,6) = {1,2,3,4}	C (3,6) = {1,2,3,4}	C (4,6) = {1,2,3,4}	C (5,6) = {1, 2, 3, 4}	C (6,5) = {2,4}	C (7,5) = {2,3,4}	C (8,5) = {2,3,4}
C (1,7) = {1,2,4}	C (2,7) = {1,2,3,4}	C (3,7) = {1,2,3,4}	C (4,7) = {1,2,3,4}	C (5,7) = {1, 2, 3, 4}	C (6,7) = {1,2,4}	C (7,6) = {1,2,3,4}	C (8,6) = {1,2,3,4}
C (1,8) = {1,2,4}	C (2,8) = {1,2,3,4}	C (3,8) = {1,2,3,4}	C (4,8) = {1,2,3,4}	C (5,8) = {1, 2, 3, 4}	C (6,8) = {1,2,4}	C (7,8) = {1,2,3,4}	C (8,7) = {1,2,3,4}

Table A.7 Discordance sets for SPIs in the economic dimension

Lower value							
D (1,2) = {1,3,4}	D (2,1) = 0	D (3,1) = {2}	D (4,1) = 0	D (5,1) = 0	D (6,1) = 0	D (7,1) = 0	D (8,1) = {2}
D (1,3) = 0	D (2,3) = 0	D (3,2) = {1,2,3}	D (4,2) = {1}	D (5,2) = {3}	D (6,2) = {1,3,4}	D (7,2) = {1}	D (8,2) = {1,2,3}
D (1,4) = {3,4}	D (2,4) = 0	D (3,4) = {2,3}	D (4,3) = 0	D (5,3) = 0	D (6,3) = {4}	D (7,3) = 0	D (8,3) = 0
D (1,5) = {1,4}	D (2,5) = 0	D (3,5) = {1,2}	D (4,5) = {1}	D (5,4) = {3}	D (6,4) = {3,4}	D (7,4) = 0	D (8,4) = {2,3}
D (1,6) = 0	D (2,6) = 0	D (3,6) = {2}	D (4,6) = 0	D (5,6) = 0	D (6,5) = {1,4}	D (7,5) = {1}	D (8,5) = {1,2}
D (1,7) = {3,4}	D (2,7) = 0	D (3,7) = {2,3}	D (4,7) = 0	D (5,7) = {3}	D (6,7) = {3,4}	D (7,6) = 0	D (8,6) = {2}
D (1,8) = {4}	D (2,8) = 0	D (3,8) = 0	D (4,8) = 0	D (5,8) = 0	D (6,8) = {4}	D (7,8) = 0	D (8,7) = {2,3}
Middle value							
D (1,2) = {1,3,4}	D (2,1) = 0	D (3,1) = {2}	D (4,1) = 0	D (5,1) = 0	D (6,1) = 0	D (7,1) = 0	D (8,1) = {2}
D (1,3) = {3,4}	D (2,3) = 0	D (3,2) = {1,2,3}	D (4,2) = {1}	D (5,2) = {3}	D (6,2) = {1,3,4}	D (7,2) = {1}	D (8,2) = {1,2,3}
D (1,4) = {3 4}	D (2,4) = 0	D (3,4) = {2,3}	D (4,3) = 0	D (5,3) = 0	D (6,3) = {3,4}	D (7,3) = 0	D (8,3) = 0
D (1,5) = {1,3,4}	D (2,5) = 0	D (3,5) = {1,2}	D (4,5) = {1}	D (5,4) = {3}	D (6,4) = {3,4}	D (7,4) = 0	D (8,4) = {2,3}
D (1,6) = 0	D (2,6) = 0	D (3,6) = {2}	D (4,6) = 0	D (5,6) = 0	D (6,5) = {1,3,4}	D (7,5) = {1}	D (8,5) = {1,2}
D (1,7) = {3,4}	D (2,7) = 0	D (3,7) = {2,3}	D (4,7) = 0	D (5,7) = {3}	D (6,7) = {3,4}	D (7,6) = 0	D (8,6) = {2}
D (1,8) = {3,4}	D (2,8) = 0	D (3,8) = 0	D (4,8) = 0	D (5,8) = 0	D (6,8) = {3,4}	D (7,8) = 0	D (8,7) = {2,3}
Upper value							
D (1,2) = {1,3}	D (2,1) = 0	D (3,1) =0	D (4,1) = 0	D (5,1) = 0	D (6,1) = 0	D (7,1) = 0	D (8,1) = 0
D (1,3) = {3}	D (2,3) = 0	D (3,2) = {1}	D (4,2) = {1}	D (5,2) = 0	D (6,2) = {1,3}	D (7,2) = {1}	D (8,2) = {1}
D (1,4) = {3}	D (2,4) = 0	D (3,4) = 0	D (4,3) = 0	D (5,3) = 0	D (6,3) = {3}	D (7,3) = 0	D (8,3) = 0
D (1,5) = {1,3}	D (2,5) = 0	D (3,5) = { 1}	D (4,5) = {1}	D (5,4) = 0	D (6,4) = {3}	D (7,4) = 0	D (8,4) = 0
D (1,6) = 0	D (2,6) = 0	D (3,6) = 0	D (4,6) = 0	D (5,6) = 0	D (6,5) = {1,3}	D (7,5) = {1}	D (8,5) = {1}
D (1,7) = {3}	D (2,7) = 0	D (3,7) = 0	D (4,7) = 0	D (5,7) = 0	D (6,7) = {3}	D (7,6) = 0	D (8,6) = 0
D (1,8) = {3}	D (2,8) = 0	D (3,8) = 0	D (4,8) = 0	D (5,8) = 0	D (6,8) = {3}	D (7,8) = 0	D (8,7) = 0

The concordance and discordance indices calculated using Equation 4.13 and Equation 4.14 are given in following matrices.

Concordance index

For lower values (l)

C =

-	0.69	0.30	0.69	0.50	0.50	0.69	0.30
0.20	-	0.11	0.32	0.50	0.20	0.32	0.11
0.39	0.69	-	0.69	0.50	0.39	0.69	0.50
0.39	0.50	0.30	-	0.50	0.39	0.50	0.30
0.20	0.69	0.11	0.50	-	0.20	0.50	0.11
0.50	0.69	0.30	0.69	0.50	-	0.69	0.30
0.39	0.50	0.30	0.50	0.50	0.39	-	0.30
0.39	0.69	0.50	0.69	0.50	0.39	0.69	-

For middle value (m)

C =

-	1.00	0.73	1.00	1.00	1.00	1.00	0.73
0.27	-	0.17	0.68	0.76	0.27	0.68	0.17
0.59	1.00	-	1.00	1.00	0.59	1.00	1.00
0.59	1.00	0.49	-	0.76	0.59	1.00	0.49
0.27	1.00	0.41	0.68	-	0.27	0.68	0.41
1.00	1.00	0.73	1.00	1.00	-	1.00	0.73
0.59	1.00	0.49	1.00	0.76	0.59	-	0.49
0.59	1.00	1.00	1.00	1.00	0.59	1.00	-

For upper values (u)

C =

-	1.45	1.45	1.45	1.45	1.45	1.45	1.45
0.62	-	0.97	0.97	1.45	0.62	0.97	0.97
1.10	1.45	-	1.45	1.45	1.10	1.45	1.45
1.10	1.45	1.45	-	1.45	1.10	1.45	1.45
0.62	1.45	0.97	0.97	-	0.62	0.97	0.97
1.45	1.45	1.45	1.45	1.45	-	1.45	1.45
1.10	1.45	1.45	1.45	1.45	1.10	-	1.45
1.10	1.45	1.45	1.45	1.45	1.10	1.45	-

Discordance index

For lower values (l)

D =

-	0.00	0.57	0.00	0.00	0.00	0.00	0.57
1.00	-	1.00	1.00	1.00	1.00	1.00	1.00
0.43	0.00	-	0.00	0.00	0.43	0.00	0.00
1.00	0.00	1.00	-	0.66	1.00	0.00	1.00
1.00	0.00	1.00	0.34	-	1.00	0.34	1.00
0.00	0.00	0.57	0.00	0.00	-	0.00	0.57
1.00	0.00	1.00	0.00	0.66	1.00	-	1.00
0.43	0.00	0.00	0.00	0.00	0.43	0.00	-

For middle values (m)

D =

-	0.00	0.30	0.00	0.00	0.00	0.00	0.30
1.00	-	1.00	1.00	1.00	1.00	1.00	1.00
0.70	0.00	-	0.00	0.00	0.70	0.00	0.00
1.00	0.00	1.00	-	0.66	1.00	0.00	1.00
1.00	0.00	1.00	0.34	-	1.00	0.34	1.00
0.00	0.00	0.30	0.00	0.00	-	0.00	0.30
1.00	0.00	1.00	0.00	0.66	1.00	-	1.00
0.70	0.00	0.00	0.00	0.00	0.70	0.00	-

For upper values (u)

D =

-	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1.00	-	1.00	1.00	0.00	1.00	1.00	1.00
1.00	0.00	-	0.00	0.00	1.00	0.00	0.00
1.00	0.00	0.00	-	0.00	1.00	0.00	0.00
1.00	0.00	1.00	1.00	-	1.00	1.00	1.00
0.00	0.00	0.00	0.00	0.00	-	0.00	0.00
1.00	0.00	0.00	0.00	0.00	1.00	-	0.00
1.00	0.00	0.00	0.00	0.00	1.00	0.00	-

The final concordance (C^*_{pq}) and discordance (D^*_{pq}) indices were computed using Equation 4.15. The average values of C^*_{pq} and D^*_{pq} are respectively 0.7426 and 0.3151. An outranking relationship was obtained by comparing the C^*_{pq} and D^*_{pq} with their averages using Equation 4.16 and Equation 4.17. The outranking diagram is given in Figure 4.5 (Chapter 4). The net concordance index (Cp) and the net discordance index (Dp) calculated for each SPI using Equation 4.18 and Equation 4.19 are given in Table A.8 with their ranks. Cp and Dp are also plotted in Figure A.1 with their final preferences.

Table A.8 Ranking of the SPIs of economic dimension

Rank	SPI	Cp	Dp	Combined value (Cp + [-1*Dp])
1	EC2	3.62	-6.00	9.62
2	EC5	2.34	-4.98	7.31
3	EC4	1.09	-0.51	1.61
3	EC7	1.09	-0.51	1.61
5	EC3	-1.98	0.67	-2.65
5	EC8	-1.98	0.67	-2.65
7	EC1	-2.09	5.33	-7.43
7	EC6	-2.09	5.33	-7.43

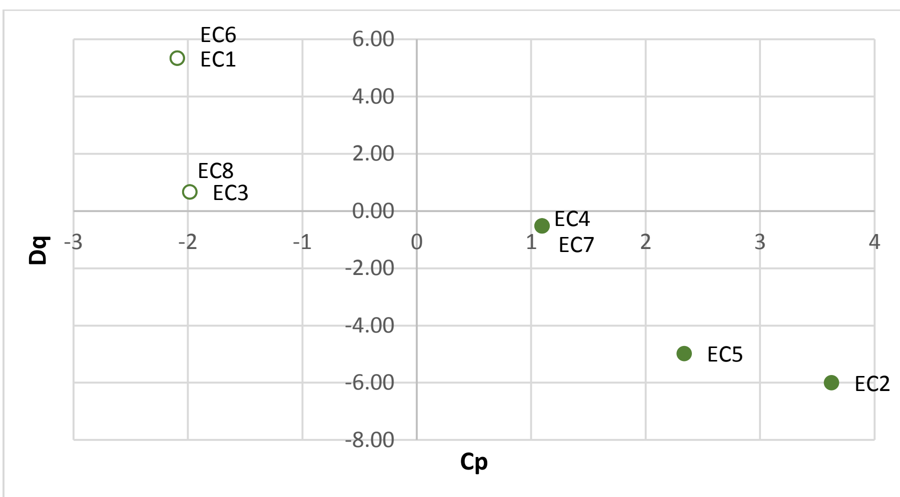

Figure A.1 Net concordance (Cp) and discordance (Dq) indices of the SPIs with the final preferences represented by solid circles in the economic dimension

Appendix B Additional Information on WEC Modelling

B.1 Stock and flow diagrams of the WEC model

The water-energy-carbon (WEC) model is based on the stock and flow diagrams of three modules: water, energy, and carbon. These modules were constructed in five sub-modules for the ease of mapping and presentation as shown in Figure B.1. The sub-modules are as follows:

i. Resiwater sub-module: Residential water and total water footprint modelling
ii. NonResiwater sub-module: Non-residential (commercial, institutional, and industrial, CII) water modelling
iii. Direct energy sub-module: Operational energy modelling for residential and non-residential water
iv. IWEC sub-module: Indirect Water, Energy and Carbon (IWEC) modelling respectively for indirect water footprint, embodied energy and carbon footprint of energy and major chemicals
v. Direct carbon sub-module: Carbon emission modelling of operational energy and total carbon emisssions

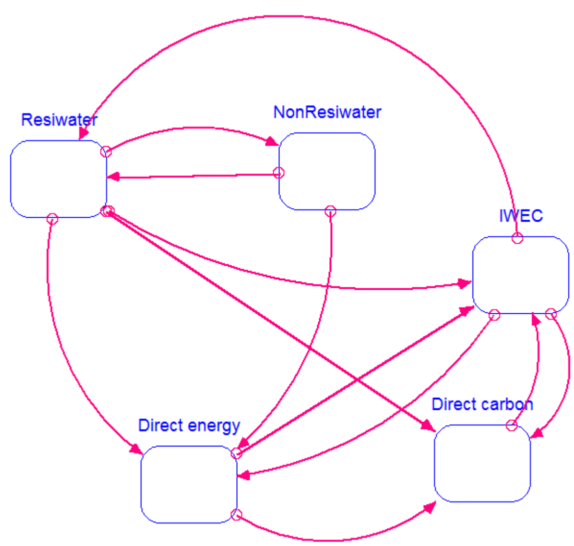

Figure B.1 Overview of five sub-modules of the WEC model

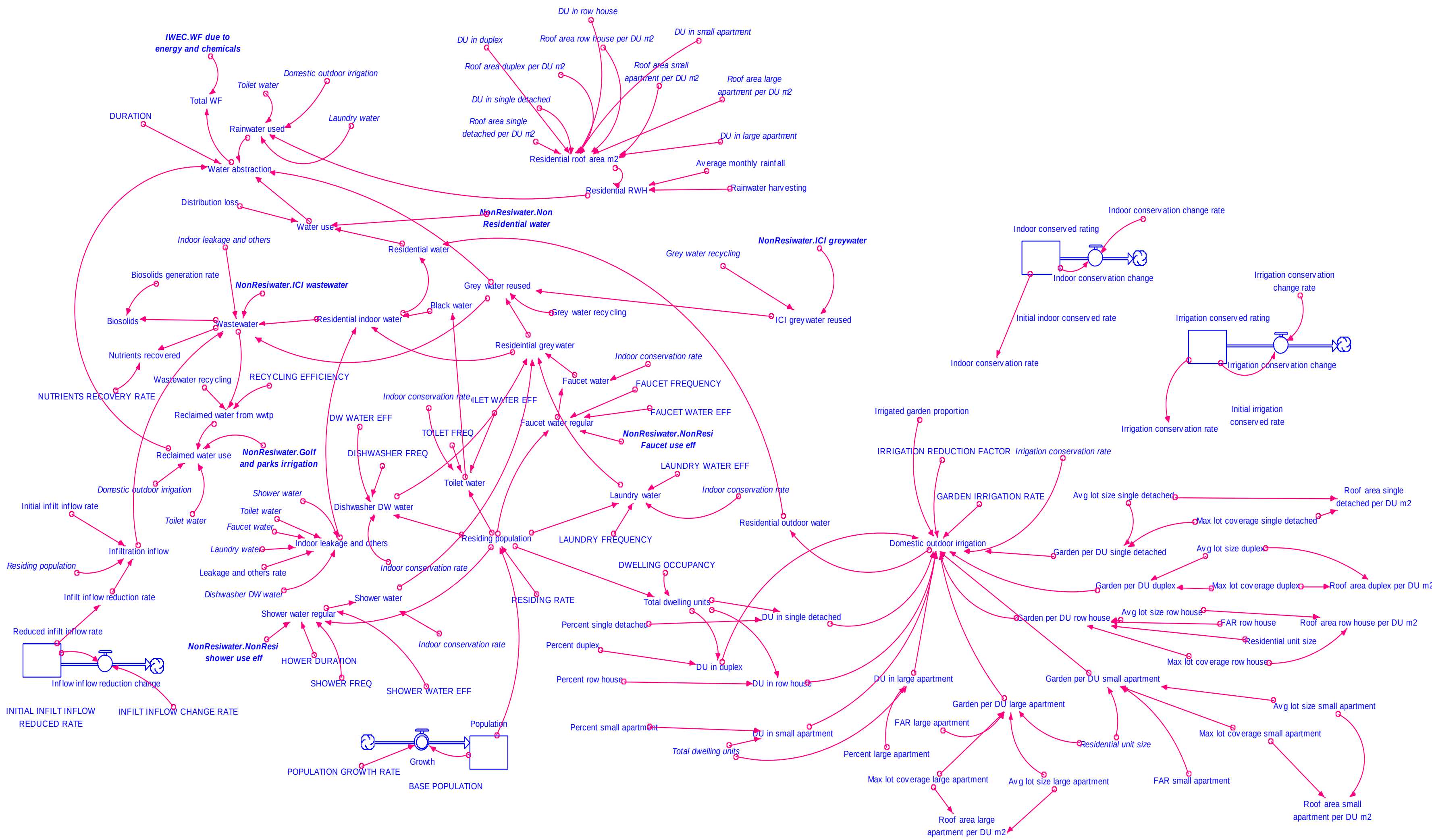

Figure B.2 Stock and flow diagram of "Resiwater" sub-module of the WEC model

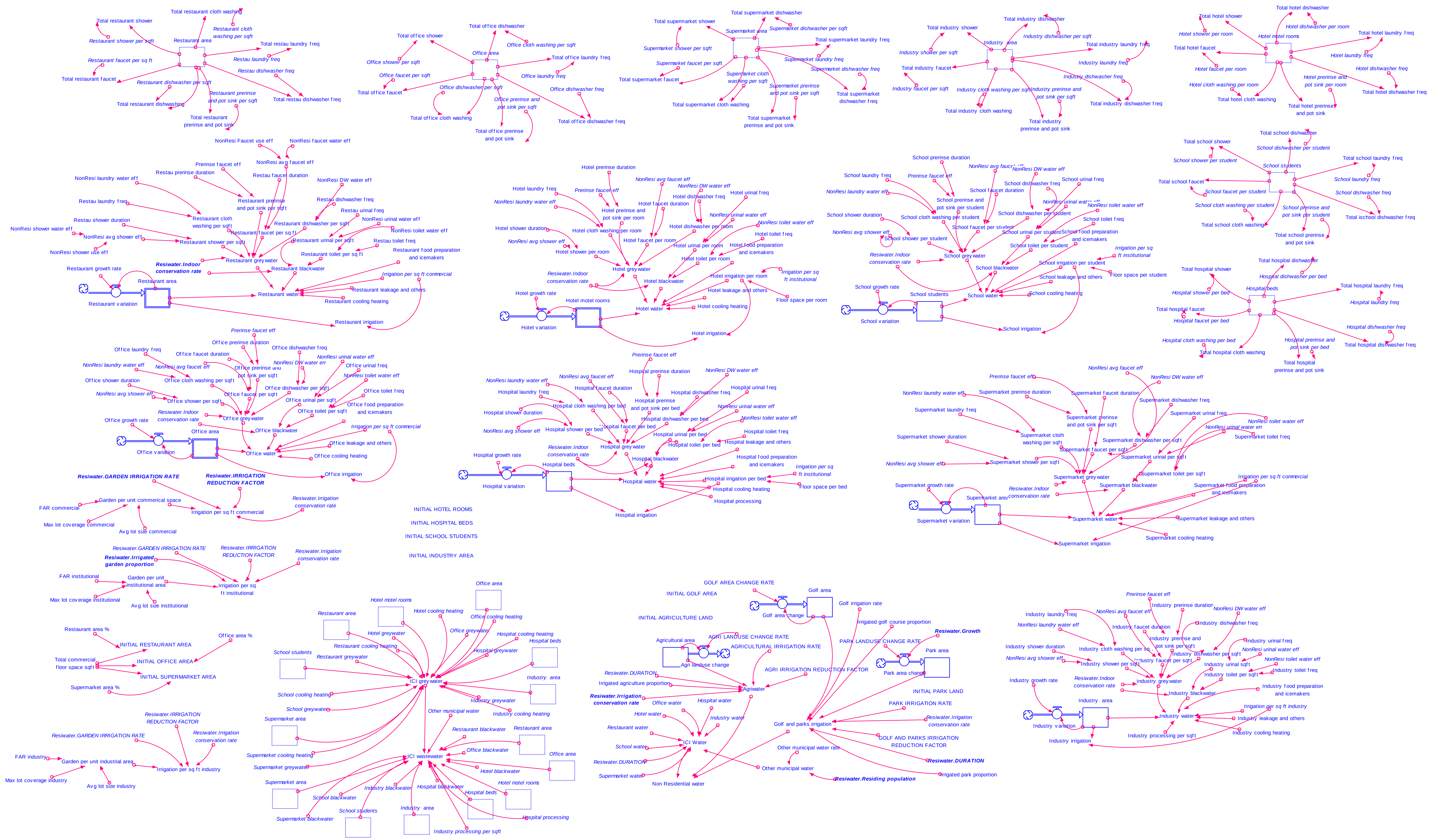

Figure B.3 Stock and flow diagram of “NonResiwater” sub-module of the WEC model

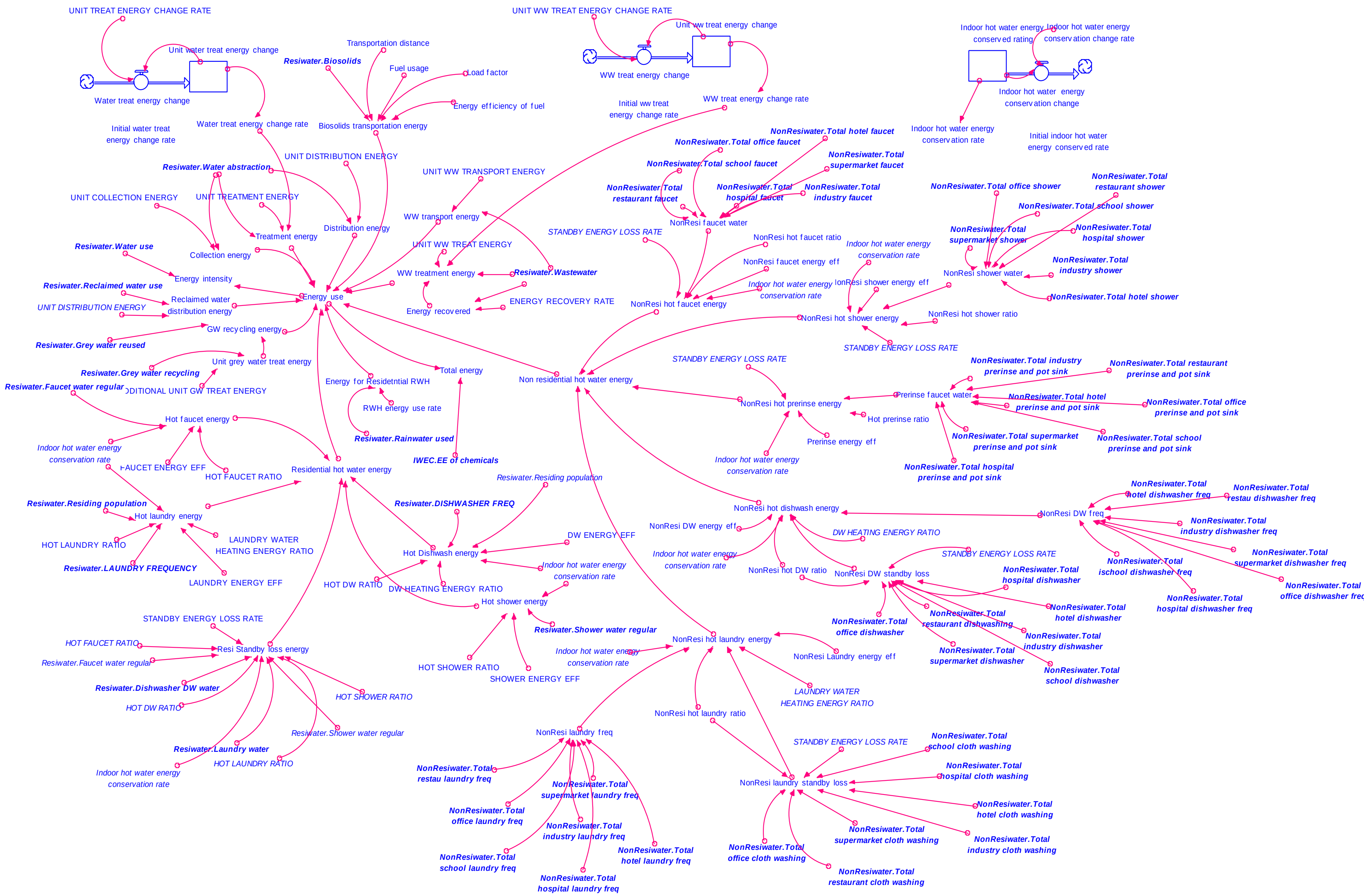

Figure B.4 Stock and flow diagram of "Direct energy" sub-module of the WEC model

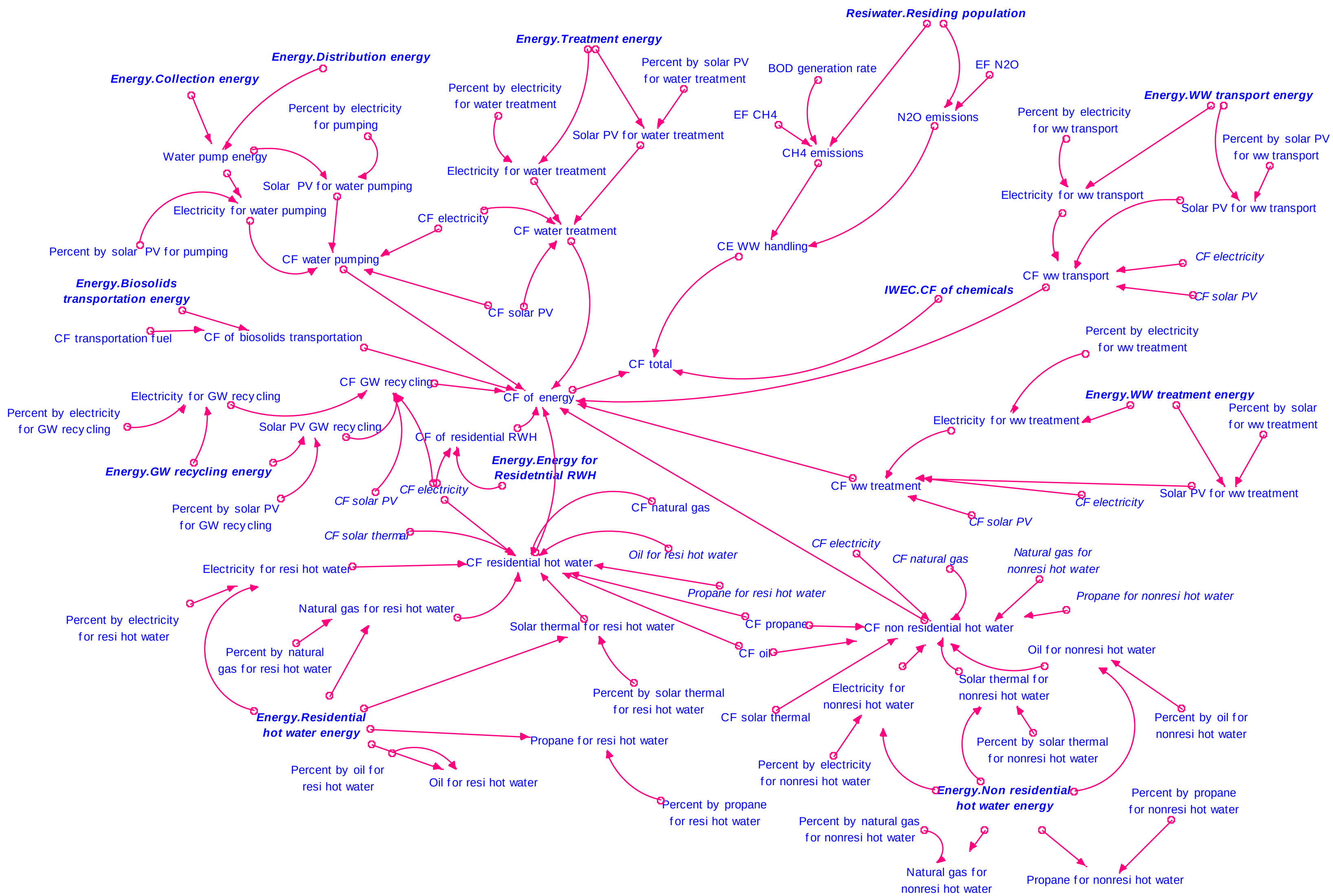

Figure B.5 Stock and flow diagram of "Direct carbon" sub-module of the WEC model

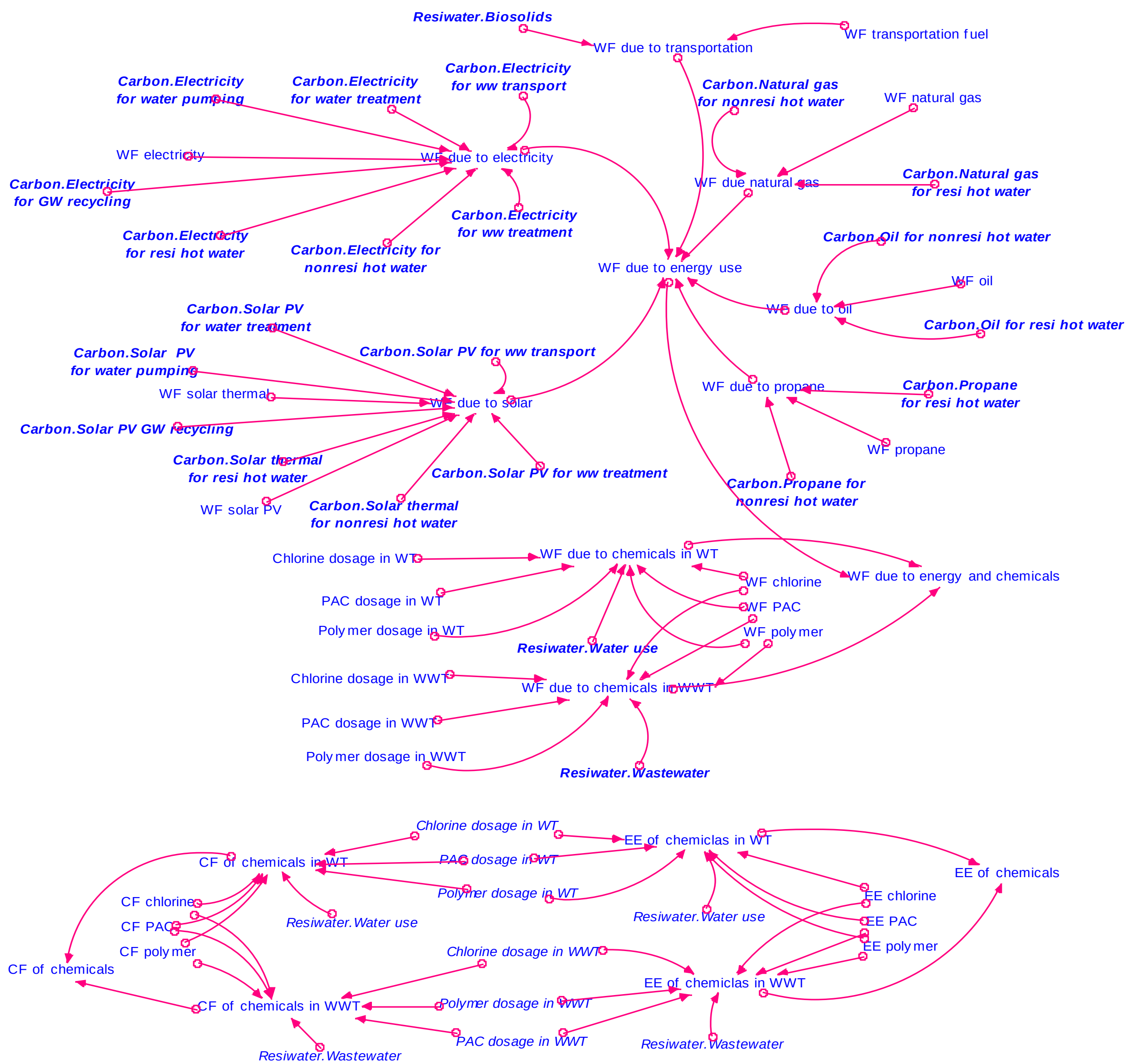

Figure B.6 Stock and flow diagram of "IWEC" sub-module of the WEC model

B.2 Data requirements

B.2.1 Data estimation method

a) Water and wastewater

The rates, i.e., frequency and duration of water fixture and appliance uses in various residential indoor consumption were obtained from the extensive study by Mayer et al. (1999), which includes two Canadian cities. The water efficiencies of conventional and efficient water fixtures and appliances were obtained from Mayer et al. (1999), ENERGY STAR (2014a), and ENERGY STAR (2014b). Similarly, the end use rates of water by the commercial and institutional (CI) sector were obtained from the CI water use studies by the US EPA (2009b) and Dziegielewski et al. (2000). The average industrial water use rate was obtained from (Briere 2010). Moreover, the average lot coverages (%) for different residential houses and CII buildings were estimated by using Google Earth and was verified with the zoning bylaws. Furthermore, the irrigation rates for lawns, golf courses, and agriculture land were obtained from the regional database for Okanagan (OBWB 2010).

The rate of change of indoor water conservation rate (r) used was 0.002/month for the City of Penticton. The rate was calculated by considering 50% increment to the water conservation rate estimated from the Penticton water use during 1998-2004 (City of Penticton 2006) to accommodate intensive water conservation, including toilet rebate program. Similarly, the monthly average infiltration and inflow rate was estimated as the monthly average difference between influent to a wastewater treatment plant and wastewater generated from the residential and CII sector by using the historical Penticton wastewater data. However, in course of time, the monthly infiltration and inflow rate may increase in deteriorating sewer networks or may decrease in regularly maintained and upgraded sewer networks for infiltration-inflow control. Penticton has a decreasing infiltration and inflow rate due to the intensive maintenance of the sewer network using smoke testing to reduce infiltration-inflow according to the municipality (Pers. comm.). The average decreasing rate of change of monthly infiltration-flow rate (r) was 0.00363/month for 2006-2007; however, for the validation period, the r value with 2.5 times higher than the average, i.e., 0.0059/month was considered to incorporate the intensive maintenance in that duration. The decreasing rate factor (X_t) was estimated as $X_t = X_0 * e^{-rt}$, which was multiplied to the monthly average infiltration-inflow rate to obtain the infiltration-inflow rate of that month. Moreover, the water footprint of major chemicals used in water and wastewater treatments, such as chlorine, poly aluminum chloride, and polymers were obtained by conducting a life cycle assessment (LCA). The LCA was performed for chemical production (gate to gate phase) by using the SimaPro 8.0.5 software (Risch et al. 2014).

b) Energy

The rate of change of indoor hot water energy conservation rate (r) was estimated to be 0.001/month based on the water heating energy data from 1990 to 2011 (Natural Resources Canada 2014). Moreover, the monthly average energy consumption rates were obtained separately for raw water collection, water treatment, distribution, wastewater transport, wastewater treatment, and average dosages of major chemicals consumed from the Penticton water treatment plant and wastewater treatment plant. In particular, the monthly energy use rate was increasing in these utilities mainly due to plant upgrading and a large energy intensity of producing a high quality water (Santana et al. 2014) to meet the increasingly stringent standards of drinking water quality and effluent discharge quality. The average rate of increase of monthly energy consumption rate (r) was 0.00028/month for water treatment and 0.01034/month for wastewater treatment based on the data of 2007-2012. The increasing rate factor (X_t) was estimated as $X_t = X_0 * e^{rt}$, which was multiplied to the monthly average to obtain the monthly energy consumption rate in that month for both water and wastewater energy.

B.2.2 Data Table

The data used in the modelling and simulation are categorized into three levels: regional (R) containing regional and site-specific data; national (N), and global (G). The data are given in Table B.1. Although the required dataset is large, the regional data constituting site-specific data is about one-third. Important data can be input from the interface of the WEC tool and if all or detailed data are available, the data can also be imported from a spreadsheet linked to the SDM.

Table B.1 Data used for the parameters of the WEC model

Parameters	Values	Data features	Level	Reference
Population growth rate (%)*	0.006/yr	Penticton data	R	Statistics Canada (2015a)
Base population	31624 persons	Penticton data	R	”
Residing rate	1	Seasonal migration not considered	R	-
Dwelling occupancy	2.20 persons/ dwelling unit	Penticton data	R	Statistics Canada (2015a)
Percent single detached (%)	0.518	Penticton data	R	”
Percent duplex (%)	0.037	Penticton data	R	”
Percent row house (%)	0.1	Penticton data	R	”
Percent small apartment (%)	0.289	Penticton data	R	”
Percent large apartment (%)	0.056	Penticton data	R	”
Avg lot size single family	0.0776 ha	Estimated from municipal data	R	GIS database
Avg lot coverage single family	0.43 ha	Estimated	R	City of Penticton (2015a); Google Earth
Avg lot size duplex	0.06 ha	Estimated from municipal data	R	GIS database
Avg lot coverage duplex	0.45 ha	Estimated	R	City of Penticton (2015a); Google Earth
Avg lot size row house	0.2583 ha	Estimated from municipal data	R	GIS database
Avg lot coverage row house (%)	0.70	Estimated	R	City of Penticton (2015a); Google Earth
Floor area ratio (FAR) row house	1.38	Zoning bylaw	R	City of Penticton (2015a)
Residential unit size	1900 sqft	National average	N	CHBA (2012)
Avg lot size small apartment	0.18 ha	Estimated from municipal data	R	Municipal GIS database
Avg lot coverage small apartment (%)	0.70	Estimated	R	City of Penticton (2015a); Google Earth
FAR small apartment	1.38	Penticton data	R	City of Penticton (2015a)
Avg lot size large apartment	0.18 ha	Estimated from municipal data	R	Municipal GIS database
Avg lot coverage large apartment (%)	0.63	Estimated	R	City of Penticton (2015a); Google Earth
Floor area ratio (FAR) large apartment	1.60	Zoning bylaw	R	City of Penticton (2015a)
Garden irrigation rate	Monthly rate for 12 months	-	R	OBWB (2010); City of Penticton (2014a)
Irrigation reduction factor (%)	0.50	For xeriscaping	N,G	Boot and Parchomchuk (2009)
Laundry frequency	11.25 per mo/cap	Average from a North American Study	N,G	Mayer et al. (1999)
Laundry water eff	104.60 L/load	Average from a North American Study	N,G	”
Hot laundry ratio	0.28	National average		REUM model - Aguilar et al. (2005)
Laundry energy eff	1.49 kWh/load	National average	N,G	”
Laundry water heating ratio	0.92	National average	N	”
Laundry energy eff	1.49 kWh/load	National average	N,G	”
Laundry water heating ratio	0.92	National average	N	”
Standby energy loss rate	0.007536 kWh/L hot water	National average	N,G	”

Faucet frequency	246.321 min/mo/cap	Average from a North American Study	N,G	Mayer et al. (1999)
Faucet water eff	7.6 L/min	Average from a North American Study	N,G	”
Faucet energy eff	0.0449 kWh/L	National average	N,G	REUM model - Aguilar et al. (2005)
Hot faucet ratio	0.727	National average	N	”
Toilet freq	153.5705 flush/mo/cap	Average from a North American Study	N,G	Mayer et al. (1999)
Toilet water eff	10.7 L/flush	Average from a North American Study	N,G	”
Dishwasher freq	3.041 wash/mo/cap	Average from a North American Study	N,G	”
DW water eff	25.4 L/cycle	Average from a North American Study	N,G	”
Hot DW ratio	1	National average	N	REUM model Aguilar et al. (2005)
DW energy eff	3.05502 kWh/hot cycle	National average	N,G	Aguilar et al. (2005)
DW heating energy ratio	0.88	National average	N,G	”
Shower freq	22.8075 per mo/cap	Average from a North American Study	N,G	Mayer et al. (1999)
Shower water eff	10.8 L/min	Average from a North American Study	N,G	”
Shower duration	8.2 min/shower	Average from a North American Study	N,G	”
Shower energy eff	0.044879 kWh/L	National average	N,G	REUM model Aguilar et al. (2005)
Hot shower ratio	0.782	National average	N,G	”
Leakage and others rate (%)	0.05	The study gave 13% leakage but considered a low value of 5%	N	Mayer et al. (1999)
Unit collection energy	Monthly rate for 12 mos in kWh/L	Based on 2010 to 2012	R	City of Penticton (2015b)
Unit treatment energy	Monthly rate for 12 mos in kWh/L	Based on 2010 to 2012	R	”
Unit distribution energy	Monthly rate for 12 mos in kWh/L	Based on 2010 to 2012	R	”
Unit WW transport energy	Monthly rate for 12 mos in kWh/L	Based on 2011 to 2012	R	City of Penticton (2015c)
Unit WW treat energy	Monthly rate for 12 mos in kWh/L	Based on 2011 to 2012	R	”
Recycling efficiency	0.95	-	R	Annual reports of the Penticton wastewater treatment plant
Park irrigation rate	Monthly rate in L/ha/mo	Monthly rate	R	OBWB (2010); City of Penticton (2014a)
Initial park land	108.98 ha	Official community plan	R	City of Penticton (2014b)
Park landuse change rate	4.05E-03 ha/cap	Official community plan	R	City of Penticton (2014b)
Golf and parks irrigation reduction factor (%)	0.50	For xeriscaping	N	Boot and Parchomchuk (2009)
AVG lot size commercial	0.31 ha	Commercial (ha)= restaurant and offices	R	GIS database
Avg lot coverage commercial (%)	0.61	Estimated	R	City of Penticton (2015a); Google Earth
FAR commercial	2.00	Penticton Zoning bylaw	R	City of Penticton (2015a)

Avg lot size institutional	0.73 ha	Institutional: schools, hospitals, hotels	R	GIS database
Avg lot coverage institutional (%)	0.73	Estimated	R	City of Penticton (2015a); Google Earth
FAR institutional	3.60	Penticton Zoning bylaw		City of Penticton (2015a)
Avg lot size industry	1.63 ha	Estimated from municipal database	R	GIS database
Avg lot coverage industry (%)	0.90	Estimated	R	City of Penticton (2015a); GoogleEarth
FAR industry	3.00	Penticton Zoning bylaw	R	City of Penticton (2015a)
Total commercial floor space sqft	4,912,941 sqft	-	R	City of Penticton (2015d)
Restaurant area %	0.07	Estimated from GIS data	R	GIS database
Office area %	0.38	Estimated from GIS data	R	”
Supermarket area %	0.55	Estimated from GIS data	R	”
Restaurant growth rate	0.60%/yr	Considered same as population growth rate	R	-
Office growth rate	0.60%/yr	Considered same as population growth rate	R	-
Supermarket growth rate	0.60%/yr	Considered same as population growth rate	R	-
Initial industry area	1,793,247 sqft	Estimated from GIS data	R	GIS database
Industry growth rate	0.60%/yr	Considered same as population growth rate	R	-
Initial hotel rooms	1783 rooms	Number of rooms as per BC Stats	R	City of Penticton (2015f)
Hotel growth rate	0.60%/yr	Considered same as population growth rate	R	-
Initial hospital beds	129 beds	-	R	Interior Health Authority (2015)
Hospital growth rate	0.60%/yr	Considered same as population growth rate	R	-
Initial school students	4970 students	-	R	Okanagan College (2011); City of Penticton (2015g)
School growth rate	0.60%/yr	Considered same as population growth rate	R	-
Floor space per room	290 sqft/room	-	N,G	Colorado Waterwise Council (2007)
Floor space per bed	460 sqft/bed	-	N,G	”
Floor space per student	144 sqft/student	-	N,G	”
Restau Laundry freq	0.0407 cycle/sqft/mo	Estimated based on the North American study	N,G	Dziegielewski et al. (2000); Gleick et al. (2003); US EPA (2009b)
Hotel laundry freq	34.36 cycles/room/ mo	”	N,G	”
Office laundry freq	0 cycle/sqft/mo	”	N,G	”
Hospital laundry freq	34.92 cycles/bed/mo	”	N,G	”
Industry laundry freq	0 cycles/sqft/mo	”	N,G	”
School laundry freq	0.06241 cycle/ student/mo	”	N,G	”
Supermarket laundry freq	0.01827 cycle/sqft/mo	”	N,G	”

NonResi Laundry water eff	104.60 L/cycle	”	N,G	”
Restau faucet duration	0.25907 min/sqft/mo	”	N,G	”
Hotel faucet duration	52.8585 min/room/mo	”	N,G	”
Office faucet duration	0.01801 min/sqft/mo	”	N,G	”
Hospital faucet duration	114.59 min/bed/mo	”	N,G	”
Industry faucet duration	0.018014 min/sqft/mo	”	N,G	”
School faucet duration	11.5753 min/student/mo	”	N,G	”
Supermarket faucet duration	0.04122 min/sqft/mo	”	N,G	”
NonResi Faucet water eff	7.6 L/min	”	N,G	”
NonResi Faucet use eff (%)	0.9175	”	N,G	”
Restau toilet freq	2.086 flush/sqft/mo	”	N,G	”
Hotel toilet freq	169.61 flush/room/mo	”	N,G	”
Office toilet freq	0.14506 flush/sqft/mo	”	N,G	”
Hospital toilet freq	922.82 flush/bed/mo	”	N,G	”
Industry toilet freq	0.14506 flush/sqft/mo	”	N,G	”
School toilet freq	25.7728 flush/student/mo	”	N,G	”
Supermarket toilet freq	0.12072 flush/sqft/mo	”	N,G	”
NonResi Toilet water eff	11.4 L/flush	”	N,G	”
NonResi Urinal water eff	6.1 L/flush	”	N,G	”
Restau urinal freq	0.92057 flush/sqft/mo	”	N,G	”
Hotel urinal freq	74.844 flush/room/mo	”	N,G	”
Office urinal freq	0.0640 flush/sqft/mo	”	N,G	”
Hospital urinal freq	407.2045 flush/bed/mo	”	N,G	”
Industry urinal freq	0.0640 flush/sqft/mo	”	N,G	”
School urinal freq	11.37 flush/student/mo	”	N,G	”
Supermarket urinal freq	0.05327 flush/sqft/mo	”	N,G	”
Restau dishwasher freq	0.56885 cycle/sqft/mo	”	N,G	”
Hotel dishwasher freq	34.979 cycle/room/mo	”	N,G	”
Office dishwasher freq	0.000638 cycle/sqft/mo	”	N,G	”
Hospital dishwasher freq	32.50 cycle/bed/mo	”	N,G	”
Industry dishwasher freq	0.000638 cycle/sqft/mo	”	N,G	”

School dishwasher freq	0.70600 cycle/sqft/mo	”	N,G	”
Supermarket dishwasher freq	0.02125 cycle/sqft/mo	”	N,G	”
NonResi DW water eff	25.4 L/cycle	”	N,G	”
NonResi Shower water eff	10.9 L/min	”	N,G	”
NonResi Shower use eff (%)	0.94		N,G	
Restau shower duration	0.2785 min/sqft/mo	”	N,G	”
Hotel shower duration	491.53 min/room/mo	”	N,G	”
Office shower duration	0.0193 min/sqft/mo	”	N,G	”
Hospital shower duration	123.2291 min/bed/mo	”	N,G	”
Industry shower duration	0.01937 min/sqft/mo	”	N,G	”
School shower duration	0.7866 min/student/mo	”	N,G	”
Supermarket shower duration	0 min/sqft/mo	”	N,G	”
Prerinse faucet eff	11.9 L/min	”	N,G	”
Restau prerinse duration	1.5150 min/sqft/mo	”	N,G	”
Hotel prerinse duration	93.16 min/sqft/mo	”	N,G	”
Office prerinse duration	0.0017 min/sqft/mo	”	N,G	”
Hospital prerinse duration	86.56 min/bed/mo	”	N,G	”
Industry prerinse duration	0.00170 min/sqft/mo	”	N,G	”
School prerinse duration	1.8803 min/sqft/mo	”	N,G	”
Supermarket prerinse duration	0.0566 min/sqft/mo	”	N,G	”
Restaurant cooling heating	0.3516 L/sqft/mo	”	N,G	”
Hotel cooling heating	815.4750 L/room/mo	”	N,G	”
Office cooling heating	0.53057 L/sqft/mo	”	N,G	”
Hospital cooling heating	1567.07 L/bed/mo	”	N,G	”
Industry cooling heating	0.5305 L/sqft/mo	”	N,G	”
School cooling heating	32.9176 L/student/mo	”	N,G	”
Supermarket cooling heating	2.0344 L/sqft/mo	”	N,G	”
Restaurant leakage and others	8.5247 L/sqft/mo	”	N,G	”
Hotel leakage and others	2920.43 L/room/mo	”	N,G	”
Office leakage and others	0.5168 L/sqft/mo	”	N,G	”
Hospital leakage and others	2922.29 L/bed/mo	”	N,G	”
Industry leakage and others	0.5167 L/sqft/mo	”	N,G	”
School leakage and others	54.409 L/student/mo	”	N,G	”
Supermarket leakage and others	0.9616 L/sqft/mo	”	N,G	”
Restaurant food preparation and icemakers	18.66 L/sqft/mo	”	N,G	”

Hotel food preparation and icemakers	1147.95 L/bed/mo	”	N,G	”
Office food preparation and icemakers	0.02096 L/sqft/mo	”	N,G	”
Hospital food preparation and icemakers	1066.63 L/bed/mo	”	N,G	”
Industry food preparation and icemakers	0.02096 L/sqft/mo	”	N,G	”
School food preparation and icemakers	23.17 L/student/mo	”	N,G	”
Supermarket food preparation and icemakers	0.6976 L/sqft/mo	”	N,G	”
NonResi laundry energy eff	1.43 kWh/hot cycle	Considered same as the residential sector	N,G	Aguilar et al. (2005)
NonResi hot laundry ratio	0.28	”	N,G	”
NonResi faucet energy eff	0.044879 kWh/L	”	N,G	”
NonResi hot faucet ratio	0.727	”	N,G	”
NonResi DW energy eff	2.5502 kWh/hot cycle	”	N,G	”
NonResi Hot DW ratio	1	”	N,G	”
NonResi shower energy eff	0.044879 kWh/L	”	N,G	”
NonResi hot shower ratio	0.782	”	N,G	”
Prerinse energy eff	0.044879 kWh/L	”	N,G	”
Hot prerinse ratio	0.727	”	N	”
Average monthly rainfall	Monthly data for 12 months	”	R	Government of Canada (2015)
Rainwater harvesting	0 (No)	-	R	-
Agricultural irrigation rate	Monthly data for 12 months	Monthly average	R	OBWB (2010); City of Penticton (2014a)
Initial agriculture land	0 ha	Drinking water not for agricultural irrigation	R	-
Agri landuse change rate	0.00	-	R	-
Agri irrigation reduction factor	0.00	-	R	-
Initial indoor conserved rate	1.000	Initial factor	R	-
Indoor conservation change rate	0.002/mo	Rate of decrease in per capita demand during 1998 -2004 with the decrement rate scaled up by 50% to accommodate intensive conservation including toilet rebate program	R	City of Penticton (2006)
Initial irrigation conserved rate	1.00	Initial factor	R	-
Irrigation Conservation change rate	0.002/mo	Considered same rate as of indoor conservation rate	R	-
Other municipal water rate	421.5 L/cap/mo	Fire-fighting, street cleaning, flushing, etc. (Obtained from Peachland)	R	District of Peachland (2014b)
Initial golf area	0.00 ha	Golf courses use private water (underground water) for irrigation	-	-
Golf irrigation rate	Monthly data for 12 months	Monthly rate	R	OBWB (2010); City of Penticton (2014a)
Carbon footprint (CF) electricity	0.003 kg CO_2e/kWh	For FortisBC	R	Ministry of Environment (2013)
CF solar thermal	0.01735 kg CO_2e/kWh	Carbon footprint of solar thermal energy	G	Menzies and Roderick (2010)

CF solar PV	0.04991 kg CO2e/kWh	Carbon footprint of solar PV energy	G	Nugent and Sovacool (2014)
CF natural gas	0.18 kg CO2e/kWh	Fortis BC	N	Ministry of Environment (2013)
CF oil	0.25219 kg CO2e/kWh	Provincial/national average	N	”
CF propane	0.22 kg CO2e/kWh	Provincial/national average	N	”
Percent by electricity for pumping (%)	1	All pumping by electricity	N	City of Penticton (2015b)
Percent by solar PV for pumping (%)	0.00	-	N	-
Percent by electricity for water treatment (%)	1	All treatment energy by electricity	N	City of Penticton (2015b)
Percent by solar PV for water treatment (%)	0.00	-	N	-
Percent by electricity for ww transport (%)	1	All energy by electricity	N	City of Penticton (2015c)
Percent by solar PV for ww transport (%)	0.00	-	N	-
Percent by electricity for ww treatment (%)	1	All treatment energy by electricity	N	City of Penticton (2015b)
Percent by solar PV for ww treatment (%)	0.00	-	N	-
Percent by electricity for GW recycling (%)	1	-	-	-
Percent by solar PV for GW recycling (%)	0.00	-	-	-
Percent by electricity for resi hot water (%)	0.269	National average for 2005 to 2011	N	Natural Resources Canada (2014)
Percent by solar thermal for resi hot water (%)	0	”	N	”
Percent by natural gas for resi hot water (%)	0.668	”	N	”
Percent by oil for resi hot water (%)	0.058	”	N	”
Percent by propane for resi hot water (%)	0.005	”	N	”
Percent by electricity for nonresi hot water (%)	0.269	Considered same as for residential indoor hot water	N	”
Percent by solar thermal for nonresi hot water (%)	0	”	N	”
Percent by natural gas for nonresi hot water (%)	0.668	”	N	”
Percent by oil for nonresi hot water (%)	0.058	”	N	”
Percent by propane for nonresi hot water (%)	0.005	”	N	”
WF electricity	19.7 L/kWh	Consumptive water use for FortisBC electricity	R	Okadera et al. (2014)
WF solar thermal	3.975 L/kWh	Consumptive water use (blue water footprint) for solar thermal energy	N,G	Fulton and Cooley (2015)
WF solar PV	0.04 L/kWh	Consumptive water use	N,G	Okadera et al. (2014)
WF natural gas	0.4 L/kWh	Consumptive water use	N,G	”
WF oil	1.22 L/kWh	Consumptive water use	N,G	”
WF propane	1.3 L/kWh	Petroleum	N,G	”

Chlorine dosage in WT	Monthly average dosage in mg/L	Average dosage for 2005 to 2009	R	Annual reports of Penticton water treatment plants from 2005 to 2014
PAC dosage in WT	16.98 mg/L	Average dosage for 2008 to 2011	R	”
Polymer dosage in WT	2.11 mg/L	Average dosage for 2008 to 2009	R	”
Chlorine dosage in WWT	0 mg/L	-	R	-
PAC dosage in WWT	19.9 mg/L	Average dosage for 2014	R	”
Polymer dosage in WWT	0 mg/L	-	R	”
WF chlorine	1.03E-05 L/mg	Ecoinvent 3 (Canada) database	N	LCA by SimaPro software
WF PAC	2.71E-06 L/mg	Ecoinvent 3 (Global) database	G	”
WF polymer	1.28 E-04 L/mg	Ecoinvent 3 (Global) database	G	”
CF chlorine	1.39E-06 kgCO_2e/mg	Ecoinvent 3 (Canada) database	G	”
CF PAC	4.10E-07 kgCO2e/mg	Ecoinvent 3 (Global) database	G	”
CF polymer	2.83E-06 kgCO_2e/mg	Ecoinvent 3 (Global) database	G	”
EE chlorine	7.35E-06 kWh/mg	Ecoinvent 3 (Canada) database	G	”
EE PAC	5.421E-07 kgCO_2e/mg	Ecoinvent 3 (Global) database	G	”
EE polymer	3.204E-07 kgCO_2e/mg	Ecoinvent 3 (Global) database	G	”
Biosolids generation rate	2.02E-03 kg/L	Average value from 2013 -2014	R	Annual reports of Penticton wastewater treatment plants from 2005 to 2014
Transportation distance	8 km	Estimation	R	GIS database
Fuel usage	4.28E-05 L/kg-km	Average	N	US DOE and Oak Ridge National Lab (2015)
Load factor (%)	0.5	Vehicles will travel empty during a return trip; used in LCA studies	N,G	Pre Consultants (2014)
Energy efficiency of fuel	10.64 kWh/L		N	Ministry of Environment (2013)
Carbon footprint (CF) transportation fuel	0.2522 KgCO_2e/kWh	CF of diesel	N	Ministry of Environment (2013)
WF transportation fuel	1.22 L/kWh	-	N,G	Okadera et al. (2014)
Emission factor (EF) CH4	0.03 kg CH_4/kg of BOD	Global average	N	IPCC (2006)
BOD generation rate	1.824 kg/cap/mo	Global average	R	IPCC (2006)
Emission factor (EF) N2O	3.17E-04 kg/cap/mo	-	G	IPCC (2006)
Initial infilt inflow rate	124 L/cap/day	Based on 2005 to 2007 data which is comparable to 110 L/cap/day given by (Briere 2010)	R	Annual reports of Penticton wastewater treatment plants2005 - 2007
Initial infilt inflow reduced rate	1.00	Initial factor	R	-
Infilt inflow change rate	0.00590/mo	Value assumed 2.5 times the rate of 2006 to 2007 data to incorporate intensive maintenance of cross connection control	R	Annual reports of Penticton wastewater treatment plants 2006 -2007
Initial water treat energy change rate	1	Initial factor	R	Annual reports of Penticton water treatment plants

Unit treat energy change rate	0.00028/month	-	R	Penticton WTP repots 2007-2012
Initial ww treat energy change rate	1.07	Assumed 7 % higher rate to accommodate high energy demand in the upgraded plant	R	Annual reports of Penticton wastewater treatment plants
Unit ww treat energy change rate	0.01034/month	-	R	Penticton WWTP repots 2007-2012
Distribution loss (%)	0.05	Loss only in distribution lines (indoor loss considered in indoor demand calculation)	R	Assumed
Irrigated garden proportion (%)	0.75	Considered 25% lawns not irrigated as 35% lawns contained not irrigated and partially irrigated areas	R	OBWB (2010); Maurer (2010)
Irrigated park proportion (%)	0.8	Considered 5% area more irrigated than lawns	R	”
Irrigated golf course proportion (%)	0	Golf course irrigation by private pumps (underground water)	R	-
Irrigated agriculture proportion (%)	0	Municipal water not used for agricultural irrigation	R	-
RWH energy use rate	0.0056 kWh/L	-	G	Wang and Zimmerman (2015)
Initial indoor hot water energy conserved rate	1.000	Initial factor	N	-
Indoor hot water energy conservation change rate	0.001/mo	Rate of decrease of water heating energy during 1990 -2011	N	Natural Resources Canada (2014)
Industry processing per sqft	5.52 L/sqft/mo	Industrial water demand considered to be 1.25 times the average industrial wastewater generation rate; industrial processing demand is the water after deducting non-industrial processing water from industrial water	N	Brière (2010)
Hospital processing	Estimated based on the North American study	N,G	N,G	Dziegielewski et al. (2000); Gleick et al. (2003); US EPA (2009)

Note: *percent values (%) are expressed out of 1 instead of 100, mo: month, cap: capita, eff: efficiency, sqft: square foot, ww: wastewater, WT: Water treatment, WWT: Wastewater treatment, RWH: Rainwater harvesting, EE: Embodied energy, PV: Photo voltaic,.

B.3 Validation figure for water consumption

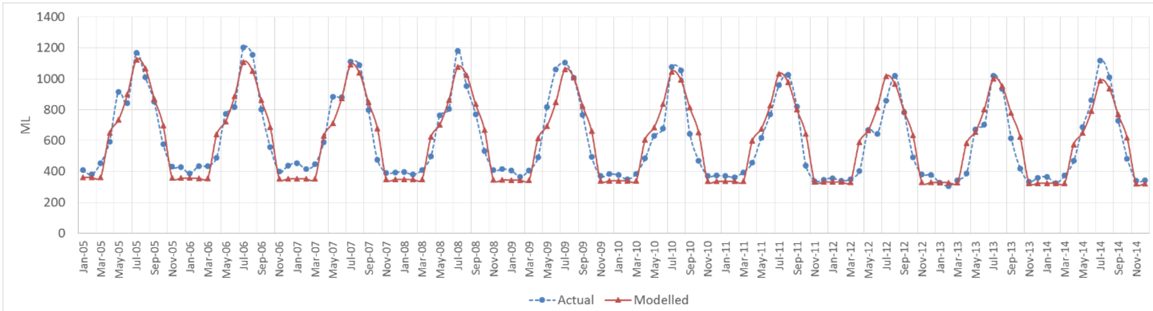

Figure B.7 Comparison of modelled and acutual water consumption in the City of Penticton

B.4 Sensitivity analysis framework and input data

The framework of sensitivity analysis show in Figure B.8. At first, the sensitivity of aggregated inputs to output parameters were analyzed. Later, the sensitivity of basic inputs to the aggregated inputs were analyzed for the important aggregated inputs.

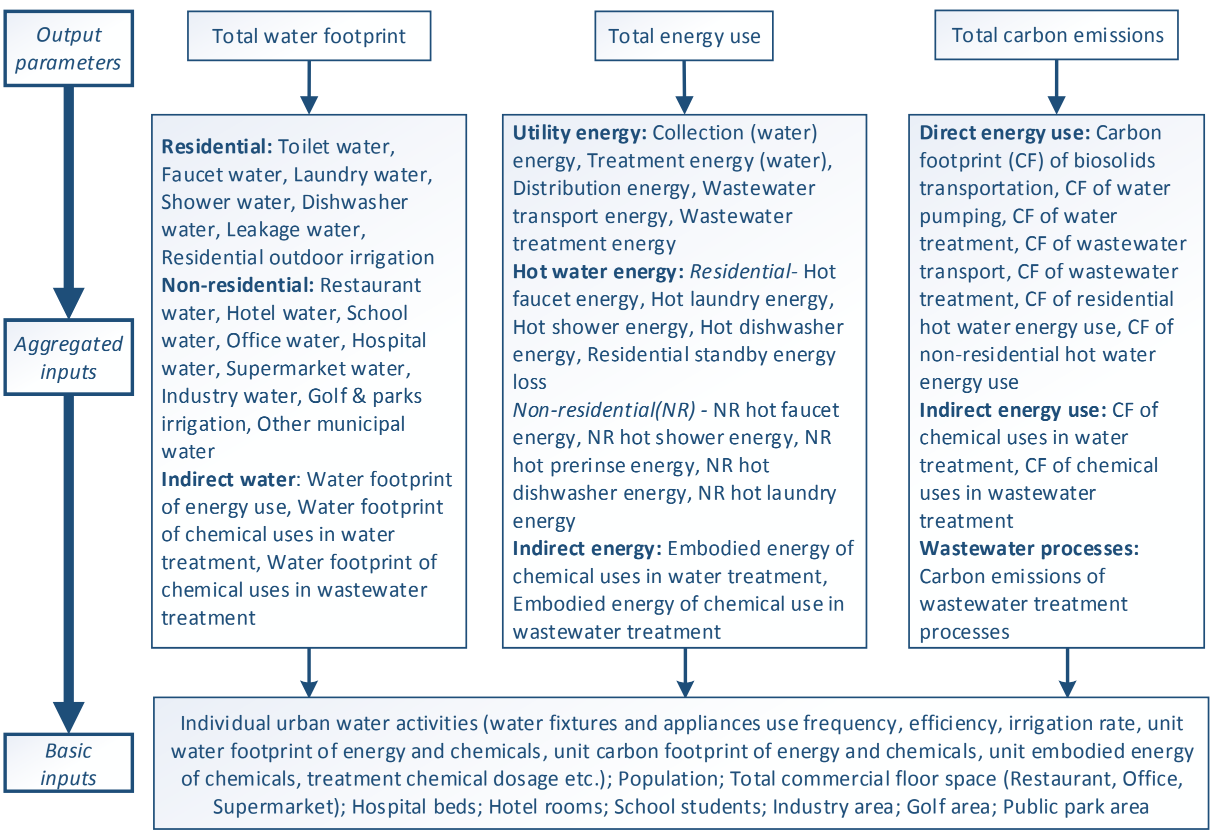

Figure B.8 Sensitivity analysis framework

Table B.2 Input paramters with distribution for sensivitity analysis

SN	Parameters	Distribution*	Reference
1	Population growth rate (%/yr)	U ~ (0.54, 0.66)	Statistics Canada (2015a)
2	Base population	U ~ (28461.6, 34786.4)	”
3	Dwelling occupancy (no./unit)	U ~ (2, 2.5)	”
4	Percent single detached (%)	U ~ (40, 60)	”
5	Percent small apartment (%)	U ~ (25, 33	”
6	AVG lot size single family (ha)	U ~ (0.07, 0.085)	GIS database
7	Avg lot coverage single family (ha)	U ~ (0.4, 0.5)	City of Penticton (2015a); Google Earth
8	Residential unit size (sqft)	U ~ (1800, 2000)	CHBA (2012)
9	AVG lot size small apartment (ha)	U ~ (0.16, 0.2)	Municipal GIS database
10	Avg lot coverage small apartment (%)	U ~ (60, 80)	City of Penticton (2015a); Google Earth
11	Irrigation reduction factor	U ~ (0.1, -0.1)	10% variation assumed
12	Laundry frequency (no./mo)	U ~ (10, 12.5)	Mayer et al. (1999)
13	Laundry water eff (L/load)	U ~ (95, 115)	Mayer et al. (1999)
14	Hot laundry ratio	U ~ (0.24, 0.32)	REUM model -Aguilar et al. (2005)
15	Laundry energy eff (kWh/load)	U ~ (1.35, 1.65)	REUM model -Aguilar et al. (2005)
16	Laundry water heating energy ratio	U ~ (0.8, 1)	REUM model -Aguilar et al. (2005)
17	Standby energy loss rate (Kwh/L hot water)	U ~ (0.0065, 0.0085)	REUM model -Aguilar et al. (2005)
18	Faucet frequency (min/mo/cap)	U ~ (220, 270)	Mayer et al. (1999)
19	Faucet water eff (L/min)	U ~ (7, 8)	Mayer et al. (1999)
20	Faucet energy eff (Kwh/L)	U ~ (0.04, 0.05)	Mayer et al. (1999)
21	Hot faucet ratio	U ~ (0.65, 0.8)	REUM model -Aguilar et al. (2005)
22	Toilet freq (Flush/mo/cap)	U ~ (145, 165)	Mayer et al. (1999)
23	Toilet water eff (L/flush/mo)	U ~ (9.5, 11.8)	Mayer et al. (1999)
24	Dishwasher freq (cycle/mo/cap)	U ~ (2.5, 3.5)	Mayer et al. (1999)
25	DW water eff (L/cycle)	U ~ (22, 30)	Mayer et al. (1999)
26	Hot DW ratio	U ~ (0.9, 1)	REUM model- Aguilar et al. (2005)
27	DW energy eff (Kwh/hot cycle)	U ~ (2.5, 3.5)	REUM model- Aguilar et al. (2005)
28	DW HEATING ENERGY RATIO	U ~ (0.8, 0.95)	Aguilar et al. (2005)
29	Shower freq (mo/cap)	U ~ (20, 25)	Mayer et al. (1999)
30	Shower water eff (L/min)	U ~ (9.5, 12)	Mayer et al. (1999)
31	Shower duration (min/shower)	U ~(7.5, 9)	Mayer et al. (1999)
32	Shower energy eff (kWh/L)	U ~ (0.04, 0.05)	REUM model- Aguilar et al. (2005)
33	Hot shower ratio	U ~ (0.7, 0.85)	REUM model- Aguilar et al. (2005)

34	Leakage and others rate	U ~ (0.04, 0.06)	Mayer et al. (1999)
35	Unit collection energy (variation factor)	U ~ (0.9, 1.1)	10% variation assumed
36	Unit treatment energy (variation factor)	U ~ (0.9, 1.1)	10% variation assumed
37	Unit distribution energy (variation (factor)	U ~ (0.9, 1.1)	10% variation assumed
38	Unit WW transport energy (variation factor)	U ~ (0.9, 1.1)	10% variation assumed
39	Unit WW treat energy (variation factor)	U ~(0.9, 1.1)	10% variation assumed
40	Golf and parks irrigation reduction factor	U ~ (0.1, -0.1)	10% variation assumed
41	AVG lot size commercial (ha)	U ~ (0.27, 0.35)	GIS database
42	Avg lot coverage commercial (%)	U ~ (55, 65)	City of Penticton (2015a); Google Earth
43	Total commercial floor space (sqft)	U ~ (4,421,647, 5,404,235)	City of Penticton (2015d)
44	Office area %	U ~ (34, 42)	GIS database
45	Supermarket area %	U ~ (50, 60)	GIS database
46	Initial hotel rooms	U ~ (1605, 1961)	City of Penticton (2015e)
47	Initial hospital beds	U ~ (116, 142)	Interior Health Authority (2015)
48	Hotel laundry freq (no./room/mo)	U ~ (31, 38)	Dziegielewski et al. (2000); Gleick et al. (2003); US EPA (2009)
49	NonResi Laundry water eff (L/load)	U ~ (95, 115)	”
50	Restau faucet duration (min/sqft/mo)	U ~ (0.21, 0.29)	”
51	Hotel faucet duration (min/room/mo)	U ~ (47, 57)	”
52	Hospital faucet duration (min/bed/mo)	U ~ (103, 126)	”
53	NonResi Faucet water eff (L/min)	U ~ (7, 8)	”
54	NonResi Faucet use eff	U ~ (0.85, 0.95)	”
55	Restau toilet freq (no./sqft/mo)	U ~ (1.8, 2.2)	”
56	Hotel toilet freq (no./room/mo)	U ~ (155, 185)	”
57	Office toilet freq (no./room/mo)	U ~ (0.13, 0.16)	”
58	NonResi Toilet water eff (L/flush)	U ~ (10.2, 12.6)	”
59	NonResi Urinal water eff (L/flush)	U ~ (5.5, 6.7)	”
60	NonResi DW water eff (L/load)	U ~ (22, 30)	”
61	NonResi Shower water eff (L/min)	U ~ (9.5, 12)	”
62	NonResi Shower use eff	U ~ (0.9, 0.98)	”
63	Hotel shower duration (min/room/mo)	U ~ (440, 530)	”
64	Prerinse faucet eff (L/min)	U ~ (11, 13)	”
65	Restau prerinse duration (min/sqft/mo)	U ~ (1.36, 1.67)	”
66	Restaurant food preparation and icemakers (L/sqft/mo)	U ~ (16, 20)	,,
67	Hotel food preparation and icemakers (L/room/mo)	U ~ (1035, 1265)	,,
68	NonResi laundry energy eff (kWh/hot cycle)	U ~ (1.3, 1.55)	Aguilar et al. (2005)
69	NonResi hot laundry ratio	U ~ (0.22, 0.35)	”
70	NonResi faucet energy eff (kWh/L)	U ~ (0.04, 0.05)	”
71	NonResi hot faucet ratio	U ~ (0.65, 0.8)	”

72	NonResi DW energy eff (kWh/hot wash)	U ~ (2, 3)	”
73	NonResi Hot DW ratio	U ~ (0.9, 1)	”
74	NonResi shower energy eff (kWh/L)	U ~ (0.04, 0.05)	”
75	NonResi hot shower ratio	U ~ (0.7, 0.85)	”
76	Hot prerinse ratio	U ~ (0.65, 0.8)	”
77	Indoor conservation change rate	U ~ (0.0018, 0.0022)	10% variation based on City of Penticton (2006)
78	Irrigation Conservation change rate	U ~ (0.0018, 0.0022)	”
79	Other municipal water rate (L/cap/mo)	U ~ (380, 465)	District of Peachland (2014b)
80	Carbon footprint (CF) electricity (kgCO2e/kWh)	U ~ (0.0027, 0.0033)	10% variation on Ministry of Environment (2013)
81	CF natural gas (kg CO_2e/kWh)	U ~ (0.162, 0.198)	”
82	Percent by electricity for resi hot water	U ~ (23, 30)	Natural Resources Canada (2014)
83	Percent by natural gas for resi hot water	U ~ (60, 75)	Natural Resources Canada (2014)
84	Percent by electricity for nonresi hot water	U ~ (23, 30)	Natural Resources Canada (2014)
85	Percent by natural gas for nonresi hot water	U ~ (60, 75)	Natural Resources Canada (2014)
86	WF electricity (L/kWh)	U ~ (17, 23)	Okadera et al. (2014)
87	WF natural gas (L/kWh)	U ~ (0.35, 0.45)	Okadera et al. (2014)
88	Chlorine dosage in WT	U ~ (0.9, 1.1)	Annual reports of Penticton water treatment plants from 2005 to 2014
89	PAC dosage in WWT (mg/L)	U ~ (0.9, 1.1)	”
90	Biosolids generation rate (kg/L)	U ~ (1.82E-03, 2.22E-03)	”
91	Transportation distance (km)	U ~ (6, 10)	GIS database
92	CF transportation fuel (kg CO2e/L)	U ~(0.22, 0.28)	Ministry of Environment (2013)
93	BOD generation rate (kg/cap/mo)	U ~ (1.6, 2)	IPCC (2006)
94	Initial infilt inflow rate variation factor	U ~ (0.9, 1.1)	10% variation assumed
95	INFILT INFLOW CHANGE RATE	U ~ (-0.00531, -0.00649)	Annual reports of Penticton wastewater treatment plants 2006 -2007
96	Unit treat energy change rate variation factor	U ~ (0.9, 1.1)	10% variation assumed
97	Unit ww treat energy change rate variation factor	U ~ (0.9, 1.1)	”
98	Distribution loss	U ~ (0.045, 0.055)	Assumed
99	Irrigated garden proportion	U ~ (0.7, 0.8)	OBWB (2010); Maurer (2010)
100	Irrigated park proportion	U ~ (0.75, 0.85)	OBWB (2010); Maurer (2010)
101	Indoor hot water energy conservation change rate	U ~(0.0009, 0.0011)	Natural Resources Canada (2014)
102	Hospital processing (L/bed/mo)	U ~(5262, 6431)	Dziegielewski et al. (2000); Gleick et al. (2003); US EPA (2009)

*U is to a uniform distribution with (low, high), T is triangular distribution with (lower most, most probable, upper most), LN is lognormal distribution with (mean, standard deviation)

Appendix C Additional Information on Water Distribution and Residential Landscaping System

C.1 Alternative Designs and their Characteristics

Table C.1 Alternative neighbourhood designs and net residential density

Alternative designs	Zone A		Zone B		Zone C		Population	Net residential Density (persons/ha)
	SF	MF	SF	MF	SF	MF		
D1	40	0	61	0	12	0	283	8.7
D2	40	0	61	0	0	439	1,350	41.5
D3	40	0	56	471	0	439	2,516	77.3
D4	40	0	51	943	0	439	3,682	113.2
D5*	40	0	46	1,414	0	439	4,848	149.0
D6	40	0	26	2,927	0	439	8,580	263.8
D7	40	0	0	4,200	0	439	11,697	359.6
D8	30	1,904	0	4,200	0	439	16,430	505.1
D9	20	3,807	0	4,200	0	439	21,164	650.6
D10	10	5,711	0	4,200	0	439	25,898	796.1
D11	0	7,614	0	4,200	0	439	30,632	941.6

* Base design (developer's plan); SF: Single-family building; MF: Multi-family building

Table C.2 Estimated WEC nexus and life cycle cost for alternative designs

Design	Water use (L/p/d)	Energy use (kWh/p/d)	Carbon ($gCO_2e/p/d$)			LCC ($/p/yr)	Ecological footprint (gha/p/yr)			
			Emission	Sequestration	Net -ve emission		Water	Energy	Net carbon	Total
D1	2,373.4	2.82	8.46	269.33	260.87	96.20	9.4E-02	4.0E-03	-2.1E-02	7.7E-02
D2	571.4	0.66	1.97	111.52	109.55	21.99	2.3E-02	9.4E-04	-9.0E-03	1.5E-02
D3	363.2	0.39	1.18	62.73	61.56	13.01	1.4E-02	5.6E-04	-5.0E-03	9.9E-03
D4	284.3	0.30	0.89	41.83	40.94	9.71	1.1E-02	4.2E-04	-3.4E-03	8.3E-03
D5	243.0	0.25	0.74	30.53	29.79	7.90	9.6E-03	3.5E-04	-2.4E-03	7.5E-03
D6	188.4	0.19	0.56	17.21	16.65	5.65	7.5E-03	2.6E-04	-1.4E-03	6.4E-03
D7	171.1	0.15	0.46	11.86	11.40	4.59	6.8E-03	2.2E-04	-9.3E-04	6.1E-03
D8	155.0	0.13	0.38	6.42	6.05	3.65	6.1E-03	1.8E-04	-5.0E-04	5.8E-03
D9	146.2	0.11	0.33	3.42	3.09	3.14	5.8E-03	1.6E-04	-2.5E-04	5.7E-03
D10	140.4	0.10	0.30	1.39	1.10	2.78	5.6E-03	1.4E-04	-9.0E-05	5.6E-03
D11	136.5	0.09	0.28	0.00	-0.28	2.59	5.4E-03	1.3E-04	2.3E-05	5.6E-03

C.2 Water Demand, Energy Use, Carbon Emissions, and Cost Estimation

The efficiency values of the water fixtures to be used in the buildings were obtained from the developer's plan, ENERGY STAR (2014a), and ENERGY STAR (2014b). The use frequency of various indoor water fixtures and appliances of the residential sector were obtained from the extensive study conducted by Mayer et al. (1999), which includes two Canadian cities. Based on the efficiency and use frequency of the fixtures, the residential indoor water demand was estimated (Table C.3). The residential outdoor irrigation demand was estimated by using the irrigation rate of residential landscaping of 991 L/m^2/yr of the Okanagan Valley obtained from Okanagan Basin Water Board (OBWB 2010). OBWB developed Agricultural Water Demand Model to estimate the irrigation rate based on crop types, irrigation systems, soil type, and climate data.

Table C.3 Indoor water demand rate for residential sector

Indoor use	Fixture use rate	Fixture efficiency	Demand (L/p/d)
Shower & bath	0.71 shower/day; 7.77 min/shower	7.6 L/min	42.02
Toilet	4.71 flush/day	4.8 L/flush	22.61
Lavatory faucets	3.65 min/cap/day	5.7 L/min	20.81
Kitchen faucets	3.65 min/cap/day	5.7 L/min	20.81
Laundry	0.345load/cap/day	54.3 L/cycle	18.73
Dishwashers	0.1 load/cap/day	13.4 L/load	1.21
Leakage	5%		6.31
Total			**132.5**

Source: Adapted from Mayer et al. (1999), ENERGY STAR (2014a), ENERGY STAR (2014b)

For the commercial and institutional (CI) sector, the use frequency of water fixtures was obtained from the CI water demand studies by the US EPA (2009b) and Dziegielewski et al. (2000). The efficiency of water fixtures and appliances in the CI buildings were considered the same as in the residential buildings. Based on the obtained efficiency and use frequency of water fixtures and appliances, the indoor water demand in the CI sector was estimated and given in Table C.4. The public park irrigation was estimated by using the park irrigation rate of 977 L/m^2/yr for the Okanagan Valley (OBWB 2010).

Table C.4 Indoor water demand rate for commercial and institutional sectors

Commercial and institutional sectors	Indoor water demand
Office	1.59 L/m^2/d
School	1.73 L/m^2/d
Retail	8.68 L/m^2/d
Hotel	620.74 L/room/day
Office and retail	5.13 L/m^2/d

Source: Adapted from US EPA (2009b) and Dziegielewski et al. (2000)

The energy related carbon emissions were estimated by using the emission factor of 0.003 kg CO_2e/kWh for the grid electricity in the region (Ministry of Environment 2013). Furthermore, the LCC was estimated by using Equation 6.13 (Chapter 6). The costs of water mains installment and repair/replacement was estimated (Tables C.5 and C.6). The price of electricity of 9.921 cents/kWh was used (Fortis BC 2016). The cost of water pumps was obtained from a pump company (Franklin Electric 2012) and that of valves were obtained from a local consulting firm (Focus Engineering 2014).

Table C.5 Water mains installation cost

Diameter (mm)	GEID[1]	RDNO[1]	Native backfill[2]	Imported backfill[2]	Average
50	65	-	-	-	65.0
100	70	230	94	131	131.3
150	80	250	105	140	143.8
200	110	300	118	162	172.5
250	135	350	164	200	212.3
300	150	400	198	236	246.0
350	200	465	233	316	303.5
400	250	515	274	366	351.3
450	300	565	317	417	399.8
500	350	615	-	-	482.5
Average					**250.8**

Source: 1. (Kabir 2016), 2.(Focus Engineering 2014)
Note: GEID: Glenmore-Ellison Improvement District; RDNO: Regional District of North Okanagan

Table C.6 Water mains repair and replacement cost

Repair (site condition)[1]	Unit cost ($/breakage)	Breakage rate (no./km/yr)	Total cost ($/km/yr)
Gravel	2500	0.4	1000
Concrete pavement	5500	0.4	2200
Agricultural/no pavement/native soil	2000	0.4	800
Unit cost in network			1333.3
Replacement		**Quantity**	**Unit**
Replacement breakage		0.08*	no./km/yr
Average pipe length/breakage		9**	m/breakage
Avg. replacement length		0.72	m/km/yr
Unit pipe replacement cost#		250.8	$/m
Unit cost in network		180.6	$/km/yr
Unit repair & replacement cost		1513.9	$/km/yr

Note: 1 (Kabir 2016) * Repair: replacement = 5: 1 assumed as per municipal engineers # From Table C.5
** Average of 6 m-long two pipes; breakage at joint affects 2 pipes but at pipe-middle affects 1 pipe

C.3 Carbon Sequestration of Residential Landscaping

The mean net soil organic carbon (SOC) sequestration rate of a residential lawn in US is 129.9 g/m^2/yr as an average of two management practices: "Do-It-Yourself (DIY)" and Best Management Practices (BMP) (Zirkle et al. 2011). Similarly, the average carbon sequestration rate of trees is 4.65 kg C/tree/yr and that of shrubs is 0.15 kg C/shrub/year in residential landscaping (Zirkle et al., 2012). The average number of trees is 4 trees/yard with a maximum of 6 trees and that of shrub is 16 shrubs/yard with a maximum of 35 shrubs for a typical yard of 1541 m^2 in a SF house in the US (Zirkle et al., 2012).

C.4 Xeriscaping Design

A xeriscape was designed, which was comprised of 15% turf grass as kids' play zone and the remaining ground covered by water conserving species. The estimated irrigation demand of the xeriscape is approximately 489 L/m^2/yr considering the irrigation rate of 991 L/m^2/yr for turf (OBWB 2010) and 400 L/m^2/yr for water conserving species (e.g., blue oat grass, feather grass, aster, and globe thistle) (City of Kamloops 2016). The estimated irrigation demand of the xeriscape is 51% reduced than that of a typical landscaping. In addition, xeriscaping reduces fertilizer, fuel, herbicides, and labour (Gleick et al. 2003). The number of trees and shrubs were considered the same as of the typical landscaping design of a single-family (SF) building. Decreased irrigation in xeriscaping reduces the carbon sequestration of soil organic carbon but for trees and shrubs, carbon sequestration was assumed to be same as of typical landscaping (Zirkle, Lal, Augustin, & Follett, 2012). The estimation of carbon sequestration of xeriscaping is shown in Table C.7.

Table C.7 Estimated soil organic carbon sequestration in xeriscaping

Carbon dynamics	DIY ($g/m^2/yr$)		BMP ($g/m^2/yr$)		Remarks
	Low	High	Low	High	
SOC	53.0	99.7	53.0	99.7	Excluding tropical grasslands in avg. SOC rate[a]
Fertilizer SOC*	30.4	30.4	30.4	38.2	39%[b] of avg. SOC rate[a]
Irrigation SOC**	0.2	0.7	2.5	4.9	49%[b] of avg. SOC rate[a]
Gross SOC	*83.7*	*130.9*	*85.9*	*142.9*	
Mowing CE	7.2	11.5	7.2	11.5	56%[b] of avg. SOC rate[a]
Irrigation CE	0.0	0.1	0.8	0.8	49%[b] of avg. SOC rate[a]
Fertilizer CE	3.9	8.0	6.0	19.3	39%[b] of avg. SOC rate[a]
Pesticide CE	0.3	2.0	0.6	4.4	78%[b] of avg. SOC rate[a]
Gross CE	*11.5*	*21.7*	*14.7*	*36.0*	
Net sequestration	**62.0**	**119.4**	**49.9**	**128.2**	**Net avg.: 89.9 $g/m^2/yr$**

Source: a. (Gleick et al. 2003) b. (Zirkle et al. 2011)
DIY: Do-it-yourself lawn BMP: Best Management Practice lawn
*SOC increment due to the use of fertilizer; **SOC increment due to irrigation
CE: Carbon emissions by an activity (mowing, irrigation) and production and use of a product (fertilizer and pesticide)

Appendix D Additional Information on Microbial Water Quality Guidelines Development

D.1 Sensitivity analysis results

Table D.1 Effects of input on mean output with 5% or more effects (in %)

Factors	Toilet_urinal flushing (%)	Park irrigation (%)	Golf course irrigation (%)	Agri irrigation (%)	Park to lawn to & golf irrigation (%)	Park to lawn to golf to & agri irrigation (%)	Vehicle washing (%)	Laundry machine (%)	Fire fighting (%)	Toilet_urinal flushing & laundry (%)	Non-potable urban uses (%)
Pathogenic *E. coli* ratio	-43 to 90	-44 to 89	-43 to 90	-44 to 89	-43 to 90	-44 to 88	-43 to 91	-44 to 88	-44 to 90	-44 to 91	-44 to 89
Morbidity (Pill|inf)	-37 to 66	-38 to 67	-37 to 65	-37 to 7	-37 to 66	-38 to 65	-37 to 66	-37 to 65	-37 to 67	-37 to 66	-38 to 68
Sus. fraction	-9 to 13	-11 to 10	-9 to 1	-10 to 10	-9 to 10	-9 to 9	-10 to 11	-10 to 11	-9 to 10	-9 to 12	-11 to 11
Dis. Burden/case	-8 to 9	-8 to 10	-8 to 10	-9 to 11	-8 to 9	-9 to 9	-8 to 10	-8 to 8	-10 to 10	-9 to 8	-10 to 9
Cross connection/ person	-7 to 10	-5 to 8	-7 to 7	-6 to 7	-	-	-	-7 to 11	-	-8 to 8	-
Annual cross connection vol.	-9 to 9	-6 to 6	-5 to 6	-6 to 7	-	-	-	-9 to 9	-	-9 to 7	-
Golf irrigation freq	-	-	-4 to 6	-	-	-	-	-	-	-	-
Log reduction_ cleaning (D)	-	-	-	-7 to 5	-	-	-	-	-	-	-
Days_irrigation to consumption	-	-	-	-13 to 13	-	-5 to 5	-	-	-	-	-
Log reduction_ transport/storage (D)	-	-	-	-11 to 14	-	-	-	-	-	-	-
Car washing unit vol.	-	-	-	-	-	-	-9 to 9	-	-	-	-4 to 5
Car washing freq	-	-	-	-	-	-	-8 to 8	-	-	-	-4 to 5
Firefighting freq	-	-	-	-	-	-	-	-	-7 to 8	-	-4 to 5
Firefighting unit vol	-	-	-	-	-	-	-	-	-7 to 8	-	-

Pill|inf: Probability of ill given infection, Sus. Fraction: Susceptibility fraction, freq.: frequency, vol.: volume

D.2 Raw wastewater microbiology

Table D.2 Microbial concentration for raw wastewater

Microorganisms	Concentration (no./100 mL)	Reference
E coli	10^5 - 10^{10}	Health Canada (2010)
Campylobacter	10^2- 10^5	EPHC/NHMRC/NRMMC (2008)
Salmonella spp.	10^3 - 10^5	Health Canada 2010); EPHC/NHMRC/NRMMC (2008)
Adenovirus	10 - 10^4	Health Canada 2010); EPHC/NHMRC/NRMMC (2008)
Norovirus	10 - 10^4	Health Canada 2010); EPHC/NHMRC/NRMMC (2008)
Rotavirus	10^2 - 10^5	Health Canada (2010)
Cryptosporidium parvum	0 - 10^4	Health Canada 2010); EPHC/NHMRC/NRMMC (2008)
Giardia spp.	10^2- 10^5	Health Canada (2010)

D.3 Log removal of treatment processes

Table D.3 Microbial removal efficiency of different treatment processes (in terms of log removal)

Components	Bacteria	Virus	*Cryptosporidium parvum*	*Giardia* spp.
Primary sedimentation	Unif. (0, 1) (EPHC/NHMRC/NRMMC 2008), (Metcalf and Eddy 2003)	Unif. (0, 1) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2011)	Unif.(0.5, 1) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)	Unif. (0, 0.5) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)
Biological Nutrients Removal (BNR)	Unif. (1, 2) (EPHC/NHMRC/NRMMC 2008), (Metcalf and Eddy 2003)	Unif. (0, 2) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010) with lowest boundary	Unif. (0, 1) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010) with lowest boundary	Unif. (0, 1) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010) with lowest boundary
Microfiltration	Unif. (4, 6) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2013b)	Unif. (2.5, 6) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)	Unif. (6, 8) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2012)	Unif. (6, 8) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2012)
Chlorination	Unif. (2, 6) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)	Unif. (2, 4) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (US EPA 2006)	Unif. (0, 1.5) (Health Canada 2012), (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)	Unif. (0, 1.5) (Health Canada 2012), (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)
UV radiation*	Unif. (2, 4) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010)	Unif. (0.25, 4) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2011)	Unif. (3, 4) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2012)	Unif. (3, 4) (EPHC/NHMRC/NRMMC 2008), (Health Canada 2010), (Health Canada 2012)

Note: Unif. means a uniform distribution with the minimum and maximum values in parenthesis; * UV of 40 mJ/cm^2 is common

Appendix E Additioal Information on FitWater Development

E.1 Fuzzy data on microbial concentration of wastewater

Table E.1 Default microbial concentration in raw wastewater used in Fitwater

Microorganisms	Concentration in TFNs (no./100 mL)	Reference
E. coli	(177,828; 31,622,777; 5,623,413,252)	Health Canada (2010)
Campylobacter	(141; 3162; 70,795)	EPHC/NHMRC/NRMMC (2008)
Salmonella spp.	(1,259; 10,000; 79,433)	Health Canada (2010) EPHC/NHMRC/NRMMC (2008)
Adenoviruses	(14; 316; 7,079)	Health Canada (2010) EPHC/NHMRC/NRMMC (2008)
Noroviruses	(14; 316; 7079)	Health Canada (2010) EPHC/NHMRC/NRMMC (2008)
Rotavirus	(141; 3162; 70,795)	Health Canada (2010)
Cryptosporidium	(2; 100; 6310)	Health Canada (2010) EPHC/NHMRC/NRMMC (2008)
Giardia spp.	(141; 3162; 70,795)	Health Canada (2010)

E.2 Fuzzy data on microbial concentration of greywater

Table E.2 Default microbial concentration in raw greywater used in Fitwater

Microorganisms	Concentration in TFNs (no./100 mL)	Reference
E. coli	(16, 1585, 15,1356)	Health Canada (2010)
Salmonella spp.	(270, 2700, 5130)	Kim et al. (2009)
Giardia spp.	(0.055, 0.100, 0.145)	NASEM (2016)

E.3 FitWater Data Entry and Flow

FitWater consists of nine spreadsheets. For the application of FitWater, data are entered in Sheets 1 and 2 as given in Figure E.1. In Sheet 1, information on wastewater quantity, quality, energy source, water and wastewater charges, and weights for multi-criteria decision analysis (MCDA) are entered. In Sheet 2, wastewater treatment trains are prepared for each alternative. Also, the information on conveyance infrastructure of wastewater collection and reclaimed water distribution are entered. The necessary calculations are performed in Sheets 3 to 7 and results are displayed in Sheets 8 and 9.

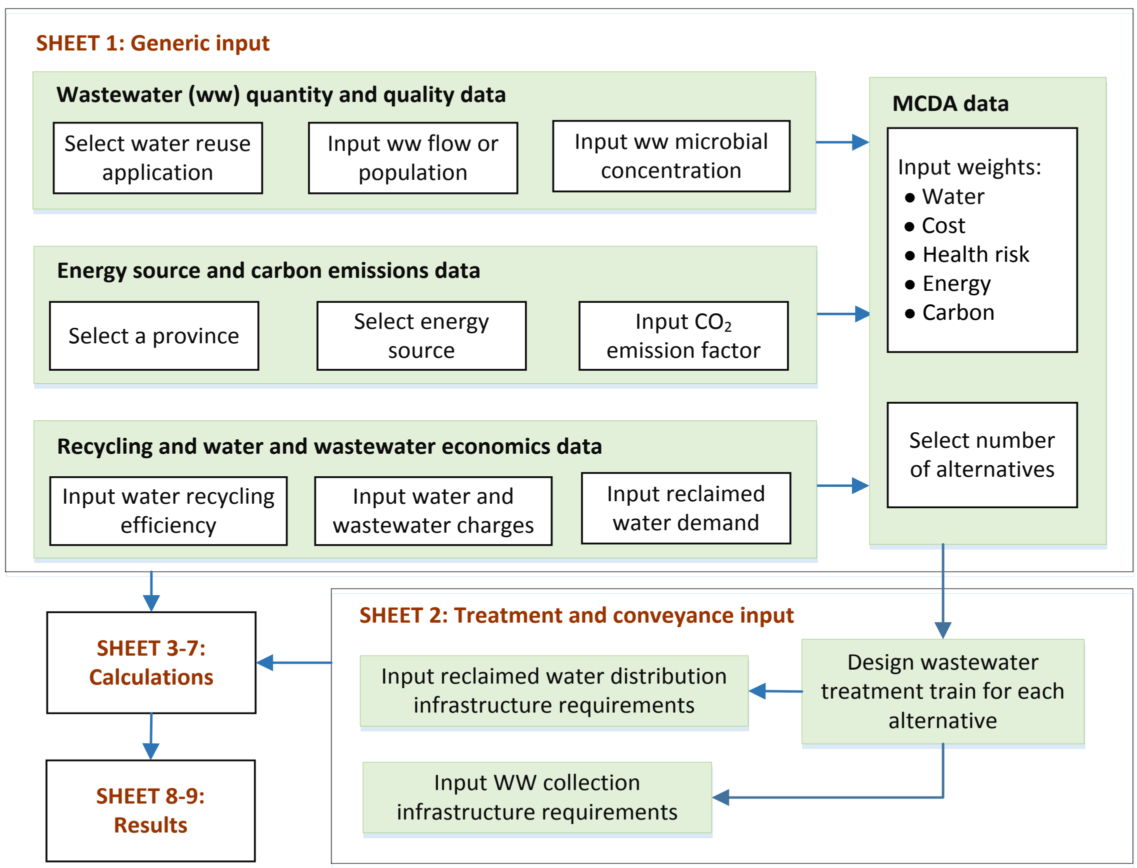

Figure E.1 Entry and flow of data in FitWater

Appendix F Additional Information on NZW Analysis

F.1 Stock and flow diagram of cost module

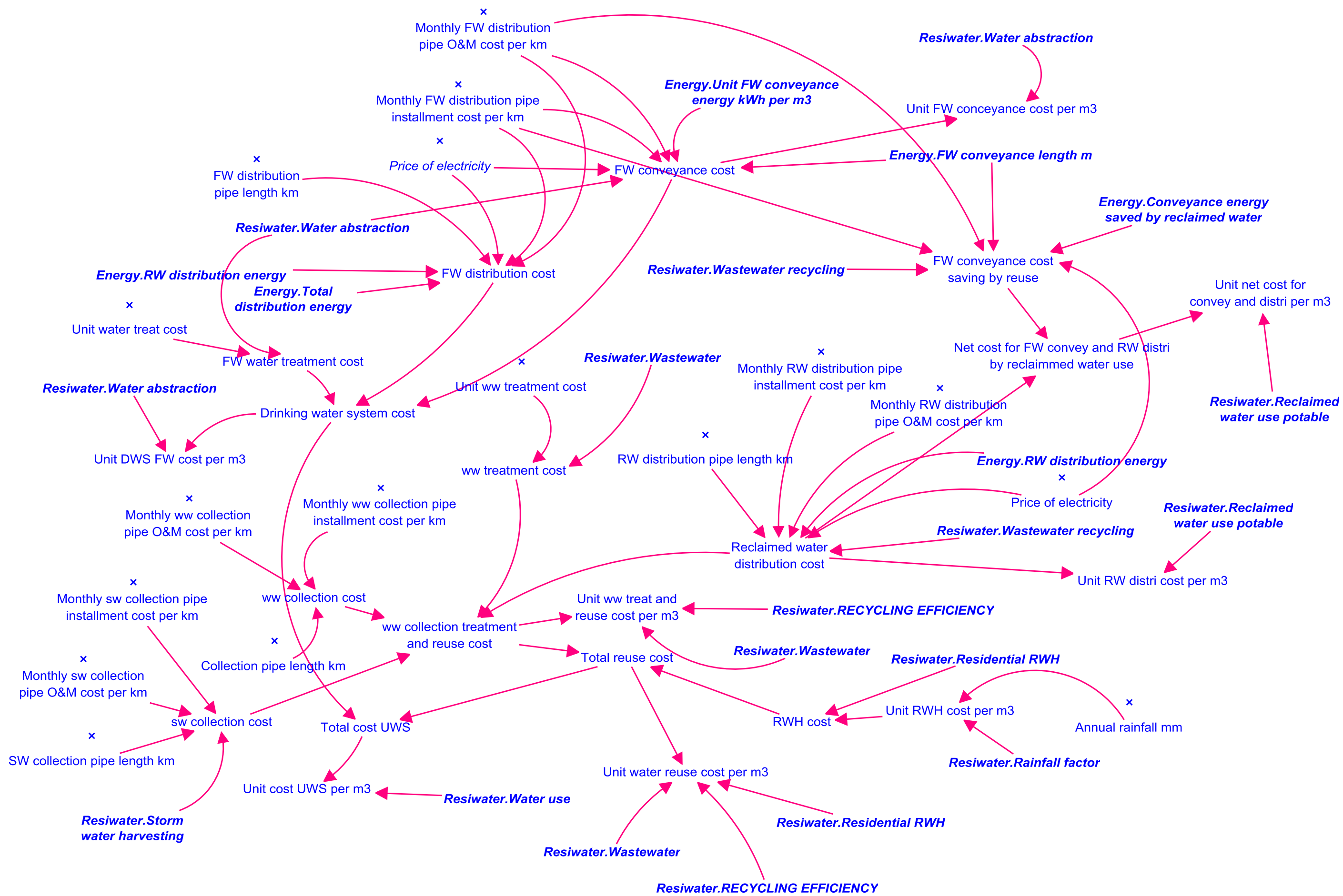

Figure F.1 Stock and flow diagram of cost module

F.2 Input data for Monte Carlo Simulations

Table F.1 Input paramters with distribution used in Monte Carlo Simulations

SN	*Parameters	Distribution**	Reference
103	Average annual rainfall (mm)	LN ~(302.3, 61.2)	Government of Canada (2016).
104	Avg monthly rainfall (mm)	LN ~(25.2, 10)	Government of Canada (2016).
105	RW harvestable roof proportion	T ~(0.75, 0.8, 0.85)	Assumed
106	Rainwater filtration efficiency	T ~(0.85, 0.9, 0.95)	Wang and Zimmerman (2015)
107	Runoff coefficient	T ~ (0.8, 0.85, 0.9)	Kim et al. (2016)
108	Urban runoff coefficient	T ~(0.65, 0.7, 0.75)	RiverLink (2014)
109	Friction factor	T ~(0.35, 0.4, 0.45)	Cheng (2002)
110	Pump safety factor	U ~(0.1, 0.15)	Cheng (2002)
111	Pump efficiency	U ~(0.7, 0.75)	Cheng (2002)
112	Velocity (m/s)	U ~(1, 1.5)	Briere (2010)
113	Pressure at consumer (m)	T ~(29.3, 32.5, 35.8)	Briere (2010)
114	Average slope degree (^{0})	U ~(0.9, 1.1)	Estimateh from GoogleEarth
115	Monthly FW distribution pipe O&M cost per km ($/km)	U ~(167.6, 204.8)	Kabir (2016); Focus Engineering (2014)
116	Monthly FW distribution pipe installment cost per km ($/km)	U ~ 284, 347)	Kabir (2016); Focus Engineering (2014)
117	Unit water treatment cost ($/L)	U ~(0.00045, 0.00055)	AECOM (2012)
118	Reclaimed water distribution pipe length (km)	U ~(147, 179)	10% variation on 163 km (City of Penticton 2014a)
119	Monthly RW distribution pipe O&M cost per km ($/km)	U ~(133.6, 138.8) For Scenario 4; U ~(136.3, 166.5); For Scenario5 (167.6, 204.8)	Kabir (2016); Focus Engineering (2014)
120	Monthly RW distribution pipe installment cost per km ($/km)	U ~(192.4, 235.1); Scenario 4; U ~(231, 282.2), For Scenario 5 U ~(284, 347)	Kabir (2016); Focus Engineering (2014)
121	Unit ww treatment cost ($/m^3)	U ~(0.00110, 0.00134); for Scenario 2 to 5 U ~(0.0021, 0.0026)	City of Penticton (2014a, 2015c); Guo et al. (2016)

* Parameters 1 to 102 are given in Appendix B.4 (Table B.2)
**U is to a uniform distribution with (low, high), T is triangular distribution with (lower most, most probable, upper most), LN is lognormal distribution with (mean, standard deviation)