WATEREUSE RESEARCH

Feasibility Study on Model Development to Estimate and Minimize Greenhouse Gas Concentrations and Carbon Footprint of Water Reuse and Desalination Facilities

WateReuse Research Foundation

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About the WateReuse Research Foundation

The mission of the WateReuse Research Foundation is to conduct and promote applied research on the reclamation, recycling, reuse, and desalination of water. The Foundation's research advances the science of water reuse and supports communities across the United States and abroad in their efforts to create new sources of high-quality water through reclamation, recycling, reuse, and desalination while protecting public health and the environment.

The Foundation sponsors research on all aspects of water reuse, including emerging chemical contaminants, microbiological agents, treatment technologies, salinity management and desalination, public perception and acceptance, economics, and marketing. The Foundation's research informs the public of the safety of reclaimed water and provides water professionals with the tools and knowledge to meet their commitment of increasing reliability and quality.

The Foundation's funding partners include the Bureau of Reclamation, the California State Water Resources Control Board, the California Energy Commission, and the California Department of Water Resources. Funding is also provided by the Foundation's subscribers, water and wastewater agencies, and other interested organizations.

Feasibility Study on Model Development to Estimate and Minimize Greenhouse Gas Concentrations and Carbon Footprint of Water Reuse and Desalination Facilities

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Abbreviations and Acronyms

AOP	advanced oxidation process
AWWARF	American Water Works Association Research Foundation
BOD	biochemical oxygen demand
CAM	Clean Air Markets
CAS	conventional activated sludge
CCAR	California Climate Action Registry
CHEApet	Carbon Heat Energy Analysis plant evaluation tool
CO_2eq	carbon dioxide equivalents
COD	chemical oxygen demand
CMF	continuous microfiltration
DAF	dissolved air flotation
DO	dissolved oxygen
eGRID	Emissions & Generation Resource Integrated Database
EIO-LCA	economic input–output life cycle assessment
ERD	energy recovery device
ERWT	Ebro River Water Transfer
GAC	granular activated carbon
GHG	greenhouse gas
GWP	global warming potential
IPCC	Intergovernmental Panel on Climate Change
LCA	life cycle assessment
LGOP	Local Government Operational Protocol
LTO	landing and takeoff
MBR	membrane biological reactor
MED	multi-effect desalination
MSF	multistage flash
NGERS	National Greenhouse and Energy Reporting System
O&M	operation and maintenance
PV	photovoltaic
RO	reverse osmosis
SPC	shadow price of carbon
TOC	total organic carbon
UAE	United Arab Emirates
UF	ultrafiltration
UV	ultraviolet
USEPA	United States Environmental Protection Agency
VOC	volatile organic compound
WEST	water energy sustainability tool
WESTWeb	water energy sustainability tool Web-based version
WSP	wastewater stabilization pond
WWEST	wastewater energy sustainability tool

Foreword

The WateReuse Research Foundation, a nonprofit corporation, sponsors research that advances the science of water reclamation, recycling, reuse, and desalination. The Foundation funds projects that meet the water reuse and desalination research needs of water and wastewater agencies and the public. The goal of the Foundation's research is to ensure that water reuse and desalination projects provide high quality water, protect public health, and improve the environment.

An Operating Plan guides the Foundation's research program. Under the plan, a research agenda of high priority topics is maintained. The agenda is developed in cooperation with water reuse and desalination communities, including water professionals, academia, and Foundation subscribers. The Foundation's research focuses on a broad range of water reuse research topics including:

- Definition of and addressing emerging contaminants
- Public perceptions of the benefits and risks of water reuse
- Management practices related to indirect potable reuse
- Groundwater recharge and aquifer storage and recovery
- Evaluation and methods for managing salinity and desalination
- Economics and marketing of water reuse

The Operating Plan outlines the role of the Foundation's Research Advisory Committee (RAC), Project Advisory Committees (PACs), and Foundation staff. The RAC sets priorities, recommends projects for funding, and provides advice and recommendations on the Foundation's research agenda and other related efforts. PACs are convened for each project and provide technical review and oversight. The Foundation's RAC and PACs consist of experts in their fields and provide the Foundation with an independent review, which ensures the credibility of the Foundation's research results. The Foundation's Project Managers facilitate the efforts of the RAC and PACs and provide overall management of projects.

This report reviews existing literature and models that estimate greenhouse gas (GHG) emissions for water reuse and desalination facilities. A thorough analysis of available information required to estimate the carbon footprint and a summary of previously documented GHG and carbon dioxide emissions is provided. In addition, various emission models are reviewed by assessing their applicability to water reuse and desalination. Knowledge gaps preventing a robust, accurate, and precise model are also analyzed. Finally, the availability and implementation of off-the-shelf models that allow utilities to assess the potential GHG emissions of current, planned, or potential desalination or recycled water facilities are evaluated.

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Executive Summary

This report critically interprets existing literature that will assist utilities employing water reuse and desalination in estimating greenhouse gas (GHG) emissions and carbon footprint while also recommending accessible models to provide estimations. GHG estimation models can aid water reuse and desalination utilities in estimating (1) direct emissions from unit processes, (2) indirect emissions associated with energy consumption, and (3) indirect emissions associated with material consumption (i.e., indirect energy consumption). To achieve this goal, this report is divided into six research tasks, organized into separate chapters, as summarized herein.

Summary of Chapter 2: Literature Review

The literature review focused on studies estimating GHG emissions of (1) desalination, (2) water reuse, and (3) both water reuse and desalination technologies. Within the literature, the majority of the studies used commercial life cycle assessment (LCA) software (e.g., SimaPro and Gabi) to quantify GHG emissions and the carbon footprint. LCA is a methodology used to assess the environmental impact of a product or process over its entire lifetime (cradle to grave). Other tools used to assess GHG emissions included hybrid LCA models, specific models, and other related models. For purposes of this study, a carbon footprint is defined as the total set of greenhouse gas emissions, in carbon equivalents, caused by an organization, event, product, or person. Utilities employing water reuse and desalination can mitigate their carbon footprint) by reducing their electricity usage through technology selection, energy recovery, and process modifications; however, it is equally important for energy providers to reduce the GHG emissions from electricity production and consider renewable energy sources. Because of variations in system boundaries, parameters considered, technologies evaluated, underlying assumptions, electricity mix, and GHG estimation methodologies, caution must be taken when comparing GHG emission findings from different literature sources. Throughout the literature, GHG emissions were reported in metric units, the approach adopted in this report.

Desalination

Studies indicate that the following factors impact the GHG emissions of desalination systems, contributing to the varied range in carbon footprint: treatment technology, energy source, electricity mix, energy consumption level, raw water source, and pretreatment process. Reverse osmosis (RO) technologies were reported to have lower carbon dioxide (CO_2) emissions than thermal technologies (e.g., multi-effect distillation [MED], multistage flash [MSF]) for an equivalent volume of water processed under similar electricity mixes. In terms of energy source, natural gas had a lower environmental impact than other fossil-based fuel types (e.g., oil and coal), and energy mixes with renewable energy sources were found to reduce CO_2 emissions at desalination facilities by 69 to 80%. The raw water source was another important factor, in which two studies found that seawater desalination had higher GHG emissions than the treatment of other brackish water sources (e.g., groundwater and surface water) because of higher energy consumption to reduce total dissolved solids (TDS). Seawater has a TDS of approximately 35,000 to 40,000 mg/L, and brackish groundwater may have TDS ranging from 1000 to 10,000 mg/L. It is important to note that there are now energy recovery devices (ERDs) gaining favor that are reducing the energy requirements of desalination technologies.

Water Reuse

Water reuse is a term for different applications including (1) direct potable reuse, (2) nonpotable reuse for irrigation, (3) indirect potable reuse via surface spreading, and (4) indirect potable reuse via groundwater injection. All these applications have different treatment requirements (and associated energy requirements and GHG emissions) ranging from tertiary filtration to advanced treatment such as RO plus advanced oxidation process (AOP). The most widely studied tertiary treatment methods evaluated within the water reuse literature were filtration- and membrane-based technologies.

Water reuse studies identified several impacting factors, including operational electricity usage, treatment processes, electricity mix, and the GHG abatement potential through water reuse or resource recovery (e.g., energy recovery). Electricity consumption in the operation stage was the dominant contributor to higher GHG emissions at water reuse facilities, accounting for 60 to 90% of the GHG emissions from a water reclamation facility.

Conflicting results emerged when comparing tertiary and secondary treatment. This is largely dependent on the type of treatment processes used in these treatment stages, highlighting the importance of considering both stages in GHG emission estimation. Similar to desalination facilities, the electricity mix of the energy provider was found to influence GHG emissions from water reuse facilities, whereas electricity mixes dominated by renewable sources contributed to lower emissions than mixes dominated by fossil fuels, as expected. Other studies focused on the GHG abatement potential through water reuse and other forms of resource recovery (e.g., on-site energy recovery). For example, one study found that combining fertilizer replacement and biogas recovery reduced GHG emissions by 55% for secondary treatment levels and 23% for tertiary treatment levels. These studies highlight how GHG emissions vary when the system boundary is expanded to include the credit of resource recovery alternatives.

Comparison of Water Reuse and Desalination

Comparison studies concluded that GHG emissions from desalination were higher than those for water reuse. In these studies, the estimated carbon footprint associated with seawater and brackish water RO desalination facilities ranged from 0.4 to 6.7 kg CO_2eq/m^3, whereas for water reuse facilities it ranged from 0.1 to 2.4 kg CO_2eq/m^3. This wide range arises not only from technology selection but also off-site considerations such as energy source and electricity mix, as discussed previously. Indirect emissions from chemical and material production were also found to be an important contributor. One study found that replacing materials such as RO membranes and cartridge filters contributed to 30 to 44% of the carbon footprint. In assessing water supply stages, approximately 85% of the energy consumption for desalination was found to be associated with treatment, whereas 61 to 74% of the energy used by water reuse facilities was consumed during distribution. This highlights the importance of considering all water supply phases (e.g., collection, treatment, and distribution) in GHG emission studies.

Summary of Chapter 3: Parameters Used in Previous Studies and Models

Chapter 3 discusses life cycle stages and types of GHG emissions considered in previously published literature and parameters used in these studies and available emission models.

Water reuse and desalination usually consist of two life cycle stages: construction and operation and maintenance (O&M). A review of existing literature found that all studies contain an O&M stage, but fewer than half consider the construction stage. Within each life cycle stage, different parameters are considered. These parameters are the specific activities responsible for or energy use that will lead to GHG emissions. Almost all studies included on-site energy use during the O&M stage. Most studies also included the energy use associated with the production and transport of materials used during the construction phase and chemicals during the O&M phase. In reviewed studies, GHGs commonly considered include CO_2, methane (CH_4), and nitrous oxide (N_2O). A comprehensive measurement commonly used is CO_2 equivalents (CO_2eq), which is an aggregate of the weighted amounts of CO_2, CH_4, and N_2O. Nearly all of the reviewed studies estimated the emissions either in CO_2 or CO_2eq.

Summary of Chapter 4: Previously Documented Emissions

Chapter 4 documents GHG and CO_2 emissions reported in reviewed studies. Reported GHG and CO_2 emissions from reviewed studies were categorized with respect to location (as geography can influence availability of technology and energy source), capacity, technology, and energy mix.

Of the four desalination technologies compared (MSF, MED, RO, and solar still), RO had the lowest intensity of electricity use and associated GHG emissions. The range of GHG emissions for RO (0.4–6.7 kg CO_2eq/m^3) also highlighted the importance of the type of water used, as the less saline brackish water yielded lower GHG emissions than seawater. Differences in energy mix also greatly affected CO_2 emissions, with higher emissions associated with a fossil fuel–powered desalination process than one powered by renewable energies. Capacity plays a small part, as desalination scenarios with plant capacities less than 5 MGD reported a higher GHG emissions range than those in the 5- to 10-MGD range; however, this trend does not continue with higher plant capacities (>10 MGD).

For water reuse, GHG emissions ranged from 0.1 to 2.4 kg CO_2eq/m^3 depending on capacity, energy mix, and technology. An analysis of the values from two California-based studies by the same authors, using RO as tertiary treatment, showed that a capacity increase was accompanied by an increase in GHG emissions. Water reuse scenarios based in countries with a higher dependence on renewable energy sources, nuclear energy, or both had lower CO_2 emissions than those based in countries that are more dependent (80%) on fossil fuels for energy. For water reuse, technological comparisons were hard to make, as studies were diverse in terms of processes included (i.e., primary and secondary treatment prior to tertiary treatment for reuse) and the inclusion of reclaimed water distribution in the overall analysis.

Summary of Chapter 5: Available Model Review

Chapter 5 provides a review of available emission models with varying levels of applicability to water reuse and desalination. Models identified may be classified as (1) LCA, (2) hybrid LCA, (3) specific, and (4) other. This chapter highlights the system boundaries, data sources, input parameters, calculation method, output parameters, limitations, and applicability of these model types.

Traditional LCA software (e.g., SimaPro and Gabi) is the most commonly used methodology to estimate GHG emissions despite the fact that it is not specifically designed for water reuse or desalination facilities. Hybrid LCA models reviewed include (1) water energy sustainability tool (WEST), (2) wastewater energy sustainability tool (WWEST) , and (3) WEST and WWEST web version (WESTWeb). Hybrid LCA models appear to be the most applicable to water reuse and desalination because they have previously been applied to these facilities and contain regionally transferable emission factors using Emission & Generation Resource Integrated Database (eGRID) data and the most comprehensive tool kit for estimating GHG emissions (e.g., inclusion of direct process emissions for water reuse).

Specific models reviewed, the Tampa Bay Water Model and the Johnston Model, differ from hybrid LCA models in that they focus solely on the O&M phase using input parameters specific to a utility. These models include GHG estimation tools with potentially beneficial attributes for a robust and accurate water reuse or desalination model. A brief description of several other related models, including Carbon Heat Energy Analysis plant evaluation tool (CHEApet), Environment Agency Model, Bridle/BSM2G Models, System Dynamics, GPS-X Model, Carbon Accounting Workbook, and mCO2, is provided in Chapter 5. These models were deemed to be less applicable than hybrid LCA or specific models because of certain constraints, including the applicability of the model to water reuse or desalination, regional transferability issues, and limited information available in the public domain.

Summary of Chapter 6: Information Gap Identification

Chapter 6 identifies current research gaps and elements beneficial to robust modeling by comparing the commonalities and differences in system boundaries, emissions considered, limitations, and applicability of both hybrid LCA and specific models.

Major knowledge gaps include the following:

1. A model with limited data inputs consistent with the functionality of a water reuse or desalination facility
2. Separation of direct emissions associated with processes (Scope 1), indirect emissions associated with electricity use (Scope 2), and indirect emissions associated with material consumption and other related activities (Scope 3)
3. Consistent framework for water reuse and desalination system boundary selection
4. Energy and direct emission estimation equations for unit processes specific to water reuse and desalination facilities
5. Indirect emissions associated with membrane production and disposal of brine effluent
6. GHG emissions associated with on-site renewable energy generation and integrated resource recovery

7. Regionally transferable models specific to water reuse or desalination
8. Assessments of GHG emissions for different types of water reuse (agricultural, direct potable, indirect potable)

Summary of Chapter 7: Availability and Application of Emission Models

This chapter provides information on type (e.g., software, MS Excel, web based), availability (e.g., commercial, public, upon request), and Website or contact information for different models. LCA-based models use commercial software, whereas specific models use spreadsheets that are available upon request. Hybrid LCA models use both spreadsheets available upon request and a publicly available web-based model. The type and availability of other related models vary.

The applicability of hybrid LCA and specific models was also assessed by sending a survey to utility partners containing input data requirements for GHG estimation models (e.g., WEST, WESTWeb, and the Tampa Bay Water Model). Survey results determined that partner utilities did not collect enough model input data for full utilization of most of the GHG estimation models. This reveals that data collected by utilities may be a limiting factor to the successful implementation of GHG estimation models.

Two models were compared in a case study: Tampa Bay Water Model and WEST Model. The Tampa Bay Water Model is the simplest one, requiring minimum data input, and the hybrid LCA models (WEST, WWEST, and WESTWeb) models are the most sophisticated, requiring extensive data input. It is recommended that existing facilities extend their data collection efforts to include, at a minimum, the information on electricity providers in addition to the amount of water pumped and produced and facility-wide electricity usage. For both existing and future facilities, the recommendations are to establish a standard data collection template and collect the following data: amount of water pumped and produced, name of electricity providers, electricity consumption associated with specific unit processes and entire facility, chemical consumption, material consumption, process equipment usage, and on-site renewable energy production. In terms of model development, a user-friendly and robust model should be developed that (1) would allow utilities and design firms to plan, design, and manage water reuse and desalination facilities; (2) is applicable to different geographical regions; and (3) has an option that would require different levels of sophistication related to required input parameters.

Chapter 1

Introduction

Water and wastewater utilities are increasingly discovering a need to adapt to increased climate variability and associated supply reliability issues. In many parts of the United States, there have been periods of prolonged drought in recent years combined with increasing pressures on traditional water resources due to increases in population and changes in land use. Accordingly, some locations have turned to utilization of alternative water supplies and treatment technologies, such as water reuse and desalination, to meet water demand.

Although use of alternative water supplies is beneficial, there are concerns that energy-intensive water reuse and desalination treatment processes have larger environmental footprints than conventional water and wastewater treatment. For example, utilization of these energy-intensive treatment processes is associated with greenhouse gas (GHG) emissions and other atmospheric pollutants from electricity generation.

To address the problem of elevated carbon footprints and climate change impacts, many states have taken legislative action to mandate a reduction in GHG emissions. In addition, there have been a number of studies conducted to assess the environmental impacts of water reuse and desalination facilities. Although these studies may provide designers, managers, and researchers with guidance related to best alternatives, the contribution of desalination and water reuse technologies and their unit processes to GHG emissions remains unclear. To assist facilities in estimating GHG emissions, identification or development of a model for estimating carbon footprints specific to water reuse and desalination is necessary.

In response to this need, the WateReuse Research Foundation contracted with the University of South Florida (Tampa) and Trussell Technologies, Inc. (Pasadena, CA) to lead the "Feasibility Study on Model Development To Estimate and Minimize the Greenhouse Gas Concentrations and Carbon Footprint of Water Reuse and Desalination Facilities." The primary objective of this project is to gather information that will help utilities estimate the GHG emissions and carbon footprint of existing and future water reuse and desalination facilities. For purposes of this study, a carbon footprint is defined as the total set of greenhouse gas emissions, in carbon equivalents, caused by an organization, event, product, or person. The term carbon footprint is also used interchangeably with GHG emissions and global warming potential (GWP) in this report. For water supply systems, relevant GHG emissions include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Specifically, work encompasses U.S. and international activities related to model development to estimate and minimize GHG emissions and the carbon footprint of water reuse and desalination.

To achieve the primary objective, this project is divided into six tasks, each with an associated chapter.

Task 1: Literature Review. Review existing literature, research, and other sources, some obtained through visits and communication with partner utilities (Chapter 2).

Task 2: Parameters Used in Previous GHG Studies and Models. Perform a thorough analysis of available information required to estimate the carbon equivalent footprint of desalination and water reuse facilities (Chapter 3).

Task 3: Previously Documented Emissions. Produce a summary of previously documented GHG and CO_2 emissions comparing those associated with water recycling and desalination with respect to location, facility capacity, technology, and energy mix (Chapter 4).

Task 4: Available Model Review. Review available emission models and tools that run aspects of the parameters developed or the emissions documented previously related to water reuse and desalination (Chapter 5).

Task 5: Information Gap Identification. Identify any gaps in knowledge preventing the creation of a robust, accurate, and precise model for water reuse or desalination facility GHG emissions (Chapter 6).

Task 6: Availabilty and Implementation of Emission Models. Detail the availability and implementation of off-the-shelf models that allow utilities to accurately assess the potential GHG emissions of current, planned, or potential desalination or recycled water facilities (Chapter 7).

The results obtained from this study will provide water utilities and other interested parties with a list of currently available tools or models to assess the carbon footprint for water reuse and desalination facilities and provide utilities with a recommendation for a tool that utilities are able to readily use that may support the development of a more accurate and applicable carbon footprint model for water reuse and desalination.

Chapter 2

Literature Review

This chapter provides a review of current literature aimed at quantifying GHG emissions and carbon footprints related to water reuse and desalination utilities. Specifically, a detailed overview of the types of technology studied, goals of each study, and relevant findings related to desalination and water reuse is provided. Some items that have been evaluated in the literature include (1) the carbon footprint and GHG emissions of alternative water supply options, including water reuse and desalination technologies (Stokes and Horvath, 2006; Lyons et al., 2009; Muñoz et al., 2010); (2) the application of alternative energy supply options (e.g., wind, solar; Ortiz et al., 2007; Biswas, 2009; Stokes and Horvath, 2009; Jijakli et al., 2011); and (3) the comparison of GHG emissions from desalination or water reclamation technologies (Raluy et al., 2006; Friedrich et al., 2009; Lyons et al., 2009).

The majority of the studies included in the literature review used life cycle assessment (LCA) to quantify GHG emissions from water reuse or desalination facilities. LCA models can be used to evaluate the environmental impact of any process or product over its lifetime (ISO, 1997) and are not specific to water supply systems. Other models were specifically designed to estimate GHG emissions from water supply systems. These include hybrid LCA models, specific models, and other related models. The hybrid LCA models use combined process-based and economic input–output LCA (EIO-LCA) to estimate emissions from water supply systems (Stokes and Horvath, 2006, 2009; Mo and Zhang, 2012). These models have been applied previously to water reuse and desalination facilities and are currently best suited to estimate life cycle GHG emissions from them. Specific models use specific utility parameters to estimate operational emissions from water supply systems and have attributes applicable to water reuse and desalination. Other related models contain some transferable elements that would be useful to the development of an accurate and transferable model but have less applicability to water reuse and desalination because of certain limitations. Studies on GHG emissions of water reuse and desalination facilities are discussed in Chapter 4, and model literature is discussed in Chapter 5.

This chapter presents a literature review of studies that quantify GHG emissions for desalination and water reuse facilities. Section 2.2 provides a detailed description of the studies that focused solely on desalination facilities. This section first presents studies that compare reverse osmosis (RO) with thermal desalination technologies and then describes studies that focused predominately on desalination RO. Section 2.3 describes studies focused specifically on water reuse technologies. All of these studies evaluated GHG emissions from water reuse facilities that focus on filtration- or membrane-based technologies. Finally, a review of the papers applicable to both water reuse and desalination is presented in Section 2.4. These studies include papers comparing RO desalination to various water reuse alternatives as well as some that compare unspecified water reuse and desalination technologies.

2.1 Literature Collection

To identify available models and assess their applicability for estimating the carbon footprint of water reuse and desalination facilities, an extensive review of the literature was conducted using online resources and personal communication. Online resources included search engines (e.g., Google Scholar and Web of Science) and the library database system at the University of South Florida (USF). Through the USF database, Compendex, Science Direct, Wiley Online Library, Springer Link, and IEEE Explore were used to identify existing studies. Gray literature (not peer reviewed) was obtained using Google Scholar and through personal communication with researchers and practitioners. In total, 24 papers in which GHG emissions were estimated at desalination or water reuse facilities were identified and reviewed.

In addition, utility partners and more than 20 practitioners and researchers were asked if they were aware of any existing models or methodologies for estimating carbon footprints at water reuse and desalination facilities. A list of these contacts is presented in Appendix A. Responses received from these individuals verified that (1) all available literature was identified through the literature search, (2) few partners had their own models, and (3) there is a need for a tool to estimate GHG emissions effectively at water reuse and desalination facilities. Information obtained from the literature review was catalogued in a standardized database to document existing models and their estimates. This database was developed using Microsoft Excel and is structured based on the first six tasks outlined in Chapter 1.

2.2 Desalination Literature

Review of the desalination literature resulted in the identification of nine LCA-based papers. GHG studies on desalination technologies took place in Australia, Spain, and the United Arab Emirates (UAE). Of these papers, the most widely studied desalination technology was RO. In contrast, the multi-effect distillation (MED) and multistage flash (MSF) desalination treatment technologies were considered in only three of the papers reviewed. Because of the high number of studies focused on RO desalination, the findings are presented in two categories: studies comparing different technologies and RO-based studies.

2.2.1 Different Desalination Technologies

Desalination technologies within the literature included (1) MSF, (2) MED, and (3) RO (Raluy et al., 2004, 2005a, 2006). Of these three methods, MSF and MED are considered thermal desalination processes. MSF and MED facilities rely on energy-intensive technologies (e.g., heat boilers, generators) to fuel the desalting process as opposed to RO desalination, which uses membrane separation processes to treat water.

Table 2.1. Overview of Comparative Desalination Literature

Focus of Study	Capacity of Facility	Population Served	LCA Tools Used	Reference
Compares desalination technologies when integrated with energy production systems	45,500 m^3/day (12.02 MGD)	not provided	SimaPro 5.0	Raluy et al., 2004
Compares desalination technologies when integrated with electricity mixes	45,500 m^3/day (12.02 MGD)	not provided	SimaPro 6.0	Raluy et al., 2005a
Compares desalination technologies with varying electricity consumption	45,500 m^3/day (12.02 MGD)	not provided	SimaPro 6.0	Raluy et al., 2006

Table 2.1 presents a general overview of studies using LCA-based models to estimate GHG emissions of different desalination technologies. These studies are presented by the major areas of focus covered. The focus of study for these papers includes factors identified in the table that can impact the GHG emissions for all three technologies (e.g., energy production systems, electricity mixes, and electricity consumption level). Unfortunately, none of these studies documented estimation of GHG emissions associated with brine disposal at an inland brackish water desalination facility. This may be important because brine disposal options can be energy intensive (e.g., thermo crystallizing technology), although one study recommends combining brackish water concentrate with seawater feed to reduce salinity levels and power requirements at seawater RO facilities (Wilf et al., 2012).

For the studies comparing MSF, MED, and RO technologies, CO_2 emissions for MED systems range from 0.3 to 26.9 kg CO_2/m^3. Emissions from MSF technologies range from 0.3 to 34.7 kg CO_2/m^3, and RO emissions ranged from 0.08 to 4.3 kg CO_2/m^3 (Raluy et al., 2004, 2005a, 2006; Lyons et al., 2009). Therefore, RO technologies were found to be less energy intensive and have lower CO_2 emissions than MSF and MED. The studies in Table 2.1 compare the MSF, MED, and RO technologies by selecting a functional unit, as is typically done in LCA studies. The functional unit is used to compare different technologies based on the function or purpose of the system over its useful lifetime. For these studies, the functional unit is the production of 45,500 m^3/day (12.0 MGD) over a period of 8000 hours needed for annual operation over a 25-year lifetime (Raluy et al., 2004, 2005a, 2006). Collectively, these studies highlight the importance of variations in energy production systems, electricity mixes, and energy consumption levels.

Raluy et al. (2004) compares desalination technologies when integrated with energy production systems. This study found that a hybrid plant using a combined cycle had lower CO_2 emissions than conventional steam, cogeneration with combined cycle, and a hybrid plant with conventional steam. In addition, when comparing different fuel types, the study found that natural gas had lower CO_2 emissions than oil or coal.

In another study, the same author compares desalination technologies integrated with renewable energy sources. Raluy et al. (2005a) found that incorporating renewable energy sources led to a 69 to 80% reduction of CO_2 emissions for three technology types, compared to a conventional fossil fuel–based energy production system. In comparing various renewable energy sources (e.g., wind, solar thermal, drive waste heat, photovoltaic [PV], and hydropower), hydropower was found to have the greatest CO_2 reduction potential.

A third paper from this research group expands on previous research to investigate the impact of varying electricity consumption levels for different desalination technologies (Raluy et al., 2006). A brief overview of each study is described in Section 2.2.1. It is important to note that MSF and MED processes are driven by heat. Therefore, an appropriate unit for comparison for RO, MSF, and MED is the equivalent energy requirement because this is directly related to GHG emissions. Table 2.2 compares total electrical energy requirements for thermal and nonthermal desalination processes. The equivalent energy requirement for MSF is three to five times higher than that of the seawater RO.

Production cost and energy consumption are related for RO desalination. For example, Gude (2011) found that energy consumption costs contribute to 69% of the total production cost for RO desalination, compared to membrane and filter replacement (21% of total production cost) and chemical costs (10% of total production cost). In addition, Gude reports that high pressure pumps have the highest percent contribution to energy consumption at 84.4%, followed by product transfer and supply (6.7%), seawater supply (4.5%), pretreatment (2.6%), and posttreatment (1.8%). Given the high operational phase energy consumption costs, efforts have been made to reduce the energy consumption at RO desalination facilities using different types of energy recovery devices (ERD) and specific operating strategies. ERDs reduce energy consumption by transferring captured energy to provide a pressure boost to second-stage feed pumps in RO desalination systems (Littrell et al., 2012). These authors also found that ERDs reduced the energy consumption by 25% for a brackish RO treatment facility in Port St. Lucie, Florida.

Table 2.2. Electricity Consumption of Different Desalination Technologies

Desalination Technology	**Electrical Energy (kWh/m3)**	**Electrical Equivalent Thermal Energy (kWh/m^3)**	**Total Electrical Energy (kWh/m^3)**
Brackish water RO	1	0	1
Brackish water EDR	1.5	0	1.5
Seawater RO	4–4.5	0	4.0–4.5
MED	1.0–1.5	5.0–8.5	6.0–10
Multistage flush	4	9.5–19.5	13.5–23.5
MED-thermal vapor compression	1.0–1.5	9.5–25.5	10.5–27.0
Mechanical vapor compression	7.0–12.0	0	7.0–12.0

Source: Erdal et al., 2012
Notes: EDR=energy recovery devices; MED=multi-effect distillation; RO=reverse osmosis

2.2.1.1 Energy Production Systems

Raluy et al. (2004), Life-cycle Assessment of Desalination Technologies Integrated with Energy Production Systems. In this study, the environmental impact of desalination technologies integrated with various energy production systems was assessed using an LCA approach. Specifically, CO_2 emissions from four different systems using natural gas, coal, and oil as a fuel source were evaluated. The four systems investigated in this study were conventional steam, cogeneration with combined cycle, hybrid plant with conventional steam, and a hybrid plant with combined cycle.

Comparison of MED, MSF, and RO showed that RO desalination technologies were associated with lower CO_2 emissions compared to thermal technologies primarily from lower electricity consumption. Using an average European Union electricity mix (43.3% thermal, 40.3% nuclear 16.4% hydroelectric), RO emissions ranged from 1.2 to 1.8 kg CO_2/m^3, whereas MED emissions were 18.1 kg CO_2/m^3, and MSF emissions were 23.4 kg CO_2/m^3. It is important to note that this study does not evaluate all GHG emissions as CO_2 equivalents but solely focuses on CO_2 emissions.

Table 2.3. Desalination CO_2 Emissions from Different Energy Sources

Technology	Configuration	Natural Gas (kg CO_2/m^3)	Coal (kg CO_2/m^3)	Oil (kg CO_2/m^3)
MSF	non-integrated	23.4	34.7	30.5
	cogeneration with steam cycle	17.2	26.1	22.8
	cogeneration with combined cycle	9.4	14.2	12.4
	hybrid plant with steam cycle	9.5	13.9	12.2
	hybrid plant with combined cycle	5.6	8.0	7.1
	driven by residual heat	2.0	not provided	not provided
MED	non-integrated	18.1	26.9	23.6
	cogeneration with steam cycle	12.9	19.5	17.0
	cogeneration with combined cycle	7.0	12.1	3.7
	hybrid plant with steam cycle	7.3	11.4	3.6
	hybrid plant with combined cycle	4.4	1.2	2.1
	driven by residual heat	1.2	not provided	not provided
RO	European electricity production scenario[1]	1.8	not provided	not provided
	steam cycle	2.8	not provided	not provided
	internal combustion engine	2.1	not provided	not provided
	combined cycle	1.8	not provided	not provided

Source: Raluy et al., 2004
Notes: 1=European Union electricity production scenario: 43.3% thermal; 40.3% nuclear; 16.4% hydroelectric; MED=multi-effect distillation; MSF=multistage flash; RO=reverse osmosis

In the case of fuel types, natural gas was the cleanest fuel, followed by oil, then coal. For example, emissions from the MSF desalination technology associated with natural gas, coal, and oil energy sources ranged from 2.0 to 23.4 kg CO_2/m^3, 8.0 to 34.7 kg CO_2/m^3, and 7.1 to 30.5 kg CO_2/m^3, respectively. MED-related emissions were similar, with natural gas, coal, and oil ranging from 1.2 to 18.1 kg CO_2/m^3, 1.2 to 26.9 kg CO_2/m^3, and 2.1 to 23.6 kg CO_2/m^3, respectively. Analysis of individual scenarios revealed that a hybrid plant using a combined cycle was the best alternative for reducing the environmental load and CO_2 emissions (see Table 2.3 for details).

2.2.1.2 Electricity Mixes

Raluy et al. (2005a), Life Cycle Assessment of Desalination Technologies Integrated with Renewable Energies. Similar to studies by Jijakli et al. (2011) and Biswas et al. (2009), this study evaluates the role of renewable energies in reducing CO_2 emissions. RO, MED, and MSF desalination facilities were assessed. The overall objective of these studies was to estimate the environmental impact and emissions associated with each type of desalination facility when integrated with different renewable energy sources. Renewable energy sources evaluated included wind, solar thermal, drive waste heat, PV, and hydropower.

Consistent with other studies, RO desalination, when integrated with renewable energy mixes, produced lower CO_2 emissions than MED and MSF desalination. For example, RO, MED, and MSF facilities using hydropower energy produced 0.1, 0.3, and 0.3 kg CO_2/m^3 water, respectively. In terms of renewable energy, Raluy et al. (2005a) report that integration of hydropower provided the greatest reduction in airborne emissions compared to other alternatives (e.g., wind energy, PV energy, solar thermal). The renewable energy with the least CO_2 reduction potential was solar thermal. A solar thermal power source produced emissions of 1.8, 8.3, and 11.0 kg CO_2/m^3 water for RO, MED, and MSF, respectively.

Overall, this study confirmed that, as expected, integration of renewable energy with existing desalination technologies will decrease CO_2 emissions. The incorporation of renewable energies can reduce CO_2 emissions by 69 to 80% on average, compared to baseline facilities using an average European Union electricity mix (43.3% thermal, 40.3% nuclear, 16.4% hydroelectric).

2.2.1.3 Electricity Consumption Levels

Raluy et al. (2006) Life Cycle Assessment of MSF, MED, And RO Desalination Technologies. Previous studies have suggested that RO desalination processes produce lower GHG emissions and environmental impacts compared to thermal desalination technologies (Raluy et al., 2004, 2005a). In this study, the CO_2 emissions and environmental impacts associated with RO, MED, and MSF desalination technologies were evaluated with varying levels of electricity consumption. For MSF and MED, the electrical energy consumption level was 4 and 2 kWh/m^3. For RO, five electrical energy consumption levels ranging from 2 to 4 kWh/m^3 were evaluated.

Results were consistent with previous findings: RO desalination has the lowest CO_2 emissions. For example, CO_2 emissions from RO ranged from 0.08 to 3.1 kg CO_2/m^3, whereas emissions from MED and MSF systems ranged from 0.3 to 26.9 kg CO_2/m^3 and 0.3 to 34.7 kg CO_2/m^3 (Raluy et al., 2004, 2005a, 2006). See Table 2.2 for details on emissions for specific RO technologies associated with different fuel sources. In addition, the study found that higher energy consumption levels led to higher GHG emissions, as expected.

For example, a 4-kWh/m^3 RO system produced 1.8 kg CO_2/m^3, and a 2-kWh/m^3 system reduced emissions to 0.9 kg CO_2/m^3. These results showed that a 47% reduction in CO_2 emissions was possible when energy consumption rates of the RO facility were reduced from approximately 4 to 2 kWh/m^3. Thus, it is suggested that new recovery systems designed to reduce energy consumption may be important to decrease the overall environmental impacts at RO desalination facilities.

2.2.2 Reverse Osmosis Desalination

The majority of desalination studies focus on RO desalination technologies. GHG emissions from studies focusing on RO desalination used LCA software, such as SimaPro and Gabi, to estimate GHG emissions. The emissions of RO desalination facilities ranged from 0.4 to 6.7 kg CO_2eq/m^3, depending on the capacity of the facility and other factors. In addition to capacity, major factors that impact GHG emissions include (1) water supply alternative selected and quality of the water source treated (e.g., level of total dissolved solids (TDS); (2) evaluation of facilities with varying electricity mixes; and (3) assessment of pretreatment processes for desalination. Table 2.4 provides an overview of studies focusing on RO for desalination.

Table 2.4. Overview of Reverse Osmosis Desalination Studies

Focus of Study	Capacity of Facility	Population Served	LCA Tools Used	Reference(s)
Evaluate water supply alternatives for desalination	7500–20,000 m^3/day (2.0–5.3 MGD)	1800–1,400,000	Gabi, SimaPro 5.0	Peters and Rouse, 2005; Raluy et al., 2005b[1]; Muñoz and Fernández-Alba, 2008
Evaluate desalination facilities with varying electricity mixes	not provided	not provided	SimaPro	Biswas, 2009; Jijakli et al., 2011
Assess pretreatment process for desalination	189,000 m^3/day (50 MGD)	not provided	Gabi 4	Beery et al., 2010

Notes: 1=Also evaluates electricity mixes and energy consumption levels; Peters and Rouse (2005), Munoz and Fernández-Alba (2008), and Jijakli (2011) compare emissions associated with brackish water and seawater desalination. Raluy et al (2005b), Biswas et al. (2009), and Beery et al. (2010) evaluate seawater only. Most studies did not provide total dissolved solid level or level of required salt removal.

To understand the environmental impact of desalination treatment, a number of studies evaluate different water supply alternatives for desalination. These studies include those by Peters and Rouse (2005), Muñoz and Fernández-Alba (2008), and Raluy et al. (2005b), which aim to measure GHG emissions associated with RO desalination from different water sources (e.g., seawater, brackish groundwater) and supply systems (e.g., desalination versus water transfer). Peters and Rouse (2005) found that seawater desalination produces more GHG emissions than either imported water or brackish surface water desalination (imported water is pumped 700 km). In addition, Muñoz and Fernández-Alba (2008) investigated brackish groundwater (influent TDS of 15,000 mg/L) and seawater (influent TDS of about 36,000 mg/L) desalination and determined that GHG emissions from these two water sources were 1.1 and 1.9 kg CO_2eq/m^3 for the same electricity mix. A Project Advisory Committee member suggested that brackish water has an energy requirement of approximately 1 to 1.5 kWh/m^3, whereas seawater RO has an energy requirement of 4 to 4.5 kWh/m^3. In addition, seawater RO also requires microfiltration (MF), ultrafiltration (UF), or granular media filtration (GMF) as pretreatment. Accordingly, one would expect higher GHG emissions for seawater RO than brackish RO.

It is also important to note that in this study, brine was discharged to the sea, which may minimize GHGs associated with concentrate management and disposal. These two studies reveal that seawater desalination has a higher carbon footprint than the treatment of other brackish water sources (e.g., surface water and groundwater), mainly because of higher electricity requirements. Furthermore, in contrast to Peters and Rouse (2005), Raluy et al. (2005b) found that importation and RO desalination alternatives had comparable CO_2 emissions (imported water was transferred in two routes of 172 and 742 km, mostly in open channels but also with a few pump stations). Factors impacting GHG emissions in this study were energy consumption levels and electricity mixes.

Additional studies that evaluate the impact of the electricity mix at RO facilities were conducted by Biswas et al. (2009) and Jijakli et al. (2011). Jijakli compared desalination facilities powered by PV, solar still, and delivery of desalinated water via truck. Among these options, PV-powered facilities were found to have the lowest CO_2 emissions, at 0.8 kg CO_2/m^3. Biswas investigated other electricity sources and mixes (e.g., wind, existing electrical grid, both combined) and found that renewable mixes provide the greatest reduction in GHG emissions, as expected. In addition, Biswas investigated unit processes within a seawater desalination facility (e.g., extraction, pretreatment, RO treatment, posttreatment, and water delivery). RO treatment was found to be the most energy- and GHG-intensive process. Compared to the other unit processes, RO treatment had a 75% contribution to GHG emissions. These studies highlight the importance of renewable energy generation to reduce emissions associated with RO desalination.

Finally, Beery et al. (2010) conducted a study on the impact of pretreatment processes for seawater desalination. This study found that conventional media filtration was less GHG-intensive than UF membrane-based filtration. This illustrates that pretreatment selection can impact the carbon footprint of a desalination system. Additional details on the background and findings of studies focusing on RO technologies are discussed in the following sections.

2.2.2.1 Water Supply

Peters and Rouse (2005), Environmental Sustainability in Water Supply Planning—An LCA Approach for the Eyre Peninsula, South Australia. In this study, an LCA-based assessment was conducted to determine the best approach for supplying potable water to the Eyre

Peninsula in Southern Australia. In this region, the Southern Australia Water Company provides an average of 7500 m^3 potable water each day (2.0 MGD). To sustain its water demand and identify treatment options with less environmental impact, the company evaluated the GHG emissions of three treatment options: (1) desalination treatment of brackish surface water collected from the Tod Reservoir, (2) desalination treatment of seawater collected from the Spencer Gulf, and (3) extension of the Morgan-Whyalla water system pipeline, which transports treated water from the Murray River.

The results indicated that the desalination treatment of brackish surface water from the Tod Reservoir had lower GHG emissions than seawater desalination and importation for the same electricity mix. The carbon footprint of the Tod Reservoir desalination was 2.0 kg CO_2eq/m^3. This was lower than seawater desalination (6.7 kg CO_2eq/m^3) and extension of the Morgan-Whyalla system (2.2 kg CO_2eq/m^3).

Muñoz and Fernández-Alba (2008), Reducing the Environmental Impacts of Reverse Osmosis Desalination by Using Brackish Groundwater Resources. This study assessed RO desalination by comparing brackish groundwater and seawater sources. The RO desalination facility for brackish groundwater is a medium-sized plant in Almeria, Spain, with a capacity of 20,000 m^3/day (5.3 MGD). The primary purpose of the plant is to supply water to farmers for crop irrigation. To compare GHG emissions and assess the overall environmental impact of the brackish groundwater facility to that of a seawater facility, Muñoz and Fernández-Alba used the Cuevas del Almanzora plant and seawater facility described by Raluy (2003).

LCA results indicate that GHG emissions associated with RO brackish groundwater desalination (source TDS of 15,000 mg/L) were 1.1 kg CO_2eq/m^3, whereas emissions associated with seawater desalination (source TDS of 36,000 mg/L) were 1.9 kg CO_2eq/m^3 for the same electricity mix. In this study, 11,000 m^3 of brine (TDS of 35,000 mg/L) was produced daily and discharged to the ocean. In addition, the assessment showed that electricity demand contributed to more than 95% of the impact for GWP over the life cycle.

Raluy et al. (2005b), Life Cycle Assessment of Water Production Technologies. In this study, the environmental impact associated with RO seawater desalination technologies and water transfer from the Ebro River was evaluated. The main objective was to determine what effect the Spanish National Hydrologic Plan to transfer water to basins located along the Spanish Mediterranean Region would have on environmental loads. The total amount of water expected to be transferred was 1.05 billion m^3 per year (760 MGD).

Results showed that neither alternative proved to be better. The water transfer scenario had lower energy values compared to RO desalination; however, the amount of construction material required for the Ebro River water transfer was higher, thus making the overall CO_2 emission rates comparable over both 25- and 50-year lifespans considered. For RO desalination, emissions ranged from 1.2 to 2.3 kg CO_2/m^3 produced, based on RO electricity consumption levels of 2 and 4 kWh/m^3. Emissions associated with the Ebro River water transfer scheme ranged from 1.4 kg CO_2/m^3 for the 25-year lifespan and 1.6 kg CO_2/m^3 for the 50-year lifespan. This study also determined that the electricity mix can impact overall CO_2 emissions. For varying electricity mixes, CO_2 emissions for RO and Ebro River Water Transfer (ERWT) ranged from 0.2 to 3.2 kg CO_2/m^3 and 0.1 to 2.0 kg CO_2/m^3.

2.2.2.2 Electricity Mixes

Another strategy that can be used to lower energy consumption and the associated carbon footprint of a desalination facility is the incorporation of renewable energy alternatives (Jijakli et al., 2011). As the development and availability of renewable energy increases, some desalination facilities are beginning to investigate these alternative energies to decrease the amount of electricity consumed. Renewable energies targeted include wind, solar still, and PV.

To assess the environmental impact of renewable energies on current conventional RO desalination technologies, an environmental assessment using LCA-based approaches was conducted by Jijakli et al. (2011) and Biswas et al. (2009). Results from both studies demonstrated that, as expected, incorporation of at least one renewable energy type or a mix of energies would reduce the carbon footprint associated with RO desalination. A brief overview of each study is described next.

Jijakli et al. (2011), How Green Solar Desalination Really Is? Environmental Assessment Using Life-Cycle Analysis (LCA) Approach. In this study, the carbon footprint of renewable energy–powered decentralized desalination plants was estimated using an LCA-based model. In the UAE, MSF and MED are the most common types of desalination technologies used, although recent advancements in membrane technologies have led to an interest in RO desalination for new facilities. In efforts to reduce GHG emissions and environmental loads, the UAE is considering the incorporation of renewable energies at desalination facilities. Solar still and PV energy are of particular interest. Preference for these energies is due to the ability of the UAE to harness high solar irradiations.

To evaluate the environmental impacts of renewable energy, three scenarios were considered: PV-powered RO desalination (PV-RO), solar still–powered RO desalination, and truck delivery of RO desalinated water. In this study, brackish groundwater was the source for PV-RO, and solar still and seawater were the sources for the central RO facility. This study demonstrated that PV-RO, solar still, and truck delivery produced 0.8, 3.1, and 3.2 kg CO_2/m^3, respectively. Therefore, the assumed PV-powered RO desalination case was the alternative with the lowest CO_2 emissions.

Biswas (2009), Life Cycle Assessment of Seawater Desalinization in Western Australia. In this study, Biswas conducted an LCA-based study to quantify the amount of GHG emissions emitted by RO desalination facilities powered by renewable (e.g., wind) and conventional (e.g., national grid) energy sources. Electricity mixes evaluated included a facility powered entirely by wind energy, wind and national grid electricity mixes, and the existing electricity grid. Results of the study demonstrate that switching from the national grid to wind energy will reduce GHG emissions by approximately 90%. This would reduce GHG emissions at the Perth facility from 3.9 to 0.4 kg CO_2eq/m^3. In addition, when wind energy is combined with the national grid at 75%, 50%, and 25%, GHG emissions are reduced by approximately 68%, 45%, and 23%, respectively.

In addition to the types of energy used by the system, emissions generated by various unit processes were evaluated. Processes considered in the study included extraction, pretreatment, RO treatment, posttreatment, and water delivery/waste treatment. CO_2 emissions were 0.3 kg CO_2eq/m^3 (7%) for extraction of seawater, 0.06 kg CO_2eq/m^3 (1%) for membrane-based pretreatment, 2.9 kg CO_2eq/m^3 (75%) for RO treatment, 0.05 kg CO_2eq/m^3 (1%) for chemical post treatment, and 0.6 kg CO_2eq/m^3 (16%) for water delivery/waste treatment. Of the five processes, RO treatment was shown to contribute the most to CO_2 emissions, accounting for 75%.

2.2.2.3 Pretreatment Processes

Beery et al. (2010), Sustainable Design of Different Seawater Reverse Osmosis Desalination Pretreatment Processes. The last factor impacting the amount of GHG emissions associated with seawater RO desalination is the type of pretreatment process established at the facility. Energy requirements and types of pretreatment processes vary with the water source. Examples of pretreatment processes associated with desalination facilities include coagulation, flocculation, and filtration. Filtration methods can use either media, cartridges, or membranes. Each of the pretreatment methods requires energy, materials, and disposal of waste. As such, each contributes to the carbon footprint and may play a significant role in the environmental loads associated with a facility.

To identify the carbon footprint of various pretreatment methodologies, Beery et al. conducted an LCA-based study using a seawater desalination plant as the system of focus. In addition to quantifying emissions, the sustainability and eco-efficiency of an RO desalination facility were evaluated using a life cycle cost assessment. Estimates for emissions and eco-efficiency were based on a seawater RO facility capable of producing 189,000 m^3 of water each day (49.9 MGD). Pretreatment methodologies considered in this study were granular media filtration (GMF), UF, and UF with ultraviolet (UV) disinfection.

In this study, it was found that pretreatment methods using UF with UV had higher GHG emissions (3.2 kg CO_2eq/m^3) than UF alone (2.4 kg CO_2eq/m^3) or GMF (2.3 kg CO_2eq/m^3). This shows that UF membrane-based technologies are more GHG intensive than conventional media filtration; however, it is important to note that membrane-based pretreatment is a relatively new technology that will improve with further design optimization over time. This is exemplified by Stokes and Horvath (2009), who found that seawater desalination with membrane pretreatment had a lower carbon footprint than conventional pretreatment.

2.3 Water Reuse Literature

Review of the water reuse literature resulted in the identification of six studies. Four are LCA-based papers; one estimates GHG emissions based on water quality data and electricity emission factors; and one estimates emissions at a statewide level. The literature on water reclamation is geographically diverse, including studies from China, South Africa, Australia, Spain, Israel, and the United States. Within this body of literature, conventional filtration technology and membrane-based filtration technology (e.g., continuous MF, UF) were the most widely studied tertiary treatment options. Other wastewater treatment options include conventional activated sludge (CAS), followed by tertiary treatment, membrane-based biological reactor (MBR) technology (e.g., immersed and external MBR), wastewater stabilization ponds, and aerated lagoons. All of the water reclamation studies include either conventional filtration or membrane-based technologies, and the findings are presented in the following section. Refer to Chapter 4 for additional details on previously documented GHG emissions.

2.3.1 Treatment Technologies for Water Reuse

LCA-based tools (e.g., Gabi and combined process-based and input output–based LCA) and alternative estimation techniques were used to evaluate the GHG emissions of filtration- and membrane-based tertiary treatment options for water reclamation. Water reclamation facilities from 3000 to 50,000 m^3/day (0.8–13.2 MGD) were assessed to evaluate the GHG emissions of urban water supply options. The environmental impact of wastewater treatment facilities with varying secondary/tertiary treatment alternatives (e.g., UF versus different types of MBR) and the carbon footprint of water reuse alternatives at different treatment levels (e.g., secondary versus tertiary treatment) were also evaluated.

Because of variations in system boundaries, parameters considered, underlying assumptions, and GHG estimation methodologies, it is important to understand the limitations of comparing GHG emission findings from different literature sources and technologies. Other important considerations that may impact GHG emissions include the electricity mix (which impacts indirect emissions), influent water quality (which impacts direct process emissions), and intended use of the reclaimed water. The studies on water reuse provide comparisons of tertiary treatment alternatives, demonstrate the importance of renewable energy, and highlight the benefit of water reclamation. An overview of all studies on water reclamation is presented in Table 2.5.

Table 2.5. Overview of Water Reuse Literature

Focus of Study	Capacity of Facility	Population Served	Tools Used	Reference(s)
Comparison of different technologies	40,000 m^3/day (10.5 MGD)	Not provided	Combined process and input output–based LCA, Gabi 3	Tangsubkul et al., 2005; Friedrich et al., 2009
Evaluate impact of different electricity mixes	3000 m^3/day (0.8 MGD)	13,200	Gabi 3	Ortiz et al., 2007
Assess abatement potential resource recovery	50,000 m^3/day (13.2 MGD)	Not provided	Combined process and input output–based LCA; estimates GHG emissions based on typical wastewater strength	Zhang et al., 2010; Fine and Hadas, 2012
Assess abatement potential resource recovery (statewide level)	Varies	2006 population of Texas	Estimates GHG emissions based on emission factors	Stillwell and Webber, 2010

Notes: GHG=greenhouse gas; LCA=life cycle assessment

Whereas most studies focused on energy-intensive, membrane-based processes including tertiary filtration (e.g., UF, MF) or MBR (e.g., immersed or external), two studies included lagoon systems (e.g., aerated lagoon, wastewater stabilization pond), which are less energy intensive. Other examples of tertiary treatment trains include:

- Coagulation, sand/anthracite filtration, ozonation, granular activated carbon (GAC), and chlorination (Friedrich et al., 2009)
- Continuous MF and ozonation for pretreatment (Tangsubkul et al., 2005)
- MBR followed by RO (Tangsubkul et al., 2005)

Many studies found that operational electricity consumption was the largest contributor to GHG emissions (e.g., Ortiz et al., 2007; Friedrich et al., 2009). For example, Friedrich et al. found that 90% of the GHG emissions were associated with the operation of a water reuse facility (including chemical coagulation/flocculation, filtration [deep bed filter with sand and anthracite], ozonation, GAC, and chlorination) in which electricity consumption accounted for the majority of this burden. However, one study revealed that the construction phase had a contribution of 17 to 35% of the life cycle GHG emissions (Tangsubkul et al., 2005).

Ortiz et al. (2007) found that CO_2 emissions for secondary treatment were 0.4 kg CO_2/m^3 by CAS, 0.78 kg CO_2/m^3 by CAS–tertiary filtration, 0.82 kg CO_2/m^3 by external MBR, and 0.77 kg CO_2/m^3 by immersed MBR. A separate study determined that secondary treatment with activated sludge had a higher contribution to GHG emissions than tertiary treatment for water reuse (Friedrich et al., 2009). This demonstrates the importance of considering both secondary and tertiary treatment when estimating GHG emissions for water reuse scenarios.

Another important finding showed the importance of the electricity mix when estimating GHG emissions. For example, Ortiz et al. (2007) showed that electricity mixes dominated by renewable or nuclear sources reduced the GWP compared to fossil fuel–dominated mixes, as expected.

Other studies revealed the benefit of water reclamation and other forms of resource recovery in offsetting GHG emissions (Stillwell and Webber, 2010; Zhang et al., 2010; Fine and Hadas, 2012). Water reclamation has a GHG abatement potential, which can be acquired through energy savings (e.g., avoided energy needed to supply and treat sources of potable water), nutrient recovery to offset synthetic fertilizers, and energy recovery from anaerobic treatment processes, which can minimize GHG emissions at water reclamation facilities.

Sobhani and Rosso (2011) studied the contribution of advanced oxidation process (AOP) in treating N-Nitrosodimethylamine, a possible cancer-causing agent, to the overall energy and carbon footprints of an indirect potable reuse system in Orange County, CA. This reuse system consists of two facilities. First, wastewater is sent to the Orange County Sanitation District (OCSD) wastewater treatment plant, which uses an activated sludge system. Next, the effluent is sent to the Orange County Water District's water reclamation plant, which consists of MF, RO, and hydrogen peroxide–based UV AOP. The authors estimated that influent pumping contributed 3% of the total energy footprint, primary treatment contributed 4%, secondary treatment contributed 16%, MF contributed 21%, AOP contributed 7%, and RO had the largest energy footprint relative to the total, contributing 49% to the total OCSD-OCWD system's energy footprint. This suggests that, as a tertiary technology for reuse, RO is a significant energy consumer.

This section describes these studies separated by their respective focus, which includes three major categories: (1) comparison of treatment technologies, (2) impact of different electricity mixes, and (3) abatement potential of resource recovery.

2.3.1.1 Comparison of Treatment Technologies

Friedrich et al. (2009), Carbon Footprint Analysis for Increasing Water Supply and Sanitation in South Africa: A Case Study. This study assesses the GHG emissions of an urban water supply system in Durban, South Africa. The system includes impoundment, collection, distribution, water, wastewater, and water reuse facilities. This study compares two scenarios, in which 200,000 customers were added to the system in an urban and peri-urban context over the construction- and operation-phase life cycle. The impact of plant decommission was considered only for the wastewater treatment facility. The two scenarios were assessed under three water augmentation options: (1) maximizing existing assets, (2) providing water reclamation for industrial customers, and (3) building new infrastructure to supply additional water. The water reuse facility included chemical coagulation/flocculation, filtration (deep bed filter with sand and anthracite), ozonation, GAC, and chlorination.

The water reclamation option in this study had a lower environmental impact than other augmentation options for both urban and peri-urban scenarios. Within the tertiary treatment process, operational phase electricity was the dominant contributor to GHG emissions, accounting for 90% of the environmental impact, with a carbon footprint of 0.09 kg CO_2eq/m^3. In addition, ozonation was found to have the highest electricity consumption and therefore the largest contribution to GHG emissions within the tertiary treatment system. It is also important to note that, within the entire water supply system, the secondary wastewater treatment facility had the highest GHG emissions, primarily because of aeration during the activated sludge process. This highlights the importance of considering a larger system boundary for the GHG emissions associated with all processes, including primary, secondary, and tertiary treatment, to identify critical areas where mitigation efforts can be made.

A unique feature in this study was the evaluation of energy consumption from aeration within the activated sludge process. Friedrich et al. conducted a preliminary improvement analysis for aeration processes, which determined that input dissolved oxygen (DO) levels could be reduced from 2.0 to 1.5 mg/L while maintaining efficient chemical oxygen demand (COD) removal requirements. This was done using Worldwide Engine for Simulation, Training and Automation, software that can optimize DO processes. This software considers factors such as sludge age, recycle rate, waste rate, and levels of aeration.

Tangsubkul et al. (2005), Life Cycle Assessment of Water Recycling Technology. This study used an LCA approach to investigate the environmental impact of three water reuse technologies in Sydney, Australia, over the construction and operation phases. This study used process-based (Gabi3 V.2) and economic input output–based LCA (e.g., Missing Inventory Estimation Tool) to estimate various impact categories, including 100-year GWP, in kg CO_2 equivalents. The technologies included (1) continuous microfiltration (CMF) preceded by ozonation, (2) an MBR with RO, and (3) a wastewater stabilization pond (WSP). Tangsubkul et al. (2005) and Fine and Hadas (2012) are the only studies that include lagoon systems for water reclamation in their analysis, providing insight on the GHG emissions associated with less energy-intensive wastewater treatment technologies that can be integrated with agriculture reuse. Lagoons are often less feasible in coastal areas where available land is scarce and cost is at a premium.

Tangsubkul et al. (2005) compared water recycling with (1) CAS plus tertiary sand filtration plus ozonation followed by CMF and chemical disinfection; (2) MBR plus RO; and (3) a WSP system consisting of an anaerobic pond, a facultative pond, and a maturation pond. The wastewater in the study was a medium-strength wastewater per M&E (1991) with the following characteristics: 220 mg/L of five-day biochemical oxygen demand (BOD_5),500 mg/L of COD, 500 mg/L of TDS, 40 mg/L of total nitrogen, and 8 mg/L of total phosphorus. Energy consumption had the largest percent contribution (68–69%) to GWP in all treatment processes except for the WSP. In addition, energy use data for the CMF with ozonation system in this study was higher than the MBR with RO system, which is likely because the TDS of 500 mg/L in the feed was low enough that less energy-intensive RO membranes could be used for salinity reduction. Furthermore, GHGs associated with concentrate management and disposal practices for the RO system were not included in this study. For the WSP, methane emissions from the ponds contributed to 65% of the GHG emissions. Electricity consumption had a negligible contribution. In contrast to other studies (Ortiz et al., 2007; Friedrich et al., 2009), Tangsubkul et al. (2005) found that the construction phase had a 25 to 38% contribution to GWP for all three technologies.

A unique feature of Tangsubkul et al. (2005) was its inclusion of a sensitivity analysis. According to this study, the sensitivity analysis showed that a 20% change in input parameters related to construction, electricity usage, chemical consumption, effluent water quality, and sludge had an impact of ≤ 20% on results. This indicates that the impact categories were not that sensitive to input parameter changes.

2.3.1.2 Electricity Mixes

Ortiz et al. (2007), Life Cycle Assessment of Water Treatment Technologies: Wastewater and Water-Reuse in a Small Town. This study compares CAS with the addition of tertiary treatment alternatives. The treatment alternatives that followed or were integrated with the CAS process were (1) UF, (2) immersed MBR, and (3) external MBR. LCA was used to investigate the impact of alternative electricity mixes for wastewater and water reuse for a 3000 m^3/day (0.8 MGD) treatment system serving a population of 13,200 people in a small Spanish town. The authors accounted for construction, operation, and decommission life stages.

The study found that tertiary treatment alternatives had higher CO_2 emissions than secondary treatment with CAS only. Values for tertiary treatment technologies ranged from 0.77 to 0.82 kg CO_2/m^3, whereas CAS had emissions of 0.4 kg CO_2/m^3. Use of an external MBR that consisted of two trains of Zee-Weed 500 membranes was found to have higher emissions (0.82 kg CO_2/m^3) than other tertiary treatment technologies. CAS followed by UF had emissions of 0.78 kg CO_2/m^3, and an immersed MBR that followed the biological reactor had emissions of 0.77 kg CO_2/m^3. Similar to Friedrich et al. (2009), the operation phase had a higher impact than other life stage phases considered.

In assessing different electricity production scenarios, an average European Union, French, Norwegian, and Portuguese energy mix were evaluated. The average European Union mix consisted of 43.3% thermal, 40.3% nuclear, and 16.4% hydroelectric production. This mix is dominated by thermal and nuclear energy sources. The French mix was dominated by nuclear energy, consisting of 11.4% thermal, 72.9% nuclear, and 15.7% hydroelectric. The Norwegian mix was dominated by renewable energy with 0.5% thermal, 0.3% nuclear, and 99.2% hydroelectric. The Portuguese mix was dominated by fossil fuel energy sources containing 80.8% thermal, 2.6% nuclear, and 16.6% hydroelectric (Ortiz et al., 2007).

The findings of this study revealed that the French mix (nuclear dominant) and Norwegian mix (renewable dominant) provide the greatest environmental benefit in terms of reduced GHG emission. For example, the external MBR resulted in 0.82 and 0.90 kg CO_2/m^3 emission rate for the European average and Portuguese energy mix. Using the French and Norwegian energy mix, these values decrease to 0.26 and 0.16 kg CO_2/m^3. This showed that the renewable energy mix was the most sustainable option. Using the renewable energy mix also reveals that tertiary treatment can reduce CO_2 emissions to levels comparable with CAS, highlighting the importance of the energy mix when evaluating GHG emissions.

2.3.1.3 Abatement Potential of Resource Recovery

Fine and Hadas (2012), Options To Reduce Greenhouse Gas Emissions During Wastewater Treatment for Agricultural Use. This study investigated the GHG emissions associated with different treatment levels in an Israeli context. It used a unique approach to estimate emissions and abatement potential that differs from the LCA-based assessments used in the majority of the studies. Using conversion factors from published literature, GHG emissions were calculated based on typical strength of Israeli wastewater (e.g., COD) and energy usage. In this study, GHG emissions and corresponding abatement potential from secondary treatment (oxidation ponds) and tertiary treatment (membrane UF) were analyzed. The authors quantified GHG abatement potential by accounting for the amount of GHG emissions avoided. For this, they determined the GHG emissions avoided from use of chemical fertilizers that were replaced by treated effluent and biosolid recovery for agricultural reuse. The GHG abatement potential also accounted for emissions avoided through energy recovered as biogas.

This study determined that secondary treatment levels with oxidation ponds have lower GHG emissions than tertiary treatment levels, at 1.6 and 2.1 kg CO_2eq/m^3. The combined GHG abatement potential (fertilizer replacement and biogas recovery) had a total reduction potential of 55% and 23% for secondary and tertiary treatment levels. The secondary treatment level had a greater potential reduction because of its comparatively lower energy consumption and greater fertilizer value. These findings demonstrate the importance of matching the level of treatment to posttreatment use. In this case, avoiding biological nutrient removal during tertiary treatment reduced energy consumption and provided greater nutrient value to the treated effluent, which can indirectly offset chemical fertilizer use during agricultural reclamation. This highlights the importance of considering nutrient and energy recovery in GHG mitigation efforts at water reuse facilities.

Stillwell and Webber (2010), Water Conservation and Reuse: A Case Study of the Energy–Water Nexus in Texas. In this study, the potential energy savings and associated CO_2 emissions avoided by water conservation and reclamation efforts in Texas were assessed. National average energy usage values for reuse technologies from Goldstein and Smith (2002) were used to estimate the net energy savings from replacing potable water sources with reclaimed water sources. This GHG estimation method differs from the LCA-based methodology used by the majority of water reuse studies. In contrast to LCA-based models, the Stillwell and Webber (2010) method estimated GHG emissions at a statewide level, accounting for operational emissions only.

This investigation found that reclaiming water at 12% of the statewide demand using the least energy-intensive tertiary treatment technologies (e.g., advanced treatment with nitrification, advanced treatment without nitrification) could reduce yearly energy consumption by 73 to 310 million kWh, representing <0.1% of energy produced across the state. This potential energy savings represents an abatement potential of 0.04 to 0.16 million metric tons of CO_2, which was estimated using average U.S. Environmental Protection Agency (USEPA) emission factors for natural gas–fired power providers.

Zhang (2010), Application of Life Cycle Assessment for an Evaluation of Wastewater Treatment and Reuse Project—Case Study of Xi'an, China. This study conducted an LCA of a wastewater treatment facility with tertiary treatment for water reclamation focusing on energy consumption over the construction, operation, and decommission stages. The treatment train consisted of an oxidation ditch with activated sludge and a 150,000 m^3/day (39.6 MGD) capacity for secondary treatment. This was followed by tertiary treatment consisting of chemical coagulation and sand filtration with a 50,000 m^3/day (13.2 MGD) capacity. This system was designed to reuse the treated effluent for both industrial and domestic use.

This study found that energy savings can be acquired through water reclamation. By adding tertiary treatment for water reuse, 1.672×10^{12} kJ of energy was avoided through water savings. This represents the energy consumption needed for treatment if water had not been reclaimed. An additional 74.2×10^9 kJ of energy was saved by accounting for the avoided pollutant discharge associated with reclamation. Water reuse energy savings surpass the life cycle energy consumption of the tertiary treatment facility (1.042×10^{12} kJ) and is almost equivalent to the life cycle energy consumption of combined secondary and tertiary treatment processes, demonstrating the benefit of water reclamation. Although this study did not focus on GHG emissions, it provides relevant information on energy savings, which is directly linked to indirect GHG emissions associated with energy usage.

2.4 Water Reuse and Desalination Comparison

Review of literature that investigated the GHG emissions of both water reclamation and desalination facilities resulted in the identification of nine studies. Seven are LCA-based, and two are hybrid LCA, which combines traditional process-based LCA techniques and combined economic input-output (EIO) LCA to estimate the environmental impact of a system.

These studies were conducted in a wide range of geographical locations, including the United States, Spain, UAE, and Australia. In the following sections, studies that quantified GHG emissions from both water reuse and desalination facilities are separated into two major categories associated with the technologies included in each study: (1) RO desalination and water reuse technologies and (2) unspecified technologies.

In addition, all of the studies are further separated by the focus of study. Table 2.6 provides an overview of literature reviewed. Refer to Chapter 4 for additional details on previously documented GHG emissions.

2.4.1 Reverse Osmosis Desalination and Water Reuse

The following sections provide a brief description of each study that compares RO desalination and water reuse technologies. For the studies that included RO desalination and water reuse technologies, process-based LCA and hybrid LCA methodologies were used to evaluate GHG emissions.

These studies found that the GHG emissions from seawater desalination were greater than emissions from water reclamation facilities. The carbon footprint of RO desalination facilities ranged from 0.4 to 6.7 kg CO_2eq/m^3, whereas the carbon footprint of water reuse facilities ranged from 0.1 to 2.4 kg CO_2eq/m^3 for different plant capacities and technologies. Specifics of these studies are provided in Chapter 4. In addition, operational phase electricity was the dominant contributor to the carbon footprint for both facility types. This highlights the importance of the electricity mix to a facility's overall GHG emissions. Both Lyons et al. (2009) and Stokes and Horvath (2009) investigated the impact of electricity mixes on GHG emissions and found that, as expected, mixes with more renewable sources had lower emissions.

Table 2.6. Overview of Literature Comparing Desalination and Water Reuse

Technology	Focus of Study	Capacity of Facility	Population Served	Tools Used	Reference(s)
Reverse osmosis desalination and water reuse alternatives	Evaluate water supply alternatives using LCA	Not provided	Not provided	Process-based LCA, SimaPro	Muñoz et al., 2009[1]; Lyons et al., 2009[1]; Muñoz et al., 2010[2]; de Haas et al., 2011[1]
	Evaluate water supply alternatives using hybrid LCA	98,630–109,589 m^3/day (26.0–28.9 MGD)	175,000–200,000	Hybrid LCA, WEST	Stokes and Horvath, 2006[2], 2009[2]
	Evaluate disinfection methods and urban reuse alternatives	2300 m^3/day (0.61 MGD)	5700	LCA, SiSOSTAQ UA, SimaPro	Pasqualino et al., 2010[1]; Meneses et al., 2010[1]
Unspecified	Water supply system	Not provided	Not provided	LCA, Gabi	Lundie et al., 2004[1]

Notes: 1=study that focuses on seawater desalination only; 2=study that focuses on both seawater and brackish water desalination; LCA=life cycle assessment; WEST=water energy sustainability tool

In California, Stokes and Horvath (2009) found that solar PV and solar thermal energy sources had a lower impact than various other energy mixes, including state, national, and renewable mixes with 35 to 36% renewable sources. Desalinated seawater with membrane pretreatment, for example, had a carbon footprint of 4.0 and 1.9 kg CO_2eq/m^3 for a U.S. average electricity mix and a European Union mix (35% renewable sources for electricity).

These studies also highlighted the benefit of water reclamation in minimizing GHG emissions. For example, Pasqualino et al. (2010) found that replacing desalinated water with reclaimed water had the greatest benefit in GHG reduction compared to no reuse, brine dilution, and potable water replacement.

In another study, Muñoz et al. (2009) investigated specific water reuse disinfection options to find that ozonation and ozonation with peroxide tertiary treatment methods were less energy intensive than seawater desalination during the operation phase. Ozonation and desalination GHG emissions were approximately 0.3 and 2 kg CO_2eq /m^3 (Muñoz et al., 2009).

Expanding on the work of Muñoz et al. (2009), Meneses et al. (2010) compared chlorination and UV disinfection to ozonation and ozonation with peroxide. They found that GHG emissions from water reuse facilities using ozonation and ozonation with peroxide were comparable to UV and chlorination disinfection options. All of the water reuse options had a lower carbon footprint than seawater desalination that employed RO (Meneses et al., 2010).

Other studies identified critical areas with high GHG emissions within both water reclamation and desalination facilities. Stokes and Horvath (2006), for example, determined that high electricity usage and replacement materials (e.g., RO membranes) led to high GHG emissions during the treatment phase of desalination facilities in California. As a result, the desalination treatment phase had a higher percent contribution (85%) to energy use than the collection or distribution phases. For two water reuse facilities, distribution was found to dominate energy use, at 61% and 74% of the water supply phases (Stokes and Horvath, 2006). This highlights the importance of considering the location of water reclamation facilities and proximity to intended reuse locations as well as the energy efficiency of treatment technologies.

In another study, Stokes and Horvath (2009) found that, all other factors being similar, seawater desalination with conventional pretreatment of coagulation/flocculation followed by filtration had slightly higher GHG emissions (2.5 kg CO_2eq/m^3) than seawater desalination with membrane pretreatment (2.4 kg CO_2eq/m^3) or brackish groundwater desalination (1.6 kg CO_2eq/m^3). Unfortunately, the TDS for the different water sources was not provided in this study. This highlights the importance of considering GHG emissions from the collection, treatment, and distribution phases, including emissions from electricity and material sources.

2.4.1.1 Studies Using LCA

This section provides a review of studies that compared RO desalination and water reuse technologies using LCA to assess water supply systems.

Muñoz et al. (2010), Life Cycle Assessment of Water Supply Plans in Mediterranean Spain: The Ebro River Transfer Versus the AGUA Programme. This study assessed two water supply systems in Spain using an LCA-based methodology. These systems include (1) the previously used ERWT, which transfers water using open-channel aqueducts and pumping stations from the Ebro River to the Barcelona area (up to 700 km distance), and (2) the currently implemented AGUA program, which combines seawater and brackish water RO desalination facilities, various advanced water reuse facilities (e.g., disinfection, UV, MF/RO), and groundwater and surface water treatment for urban and agricultural use. An optimistic and pessimistic condition for both water supply systems (from water abstraction to wastewater treatment) was evaluated because of uncertainty in the input parameters, particularly energy consumption.

This study found that under both optimistic and pessimistic conditions, the AGUA program (uses water reuse, desalination, and other water supplies) had a lower impact on GWP per cubic meter of water than ERWT (imported water option) because of lower energy and resource consumption. Under optimistic conditions, the AGUA program and ERWT had a GWP of 1.4 and 1.9 kg CO_2eq/m^3. The pessimistic condition resulted in a GWP of 2.3 and 2.5 kg CO_2eq/m^3 for the AGUA program and ERWT. This differs from a previous study (Raluy et al., 2005b) that determined that seawater desalination produced 40% more CO_2 emissions than importation from the Ebro River because of higher energy consumption.

In expanding the system boundary to consider other water supply systems within the AGUA program (e.g., water reclamation, surface water, and groundwater treatment), Muñoz et al. (2010) found that this combination of seawater desalination and less energy-intensive technologies produced lower yet comparable emissions to the importation option. It is important to note that energy consumption and GHG emissions for water reclamation and desalination technologies investigated by Muñoz et al. (2010) were on the same order of magnitude as results in the Stokes and Horvath (2009) study, and seawater desalination had the highest contribution to GHG emissions for the AGUA program.

Muñoz et al. (2009), Life Cycle Assessment of Urban Wastewater Reuse with Ozonation as Tertiary Treatment. In another study conducted in the same region, Muñoz et al. (2009) also used LCA to evaluate alternative treatment scenarios for agricultural irrigation in Spain. The alternatives considered included RO desalination with (1) no water reclamation (baseline), (2) direct reuse with no tertiary treatment, (3) reuse with ozonation as tertiary treatment, and (4) reuse with ozonation and peroxide for tertiary treatment. Experimental data related to ozonation treatment options were collected on effluent samples from a wastewater treatment plant in Madrid to evaluate GHG emissions and toxicity related impacts of reuse alternatives.

The study found that desalination had the highest impact (approximately 2 kg CO_2eq /m^3), whereas both ozonation alternatives for water reuse had comparable GHG emissions (approximately 0.3 kg CO_2eq/m^3). Consequently, despite implementing relatively energy-intensive disinfection technologies (e.g., ozonation or ozonation with peroxide), water reuse was still found to be 85% less GHG intensive than desalination options.

Lyons et al. (2009), Life Cycle Assessment of Three Water Supply Systems: Importation, Reclamation, and Desalination. Similar to Muñoz et al. (2010), this study investigates water supply alternatives, including desalination, importation, and water reclamation. The study used SimaPro 7.10 software to conduct an LCA-based study in Scottsdale, AZ, over the construction and operation life stages. For water reuse, MF, wastewater RO, advanced treatment, and aquifer storage and recovery were evaluated. For desalination, RO was evaluated.

Inventory air emissions revealed that CO_2 from the desalination was the highest (4.3 kg CO_2/m^3), followed by imported water and reclamation (1.0 kg CO_2/m^3). The operation and maintenance (O&M) phase was dominant for all three alternatives, and energy usage was the most prevalent contributor to the environmental impact. Based on these findings, the study recommended further research to make RO treatment technologies less energy intensive. Similar to Ortiz et al. (2007), Lyons et al. (2009) found that changing the electricity mix to a predominantly nuclear mix (French) or hydroelectric mix (Norwegian) could reduce the environmental impact.

de Haas et al. (2011), Life Cycle Assessment of the Gold Coast Urban Water System. In this study, de Haas et al. investigated traditional and future water supply systems in the Gold Coast region of Southeast Queensland. Whereas traditional water supply systems were based on existing water provision through dams, future water supply systems include large-scale seawater desalination facilities and various hypothetical water reuse schemes. By investigating traditional and future water supply systems over the construction and use phases, this study compares the environmental impact of both traditional and future scenarios using SimaPro LCA-based software.

The future water supply alternatives related to wastewater reclamation and desalination include (1) water reuse in which 100% of the wastewater goes through the advanced treatment plant regardless of overall demand, (2) demand-driven water reuse (advanced treatment is matched with demand), (3) indirect potable reuse (water from the advanced treatment plant is sent to the local reservoir), and (4) seawater desalination. Demand-side graywater capture through rainwater capture is also investigated. A distinction in this study is its consideration of fugitive CH_4 and N_2O emissions, which may contribute to a higher carbon footprint. These direct process emissions could be important for GHG mitigation efforts because they can be directly controlled through process modifications.

Future water supply systems (e.g., advanced water treatment plant for water reuse and RO for desalination options) were found to have a greater environmental impact than existing water supply infrastructure (e.g., existing water treatment plants, wastewater treatment plants, and biosolid recovery) in the majority of the impact categories, including GWP. Electricity consumption and chemical usage were the major contributors to the increased impact induced by seawater desalination and water reclamation options. According to de Haas et al. (2011), the option with the lowest GWP was demand-driven water reuse at 0.6 kg CO_2eq/m^3. This was followed by the indirect potable dam water provision option (0.7 kg CO_2eq/m^3), water reuse for which 100% of the wastewater is treated with an advanced treatment system (2.4 kg CO_2eq/m^3), and the seawater desalination option (4.4 kg CO_2eq/m^3). This study clearly showed that even the most sophisticated reuse scheme (100% wastewater treated via advanced treatment) produces less GHG emission than seawater desalination.

2.4.1.2 Studies Using Hybrid Life Cycle Assessment

This section provides a review of studies that compared RO desalination and water reuse technologies using hybrid-based LCA to analyze water supply alternatives.

Stokes and Horvath (2006), Life Cycle Energy Assessment of Alternative Water Supply Systems. In this study, a hybrid LCA method was used to estimate GHG emissions, energy consumption, and other air pollutants of different water supply options. Hybrid LCA combines process-based LCA and EIO-based LCA to assess the environmental impact of a system. Stokes and Horvath developed a water energy sustainability tool (WEST) to evaluate GHG emissions of water supply alternatives, including importation, water reclamation, and desalination. GHG emissions associated with supply, treatment, and distribution were assessed over the construction and operation life stages. Refer to Chapter 5 for further details on this GHG estimation method.

This research used the WEST model to conduct a case study in Northern and Southern California. The Marin Municipal Water District in Northern California uses coagulation, filtration, and disinfection processes for water reclamation, and flocculation, filtration, RO, and disinfection processes for desalination. The Oceanside Water Department in Southern California uses filtration processes for water reclamation and filtration, and RO and disinfection processes for treatment.

This study found that, for both sites, desalination had a carbon footprint two to three times greater than water reclamation and two to five times greater than imported water. Water reclamation was found to have the lowest environmental impact. The Northern California site treated seawater (influent TDS approximately 32,000 mg/L), whereas the Southern California site treated brackish water (influent TDS approximately 1500 mg/L). Consequently, the Northern California site had a higher GWP intensity because of higher influent salinity levels. This is consistent with other studies that found seawater desalination to have a higher carbon footprint than other brackish water sources (Peter and Rouse, 2005; Muñoz and Fernández-Alba, 2008). The operation phase was the greatest contributor to GHG emissions for all systems because of the production of energy, which had a 56 to 90% contribution. The production of materials (such as RO membranes and cartridge filters) was also a significant contributor to the GWP (30–44%), particularly for the desalination facilities.

Finally, the treatment phase was the dominant contributor to GHG emissions for the desalination facilities, accounting for 85% of the energy consumption. This differed from water reclamation, in which distribution had the highest GWP intensity and energy consumption compared to other water supply phases (treatment and collection). For the two facilities investigated, distribution accounted for 61% and 74% of the energy consumed. This highlights the importance of considering GHG emissions associated with the distribution for water reclamation projects and the treatment of desalination projects.

Stokes and Horvath (2009), Energy and Air Emission Effects of Water Supply. This study assessed hypothetical alternatives in Southern California to compare different types of desalination, imported water, and water reclamation. Different electricity mixes and a projected desalination case study in Dubai, UAE, for 2030 based on current growth conditions were also analyzed. The hypothetical scenarios considered in Southern California include (1) imported water, (2) seawater desalination with conventional pretreatment, (3) seawater desalination with MF/UF membrane pretreatment, (4) brackish groundwater desalination, and (5) reclaimed wastewater treatment. The six electricity mixes analyzed include California's average mix, the U.S. average mix, all solar PV generation, all solar thermal generation, EU 2020 mix (~35% renewable sources), and low emission mix (hypothetical renewables [~36%] combined with current California mix).

This study found that, under current conditions, water reclamation and imported water were comparable in terms of GHG emissions, at 1.1 and 1.0 kg CO_2eq/m^3. Seawater desalination with conventional pretreatment had similar GHG emissions (2.5 kg CO_2eq/m^3) compared to seawater desalination with membrane pretreatment (2.4 kg $CO_2eq/m^{3)}$. In contrast, brackish groundwater desalination was estimated to have lower emissions of 1.6 kg CO_2eq/m^3. The California electricity mix was found to have lower GHG emissions than the national U.S. average mix. The EU mix and low emission mix (hypothetical renewables with California mix) were lower than the California electricity mix. Solar PV and solar thermal energy mixes were found to have the lowest GHG emissions, and solar thermal was the recommended option because this alternative also decreased other air emissions (e.g., NO_x and particulate matter). The authors also demonstrated that when desalination was combined with solar thermal energy, as expected, it was found to have a lower carbon footprint than importing or reclaiming water (using a California electricity mix).

Future projections of the Dubai water supply system, which relies on desalination for 95% of its water supply, were used to obtain an international comparison of desalination. This comparison showed that Dubai GHG emissions from desalination were 1.6 times greater than the California scenario (Stokes and Horvath, 2009).

2.4.1.3 Disinfection Methods and Urban Reuse Alternatives

This section provides an overview of studies that sought to compare disinfection methods (e.g., chlorination, UV disinfection, ozonation, ozonation with peroxide) and urban reuse alternatives (e.g., brine dilution, desalination water replacement).

Pasqualino et al. (2010), Life Cycle Assessment of Urban Wastewater Reclamation and Reuse Alternatives. This study examined the environmental impact of a 2300-m^3/day (0.6 MGD) capacity water reclamation facility serving 5700 inhabitants in a tourist region of Catalonia, Spain, using SiSOSTAQUA, an LCA-based tool for environmental management. Four urban reuse alternatives were assessed, including no reuse, brine dilution, potable water replacement, and desalinated water replacement. The no reuse and brine dilution scenarios use primary and secondary treatment, whereas the potable water replacement and desalinated water replacement include additional tertiary treatment. The tertiary treatment system includes coagulation/flocculation, chlorination, sand filtration, and UV disinfection. In contrast to other studies, which focused on both construction and operation phases (e.g., Lyons et al., 2009; Muñoz et al., 2010; de Haas et al., 2011), this study focused solely on the operation phase.

The study found that using reclaimed water instead of desalinated water has the greatest environmental benefit in terms of GHG emissions. By replacing desalination water with reclaimed water, the GHG emissions avoided are greater than the GWP generated. Because more energy is required for desalination than water reclamation, replacing desalination water with reclaimed water reduces GHG emissions by 2.1 kg CO_2eq/m^3. Potable water replacement has a lower GWP (0.7 kg CO_2eq/m^3) than no reuse or brine dilution (0.8 kg CO_2eq/m^3), and the least favorable option is not reclaiming water. Compared to conventional primary, secondary, and sludge treatment, this study found that tertiary treatment had a minimal impact on the GHG emissions. Similar to Zhang et al. (2010), this study found that secondary treatment was the largest contributor to electricity consumption (Pasqualino et al., 2010).

Meneses (2010), Environmental Assessment of Urban Wastewater Reuse: Treatment Alternatives and Applications. In this study, Meneses et al. conducted research on disinfection options for the same Spanish facility investigated by Pasqualino et al. (2010). In this study, chlorination with UV disinfection was compared to ozonation and ozonation with peroxide. Disinfection by ozonation and ozonation with peroxide came from an experimental study by Muñoz et al. (2009), whereas chlorination with UV came from current operational data. In addition, water reclamation for both agricultural and urban use was compared to potable and desalinated water sources. Meneses et al. also conducted a sensitivity analysis to account for both seasonal variability, which changes the amount of chemical usage needed for treatment, and nitrogen content variability. The nitrogen content varies the amount of fertilizers avoided through agricultural reclamation. It is important to note that construction phase impacts are ignored because infrastructure for all the scenarios are similar, and previous studies found that the operation phase dominates the environmental impact (Raluy et al., 2006; Lyons et al., 2009).

This study found that disinfection using chlorination with UV treatment had a lower GWP than both ozonation and ozonation with peroxide options; however, all disinfection options were on the same order of magnitude, and differences arose primarily from energy usage. The GWP expressed as kg of CO_2 equivalents per cubic meter of nonpotable water application was lowest for agricultural reuse because of its fertilizer replacement potential at 0.1 kg CO_2eq/m^3. Desalinated water was found to have the highest GWP at 3.4 kg CO_2eq/m^3, followed by potable water at 0.3 kg CO_2eq/m^3 and urban reuse at 0.2 kg CO_2eq/m^3. Seasonal variability revealed that GHG emissions were lower in the winter because of better water quality and lower chemical needs for tertiary treatment. Variations in nitrogen content were found to have minimal impact on the GWP.

2.4.2 Different Types of Reuse and Desalination

This section reviews one paper that did not specify water reuse and desalination technologies evaluated but provides insight on the importance of integrated water management efforts to reduce GHG emissions with the water supply sector.

Lundie et al. (2004) Life Cycle Assessment for Sustainable Metropolitan Water Systems Planning. Lundie et al. conducted an LCA on a water supply system in Sydney, including water and wastewater systems for various facilities run by Sydney Water. Future scenarios were compared to a base-case scenario that uses current operational data modified to replicate 2021 conditions. Future scenarios included desalination, demand management initiatives aimed at reducing consumption, changes in population (both increase and decrease), energy efficiency techniques (e.g., energy efficient pumps and lighting), the generation of alternative electricity (e.g., hydroelectric and biogas), energy generation from biosolids, upgrades to wastewater treatment systems in coastal regions, and a greenfield scenario that included a wide range of integrated water management techniques (e.g., rainwater harvesting, localized agricultural irrigation, nutrient removal facilities, and biosolids treatment at neighborhood and regional scales).

For the base-case scenario, the study found that wastewater treatment facilities contributed 20 to 29% of the total CO_2 emissions from the entire water supply system. In addition, water reclamation and distribution contributed 2% of these emissions. If all of Sydney's future water demands were met through desalination, CO_2 emissions would increase by five times compared to the existing water supply system. Upgrades to secondary and tertiary coastal wastewater treatment facilities for reuse increased CO_2 emissions, but to a lesser extent. These upgrades increased CO_2 emissions by 21 to 23%, while reducing eutrophication by 8 to 10%. This demonstrates an environmental trade-off associated with wastewater treatment facility upgrades for water reclamation.

Demand management, energy efficiency, energy generation, and biosolids energy generation had varying levels of emission reductions ranging from 2 to 11% (from the base-case scenario), showing the benefit of various forms of demand- and supply-side mitigation efforts. This study also confirmed that an increase in population correlates with an increase in CO_2 emissions, whereas a decrease in population has the opposite effect. Finally, the greenfield scenario (includes water reuse and nutrient recovery) reduced the GHG emissions by 18% compared to the base-case scenario, illustrating the benefit of integrated water management techniques.

2.5 Review of Emission Models

During the literature review, various models were also identified that estimate GHG emissions. Model literature comes from a wide range of Websites, user manuals, gray literature, and personal communication with model developers. The models identified include LCA-based models (discussed throughout Chapter 2), hybrid LCA-based models, specific models, and other related models (discussed in Chapter 5). Whereas LCA-based models provide a generic framework for evaluating GHG emissions from water reuse and desalination facilities, hybrid LCA-based models are specifically designed for water, wastewater, water reuse, and desalination facilities. Hybrid LCA-based models include WEST, WWEST, and WESTWeb. WEST has previously applied to water reuse and desalination facilities (Stokes and Horvath, 2006, 2009). An in-depth review of the methodology used to estimate GHG emissions using hybrid-LCA based models is provided in Section 5.3.

Specific models include the Johnston (2011) and Tampa Bay Water (2011) models. These models use facility-specific information to estimate GHG emissions for either drinking water treatment facilities or entire water supply systems (e.g., groundwater, surface water, desalination). They also contain attributes applicable to water reuse and desalination facilities. The methodology used to estimate GHG emissions from these models is reviewed in Section 5.4.

Finally, other related models include a wide range of water and wastewater models with aspects applicable to water reuse and desalination (e.g., CHEApet, Environment Agency Model, Bridle Model, BSM2G Model, System Dynamics, GPS-X Model, Carbon Accounting Workbook, and mCO2). These models were deemed to have less applicability because of their focus on conventional water or wastewater treatment. In addition, some models are less applicable because of their use of data specific to a given geographical location. Despite having less applicability to water reuse and desalination facilities, certain aspects of these models are useful to water reuse and desalination facilities and are discussed in Section 5.5.

Chapter 3

Parameters Used in Previous Studies and Models

Analysis of available information required to estimate the GHG emissions of desalination and water reuse facilities is presented in this chapter. Life stages, system boundaries, emissions considered, and input parameters vary significantly among GHG estimation methods, creating differences in results. This presents a challenge to compare GHG emissions from different facilities. Therefore, this chapter focuses primarily on life stages and parameters considered in LCA-based studies, with some inclusion of other estimation techniques found within the literature.

The entire life cycle of facilities in carbon footprint or GHG emission analysis includes (1) construction, (2) O&M, and (3) decommission. During each life stage, three types of emissions can be included: (1) direct emissions from the various unit processes, (2) indirect emissions associated with energy consumption, and (3) indirect emissions associated with material consumption (i.e., indirect energy consumption). Within each of these categories, parameters representing specific materials, processes or actions can be designated and their associated GHG emissions calculated. Table 3.1 illustrates the relationships between life stages, emission source, and parameters considered that would ultimately impact GHG emissions.

This chapter provides a review of the life cycle stages considered in the studies reviewed, lists of parameters used to represent the energy-using or GHG-emitting activities in the development of a water reuse or desalination scenario, and the specific GHG values used to estimate GHG emissions. Most of the studies referred to in this chapter are either LCA or LCA-based. The first section (Section 3.1) summarizes the life cycle stages included in each study. Section 3.2 provides more detail, discussing the parameters within each life cycle stage in the reviewed studies. Section 3.3 presents the emissions used in each study, and Section 3.4 reviews studies that may not be LCA-based but rather hybrid or off-the-shelf models. More in-depth analysis on these types of models is provided in Chapter 6.

3.1 Life Cycle Stages Considered in Previous Studies

In estimating the overall environmental impact of desalination and water reuse, scenarios are divided into three categories based on life cycle stages: construction, O&M, and decommission. Construction refers to the energy and materials that are used in building the necessary infrastructure for water reuse and desalination projects. O&M is defined as the energy and materials used when the desalination plant or advanced water treatment facility (water reuse) is online. Decommission refers to the energy required in dismantling, disposal, and, if possible, recycling of the used materials in the water reuse or desalination scenario. This section summarizes the stages that are used in the reviewed studies and highlights any notable trends. Tables presenting the life cycle stages included in each of the reviewed studies can be found in Appendix B.

3.1.1 Desalination Studies

A review of LCA-based literature related to GHG emissions at desalination facilities showed that 5 of 10 total studies analyzed the emissions associated with all three life cycle stages; however, 4 studies focused primarily on the O&M stage. Because construction is a one-time event, if the life of the plant is long, the contribution to the overall environmental impacts of the implementation of desalination can be minimal. This was the reason for the omission of the plant construction stage by Biswas (2009).

3.1.2 Water Reuse Studies

The majority of water reuse studies focused solely on the O&M stage. This differs from the desalination research literature, in which most of the studies included construction, O&M, and decommission stages. According to Tangsubkul et al. (2005), the O&M phase is responsible for about 65 to 83% of the overall GHG when only the construction and O&M phases are considered. This shows that, depending on the technology, the construction phase is an important emitter. A few studies focused only on the O&M phase (Stillwell and Webber, 2010; Fine and Hadas, 2012; Shrestha et al., 2012).

Table 3.1. Life Cycle Parameters for Water Treatment Facilities

Life Cycle Stage	Activities	Parameters
Construction	Energy consumption	Electricity generation and consumption; fuel generation and consumption (on-site during construction); transport distance; mode of transport
	Material consumption	Types of construction materials; amount of construction materials; types of pipelines and reactors (material and size); length of pipelines and size of reactors
Operation and maintenance	Direct emissions	Raw water quality; treatment efficiency
	Energy consumption	Flow rate; electricity consumption (on-site; e.g., pumping and administrative requirements); pump efficiency; fuel use by delivery vehicles and equipment; equipment type, use amount and frequency; on-site power sources; administrative transportation (fleet vehicles, business travels)
	Material consumption	Types of treatment chemicals; frequency of material replacement; material service life and cost; treatment chemical dosage; amount of sludge; sludge disposal scenarios (% for land application, % for landfill); administrative service supplies, engineering service supplies
Decommission	Energy consumption	Lifetime of facility; electricity consumption of disposal facility; electricity consumption of recycle facility
	Material consumption	Disposal scenarios (% material recycle, % landfill)

3.1.3 Studies Comparing Water Reuse and Desalination

In contrast to studies focusing only on water reuse or desalination, the majority of studies (five out of nine) comparing facilities assessed construction and operation life cycle stages.

3.2 Parameters Used in Previous Studies

Within each life cycle stage, there are specific parameters that are used to represent the energy use or GHG-emitting activities in the development of a water reuse or desalination scenario. For example, in the construction stage, impacts typically are determined by the production and transport of materials used and the fuel use of the equipment used for construction. This section reviews the parameters included in each study.

The parameters defined in the construction stage are listed in Tables 3.2 through 3.4. “Fuel and Electricity” refer to the amount of energy used by construction equipment (e.g., bulldozers, cranes) during the building process. “Materials” is defined as the building materials used as well as the impacts of their production and transport to the vendor. “Materials Delivery” is the fuel used in transporting the materials from the place of distribution to the construction site.

Table 3.2. Life Cycle Parameters Used in Desalination Studies

Life Cycle Stage	Parameter	Raluy et al., 2004	Raluy et al., 2005a	Raluy et al., 2005b	Muñoz and Fernández-Alba, 2008	Biswas, 2009	Jijakli et al., 2011	Shrestha et al., 2011
Construction	Fuel and electricity			X	X		X	
	Materials	X		X	X		X	
	Materials delivery	X			X			
Operation and maintenance	Electricity consumption	X	X[1]	X[2]	X	X		X
	Fuel consumption							
	Chemicals				X	X		
	Piping/pumping						X	X
	Materials				X	X	X	
	Wastewater emissions							
Decommission	Demolition	X		X	X		X	
	Transport of waste				X			
	Disposal	X		X	X			

Table 3.3. Life Cycle Parameters Used in Water Reuse Studies

Parameter	Tangsubkul et al., 2005	Ortiz et al., 2007	Friedrich et al., 2009	Zhang et al., 2010	Stillwell and Webber, 2010	Shrestha et al., 2012	Fine and Hadas, 2012
Fuel and energy use			X	X			
Materials		X	X	X			
Materials delivery		X		X			
Electricity consumption	X		X	X	X	X	
Fuel consumption	X			X			
Chemicals	X		X	X			
Piping/pumping			X			X	X
Materials	X						
Wastewater emissions							X

Parameters for decommission included "Demolition" or dismantling of the analyzed infrastructure; "Transport of Wastes," which includes the transport of construction material wastes to a landfill or a recycling center, and "Disposal," which refers to the energy used in processing, storing, or recycling the wastes at the final destination. Tables 3.2 and 3.4 indicate which studies contain these parameters. Decommission parameters were not included in Table 3.3 because decommission data were not provided in any of the studies.

3.2.1 Desalination Studies

The studies that include the construction stage primarily defined it as the use of energy associated with the production and transport needed to produce the materials to make the desalination processes and facility, as can be seen in Table 3.2 (Raluy et al., 2004, 2005b). Two studies also included the parameter of terrain excavation prior to installation or construction of components for the studied desalination facilities (Muñoz and Fernández-Alba, 2008; Jijakli et al., 2012). Although not associated with the construction phase of the plant, Raluy et al. (2005a) also included the construction of the solar- or wind-power facilities (material inputs) that would be used to power the desalination plants in the different LCA scenarios.

Table 3.4. Life Cycle Parameters Used in Studies Comparing Water Reuse and Desalination

Life Cycle Stage	Parameter	Lundie et al., 2004	Stokes and Horvath, 2006	Lyons et al., 2009	Muñoz et al., 2009	Stokes and Horvath, 2009	Meneses et al., 2010	Muñoz et al., 2010	Pasqualino et al., 2010	de Haas et al., 2011
Construction	Fuel and electricity		X							
	Materials	X	X	X		X		X		
	Materials delivery					X				
Operation and maintenance	Electricity consumption	X	X	X	X	X	X	X	X	X
	Fuel consumption		X			X				
	Chemicals	X	X	X	X		X	X	X	X
	Piping/pumping							X		X
	Materials	X	X			X		X		
	Wastewater emissions				X					
Decommission	Demolition									
	Transport of waste						X			
	Disposal						X			

In addressing electricity and fuel use, the studies by Raluy et al. (2004, 2005a, 2005b) use the term "energy use" without specifying if the fuel consumption, in addition to electricity consumption, was included. It was inferred that the authors referred to electricity use, as in other studies (Biswas, 2009; Shrestha et al., 2011), because desalination processes rely mainly on electricity as their source of energy. A few studies also specified the materials used during the O&M stage because some processes may require a continual chemical or material input. For instance, chemical additive considerations such as anti-scaling agents and chlorine were considered in one study (Muñoz and Fernández-Alba, 2008). Biswas (2009) included the impacts of the production and transport of materials that needed to be frequently replaced, such as membranes. Although pumping affects electricity use, it was also included as a separate parameter because of its importance in water treatment and distribution (if included). This was the case for the study by Shrestha et al. (2011), which based its electricity use calculations on pumping for distribution throughout the Las Vegas Valley and from California (where the modeled desalination plant was located) to Nevada.

The studies that took into account the decommission stage included the energy and materials used to dismantle the plant. One study (Muñoz and Fernández-Alba, 2008) also included the energy used to transport the waste to a final impoundment facility.

3.2.2 Water Reuse Studies

Similar to the desalination research, water reuse studies that included the construction phase consisted mainly of the off-site production and transport energy of the materials used to build the water reuse or desalination infrastructure (Table 3.3); however, Ortiz et al. (2007) and Zhang et al. (2010) also specifically included the amount of fuel used to transport materials to the construction site. Although lacking detail, it could be inferred that Friedrich et al. (2009) incorporated the fuel and energy used during construction because the construction phase was described for a dam, which was not a component of the water reuse phase, but the same procedure may have been used for other structures such as the water recycling plant and the wastewater treatment facility, which were included.

Defined parameters that composed the O&M phase included the electricity used to power the specific processes, the pumping of water, and the chemicals added for treatment (Table 3.3). Electricity-based parameters such as pumping and other electricity use by the plant or system were linked to emissions using a national energy mix for countries like Australia and South Africa (Tangsubkul et al., 2005; Friedrich et al., 2009) or varying energy-mix scenarios (see Chapters 2 and 4; Ortiz et al., 2007). Several studies also specifically mentioned the energy associated with material and chemical usage during treatment process operation (Tangsubkul et al., 2005; Friedrich et al., 2009). Parameters characterizing the decommission stage have not been specified in the analyzed water reuse studies.

3.2.3 Studies Comparing Water Reuse and Desalination

Although there are five studies that include a construction phase, only a few provide details of the parameters used in calculating the environmental impacts (Table 3.4). Lyons et al. (2009) calculated the energy consumed per unit of concrete and steel used to build the infrastructure for the water reuse and desalination scenarios. Stokes and Horvath (2006) were more detailed, as the construction in the WEST model took into account not only materials production and transport to the proposed site of the water reuse or desalination infrastructure, but also the fuel use by the construction equipment, as well as the production of fuel needed for construction. Because Stokes and Horvath (2011a) produced a hybrid LCA GHG estimation model (WWEST), a more in-depth discussion on this model is included in Section 6.2.

Parameters included in the O&M stages include electricity and fuel use, the production and transportation of chemicals and materials used for water reuse, desalination, or both, and the pumping energy requirement. It is not surprising that all studies included electricity use considerations in their O&M emissions calculations, whereas most included the production and transport of chemicals used during treatment. Pasqualino et al. (2011) created a table that not only outlined the electricity consumption of every treatment step but also the amount of chemicals and materials that would be consumed per functional unit of water produced. The scope of the study also played a role, as the studies by Muñoz et al. (2010) and Lyons et al. (2009) include the materials and chemicals for the construction of pipelines as well as the energy needed to pump water to users. Fine and Hadas (2012) took into account the direct CO_2 emissions from the wastewater due to the concentrations of COD. One study (Meneses et al., 2010) included disposal in its estimation of environmental impacts; ;however, specific data were not given on the parameters included.

3.3 Types of Emissions Considered in Previous Studies

Once the parameters are defined and quantified, the emissions can be calculated. CO_2 is the most prominent GHG emission, and it is used exclusively in many studies as a measure of how much a scenario will contribute to global climate change; however, there are other GHGs that, although not emitted in as great amounts as CO_2, can exert a higher GWP per unit emitted compared to CO_2. These include CH_4 and N_2O. The GHG measurement unit CO_2eq is the sum of weighted values for CO_2, CH_4, and N_2O. Because CO_2eq accounts for multiple GHGs and not only CO_2, the measurement provides a more accurate value for overall carbon footprint.

3.3.1 Desalination Studies

Most desalination studies reviewed herein focus exclusively on CO_2 emissions (Table 3.5). One exception was Biswas (2009), who also estimated emissions of CH_4 and N_2O; however, the results from that study showed that CO_2 emissions were much higher. For example, the RO stage of the Southern Seawater Desalination Plant in Australia was reported to be responsible for 2.9 kg CO_2/m^3 water produced, while emissions values for CH_4 and N_2O were 0.02 and 0.005 kg/m^3 water produced.

Table 3.5. Reported GHG Emissions in Desalination Studies

Study	CO_2	CH_4	N_2O	CO_2eq
Raluy et al., 2004	X			
Peters and Rouse, 2005	X			
Raluy et al., 2005a	X			
Raluy et al., 2005b	X			
Raluy et al., 2006	X			
Muñoz and Fernández-Alba, 2008	X			
Biswas, 2009	X	X	X	X
Jijakli et al., 2011	X			
Shrestha et al., 2011	X			

Notes: CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalents; N_2O=nitrous oxide

3.3.2 Water Reuse Studies

Table 3.6 shows the emissions used to represent contributions to the literature that investigated GHG emissions during water reuse scenarios. The table shows that most studies use CO_2 as the primary emission factor; however, two studies report GHG emissions only as CO_2eq (Fine and Hadas, 2012; Shrestha et al., 2012).

3.3.3 Studies Comparing Water Reuse and Desalination

In contrast to the previous two sections, the majority of water reuse and desalination studies reported GHG emissions in CO_2eq (Table 3.7); however, Lyons et al. (2009) estimated CH_4 emissions in addition to CO_2 emissions. The emissions category "Nitrogen Oxides" was also included in the study, yet the study did not specify whether N_2O emissions were also considered within that category.

Table 3.6. Reported GHG Emissions in Water Reuse Studies

Study	CO_2	CH_4	N_2O	CO_2eq
Tangsubkul et al., 2005	X			
Ortiz et al., 2007	X			
Friedrich et al., 2009	X			
Stillwell and Webber, 2010	X			
Shrestha et al., 2012				X
Fine and Hadas, 2012				X

Notes: CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalents; N_2O=nitrous oxide

Table 3.7. Reported GHG Emissions in Studies Comparing Water Reuse and Desalination

Study	CO_2	CH_4	N_2O	CO_2eq
Lundie et al., 2004	X			
Stokes and Horvath, 2006				X
Lyons et al., 2009	X	X		
Muñoz et al., 2009				X
Stokes and Horvath, 2009				X
Meneses et al., 2010				X
Muñoz et al., 2010				X
Pasqualino et al., 2010				X
de Haas et al., 2011				X

Notes: CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalents; N_2O=nitrous oxide

3.4 Parameters Used in Reviewed Emission Models

Several studies reviewed integrate LCA) and input–output tables (hybrid LCA) or were off-the-shelf models that are able to calculate the GHG emissions of water reuse and desalination scenarios. The parameters used in these GHG calculation methods are explained in depth in Section 6.3.

Chapter 4

Previously Documented Emissions

4.1 GHG Emission Sources

In 2010, approximately 6820 MMT of GHGs (as CO_2eq) was produced in the United States. The composition of these emissions included 5710 MMT of CO_2, 670 MMT of CH_4, and 310 MMT of N_2O. Of the total annual CO_2 production, about 94% of emissions was due to combustion of fossil fuels. Within this category, electricity generation and transportation were the two highest contributors, responsible for 40% and 31% of overall CO_2 emissions. For other GHGs, natural gas systems (32%) and agricultural soil management (68%) were the most significant sources of CH_4 and N_2O (USEPA, 2012a). Although declining from a peak value of 7250 MMT CO_2eq in 2007, when compared to 1990s values, GHG emissions have increased about 10%.

Contributing to the rising GHG emissions is the implementation of desalination and water reuse. These GHGs come from the material and energy uses associated with treatment technologies applied for the purpose of water reuse or the specific unit processes used in a desalination plant over the three life cycle stages (construction, O&M, and decommission).

Electricity use is crucial in the provision of reclaimed or desalinated water. Electricity is consumed by pumping water as well as by the operation of certain unit processes such as ozonation and RO. According to Stokes and Horvath (2009), direct electricity use is responsible for about 45% of the total life cycle energy used in water reuse, whereas for desalination the percentage is between 39 and 45%. This energy use could translate into GHG emissions because of the amount of oil and coal combustion plants that compose the electricity source mix.

Fuel use is also necessary for the implementation of water reuse and desalination schemes. To carry out construction of necessary infrastructure, materials need to be transported from their place of production to the construction site. When the water reuse or desalination infrastructure is in operation, chemicals such as coagulants, chlorine, and ozone (for ozonation) have to be transported from their production facility to where they will be used in treatment. Transportation of these chemicals and materials will result in GHG emissions through the combustion of fossil fuels.

During the construction and operation phases, chemicals and materials are required for construction of infrastructure, use as additives for water treatment, or use as equipment within unit processes, such as a membrane for RO. Processing of these materials may directly emit GHGs or use fuel or electricity, which may indirectly be responsible for GHG emissions. In fact, cement and ferroalloy production are significant sources of CO_2, responsible for the emission of 30.5 and 1.7 MMT in 2007 (USEPA, 2012a).

4.2 CO_2 and GHG Emissions Associated with Desalination

This section discusses the influence of factors such as location of facility, capacity of the desalination facility, desalination technology, and the energy production source responsible for providing electricity to the facility.

Few papers have quantified emissions associated with specific unit processes at a desalination plant. As discussed in Chapter 2, Biswas (2009) conducted an LCA-based study to quantify the amount of GHG emissions emitted by an RO desalination facility. In addition to the types of energy used by the system, emissions generated by various unit processes were evaluated. Processes considered in the study included extraction, pretreatment, RO treatment, posttreatment, and water delivery/waste treatment. GHG emissions were 0.3 kg CO_2eq/m^3 for extraction of seawater, 0.06 kg CO_2eq/m^3 for membrane-based pre-treatment, 2.9 kg CO_2eq/m^3 for RO treatment, 0.05 kg CO_2eq/m^3 for chemical posttreatment, and 0.6 kg CO_2eq/m^3 for water delivery/waste treatment. Of the five processes, RO was shown to contribute the most to GHG emissions, accounting for 75%.

4.2.1 Location of Desalination Facilities and Scenarios

Studies that examine the GHG emissions of desalination have taken place mostly in water-scarce regions such as Spain (Raluy et al., 2004, 2005a; Muñoz and Fernández-Alba, 2008; Muñoz et al., 2010), UAE (Jijakli et al., 2011), Australia (Peters and Rouse, 2005; Biswas, 2009; de Haas et al., 2011), and the southwestern United States (Lyons et al., 2009; Stokes and Horvath, 2009). Table 4.1 lists the countries where these desalination studies took place and their corresponding CO_2 and GHG emissions ranges. CO_2 emissions ranged from 0.08 to 34.7 kg CO_2/m^3, and GHG emissions ranged from 0.4 to 6.7 kg CO_2eq/m^3.

Table 4.1. Desalination CO_2 and GHG Emissions Organized by Facility Location

Location	CO_2 Emissions (kg CO_2/m^3)	Reference(s)	GHG Emissions (kg CO_2eq/m^3)	Reference(s)
Spain	0.08[1]–34.7	Raluy et al., 2004, 2005a	1.1–1.9	Muñoz and Fernández-Alba, 2008
UAE	0.8–3.2	Jijakli et al., 2011	not provided	Not provided
Australia	not provided	not provided	2.0–6.7	Peters and Rouse, 2005; Biswas, 2009; de Haas et al., 2011
U.S.	4.3	Lyons et al., 2009	0.4–4.0	Stokes and Horvath, 2009

Notes: 1=Extremely low value refers to Norwegian energy mix even though study was conducted by a Spanish research group; UAE=United Arab Emirates

Regarding CO_2 emissions, the Spanish studies had a large range of emissions, from 0.08 to 34.7 kg CO_2/m^3. This variation in CO_2 emission values is attributed to the use of different national energy mixes in providing electricity to desalination plants, as discussed in Raluy et al. (2004). For instance, comparisons were made between Norwegian, French, Portuguese, and European Union energy mixes. For each technology (MED, MSF, and RO), this hypothetical connection to the Norwegian energy grid yielded the lowest CO_2 emissions (0.08–0.3 kg CO_2/m^3) compared to other national energy mixes. Also, Raluy et al. (2004, 2005a) made comparisons between distinct energy sources such as oil, natural gas, and coal as well as renewable sources, including PV, solar thermal, and wind. Desalination plants that relied exclusively on coal energy were responsible for the highest CO_2 emissions (34.7 kg CO_2/m^3).

Two other countries reported CO_2 emissions: the United States and the UAE. The UAE study (Jijakli et al., 2011) had lower emission values (0.8–3.2 kg CO_2/m^3) than the U.S. study (4.3 kg CO_2/m^3; Lyons et al., 2009). The relatively low CO_2 emission values of the UAE study are probably due to the fact that two of the three desalination scenarios were solar-powered (PV), whereas the highest value was a desalination scenario powered by natural gas. The study by Lyons et al. (2009) used the U.S. energy mix to estimate emissions from electricity sources. According to this study, 80% of electricity produced in the United States comes from fossil fuels; however, there are differences between the boundaries of the studies. Although both studies include transport to a city (Lyons et al., 2009) or storage tank (Jijakli et al., 2011), the desalination scenario investigated by Lyons et al. (2009) included the energy used in transporting water 235 km from California to Phoenix, AZ. This accounted for about 43% of the total energy use of the scenario.

Studies that analyzed GHG showed emissions ranging from 0.4 to 6.7 kg CO_2eq/m^3. The range was the lowest for the United States (0.4–4.0 kg CO_2eq/m^3) and the highest for Australia (2.7–6.7 kg CO_2eq/m^3). Spanish values were situated in the middle, yet were relatively low (1.1–1.9 kg CO_2eq/m^3). The variation in the U.S. GHG values is explained by the comparison of different energy mixes (Stokes and Horvath, 2009). The lowest value (0.4 kg CO_2eq/m^3) corresponds to the use of solar thermal energy production to provide electricity to brackish water desalination plants that use membrane pretreatment. The total CO_2 emissions of a seawater desalination plant using membrane pretreatment that is connected to the U.S. energy grid yields the highest value at 4.0 kg CO_2eq/m^3. This contrast highlights the importance of water source and energy mix. The level of energy required to treat brackish water is much lower than that for seawater because the level of salts in the source water in the case of brackish water desalination is often a fraction of the 30,000 to 40,000 TDS found in seawater.

The Spanish and Australian studies, on the other hand, mainly used national energy mixes. For instance, Muñoz and Fernández-Alba (2008) based their electricity use on the Spanish electricity mix, whereas the studies that took place in Australia primarily used nationwide Australian energy mix data for electricity provision. The scope of the study could also be an important factor, as Muñoz and Fernández-Alba (2008) estimated only the emissions associated with the desalination plant, excluding distribution to users or a reservoir, which was included to some degree in other studies (Peters and Rouse, 2005; Biswas, 2009; Stokes and Horvath, 2009; de Haas et al., 2011).

Table 4.2. Desalination CO_2 and GHG Emissions Organized by Facility Capacity

Capacity (MGD)	CO_2 Emissions (kg CO_2/m^3)	Reference(s)	Capacity (MGD)	GHG Emissions (kg CO_2eq/m^3)	Reference(s)
0.0003	0.8–3.2	Jijakli et al., 2011	2.0 & 2.6	1.2–6.7	Peters and Rouse, 2005; Stokes and Horvath, 2006
12	0.2–3.2	Raluy et al., 2005b	5.3 & 7.2	1.1–2.4	Muñoz and Fernández-Alba, 2008; Stokes and Horvath, 2006
324	4.26	Lyons et al., 2009	26 & 33	0.4–4.4	Stokes and Horvath, 2009; de Haas et al, 2011

Note: GHG=greenhouse gas

4.2.2 Desalination Capacity

All reviewed desalination studies were categorized based on the capacities of the specific desalination plants or scenarios. These capacities ranged from 0.0003 to 324 MGD, with most studies performed on plants with a capacity between 1 and 33 MGD. As can be seen in Table 4.2, desalination capacities were divided into three categories: less than 5 MGD (Peters and Rouse, 2005; Stokes and Horvath, 2006; Jijakli et al., 2011), 5 to 15 MGD (Raluy et al., 2005a, 2005b; Stokes and Horvath, 2006; Muñoz and Fernández-Alba, 2008), and greater than 15 MGD (Lyons et al., 2009; Stokes and Horvath, 2009; de Haas et al., 2011). Values were obtained for studies that looked exclusively at CO_2 emissions and GHG emissions (reported as CO_2eq). CO_2 emissions ranged from 0.2 to 4.3 kg CO_2/m^3, while GHG emissions ranged from 0.4 to 6.7 kg CO_2eq/m^3.

For CO_2 emission studies, the range of values for desalination plants or scenarios with a capacity of 12 MGD (0.2–3.2 kg CO_2/m^3) is characterized by a minimum emission value that is lower than the minimum normalized emission value from plants that have a capacity of 0.003 MGD (0.8–3.2 kg CO_2/m^3), if both groups have the same maximum emission values. However, the study by Lyons et al. (2009), which estimates the CO_2 emissions of a 324-MGD desalination plant, determined a value of 4.3 kg CO_2/m^3, which is higher than the upper limits of the CO_2 emission ranges of the smaller capacity studies. This might be caused by the fact that the 324-MGD desalination plant is located about 265 km from Phoenix, where the water will be distributed. Transport for this distance is responsible for about 43% of the overall energy used, which would significantly affect the amount of total emissions.

Similar trends can be seen in Table 4.2 for GHG emission studies when comparing capacities less than 5 MGD and those between 5 and 15 MGD. The range, corresponding to plants, of scenarios with capacities of 2.0 and 2.6 MGD (1.2–6.7 kg CO_2eq/m^3) is slightly higher than the range of values associated with capacities of 5.3 and 7.2 MGD (1.1–2.4 kg CO_2eq/m^3). There is a wider range of GHG emission values that correspond to desalination scenarios with capacities of 26 and 33 MGD, varying from 0.4 to 4.4 kg CO_2eq/m^3.

4.2.3 Desalination Technologies

From the reviewed literature, four distinct known desalination technologies have been identified. They include MED, MSF, RO, and solar still technology (Table 4.3). Most of the studies reported emissions as kg CO_2/m^3, thus mainly focusing on CO_2 emission as opposed to the total GHG emissions. The studies by Raluy et al. (2004, 2005a, 2005b, 2006) specifically compared the MED, MSF, and RO technologies.

The most notable difference is the relatively lower range of values for RO than the other three technologies. A complete analysis of studies with respect to technologies shows that CO_2 emissions for RO vary between 0.08 and 4.3 kg CO_2/m^3, which is lower than MED (0.265–26.91 kg CO_2/m^3) and MSF (0.3–34.7 kg CO_2/m^3). When comparing the three desalination technologies analyzed by Raluy et al. (2004, 2005a, 2005b, 2006), RO is associated with the least emissions, followed by MED, and then MSF (although both MED and MSF share similar minimum emission values). For a fair comparison with solar still technology, however, which in the study by Jijakli et al. (2011) was powered by PV solar energy, the emission values for PV-powered RO, MED, and MSF technologies were used. Nevertheless, solar still technology emitted the highest amount of CO_2 of all four technologies, with an emission of 3.1 kg CO_2/m^3, in contrast to the 0.3 to 1.0 kg CO_2/m^3 emitted by PV-powered RO, 0.4 to 1.1 kg CO_2/m^3 emitted by PV-powered MSF, and 0.4 to 0.7 kg CO_2/m^3 emitted by PV-powered MED. This comparison might not be fair because of the fact that the solar still scenario includes pumping from a brackish groundwater table, whereas the other three technologies do not take pumping from the source into account.

Table 4.3. Desalination CO_2 and GHG Emissions Organized by Technology

Technologies/ Processes	CO_2 Emissions (kg CO_2/m^3)	Reference(s)	GHG Emissions (kg CO_2eq/m^3)	Reference(s)
Multi-effect distillation	0.3–26.9	Raluy et al., 2004, 2005a, 2006	Not provided	Not provided
Multistage flash	0.3–34.7	Raluy et al., 2004, 2005a, 2006	Not provided	Not provided
Reverse osmosis	0.08–4.3	Raluy et al., 2004, 2005a; 2005b, 2006; Lyons et al., 2009; Jijakli et al., 2011	0.4–6.7	Peters and Rouse, 2005; Stokes and Horvath, 2006; Muñoz and Fernández-Alba, 2008; Biswas, 2009; Stokes and Horvath, 2009; Beery et al., 2010; de Haas et al., 2011
Solar still	3.1	Jijakli et al., 2011	Not provided	Not provided

RO was the only technology that reported results in kg CO_2eq/m^3; however, the variation in emission values was mainly due to the study by Stokes and Horvath (2009). This study compared desalination pretreatment technologies, water sources, and energy mixes. For instance, seawater desalination with membrane pretreatment emitted about 3% more GHGs than desalination with conventional pretreatment. Brackish water desalination (0.4–2.5 kg CO_2eq/m^3) showed no difference from seawater desalination (0.5–4.0 kg CO_2eq/m^3) based on lower-limit values; however, upper-limit values show that brackish water emits less than seawater desalination. The discrepancy is most likely based on reference energy mixes used in the studies. A desalination plant powered by solar thermal energy emitted the least GHG (0.4–0.5 kg CO_2eq/m^3), and the highest GHG emissions were associated with a plant powered by the U.S. energy mix (2.5–4.0 kg CO_2eq/m^3).

4.2.4 Energy Mix and Energy Source

Energy source is an important contribution to GHG emissions. Most of the processes in desalination plants use electricity. Therefore, the amount of GHG emitted depends on the local energy mix of the area where the desalination plant is sited. Previous studies have compared the CO_2 and GHG emissions associated with desalination plants powered by different national energy mixes as well as different energy sources (Raluy et al., 2004, 2005a, 2005b, 2006; Stokes and Horvath, 2006, 2009). Table 4.4 presents the range of values obtained for the specific energy mix or energy source.

Table 4.4. Desalination CO_2 Emissions Organized by Energy Mix and Source

Energy Mix	CO_2 Emissions (kg CO_2/m^3)	Reference(s)
European Union mix	1.1–2.0	Raluy et al., 2004, 2005b, 2006
French mix	0.5–0.7	Raluy et al., 2004, 2005a
Norwegian mix	0.08–0.3	Raluy et al., 2004, 2005a, 2005b
Portuguese mix	1.8–3.3	Raluy et al., 2004, 2005a
Spanish mix	0.9–2.4	Raluy et al., 2004, 2005a, 2006
U.S. mix	4.3	Lyons et al., 2009
Coal	1.2–35.0	Raluy et al., 2004
Waste heat driven	0.3–2.0	Raluy et al., 2005a
Hydroenergy	0.08	Raluy et al., 2005a
Natural gas	1.2–23.4	Raluy et al., 2004
Oil	1.8–30.5	Raluy et al., 2004
Photovoltaic	0.3–0.9	Raluy et al., 2005b; Jijakli et al., 2011
Solar thermal	8.2–11.0	Raluy et al., 2005b
Wind	0.01–0.2	Raluy et al., 2005b

National energy mix has a very large impact on the amount of GHG emissions associated with desalination. CO_2 emissions range from 0.08 to 3.3 kg CO_2/m^3, which is a 40-fold increase. When comparing these values, the lowest emissions are associated with the Norwegian energy mix, which relies on 99% hydropower, whereas the highest energy mix is the Portuguese mix, which is dominated (81%) by fossil fuels (Raluy et al., 2006).

Similar to national energy mix, energy source also plays an important role in the amount of CO_2 emitted. Raluy et al. (2004, 2005a, 2005b) compared the CO_2 emissions of desalination plants exclusively powered by coal, natural gas, oil, wind, and solar (PV and thermal). Results highlighted the importance of energy source in the implementation of desalination technology. For instance, an RO desalination plant exclusively powered by hydropower was estimated to emit 0.08 kg CO_2/m^3, whereas an MSF plant powered by a non-integrated coal plant was estimated to emit about 35.0 kg CO_2/m^3. When comparing fossil fuel sources, natural gas tends to emit the least (1.2–23.4 kg CO_2/m^3), and coal has the highest emissions (1.2–34.7 kg CO_2/m^3). Not surprisingly, renewable sources (hydropower, PV, solar thermal, and wind) have lower GHG emissions (0.08–11.0 kg CO_2/m^3) than fossil fuel–powered desalination plants (1.2–35.0 kg CO_2/m^3). Nevertheless, even within different energy sources, power plant configuration can have an impact on the CO_2 emissions. For instance, a hybrid coal plant with a combined cycle emits about 1.2 kg CO_2/m^3, which is drastically less than a non-integrated coal plant (35.0 kg CO_2/m^3).

Table 4.5 shows the impact of energy mix or energy source on life GHG emissions. Overall, the lowest value (0.4 kg CO_2eq/m^3) corresponded to solar thermal energy production, and the highest value was associated with the U.S. energy mix (4.0 kg CO_2eq/m^3). Although mentioned in a different study, the high GHG emissions associated with the U.S. energy mix might be due to the predominance of fossil fuel sources, as they are responsible for 80% of electricity production (Lyons et al., 2009)

Table 4.5. Desalination GHG Emissions Organized by Energy Mix and Source

Energy Mix	GHG Emissions (kg CO_2eq/m^3)	Reference(s)
European Union mix 2020	1.3–1.9	Stokes and Horvath, 2009
Australian mix	4.0	Biswas, 2009
Spanish mix	1.1–1.9	Muñoz and Fernández-Alba, 2008
California mix	1.2–2.5	Stokes and Horvath, 2006, 2009
U.S. mix	2.5–4.0	Stokes and Horvath, 2009
Photovoltaic	0.6–0.7	Stokes and Horvath, 2009
Solar thermal	0.4–0.5	Stokes and Horvath, 2009
Low emissions	1.4–2.1	Stokes and Horvath, 2009
Wind	0.4	Biswas, 2009

Comparisons between supranational (European Union), national, and state energy mixes show that the Spanish mix has the lowest amount of associated GHG emissions (1.1 kg CO_2eq/m^3). This is partially due to the scope of the study by Muñoz and Fernández-Alba (2008), which includes only the desalination plant, thus excluding distribution, which is accounted for in the studies by Stokes and Horvath (2009) and to a lesser degree by Biswas (2009). Also of note are the GHG emission values for the California mix, which are about 39 to 52% lower than those associated with the U.S. mix. According to Stokes and Horvath (2009), the difference is due to the higher contribution of renewable energy sources in the California grid.

Comparisons across energy sources highlight the relatively low GHG emissions from wind energy production (0.4 kg CO_2eq/m^3) compared to solar energy (PV and solar thermal), whereas the "low emissions" energy production scenario yielded the highest GHG emission value (2.1 kg CO_2eq/m^3). This electricity mix comprised renewable and existing California electricity sources; however, the authors did not give any details on the exact composition of this mix. One detail worth noting is that solar thermal GHG emissions (0.4–0.5 kg CO_2eq/m^3) are lower than PV emissions (0.6–0.7 kg CO_2eq/m^3). This comparison is the opposite of that in the CO_2 emissions studies (Table 4.4), in which PV emits more than 10 times less CO_2 than solar thermal electricity production. One possible explanation for the differences is that this information was obtained from three different studies, which most likely have estimated PV and solar thermal production and operations emissions rates based on different data sources (Raluy et al., 2005b; Stokes and Horvath, 2009; Jijakli et al., 2011).

4.3 CO_2 and GHG Emissions Associated with Water Reuse

This section discusses CO_2 and GHG emissions from the water reuse studies reviewed in this report to determine the effects of location of facilities or case study scenarios, capacity of facilities or case study scenarios, treatment technology applied for the purpose of water reuse, and national energy mix used to provide electricity.

No studies were identified that quantified emissions of specific unit processes associated with tertiary treatment for water reuse treatment, such as (1) phosphorus removal processes (e.g., chemical precipitation, biological); (2) residual suspended solids removal processes (e.g., MF, depth filtration, surface filtration, dissolved air flotation); (3) residual colloidal solids removal (e.g., MF, UF); (4) residual dissolved solids removal (e.g., nanofiltration, RO, electrodialysis); and (5) specific trace constituent removal (advanced oxidation, carbon adsorption, ion exchange, lime softening). Pasqualino et al. (2010) reported that GHG emissions for assumed tertiary treatment stages were for coagulation (0.0195 kg CO_2eq/m^3), flocculation (0.02 kg CO_2eq/m^3), prechlorination (0.03 kg CO_2eq/m^3), sand filtration (0.02 kg CO_2eq/m^3), UV disinfection (0.03 kg CO_2eq/m^3), and post-chlorination (0.04 kg CO_2eq/m^3). In that study, the authors concluded that these types of tertiary treatment only accounted for 2% of the total carbon footprint when compared to primary, secondary, and sludge treatment.

4.3.1 Location of Facilities and Scenarios

Location-based water reuse studies generally have been in areas known for water scarcity issues, such as Australia (Lundie et al., 2004; Tangsubkul et al., 2005; de Haas et al., 2011), Israel (Fine and Hadas, 2012), South Africa (Friedrich et al., 2009), Spain (Ortiz et al., 2007; Muñoz et al., 2009; Meneses et al., 2010; Pasqualino et al., 2010), and the United States (Lyons et al., 2009; Stokes and Horvath, 2009). Studies typically are based on existing water reuse schemes (Lundie et al., 2004; Friedrich et al., 2007, 2009; Meneses et al., 2010;

Pasqualino et al., 2010), proposed water reuse scenarios using effluent from existing wastewater treatment plants within the country of study (Ortiz et al., 2007; de Haas et al., 2011; Fine and Hadas, 2012), or completely hypothetical designs (Tangsubkul et al., 2005; Fine and Hadas, 2012). Most studies, with the exception of Ortiz et al. (2007), based their electricity sources on national or regional energy mixes. Table 4.6 compares CO_2 and GHG values by location.

The CO_2 emissions ranged from 0.1 to 1.1 kg CO_2/m^3. The Spain-based studies have the widest range (0.1–1.1). This is mainly due to the study by Ortiz et al. (2007), which compared the emissions of different water reuse technologies using four different national energy mixes (French, Norwegian, European Union, and Portuguese). For instance, an immersed MBR connected to the Norwegian energy grid was responsible for the low CO_2 emissions value of 0.1 kg CO_2/m^3, and a Portuguese mix–powered CAS system followed by filtration treatment had the highest emission value of 1.1 kg CO_2/m^3. Also, there were differences in the CO_2 emission ranges of studies based in Australia, South Africa, and the United States. South Africa had the lowest emissions, ranging from 0.1 to 0.7 kg CO_2/m^3, followed by Australia (0.4–0.8 kg CO_2/m^3) and the United States (1.0 kg CO_2/m^3). These studies used their corresponding national energy mix data to estimate emissions from electricity. Lundie et al. (2004) and Lyons et al. (2009) specified the compositions of their corresponding national energy mixes (91% coal and 80% fossil fuels); however, the U.S. CO_2 emissions (1.0 kg CO_2/m^3) were higher than the Australian range (0.4–0.8 kg CO_2/m^3) even though the New South Wales energy mix has a higher dependence on coal.

Boundaries could be an important issue, as the lower value of the study by Lundie et al. (2004; 0.4 kg CO_2/m^3) takes into account only the treatment of wastewater prior to reclamation, whereas the higher value (0.8 kg CO_2/m^3), which is closer to the value estimated by Lyons et al. (2009; 1.0 kg CO_2/m^3) also includes distribution of reclaimed water. Another factor is the incorporation of an RO tertiary step in Lyons et al. (2009). This is in contrast to Lundie et al. (2004), who assume a conventional wastewater treatment without tertiary treatment will produce a reusable effluent for nonpotable applications.

Table 4.6. Water Reuse CO_2 and GHG Emissions Organized by Facility Location

Location	CO_2 Emissions (kg CO_2/m^3)	Reference(s)	GHG Emissions (kg CO_2eq/m^3)	Reference(s)
Australia	0.4–0.8	Lundie et al., 2004	0.5–2.4	Tangsubkul et al., 2005; de Haas et al., 2011
Israel	Not provided	Not provided	2.1	Fine and Hadas, 2012
South Africa	0.1–0.7	Friedrich et al., 2009	Not provided	Not provided
Spain	0.1–1.1	Ortiz et al., 2007	-2.1–0.7	Meneses et al., 2010; Muñoz et al, 2010; Pasqualino et al., 2010
United States	1.0	Lyons et al., 2009	0.1–1.7	Stokes and Horvath, 2006, 2009

Total GHG emissions were reported for water reuse scenarios in Spain, Australia, Israel, and the United States. GHG emissions were higher for Australia, ranging from 0.5 to 4.0 kg CO_2eq/m^3, while Spain's values were lower at -2.1 to 0.7 kg CO_2eq/m^3. The negative value was due to the calculation of avoided CO_2 as Pasqualino et al. (2010) modeled a scenario in which reused water would prevent the use of desalinated water. This difference was most likely due to the scope of the studies; the study by de Haas et al. (2011), which estimated a GHG emissions range of 0.6 to 2.4 kg CO_2eq/m^3, had more extended boundaries that included conveyance to communities, whereas the Spanish studies focused only on the operations within the WWTP (Pasqualino et al., 2010) or exclusively the tertiary treatment technology (Meneses et al., 2010; Muñoz et al., 2010). Nevertheless, a more fair comparison could be made between Australian and U.S. GHG emissions because studies included collection, treatment, and distribution. The range of GHG emissions for Australian studies contains somewhat higher values (0.5–2.4 kg CO_2eq/m^3) than the U.S. studies (0.1–1.7 kg CO_2eq/m^3), thus highlighting the fact that energy mix could play an important role in the differences in the amount of GHGs emitted, although the Australian studies did not specify the exact composition of their energy mixes. It is also worth noting that the wide range of emissions of the U.S.-based studies is due to the variation of electricity mix scenarios used to provide power to the water reuse scenario. These electricity mix scenarios include solar (PV and solar thermal), California, U.S., Europe 2020 (2020 energy mix goal for the European Union), and “low emissions.”

4.3.2 Water Reuse Capacity

The capacity of the water reuse facility or case study scenario for available water reuse studies ranged from 0.07 to 26 MGD. Based on this distribution, water reuse scenarios were divided into three capacity groupings: less than 1 MGD (Ortiz et al., 2007; Meneses et al., 2010; Pasqualino et al., 2010), between 1 and 10 MGD (de Haas et al., 2011), and greater than 10 MGD (Lundie et al., 2004; Friedrich et al., 2009; Stokes and Horvath, 2009; Muñoz et al., 2010). Table 4.7 shows the ranges of CO_2 and GHG emissions associated with each capacity category.

Table 4.7. Water Reuse CO_2 and GHG Emissions Organized by Facility Capacity

Capacity (MGD)	Emissions (kg CO_2/m^3)	Reference(s)	Capacity (MGD)	Emissions (kg CO_2eq/m^3)	Reference(s)
0.8	0.2–1.1	Ortiz et al., 2007	0.07, 0.62	0.1–0.9	Stokes and Horvath, 2006; Meneses et al., 2010; Pasqualino et al., 2010
Not provided	Not provided	Not provided	1.3, 4.5, 5.6	0.5–1.2	Tangsubkul et al., 2005; de Haas et al., 2011
13	0.4–0.8	Lundie et al., 2004	10.6, 11.7, 26	0.1–2.4	Friedrich et al., 2007; Stokes and Horvath, 2009; de Haas et al., 2011

For CO_2 emissions, a capacity of 0.8 MGD covers a relatively large range of values (0.2–1.1 kg CO_2/m^3). The lowest (0.2 kg CO_2/m^3) and the highest (1.1 kg CO_2/m^3) values were estimated by Ortiz et al. (2007), who compared the emission values by water reuse technologies using four different energy mixes (French, Norwegian, Portuguese, and European Union). The variation in values for a 13-MGD plant from Lundie et al. (2004) is primarily due to the inclusion of the CO_2 emissions attributed to distribution of reused water, which doubles the CO_2 emissions from 0.4 to 0.8 kg CO_2/m^3.

For GHG emission studies, the upper limits of each range of values appear to increase with increasing plant capacity (0.1–2.4 kg CO_2eq/m^3). For a more thorough analysis, the effect of capacity on GHG emissions was investigated for studies that contained a few water reuse scenarios with different capacities. For example, de Haas et al., (2011) compared four distinct water reuse scenarios with capacities ranging from 4.5 to 11.7 MGD. When compared to GHG emission values, however, a general increase from 1.2 kg CO_2eq/m^3 for a 4.5-MGD capacity to 2.4 kg CO_2eq/m^3 for an 11.7-MGD capacity was observed. Still, this comparison may not be fair, as water reuse scenarios were composed of different configurations including treatment of wastewater using RO followed by its transport to a reservoir for potable reuse, use of effluent from an advanced water treatment plant for irrigation, and use of effluent from a standard WWTP (in a different community) for agricultural irrigation (de Haas et al., 2011). Again, the study does not break down the GHG emissions associated with the distinct configurations. Accordingly, it may be difficult to compare studies of different plant capacity because of different reuse scenarios. Stokes and Horvath (2006, 2009) also carried out studies containing distinct water reuse scenarios; however, all scenarios used the same filtration and disinfection method for treatment prior to distribution. Comparison of the three scenarios represented shows that when the capacity increases from 0.07 to 26 MGD, GHG emissions also increased, from 0.5 to 1.0 kg CO_2eq/m^3. Results from both of these studies show that for water reuse, increasing capacity generally correlated with increasing GHG emissions per unit of water treated. Nevertheless, because of a lack of data, the relationship between capacity and GHG emissions is difficult to prove. Upon comparing two water reuse scenarios studied by Stokes and Horvath (2006), the larger capacity scenario consumed almost twice as much energy in terms of supply and treatment and slightly more energy in distribution than the lower capacity scenario.

4.3.3 Energy Mix and Energy Source

The studies shown in Table 4.8 based their electricity-related emission on national, local, or hypothetical national energy mixes. Energy mixes included those from the European Union, France, New South Wales (Australia), Norway, Portugal, Israel, California, South Africa, and Spain. CO_2 emission values ranged from 0.1 to 1.1 kg CO_2/m^3, whereas GHG emissions ranged from 0.27 to 2.1 kg CO_2eq/m^3.

For CO_2 emissions, water reuse technologies connected to the Norwegian grid emitted the least (0.14–0.16 kg CO_2/m^3), and the highest emissions came from the Portuguese mix (0.7-1.1 kg CO_2/m^3). The extremely low emission values associated with the Norwegian mix were due to that country's reliance on hydropower (99%), which produces a very small amount of CO_2 per kWh of electricity produced. On the other hand, about 81% of energy in Portugal comes from fossil fuels (at the time the study was conducted), which explains the higher CO_2 emission values. It is important to note that most of the CO_2 emission scenarios in Table 4.8 are from Ortiz et al. (2007). One of the objectives of this study was to compare emissions of different tertiary treatment technologies for water reuse based on different national energy mixes. Therefore, any differences in technology, scope, and capacity are eliminated. Studies by Lyons et al. (2009) and Lundie et al. (2004) have also been included, but their CO_2 emission values seem to fall within the range of values estimated by Ortiz et al (2007).

Studies of GHG emissions have included water reuse scenarios using Israeli, Californian, South African, Spanish, and U.S. energy mixes as well as different electricity sources (PV, solar thermal, and low emissions). Unlike the CO_2 emission studies, values were taken from about six different studies. Water reuse studies responsible for emitting the least GHG used the Spanish and South African energy mixes (0.1–0.8 kg CO_2eq/m^3 and 0.1–0.7 kg CO_2eq/m^3), whereas the Israeli energy mix had the highest GHG emissions (2.1 kg CO_2eq/m^3). Although energy mix may play an important role in the amounts of GHG emissions, the studies being compared might be too distinct to show this.

The scope of the study is another important factor. For instance, for the GHG emissions of treatment and distribution of reclaimed water, Muñoz et al. (2009) focuses mainly on ozonation technology, excluding distribution, whereas Stokes and Horvath (2009) also include the energy used in water distribution. In fact, distribution energy use can be responsible for more than half of the total operation energy use if included in a water reuse scenario analysis (Stokes and Horvath, 2009). Comparison between electricity sources found that solar thermal, with a GHG emission value of 0.1 kg CO_2eq/m^3, is the lowest-emitting electricity source, whereas the "low emissions" energy source is the highest (0.9 kg CO_2eq/m^3). Most likely, the high GHG emission value is due to the inclusion of some fossil fuel electricity sources within the mix.

Table 4.8. Water Reuse CO_2 and GHG Emissions Organized by Energy Mix and Source

Energy Mix	Emissions (kg CO_2/m^3)	Reference(s)	Emissions (kg CO_2eq/m^3)	Reference(s)
European Union	0.8–1.0	Ortiz et al., 2007; Lyons et al., 2009	1.3–1.9[1]	Stokes and Horvath, 2009
France	0.23–0.27	Ortiz et al., 2007	Not provided	Not provided
New South Wales	0.4–0.8	Lundie et al., 2004	Not provided	Not provided
Norway	0.14–0.16	Ortiz et al., 2007	Not provided	Not provided
Portugal	0.7–1.1	Ortiz et al., 2007	Not provided	Not provided
Israel	Not provided	Not provided	2.1	Fine and Hadas, 2012
California	Not provided	Not provided	0.5–1.0	Stokes and Horvath, 2009
South Africa	Not provided	Not provided	0.1–0.7	Friedrich et al., 2009
Spain	Not provided	Not provided	-2.1–0.8	Muñoz et al., 2009; Meneses et al., 2010; Muñoz et al., 2010 Pasqualino et al., 2010
United States	Not provided	Not provided	1.7	Stokes and Horvath, 2009
Photovoltaic	Not provided	Not provided	0.2	Stokes and Horvath, 2009
Solar thermal	Not provided	Not provided	0.1	Stokes and Horvath, 2009
Low emissions[2]	Not provided	Not provided	0.9	Stokes and Horvath, 2009

Notes: 1=Based on Europe 2020 mix, which is composed of 35% renewable electricity production; 2=Low emissions refers to "a mix of renewable energy and current California sources" (Stokes and Horvath, 2009).

Chapter 5

Available Model Review

5.1 Identification of Models for Estimation of GHG Emissions

The estimation models reviewed for this report include a range of LCA-, spreadsheet-, and web-based models specific to estimating GHGs within the water sector. The models that were identified were categorized into one of four types of methodology used to estimate GHG emissions. As shown in Table 5.1, these can be classified as (1) LCA models, (2) hybrid LCA models, (3) emission models specifically for estimating GHGs, and (4) other related models.

Table 5.1. Four Methods for Estimating GHG Emissions

GHG Emission Estimation Method	Description of Methodology	Examples of Models That Fit This Methodology	Reference(s)
Traditional LCA	Estimates GHG emissions associated with energy consumption, materials, transport, and disposal	SimaPro, GaBi	Raluy et al., 2004, 2005a, 2005b; Tangsubkul et al., 2005; Ortiz et al., 2007; Friedrich et al., 2009; Lyons et al., 2009; Muñoz et al., 2009, 2010; Meneses et al., 2010; Pasqualino et al., 2010
Hybrid LCA-based models	Estimates GHG emissions from a life cycle perspective using both process- and input output–based inventory	WEST, WWEST, and WESTWeb	Stokes and Horvath, 2006, 2009, 2011a, 2011b
Specific models for estimating GHG emissions	Uses input parameters specific to utility to calculate GHG emissions	Johnston Model, Tampa Bay Water Model	Johnston, 2011; Tampa Bay Water, 2012
Other related models	Models identified during review that are not specifically used to estimate emissions from water reuse or desalination facilities but contain aspects that are applicable	UKWIR Model, UK Environmental Agency Model, CHEApet, Systems Dynamics, GPS-X Model, mCO2, Bridle, and BSM2G	UKWIR, 2008; Reffold et al., 2008; Crawford et al., 2011; Shrestha et al., 2011, 2012; Goel et al., 2012; MWH Global, 2012; Corominas et al. (2012)

Notes: GHG=greenhouse gas; LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

On the basis of the review, no emission models were exclusively designed for determining GHGs and carbon footprint associated with water reuse and desalination facilities; however, many models have been used to estimate GHG emissions associated with water reuse and desalination. In addition, various models contain attributes that would be beneficial to an accurate, robust, and implementable GHG estimation model specific to water reuse and desalination facilities.

LCA software such as SimaPro and Gabi has been used extensively to estimate GHG emissions of water reuse and desalination facilities. Various LCA-based studies have focused on specific case studies or hypothetical scenarios for future applications. LCA software can be used to evaluate many environmental impacts (beyond GHG emissions) of any product or process and is not a model exclusively for water reuse or desalination facilities. Despite its widespread use in academia, LCA can be time-consuming and data-intensive, which may limit the ability to implement it in practice. Consequently, LCA models are discussed only briefly in Section 5.2.

Hybrid LCA models can be applied to water reuse or desalination facilities. The hybrid LCA models (WEST, WWEST, and WESTWeb) appear to be the most applicable to water reuse and desalination facilities because of their flexibility, inclusion of regional eGRID data that account for an area's specific energy fuel mix, and previous application to water reuse and desalination facilities. The hybrid LCA models reviewed in this report contain two Excel spreadsheets and a web-based model that tracks GHG emissions. Similar to LCA-based models, the hybrid LCA models reviewed account for life cycle GHG emissions, including upstream supply chain effects. The life cycle approach included in LCA and hybrid LCA models differs from the specific models reviewed, which focus on emissions associated with the operational life stage.

The specific emission models are used to estimate operational stage GHG emissions and contain beneficial aspects for the development of a robust and accurate water reuse or desalination model. Specific models include the Tampa Bay Water model (Tampa Bay Water, 2011), which is designed to track the GHG emissions from various water utilities in the Tampa Bay region (including a desalination facility), and the Johnston Model (Johnston, 2011), which is specifically tailored to drinking water treatment facilities. Both of these models track operational GHG emissions and contain useful techniques (e.g., energy consumption estimations and benchmarking) that could be used in the water reuse and desalination sector. A detailed description of the hybrid LCA-based and specific models, as well as their applicability to water reuse and desalination technologies, is presented in Sections 5.3 and 5.4. The Johnston Model is one of the few models that report emissions as Scopes 1, 2, and 3. Scope 1 accounts for direct emissions, including those from fuel consumption, treatment processes (e.g., ozone generation), and sludge disposal. Scope 2 is indirect emissions from electricity consumption, and Scope 3 includes additional indirect emissions from the production and transport of chemicals and utility vehicle travel.

In addition, other GHG emission models were identified (referred to as "Other related models" in Table 5.1) that contain aspects that could benefit the development of a comprehensive water reuse or desalination GHG emission model. These models were found to be less applicable to water reuse and desalination processes than hybrid LCA models and specific models because of their focus on traditional drinking water and wastewater treatment facilities. Other limitations come from regional constraints or methodological approach. Despite being less applicable than hybrid or specific models, a brief description of other related models is provided in Section 5.5.

5.2 LCA Models

The first and most commonly used methodology for calculating GHG emissions was LCA, which is a method to assess the environmental impact(s) of a product, process, or system over its entire life cycle (ISO, 1997). In total, more than half of the water reuse and desalination papers reviewed used LCA methodologies, which employ process-based and input–output databases to estimate GHG emissions associated with a particular utility. Using LCA-based models, various authors (e.g., Raluy et al., 2006; Lyons et al., 2009; Friedrich et al., 2009) were able to evaluate individual facilities under different scenarios (e.g., renewable energy options) and make comparisons between different treatment technologies (e.g., desalination versus water reclamation).

These commercial software programs are used to evaluate the environmental impacts of a product or process over its lifetime and include extensive life cycle inventory databases. These databases such as Ecoinvent include supply-chain emissions for various materials, chemicals, and products to account for upstream life cycle stages (e.g., raw materials extraction, manufacturing, materials provision) and end-of-life stages. They provide country-specific or regional background data and therefore do not take into account specific facility parameters.

LCA software is not specifically designed for water reuse or desalination facilities but provides a useful model for estimating GHG emissions from these facilities. Various LCA studies have focused on case studies or hypothetical scenarios related to water supply systems. These studies vary significantly in terms of system boundary, parameters considered, and emission sources considered. Consequently, results from these studies vary significantly, making it difficult to compare results from different studies. LCA studies also highlight the importance of developing a specific model for assessing GHG emissions from water reuse and desalination facilities. Since LCA software is not an actual model designed for water reuse or desalination, a detailed description of LCA approaches is not presented in the following sections; instead, the focus is on the hybrid LCA-based models, specific models, and other related models presented in Table 5.1.

5.3 Hybrid LCA-based Models

The second type of methodology identified is the hybrid LCA-based model. Models fitting this category include WEST, WWEST, and WESTWeb. A general overview of these models is presented in Table 5.2. A hybrid LCA is a method that combines a process-based life cycle inventory with an input output–based inventory (Hendrickson et al., 2006) to assess the environmental impact(s), including GHG emissions, of a system over its entire life cycle (Mo et al., 2010). These models incorporate process-specific data from designs (e.g., construction materials) and consider the operational practices (e.g., energy consumption) of water and wastewater systems to estimate GHG emissions.

Table 5.2. Overview of Hybrid LCA-based Methods

Model	Provider	Description of Model	Utility	Applicable to Water Reuse or Desalination
Water Energy Sustainability Tool (WEST)	UC Berkeley/ CEC PIER	MS Excel spreadsheet that relies on economic input output– and process-based inventory techniques to evaluate life cycle emissions during supply, treatment, and distribution phases of a system	Water	Applicable to water reuse and desalination, depending on availability of input information
Wastewater Energy Sustainability Tool (WWEST)	UC Berkeley/ CEC PIER	MS Excel spreadsheet that relies on economic input output– and process-based inventory techniques to evaluate life cycle emissions during supply, treatment, and distribution phases of a system	Wastewater	Applicable to water reuse, depending on availability of input information; includes direct treatment process emissions and coproduct offsets
WESTWeb	UC Berkeley/ CEC PIER	User-friendly Web-based model that provides streamlined version of WEST and WWEST; allows utilities to enter information directly into a web model to estimate GHG emissions	Water, wastewater	Applicable to water reuse and desalination facilities, given that default process equipment and chemicals are available; provides less flexibility than MS Excel model (e.g., cannot customize emission factors)

5.3.1 Water Energy Sustainability Tool (WEST)

WEST uses an MS Excel platform to estimate energy use, GHG emissions (CO_2eq), and emissions of nitrous oxides (NO_x), sulfur oxides (SO_x), particulate matter (PM_{10}), volatile organic compounds (VOCs), and carbon monoxide (CO) air emissions. This model has been used to compare imported water, desalination, and water reclamation supply alternatives in California (Stokes and Horvath, 2006). WEST uses a hybrid LCA to estimate GHG emissions by considering data related to production and delivery of materials, operation of equipment, production of energy, and sludge disposal as input parameters. Typically, input data are entered as cost in 1997 USD; however, the mass of some materials is required. Results of GHG emissions are reported as grams of CO_2 equivalents. The model is available for no charge from the developer, Dr. Jennifer Stokes at the University of California, Berkeley at ucbwaterlca@gmail.com.

5.3.1.1 System Boundary

The WEST structure is shown in Figure 5.1. The system boundary includes the material provision, construction, and O&M phases for the supply, treatment, and distribution of a water supply system. Decommission of the facility is not considered, but sludge disposal is included. WEST also includes emissions associated with material production and delivery, equipment use, energy production, and sludge disposal activities.

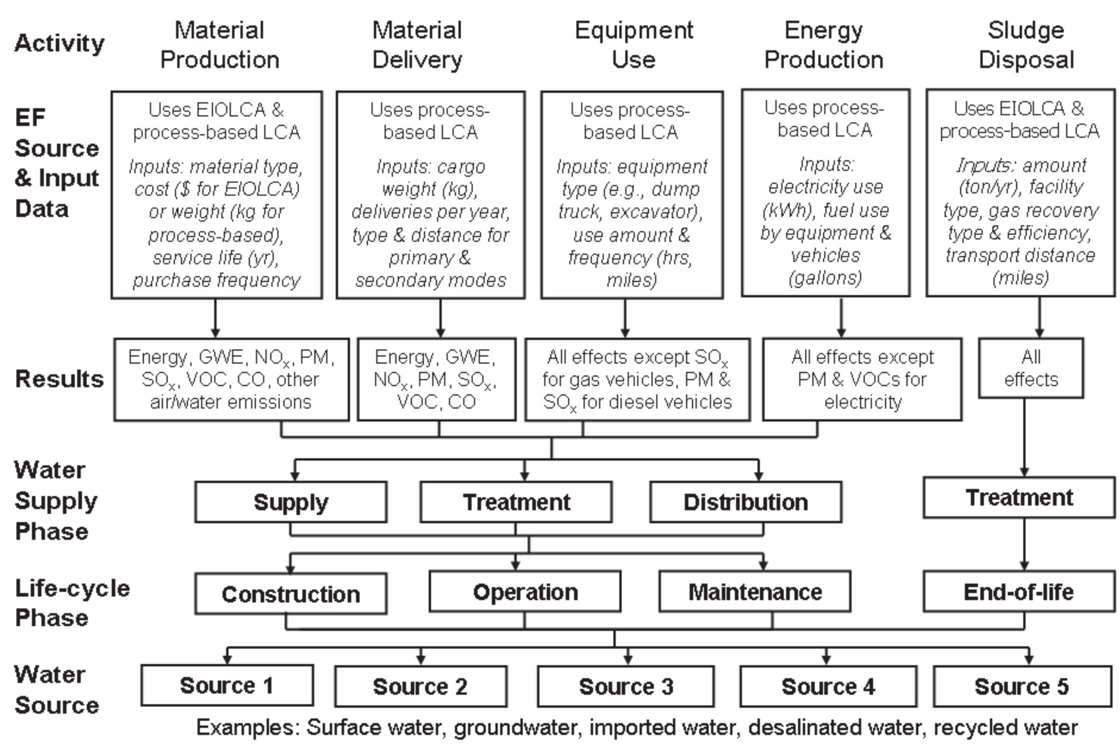

Figure 5.1. WEST model structure.

Reprinted with permission from Stokes and Horvath, 2011a; adapted from Stokes and Horvath, 2006.

As shown in the figure, "material production" represents cradle-to-grave supply-chain activities (e.g., raw material extraction, manufacturing, transport) necessary for the production of materials. "Material delivery" includes the transport of materials from the manufacturing facilities to the construction site. This may include delivery using different modes of transportation (e.g., ship, train, local or long-distance trucks). "Equipment use" activities are used to estimate direct tailpipe emissions associated with vehicles and equipment. "Energy production" data are used to estimate emissions associated with the upstream processes needed to produce electricity, fuel, and natural gas. It is important to note that, for electricity consumption, WEST allows users to estimate direct emissions associated with the electricity generation (e.g., smokestack) or life cycle emissions (e.g., upstream effects of electricity generation). Finally, "sludge disposal" activities are used to estimate emissions from the collection, conveyance, and final disposal of sludge (landfilled or incinerated).

This hybrid method relies on EIO and process-based LCA techniques to evaluate emissions during the supply, treatment, and distribution phases of a system. Both construction and O&M phases of water supply systems can be analyzed. EIO-LCA tracks emissions through an analysis method that captures interactions throughout the U.S. economy. Process details for a given facility can also be incorporated. WEST can be applied to up to five different water supply sources (e.g., water reclamation, desalination).

5.3.1.2 Data Sources

The WEST model uses both process-based LCA and EIO-LCA to estimate GHG emissions. Consequently, this hybrid model draws from a wide range of data sources to track the environmental outputs, resources, and energy inputs considered. Emissions for the production of materials come from the EIO-LCA database (www.eiolca.net), which was developed at Carnegie Mellon University. The EIO-LCA database is free and available to the public. EIO-LCA analysis is currently based on 1992, 1997, or 2002 data and purchaser price models. Process-based data from published LCAs such as Gabi (www.gabi-software.com) are used to determine the emissions from the use of equipment, the generation of energy, and the transportation of resources. Other sources for emissions come from EPA data on the electricity grid and diesel engines. Additional equipment information is from manufacturers such as Caterpillar. Emission data associated with the equipment used during construction come from the California Air Resources Board.

All of these data sources are compiled in an MS Excel document that provides further details on each data source. Further information on data sources can be obtained from the WEST user's manual (Stokes and Horvath, 2011b), the MS Excel WEST model, and previously published literature on this model (Stokes and Horvath, 2006, 2009).

5.3.1.3 Model Inputs

As shown in Figure 5.1, model inputs are separated by material production, material delivery, equipment operation, energy production, and sludge disposal. Inputs for material production include the material value (price or weight), service life, and purchase frequency. The model requires the price in 1997 USD for EIO-LCA inputs and weight in kilograms for process-based LCA inputs. The latest version of EIO-LCA has been updated to include the price value of materials in 2002 USD; however, the MS Excel spreadsheet uses 1997 data.

Material delivery inputs include cargo weight, deliveries per year (for both primary and secondary modes of transportation), mode of transportation, and distance traveled. Input data required for equipment operation include type, use amount, and frequency of use, and energy production inputs are electricity and fuel use for both vehicles and equipment. Finally, sludge disposal inputs are amount of sludge, facility type, gas recovery type, efficiency, and transport distance for sludge disposal. The WEST model also allows users to customize an electricity mix or use a state or national average mix. An option to include marginal generation sources is also available. Inputs are summarized in Table 5.3.

Table 5.3. WEST Model Inputs

Material Production	Material Delivery	Equipment Use	Energy Production	Sludge Disposal
Material type (e.g., chemicals, piping, concrete)	Cargo weight (kg)	Equipment type (e.g., dump truck, excavator)	Electricity use (kwh)	Amount (tons/year) and facility type
Material cost and year of purchase or weight (kg)	Deliveries per year	Use amount (hours)	Fuel use by equipment (gallons)	Gas recovery type and efficiency
Service life (years) and purchase frequency	Mode and distance (primary and secondary mode)	Use frequency (hours, miles)	Fuel use by vehicles (gallons)	Transport distance (miles)

Source: Stokes and Horvath, 2011a

5.3.1.4 Method for Calculations

WEST combined EIO-LCA and process-based LCA to develop a hybrid LCA model. This model calculates the GHG emissions over the life of a system and allows for comparison between alternatives based on a functional unit. The functional unit allows for two different systems to be compared on an equivalent basis. A typical functional unit used in water and wastewater LCA studies is 1 m^3 of treated water over a given period of time (typically, the lifetime of the facility). All wastes, materials, and energy consumed over the lifetime of the facility are considered to assess the environmental impacts of a system.

Process-based LCA calculations quantify the emissions associated with material inputs, energy inputs, and associated environmental outputs. WEST uses publicly available and published LCA databases that can determine the effects associated with delivering materials, operating equipment, producing energy, and manufacturing chemical and plastic products. These databases account for supply-chain effects associated with manufacturing and material provision to quantify life cycle emissions. By simply entering the mass or price of a product into the spreadsheet, WEST provides the associated GHG emissions through the use of these pre-established databases (Stokes and Horvath, 2011a).

EIO-LCA calculations use a different method to quantify supply-chain effects associated with material production. This method tracks the environmental emissions associated with material and energy inputs using economic interactions. EIO-LCA uses matrices that track the inputs and outputs between various economic sectors. Industrial economic transactions are combined with emissions data to estimate supply-chain emissions of resources. By entering the price of an item in 1997 USD, the WEST model spreadsheet is set up to provide the associated GHG emission; 1997 USD are used because the 1997 economic input–output table is used in the model. Emission factors come from both EIO-LCA and process-based LCA sources; the source of emission factors used for material production, material delivery, equipment use, energy production, and sludge disposal activities is noted in Table 5.4. For material production and sludge disposal, emission factors can come from either EIO or process-based LCA sources, depending on the material selected and sludge disposal method.

Table 5.4. Activities Analyzed in WEST and Corresponding Source of Emission Factors

Activity	Emission Factor Source	
	EIO-LCA	**Process-based LCA**
Material production	X	X
Material delivery		X
Equipment use		X
Energy production		X
Sludge disposal	X	X

Source: Adapted from Stokes and Horvath, 2011a
Notes: EIO-LCA=economic input–output life cycle assessment; LCA=life cycle assessment

WEST provides details on all the specific equations used to calculate GHG emissions from the model inputs listed in Table 5.3. This section includes examples of such equations in order to provide an understanding of how this model works.

Material Production. Emissions from materials purchased are estimated using EIO-LCA emission factors written as:

$$Emissions(Mg) = \frac{EIOLCAEF\left(\frac{Mg}{1997\$}\right) * Unit\ Cost\ (1997\$) * Units(\#) * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

where,

Emissions = emissions for material production (e.g., mg CO_2eq)

EIOLCAEF = emission factor for material from EIO-LCA database expressed in mg of emission per cost

Unit cost = cost of material in 1997 USD. If costs from other years are provided, *Engineering News Record*'s Construction Cost Index is used as a discounting factor, where Discount=1997 CCI/year of purchase CCI; this particular discount factor is built into the model.

Units = # of units purchased

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system

Volume Treated = volume of water treated annually

This calculation provides the life cycle emission of a material (e.g., concrete, materials, chemicals), accounting for supply-chain emissions. This means that for concrete, for example, supply-chain emissions from mining, water consumption, cement processing, transportation, and other manufacturing and concrete provision activities are included.

Process-based databases also account for these supply-chain effects, and materials can be calculated alternatively using these databases.

The emissions associated with a material purchased are estimated using process-based LCA emission factors as follows:

$$Emissions(kg) = \frac{GabiEF\left(\frac{kg}{kg}\right) * Unit\ Weight(kg) * Units(\#) * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

where,

Emissions = emissions for material production (e.g., kg CO_2eq)

GabiEF = emission factor for material from Gabi database expressed as kg of emission per unit weight of material

Unit Weight = weight of material in kg

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system

Volume Treated = volume of water treated annually

Embedded in the WEST Excel spreadsheet are various default materials associated with water treatment facilities and applicable to water reuse and desalination facilities. Materials not listed in the spreadsheet can be customized by users if the user has the information on emission factors.

Material Delivery. The calculations for emissions from material delivery are separated by the mode of transportation in which general equations exist for (1) local trucks, long-distance trucks, ships, and trains and (2) planes.

The calculation for material delivery emissions from trucks, ships, and trains is written as:

$$Emissions(g) = \frac{EF\left(\frac{g}{km * kg}\right) * Cargo\ Weight(kg) * Distance\ (km) * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

where,

Emissions = emissions from material delivery, expressed as g

EF = emission factor for different transportation modes expressed as g of emission per km of distance transported per kg of cargo weight

Cargo Weight = weight of material in kg (user-specified or default unit weight)

Distance = delivery distance, expressed as km

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system, expressed as years

Volume Treated = volume of water produced annually

Material deliveries for planes account for flight emission and landing and takeoff (LTO) emissions, drawing from various process-based emission factor sources. The general equation for calculating plane emissions is written as:

$$Emissions(Mg) = \frac{Flight\ emissions * LTO\ emissions * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

where,

$$Flight\ Emissions(g) = \frac{FlightEF\left(\frac{g}{km*kg}\right)*Cargo\ Weight(kg)*\ Distance(km)*Functional\ Unit\ (Vol)}{Analysis\ Period(yr)*Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

and

$$LTO\ Emissions(g) = \frac{LTOEF\left(\frac{g}{km*kg}\right)*Cargo\ Weight(kg)*\ Distance(km)*No.of\ Trips*Functional\ Unit\ (Vol)}{Analysis\ Period(yr)*Volume\ Treated\left(\frac{Vol}{yr}\right)}$$

where,

Emissions = emissions from material delivery, expressed as g

FlightEF and LTOEF = emission factor for flight and landing/takeoff

Cargo Weight = weight of material in kg (user-specified or default unit weight)

Distance = delivery distance to construction site expressed as kmNo. of Trips = cargo weight (kg)/freight capacity (kg)/trip utilization (%)

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system, expressed as years

Volume Treated = volume of water treated annually

These equations account for the impacts of material delivery, where two primary and secondary modes of transportation can be selected.

Equipment Use. Tailpipe emissions from nonroad and road equipment are calculated for the equipment use activity. Emissions from nonroad equipment include diesel use from construction equipment (e.g., excavators, cranes), gasoline use from generators, and electricity consumption from other equipment (e.g., electric saw). Emissions from road equipment include diesel-fueled trucks and gasoline consumption from cars and trucks.

Emissions for nonroad equipment are calculated as follows:

$$Emissions(g) = \frac{Use\ \ (hours) * Emission\ Factor\ \left(\frac{g}{hr}\right) * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right) * Efficiency(\%)}$$

Emissions for road equipment are written as:

$$Emissions(g) = \frac{Distance(miles) * Emission\ Factor\ \left(\frac{g}{mile}\right) * Functional\ Unit\ (Vol)}{Analysis\ Period(yr) * Volume\ Treated\left(\frac{Vol}{yr}\right) * Efficiency(\%)}$$

where,

Emissions = emissions from nonroad or road equipment, expressed as g

Use = hours of equipment use

Distance = travel distance, expressed as miles

Emission Factor = emission factor for nonroad and road equipment

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system

Efficiency = equipment or truck efficiency where default is 60% and 80%

These equations account for the tailpipe emissions from nonroad and road equipment.

Energy Production. Energy consumption calculations vary for electricity, natural gas, gasoline, and diesel. According to the WEST Website (west.berkeley.edu/), direct or life cycle emissions from electricity are calculated using emission factors from eGRID data and upstream emission data from literature, respectively. Natural gas emissions are calculated using both direct and supply-chain effects from USEPA and EIO-LCA databases, whereas gasoline and diesel use emission factors from California Climate Action Registry. The environmental effects associated with fuel and electricity consumption are calculated using the following general equation:

$$Emissions = \frac{FuelUse * FuelEF * Functional\ Unit\ (Vol)}{Volume\ Treated\ \left(\frac{vol}{yr}\right)}$$

where,

Emissions = emissions from fuel or electricity consumption

Fuel Use = consumption of fuel in MWh for electricity, MMBTU for natural gas, and volume for gasoline or diesel

FuelEF = emission factor for electricity, natural gas, gasoline, or diesel (varies depending on fuel)

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Analysis Period = service life of the system, expressed as years

Volume Treated = volume of water treated annually

Assumed distribution losses and the contribution from each electricity source are accounted for when users enter electricity grid information.

Sludge Disposal. Emissions from sludge disposal depend on whether incineration or landfills are used for sludge handling. In the case of landfills, the type of gas recovery and efficiency is also included as an input. The transport and processing of sludge is included in a separate calculation that accounts for equipment use. For sludge disposal, the following general equation is used to calculate emissions, according to the WEST Website:

$$Emissions = \frac{WasteDisposalEF\left(\frac{Mg}{ton}\right) * AnnualSludgeDisposed\left(\frac{tons}{yr}\right) * Functional\ Unit(Vol)}{Volume\ Treated\left(\frac{vol}{yr}\right)}$$

where,

Emissions = emission from sludge disposal

Waste Disposal EF = emission factor for landfill or incineration, in Mg/ton

Annual Sludge Disposed = user-entered yearly amount of sludge disposed, in ton/yr

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated water

Volume Treated = volume of water treated annually

5.3.1.5 Model Outputs

Table 5.5 summarizes the WEST model outputs. WEST outputs include energy use, the carbon footprint, and other air emissions. The carbon footprint is expressed in mg of CO_2 equivalents per functional unit, where the functional unit is typically a unit volume of treated water. GHGs (CO_2, CH_4, N_2O) are taken into account within the carbon footprint, but the results do not present values for these GHGs individually. Additional air emissions quantified by the WEST model include NO_x, PM_{10}, SO_x, CO, and VOC. WEST also offers other water and air emission calculations and a water stress indicator for California.

Table 5.5. WEST Model Outputs

Emission	Symbol	Units
Energy consumption	energy	MJ/functional unit
Carbon footprint	GWP	mg CO_2eq/functional unit
Volatile organic compounds	VOCs	mg/functional unit
Carbon monoxide	CO	mg/functional unit
Nitrous oxides	NO_x	mg/functional unit
Particulate matter	PM	mg/functional unit
Sulfur oxides	SO_x	mg/functional unit

Source: Adapted from Stokes and Horvath, 2006
Note: GWP=global warming potential

5.3.1.6 Model Limitations

A limitation to this model is that it is not designed to quantify emissions from specific unit processes within the facility. Therefore, it may be difficult for utilities to identify and mitigate the effects of specific unit processes within a treatment facility, but a user could run the model for a specific unit process given input data availability. Another limitation is its exclusion of the decommission phase of the life cycle. This limitation however, may not be as important because previous studies have shown that the decommission phase usually contributes less than 1% of the total GHG emissions (Friedrich, 2002).

It is also important to note that some utilities may not collect data at the level of detail required by WEST. WEST requires many data inputs and therefore can be very data- and time-intensive. For example, through correspondence with a utility partner in California, the authors determined that the San Elijo Water Reclamation Facility (3.0 MGD capacity) provided useful data related to electricity consumption, water quality, and water consumption, but the WEST data inputs were more extensive than the data routinely collected to reliably operate the treatment facility. Based on this information, only emissions associated with energy production could be evaluated using WEST. The lack of data availability may require making assumptions, which may reduce the accuracy of the projections, which is discussed further in Chapter 7.

GHG estimation methods that include the construction phase may be of less interest to utilities that are more concerned with avoiding GHG emissions and saving energy associated with the operational phase. In addition, plants undergoing increases in capacity and changes in unit processes must estimate the change in GHG emissions, as required by the California Environmental Quality Act. Finally, it is important to note that the web-based version of WEST is more user friendly than the Excel spreadsheet. This is due, in large part, to the fact that fewer data inputs are required for WESTWeb because it excludes material delivery and contains a smaller range of material inputs compared to the spreadsheet model.

5.3.1.7 Applicability to Water Reuse and Desalination

This model is applicable to water reuse and desalination facilities and has been used to quantify emissions from these facilities in previous studies (Stokes and Horvath, 2006, 2009). This model has flexibility in the amount of data entered. For example, utilities interested only in the effects associated with operational energy can use this model to estimate them. It can also be used to compare alternative selections of materials.

Despite the fact that WEST is not designed to assess specific unit processes, if input data for a given unit process are available, WEST could be used to assess specific unit processes as well. For example, if input information (e.g., energy consumption and other inputs) for given treatment processes (e.g., secondary treatment with nitrogen removal) is available, WEST can be used to estimate GHG emissions and compare alternatives. WWEST is another hybrid-LCA model discussed in the following section. WWEST differs from WEST in its focus on wastewater treatment facilities.

5.3.2 Wastewater Energy Sustainability Tool (WWEST)

Figure 5.2 shows the structure of WWEST. It has a lot of the same capabilities as WEST (refer to Section 5.3.1), but has additional features specifically designed for wastewater treatment systems. In addition to quantifying the life cycle GHG emissions for material production, equipment use, energy production, and sludge disposal, WWEST also allows users to estimate direct emissions from selected treatment processes. Utilities can use WWEST to estimate CH_4 and N_2O emissions from the following unit processes: poorly managed centralized aerobic treatment, well managed centralized aerobic treatment, anaerobic reactor, anaerobic shallow lagoon, anaerobic deep lagoon, septic system, or anaerobic digester, as defined by the Intergovernmental Panel on Climate Change (IPCC; 2006). Other features include the estimation of GHG emissions offset from coproducts, such as fertilizers.

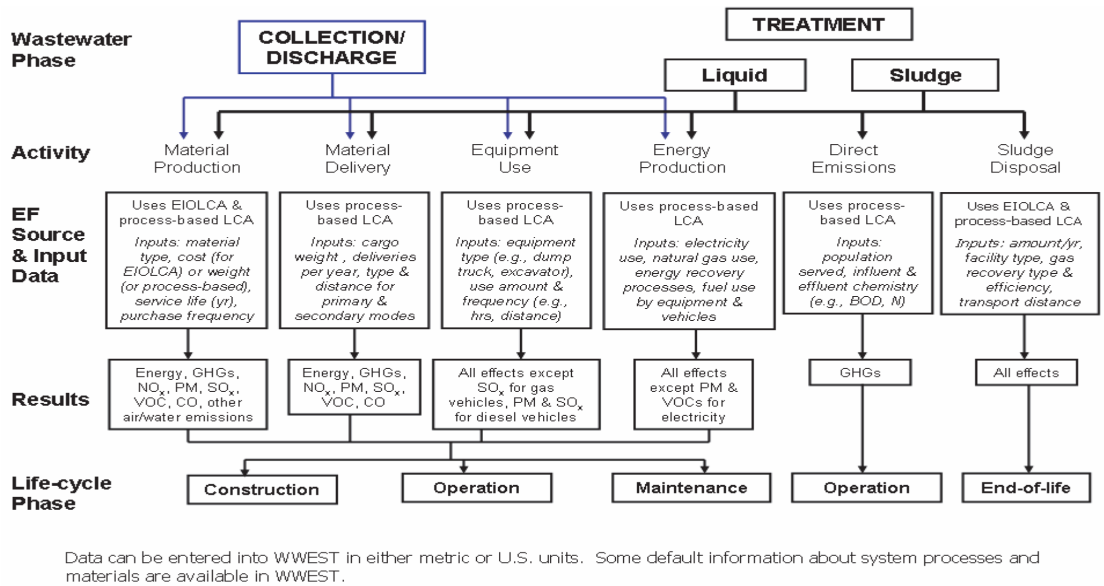

Figure 5.2. WWEST model structure.

Reprinted with permission from Stokes and Horvath, 2011b

5.3.2.1 System Boundary

Emissions are assessed for supply, treatment, and distribution water supply phases. Life cycle phases assessed include construction, operation, and end-of-life for sludge disposal. Because WESTWeb uses EIO-LCA, the entire U.S. economy is included in the system boundary. Process-specific information can be entered as well because of its inclusion of process-based LCA. The WWEST structure is very similar to that of the WEST model, with the additional inclusion of direct emissions from selected treatment processes (see Figure 5.2).

5.3.2.2 Data Sources

Most of the data sources are the same as WEST (see Section 5.3.1.2). Estimation techniques for direct emissions from wastewater treatment processes come from the IPCC (2006). Refer to the WWEST User's Manual (Stokes and Horvath, 2011a), the MS Excel WWEST model, and previously published literature on this model for further information on data sources (Stokes and Horvath, 2010).

5.3.2.3 Model Inputs

Data inputs are the same as the WEST model for material production, delivery, equipment use, energy production, and sludge disposal. Additional inputs to account for direct process emissions include the number of people served and influent and effluent water quality data (e.g., BOD concentration in mg/L).

5.3.2.4 Method for Calculations

All calculations use a similar approach to WEST, with the exception of unit treatment process emissions. CH_4, N_2O, and total direct emissions from poorly managed centralized aerobic treatment, well managed centralized aerobic treatment, anaerobic reactors, anaerobic shallow lagoons, anaerobic deep lagoons, septic systems, or anaerobic digesters are calculated using emission factors defined by IPCC (2006). As stated earlier, the IPCC is the lead international body for assessment of climate change. Emission factors vary for each treatment process defined. The following general equations can be used to calculate direct process emissions:

$$Methane = CH_4ProcessEF * \frac{InfBOD * AnnualProd - SludgeBOD * SludgeDisp}{1000} * (1 - CH_4Capture)$$

where,

Methane = methane emissions from selected wastewater treatment process, g

CH_4ProcessEF = emission factor for given treatment process, gCH_4/gBODtreated

InfBOD = user-entered influent BOD, mg/L

AnnualProd = user-entered annual production of wastewater, L

SludgeBOD = user-entered BOD of sludge, mg/kg

SludgeDisp = user-entered sludge disposed, kg

CH_4Capture = user-entered percentage of CH_4 captured

$$N_2O = N_2OProcessEF * PopulationServed * IndContribution$$

where,

N_2O = nitrous oxide emissions from wastewater treatment process, expressed as g

N_2OProcessEF = emission factor, assumed to be 3.2g/capita*yr from the IPCC

PopulationServed = user-entered number of people served by treatment facility

IndContribution = contribution from industrial users, assumed to be 1.25 as default from the IPCC

$$WWProcessGHG = \frac{(23 * Methane + 296 * N2O) * FunctionalUnit}{AnnualProduction}$$

where,

WWProcessGHG = GHG emissions from wastewater treatment process, expressed as g

Functional Unit = unit based on the function of the system that allows for alternative comparisons; for treatment systems, this is typically a unit volume of treated wastewater

Annual Production = volume of water produced annually

The CH_4 and N_2O emissions are multiplied by their IPCC 100-year GWP. These equations represent the additional calculations for direct emissions attributed to specific wastewater treatment processes. The energy production equations account for energy recovery for WWEST. These are presented in the WWEST spreadsheet.

5.3.2.5 Model Outputs

Model outputs are the same as the WEST model (see Section 5.3.1.5). Results are expressed as mg CO_2eq/functional unit, where the functional unit is a unit volume of treated wastewater.

5.3.2.6 Model Limitations

Model limitations are the same as the WEST model (see Section 5.3.1.6).

5.3.2.7 Applicability to Water Reuse and Desalination

WWEST seems to be more applicable to water reuse facilities than WEST because it includes direct emissions from treatment processes. Although WEST is recommended for water reclamation and desalination facilities, WWEST could possibly be the preferable alternative if water reuse facilities use the treatment processes included in WWEST. WWEST is not applicable to desalination facilities because direct emissions from desalination facilities will differ from water reuse or wastewater treatment facilities. Section 5.3.1.7 provides additional information on applicability; WEST and WWEST are based on a similar methodology.

5.3.3 WESTWeb Model

WESTWeb is a user-friendly, Web-based model that incorporates WEST and WWEST. According to its Website, WEST can be used for desalination and water reclamation, whereas WWEST is only applicable to wastewater treatment. WESTWeb allows utilities to enter information directly to a Web-based model to estimate GHG emissions using a hybrid LCA approach. The Website requires users to select modeling parameters, annual water or wastewater production data, infrastructure data (e.g., pipe length and material, reinforced concrete materials, and specified process equipment information), operation data (e.g., electricity mix location, energy use, and chemical consumption), and waste management data (e.g., sludge disposal). Upon running the model, carbon footprint results are presented in tabular format. Users also have the option to evaluate additional human/environmental impact potentials. Additional impact categories include carcinogens, noncarcinogens, respiratory inorganics, ozone depletion, respiratory organics, aquatic ecotoxicity, terrestrial ecotoxicity, aquatic acidification, and aquatic eutrophication.

5.3.3.1 System Boundary

The system boundaries for WESTWeb are the same as those in WEST (see Section 5.3.1.1) and WWEST (see Section 5.3.2.1).

5.3.3.2 Data Sources

The data sources for the WESTWeb model are the same as those for WEST (see Section 5.3.1.2) and WWEST (see Section 5.3.2.2).

5.3.3.3 Model Inputs

WESTWeb inputs are summarized in Table 5.6. For water and wastewater utilities, the annual production volume of water is a required input. Also, WESTWeb users must define a functional unit by selecting a volume of water or wastewater treated. For wastewater treatment specifically, utilities have the option to select the sludge disposal and ash disposal process. Energy mixes are built into the model, and users may select a national, state, or custom electricity mix that requires the percentage of fuel or energy from each source. For energy use, the annual consumption of electricity, natural gas, gasoline, and diesel is required during each water supply phase (collection, treatment, and discharge). The 2002 USD purchase price is required for piping, fittings, flow meters, and valves. Material length and pipe diameter data are needed for infrastructure piping, and the total volume of concrete is required for reinforced concrete used. Users can also enter the purchase price (2002 USD) of various process equipment summarized in Table 5.6. WESTWeb incorporates more recent 2002 EIO-LCA data, whereas WEST uses the 1997 data. Inputs also include the annual consumption of chemicals for pH adjustment, coagulation and flocculation, and disinfection (see Table 5.6).

For wastewater, WESTWeb incorporates process emission inputs, sludge data, and energy recovery information. The process information includes BOD water quality information. For sludge disposal, the concentration and annual amount of sludge disposed are required. The percentage of CH_4 recovered is another available input. The type of system must be specified using the following choices: poorly managed centralized aerobic treatment, well managed centralized aerobic treatment, anaerobic reactor, anaerobic shallow lagoon, anaerobic deep lagoon, septic system, or anaerobic digester.

5.3.3.4 Method for Calculations

WESTWeb methods for calculations are the same as WEST (see Section 5.3.1.4) and WWEST (see Section 5.3.2.4).

Table 5.6. Summary of WESTWeb Model Inputs

Sludge Disposal and Energy Production/Use	Material Production, Process Emissions, and Process Equipment	Treatment Chemical Consumption (lb/year)
Sludge disposal—sludge disposal and ash disposal process	Infrastructure materials—purchase price (2002 USD) of the following items: piping, fittings, flow meters, valves; infrastructure piping—material, length, and diameter of piping; reinforced concrete materials—total volume of concrete used	ph adjustment—lb/year of hydrochloric acid, sulfuric acid
Energy production—national, state, or custom electricity mix. Custom requires percentage of fuel/energy from source	Process emissions—number of people served, influent and effluent water quality data (e.g., bod [mg/l]), effluent bod concentration of sludge (mg bod/kg sludge), annual dry sludge disposed (lb), percentage methane captured, and type of system	Coagulants and flocculants—lb/year of aluminum sulfate, aluminum hydroxide, caustic soda, ferric chloride, polymers
Energy use—annual consumption of electricity (Mwh), natural gas (MMBTU), gasoline (gallons), diesel (gallons)	Process equipment—purchase price (2002 USD), if applicable: filter media (sand, gravel, anthracite, or other coal product), membranes, pumps, fans/blowers, motors and generators, turbines, metal tanks, UV lamps/lights, other industrial equipment, electrical, controls	Disinfectants—lb/year of chlorine, calcium hypochlorite, ozone, aqueous ammonia; other chemicals—lb/year of fluorosilicic acid, other chemicals

Source: Adapted from west.berkeley.edu
Notes: BOD=biochemical oxygen demand; UV=ultraviolet

5.3.3.5 Model Outputs

Outputs are the same as for WEST (see Section 5.3.1.5) and WWEST (see Section 5.3.2.5); however, additional impacts such as human toxicity, terrestrial and aquatic ecotoxicity, and respiratory effects can also be assessed. Results for the carbon footprint are presented in g of CO_2eq per functional unit, which is typically a unit volume of water or wastewater.

5.3.3.6 Model Limitations

WESTWeb has less flexibility for entering customized facility data compared to WEST and WWEST; however, it is more user friendly. WEST and WESTWeb are not designed to estimate emissions from specific unit processes (e.g., different treatment alternatives); however, they can be customized to do so if input data are available.

5.3.3.7 Applicability to Water Reuse and Desalination

WESTWeb is applicable to water reuse and desalination facilities provided the user recognizes that the process equipment and chemicals included in the model must be used. Examples of process equipment that might be applicable include filter media (sand, gravel, anthracite, or other coal product), membranes, and UV lamps/lights. Examples of chemicals that might be applicable include chlorine, calcium hypochlorite, and ozone. Refer to Table 5.6 for a list of process equipment and chemicals available. Because WESTWeb cannot be customized, facilities that do not use the equipment and chemicals included in the model would not be able to use WESTWeb for a comprehensive assessment.

5.4 Available Specific Models for Estimating GHG Emissions

The third category of available models identified in this study includes models that use input parameters specific to a utility to calculate GHG emissions. Examples of input parameters include amount of water pumped, amount of water produced, pumping electrical usage, utility electricity consumption, and power utility emissions. Models fitting this group include the Tampa Bay Water Model (Tampa Bay Water, 2011) and the Johnston Model (Johnston, 2011) for estimating GHG emissions. An overview of these two models is presented in Table 5.7.

Table 5.7. Overview of Tampa Bay Water and Johnston Models

Model	Provider	Description of Model	Utility	Applicable to Water Reuse or Desalination
Tampa Bay Water Model	Tampa Bay Water, 2012	Quantifies emissions associated with electricity consumption during collection, treatment, and distribution	Various facilities including water and desalination	Can be applied to estimate emissions of water produced at water reuse and desalination facilities
Johnston Model	Johnston, 2011	Estimates both direct and indirect emissions from collection, treatment, distribution, and buildings/fleet/ others	Water	Elements of model are applicable to water reuse and desalination facilities if input parameters are available

5.4.1 Tampa Bay Water Model

Tampa Bay Water treats and delivers water to various cities and counties in the Tampa Bay, FL region. It developed an MS Excel model that is used internally to track GHG emissions associated with the collection, treatment, and distribution of water to its member governments. Water includes groundwater, surface water, and desalinated water sources. All three sources are treated at separate facilities, subsequently blended, and delivered to users in Hillsborough County, Pasco County, Pinellas County, New Port Richey, and St. Petersburg. Its desalination facility uses pretreatment and RO to provide 10% of the water in the region (www.tampabaywater.org). Tampa Bay Water developed a method to quantify emissions from electricity consumption during water delivery. The model uses MS Excel to estimate CO_2, CH_4, and N_2O emissions and the carbon footprint (CO_2eq) associated with water produced (Tampa Bay Water, 2011).

5.4.1.1 System Boundary

The system boundary includes the operation stage of water treatment and distribution, specifically energy consumed to transport, treat, and deliver water to Tampa Bay Water's member governments for distribution. This includes the energy consumed during collection, treatment, and distribution.

5.4.1.2 Data Sources

The Tampa Bay Water Model uses an energy-consumption manager to collect input data necessary for its analysis. This data management system collects information from the three power utilities (TECO, Progress Energy, and WREC) that provide energy to the various Tampa Bay Water facilities. Data collected include operational flow rate, equipment run time, and energy usage during pumping. This system combines operational data with billing information from the power providers to determine the pumping electricity usage.

Additional inputs required for emission estimates include electricity mix and emission data from the EPA. Tampa Bay Water used EPA data and information from each power provider to determine the electricity mix of all three power utilities. Emission data from each regional power plant were collected from two EPA databases—eGRID and Clean Air Markets (CAM) data program. The model used eGRID emission data available for regional power plants from the 2005 calendar year (cfpub.epa.gov/egridweb). Also, 2010 CAM data were used for comparison (cfpub.epa.gov/egridweb), although CAM does not collect N_2O data.

5.4.1.3 Model Inputs

The model uses the annual amount of water pumped, the amount of water produced, pumping electrical usage, utility electricity consumption, and electric power utility emissions as input parameters. Model inputs and data sources are summarized in Table 5.8.

Table 5.8. Tampa Bay Water Model Input Parameters and Data Sources

Model Inputs	Data Source
Water pumped (MG/yr)	In house
Water produced (MG/yr)	In house
Electrical usage from pumping (kWh/yr)	In house
Gross load by power provider (MWh used/yr)	EPA eGRID or CAM
CO_2 emissions from power provider (tons/yr)	EPA eGRID or CAM
CH_4 emissions from power provider (tons/yr)	EPA eGRID or CAM
N_2O emissions from power provider (tons/yr)	EPA eGRID 2005
Electricity mix of power provider (% per source)	EPA data and utility contacts

Notes: CH4=methane; CO_2=carbon dioxide; N_2O=nitrous oxide

5.4.1.4 Method for Calculations

To estimate the emissions, the energy consumed per million gallons of water produced is calculated (Step 1). Subsequently, the annual energy consumption to produce water is calculated for each power provider (Step 2). After that, the annual emissions are calculated by first determining the emission factor from each power station serving Tampa Bay Water facilities (Step 3). Finally, the emission per million gallons of water produced is calculated (Step 4). Steps 1 through 4 were taken to calculate the CO_2, CH_4, and N_2O emissions of water produced.

Step 1: Calculate the amount of energy consumed (kWh) per MG of water produced.

$$\frac{EnergyConsumed\ (kWh)}{WaterProduced\ (MG)} = \frac{EnergyConsumed\ (kWh)}{WaterPumped\ (MG)} * \frac{AnnualWaterPumped\ (MGY)}{AnnualWaterProduced\ (MGY)}$$

Step 2: Calculate the annual energy consumption (kWh/year).

$$EnergyConsumption\left(\frac{kWh}{yr}\right) = \frac{EnergyConsumed(kWh)}{WaterProduced(MG)} * WaterProduced(MGD) * \frac{365d}{yr}$$

Step 3: Convert the energy consumed per year (kWh/year) to the amount of emission produced per year (lbs/year) using EPA emission factors (lbs/kWh).

$$Emission\left(\frac{lbs}{yr}\right) = EmissionFactor\left(\frac{lbs}{kWh}\right) * EnergyConsumption\left(\frac{kWh}{yr}\right)$$

where,

$$Emission\ Factor\ \left(\frac{lbs}{kWh}\right) = \frac{EmissionfromPowerProvider\left(\frac{tons}{yr}\right)}{EnergyUsedbyPowerProvider\ \left(\frac{MWh}{yr}\right)} * \left(\frac{2000lbs}{ton}\right) * \left(\frac{MWh}{1000kWh}\right)$$

Step 4: Convert the amount of energy consumed (kWh) per MG of water produced to the amount of emission produced per MG (lbs/MG produced).

$$Emission\left(\frac{lbs}{MG}\right) = EmissionFactor\left(\frac{lbs}{kWh}\right) * \frac{EnergyConsumed(kWh)}{WaterProduced(MG)}$$

Using the IPCC 100-year GWP factors, the GHG emissions can then be expressed in CO_2 equivalents per kWh for each power provider. In addition, the yearly electricity consumption of a specific treatment facility (e.g., the desalination plant) can be multiplied by the lbs of emission per kWh for a given power provider to obtain the yearly CO_2 equivalents in lbs per year.

5.4.1.5 Model Outputs

Results are reported as lbs of emission per kWh, lbs of emission per year, and lbs of emission per MG of water produced from Tampa Bay Water's surface drinking water treatment plant, groundwater treatment facilities, and desalination facility. Emissions calculated include CO_2, CH_4, N_2O, and CO_2eq of water produced, as summarized in Table 5.9. In addition, results for specific facilities can be reported as CO_2eq/year and CO_2eq/MG. This was done for the Tampa Bay Water desalination facility, resulting in an average of 21,175 kg CO_2eq/year and 3.08 kg CO_2eq/m^3 from 2006 to 2009 (Bracciano and Medina, 2012).

Table 5.9. Tampa Bay Water Model Outputs for Each Power Provider Serving the Region

Model Outputs
CO_2eq, CO_2, N_2O, and CH_4 emissions (lbs/kWh)
CO_2eq, CO_2, N_2O, and CH_4 emissions (lbs/yr)
CO_2eq, CO_2, N_2O, and CH_4 emissions (lbs/MG)

Notes: CH4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalent; N_2O=nitrous oxide

5.4.1.6 Model Limitations

A limitation of the Tampa Bay Water Model is that electricity consumption is the only emission source considered. This model excludes direct process emissions as well as indirect emission sources such as the production of construction materials and chemicals. Also, this model is not designed to track emissions from specific unit processes (e.g., pretreatment and RO). This may be a limitation if utilities want to track emissions specifically associated with tertiary technologies, for example. Fuel consumption during construction and operation life stages is also omitted from this model.

5.4.1.7 Applicability to Water Reuse and Desalination

The Tampa Bay Water Model can be used to estimate key GHG emissions (CO_2, N_2O, CH_4) and the carbon footprint (and CO_2eq) for the collection, treatment, and delivery of treated water. This model is intended to estimate emissions associated with water production from various sources (groundwater, surface water, and desalinated water). It has previously been applied to estimate emissions from a desalination facility and can be applied to estimate the emissions associated with electricity consumption at water reuse facilities.

5.4.2 Johnston Model

The Johnston Model (Johnson, 2011) is an MS Excel–based model that estimates the GHG emissions of a water utility. This model estimates both direct and indirect emissions from the collection, treatment, and distribution of water, and from buildings, fleets, and other sources. It assesses the GHG emissions associated with fuel and energy consumption, chemicals, buildings, and utility vehicles, and direct emissions from specific water treatment processes. Outputs include CO_2, CH_4, N_2O, and GHG emissions (expressed as CO_2eq). In addition, the Johnston Model developed energy prediction equations for utilities that do not collect electricity usage data and tested the model on seven utilities in three different states in the southern United States. It also contains a water–energy nexus tool, which uses the electricity usage at a drinking water treatment facility and an average water consumption factor from various electricity production methods (Glassman et al., 2011) to estimate the yearly water consumption for energy production and the net water production.

5.4.2.1 System Boundary

The system boundary includes the collection, treatment, and distribution of water during the operational phase. This also includes an option to assess emissions from energy consumption associated with administrative buildings and utility fleet vehicles.

5.4.2.2 Data Sources

Data come from a wide range of sources. Fuel and eGRID emission factors come from the EPA, whereas emission factors for treatment come from other literature. Emission factors for the direct emissions of specific potable water treatment technologies (e.g., ozone generation, GAC regeneration, reservoir emissions, and sludge disposal) come from Huxley et al. (2009). Additional energy use factors for specific treatment steps (e.g., pressure filtration, encased and submerged MF/UF, different types of UV, ozonation, and different types of RO) are obtained from Veerapaneni et al. (2011). Emission factors for the production of chemicals are from a master's thesis (Tripathi, 2007), and passenger car emission factors are from the EPA (2008a, 2008b). See Johnston (2011) for a comprehensive list of all data sources.

5.4.2.3 Model Inputs

Inputs to the model are summarized in Table 5.10. They include electricity grid information from the eGRID database, annual fuel consumption, annual electricity consumption, data to estimate direct emissions from specific treatment phases (e.g., collection and distribution), chemicals used yearly, and data to estimate emissions from utility vehicles. Inputs are separated by collection, treatment, and distribution. Four electricity grid input options are available for the Johnston Model. These options are presented from the most accurate to the least accurate.

Table 5.10. Johnston Model Inputs

Collection	Treatment[1,2]	Distribution	Building/Fleet/Other
Annual fuel usage (therms/yr, MMBTU/yr, or gal/yr)	Annual fuel usage (therms/yr, MMBTU/yr, or gal/yr)	Annual fuel usage (therms/yr, MMBTU/yr, or gal/yr)	Annual fuel usage (therms/yr, MMBTU/yr, or gal/yr)
Specific fuel type	Specific fuel type	Specific fuel type	Specific fuel type
Annual electricity usage (kWh/yr)	Annual electricity usage (kwh/yr)	Annual electricity usage (kwh/yr)	Annual electricity usage (kwh/yr)
Average flow rate and average purchased water flow rate (MGD)	Average flow rate (MGD) and specific treatment processes	Average flow rate (MGD)	Vehicle fuel type, fuel amount (gal/yr), mileage (miles/yr), vehicle type, model year
Electric pump horsepower (hp)	Chemicals used (lbs/yr) and sludge disposal (TOC removed/yr)	Electric pump horsepower (hp)	

Source: Adapted from Johnston, 2011.
Notes: 1=Additional inputs for drinking water treatment processes: ozone generation (m^3/year), GAC regeneration (tons/yr), reservoir (S.A. and climate region); 2=Utilities can estimate energy consumption from mixers, flocculators, settlers, dissolved air flotation, filtration, MF/UF, UV, ozone, hypochlorite, decarbonators, RO, and thermal desalination by entering the average flow rate; TOC=total organic carbon.

The best option for utilities would be to manually enter emission factors (lb of CO_2, CH_4, and N_2O/MWh) if the information is readily available from the power provider. The second option is to enter the ZIP code where the water utility is located. This will select the proper EPA subregion to use those emission factors. The next option is to use U.S. national average emission factors, and utilities that have on-site energy generation can manually enter the percent contribution of each energy/fuel source. According to Johnston (2011), this is the least preferred option because emissions are expressed as overall CO_2 equivalents as opposed to individual emissions (CO_2, CH_4, and N_2O). In addition, this option is less accurate because emission factors come from various energy sources (e.g., wind, solar, natural gas, coal) compiled by Johnston (2011), which are not site specific.

For the collection system, annual fuel usage, specific fuel type, and annual electrical usage are required. Utilities that do not collect data on electrical usage can use energy prediction equations developed by Johnston (2011). Required inputs for energy prediction equations include average flow rate, electric pump horsepower, and average purchased water flow rate.

Treatment process inputs include annual fuel usage, specific fuel type, data to estimate direct emissions from specific technologies, annual electrical usage, and chemicals used. Inputs to estimate the direct emissions of ozone generation include the volume of ozone produced per year. For GAC generation, the amount regenerated yearly is required. Surface area and climate region are needed to estimate fugitive CO_2, CH_4, and N_2O emissions from standing water in reservoirs. This information is not useful for water reuse or desalination facilities because it focuses on drinking water treatment. For sludge disposal, the TOC removed annually is needed to estimate emissions. This may require estimating solids first and then converting it to a TOC basis using the organic content of sludge (solids).

If annual electricity consumption is not available, utilities can enter the average flow rates to estimate energy use from specific unit processes. Energy use can be estimated for different types of equipment as well as treatment processes, including mixers, flocculators, settlers, dissolved air flotation (DAF), filtration, MF/UF, UV disinfection, ozone, chlorination, decarbonators, RO, and thermal desalination. Some of these processes are applicable to desalination and water reuse (e.g., MF/UF, chlorination, UV disinfection, RO).

Distribution inputs include annual fuel usage, fuel type, and annual electricity usage. Energy prediction equations were developed for distribution as well, which require average flow rate and total electric pump horsepower as inputs.

Finally, emission from buildings and fleet vehicles can be estimated with the Johnston Model. This requires annual fuel usage, fuel type, and annual electricity usage as inputs. To estimate emissions from utility vehicles, required inputs include fuel type, fuel amount, annual mileage, vehicle type, and model year. EPA emission factors are then used to estimate the emissions (USEPA, 2008a, 2008b).

5.4.2.4 Method for Calculations

To estimate the GHG emissions from the various input parameters, the Johnston Model multiplies inputs by emission factors obtained from the EPA and various literature sources. For example, the electricity consumption multiplied by the eGRID emission factor provides the emissions for electricity consumption. In addition, chemicals used multiplied by the chemical emission factor give the GHG emissions associated with chemical usage. See Johnston (2011) for a detailed explanation of emission factors used in calculations.

Energy prediction equations were also developed using various statistical methods and the SAS lasso method. SAS uses a lasso statistical method to determine which independent variables are significant in predicting the energy consumption of water utilities. Data used to develop these equations were collected through an online survey sent to water utilities and combined with previous data from an American Water Works Association Research Foundation (AWWARF) report (Carlson and Walburger, 2007). Data from 155 utilities from the AWWARF report and an additional 37 utilities were identified to assess the statistical significance of variables such as average flow, average purchased water flow, and source water pumping horsepower (Johnson, 2011).

The purpose of developing energy prediction equations is to provide a means for utilities to estimate energy consumption if data are not readily available. The Johnston Model can determine what independent variables were significant and develop energy estimation equations for collection and distribution. This research was not able to produce an estimation equation for treatment because of a lack of data on energy consumption for specific processes. A statistical analysis of water reuse and desalination facilities similar to this could be useful to estimate the GHG emissions associated with electricity consumption for specific unit processes.

The following empirical equations and corresponding R^2 value for the regression model were developed by Johnston (2011). The energy estimation equation for collection with no purchased water flow was developed from a regression model determined to have an R^2 value of 0.79:

$$\begin{aligned} &Log_{10}(Electricity\ [kWh/yr]) \\ &\quad = 3.04430 + 0.42367 * Log_{10}(Total\ Average\ Flow\ [kGD]) + \\ &\quad 0.57216 * Log_{10}(Raw\ Water\ Collection\ Pumping\ HP + 1) \end{aligned}$$

Energy estimation equation for collection with purchased water flow was developed from a regression model determined to have an R^2 value of 0.87:

$$\begin{aligned} &Log_{10}(Electricity\ [kWh/yr]) \\ &\quad = 2.91331 + 0.80696 * Log_{10}(Total\ Average\ Flow\ [kGD]) + \\ &\quad 0.51377 * Log_{10}(Raw\ Water\ Collection\ Pumping\ HP + 1) - \\ &\quad 0.35124 * Log_{10}(Average\ Purchased\ Water\ Flow\ [kGD] + 1) \end{aligned}$$

For distribution, an energy estimation equation was developed from a regression model with an R^2 value of 0.69:

$$\begin{aligned} &Log_{10}(Electricity\ [kWh/yr]) \\ &\quad = 3.6538 + 0.4259 * Log_{10}(Total\ Average\ Flow\ [MGD]) + \\ &\quad 0.6590 * Log_{10}(Finished\ Water\ Distribution\ Pumping\ HP + 1) \end{aligned}$$

After estimating energy use by the equations listed previously, utilities can use emission factors to convert energy to GHG emissions. The author developed these energy estimation equations using actual data but did not discuss calibration of these equations with actual results.

5.4.2.5 Model Outputs

Results are presented as Scope 1, Scope 2, and Scope 3 emissions. Scope 1 emissions account for direct emissions, including those from fuel consumption, treatment processes (e.g., ozone generation), and sludge disposal. Scope 2 includes indirect emissions from electricity consumption, and Scope 3 includes additional indirect emissions from the production and transport of chemicals and utility vehicle travel. Scopes 1 and 2 are called the carbon inventory, and all three scopes are considered the carbon footprint. Carbon inventory results include CO_2, CH_4, N_2O, and CO_2eq emissions in kg/year for collection, treatment, and distribution. Results are reported as individual GHGs (CO_2, CH_4, N_2O), and the cumulative carbon footprint (CO_2eq). Model outputs are summarized in Table 5.11.

5.4.2.6 Model Limitations

Limitations of this model include the exclusion of the construction phase and decommission phase emissions. Other limitations include the emission factors used for the production of chemicals. An LCA approach was used to determine the GHG emissions associated with chemicals by Tripathi (2007); however, the study did not provide a clear description of what was considered to estimate these GHG emission factors.

5.4.2.7 Applicability to Water Reuse and Desalination

Despite being designed for water utilities, the Johnston Model has elements that are applicable to water reuse and desalination. For example, this model developed energy estimation equations for the distribution and collection of water. A similar approach could be taken for water reuse and desalination facilities to estimate GHG emissions during treatment and distribution so that utilities can predict energy consumption based on known factors. Also, this model contains energy use conversion factors (in kWh/1000 gallons) for various treatment processes obtained from Veerapaneni et al. (2011). These could be useful for estimating energy consumption and GHG emissions because the only input required to use these conversion factors is average flow rate. These conversion factors may be applicable to some water reuse and desalination technologies such as membranes, seawater RO, thermal desalination, on-site hypochlorite, ozonation, different UV applications, and filtration. See Johnston (2011) for a comprehensive list of these conversion factors. It can be concluded, therefore, that aspects of this model could be adapted to water reuse and desalination facilities for tracking operational-phase GHG emissions.

Table 5.11. Johnston Model Outputs

Scope	Description	GHG Emission Outputs
Scope 1	Direct emissions (from fuel consumed, treatment processes, and sludge disposal)	CO_2, CH_4, N_2O, CO_2eq (kg/yr) for collection, treatment, and distribution
Scope 2	Indirect emissions (from the production of electricity)	
Scope 3	Additional indirect emissions (from building/fleet/other and production and transport of chemicals)	

Notes: CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalent; GHG=greenhouse gas; N_2O=nitrous oxide

5.5 Other Related Models

Table 5.12 provides an overview of several other models that are available and their applicability to estimating GHG emissions associated with water reuse and desalination facilities. Additional models investigated were deemed to have limited applicability to water reuse or desalination and are only briefly analyzed in this section. These include models that were geographically not applicable to the United States, focused on traditional wastewater or water treatment facilities, or had limited information available in the public domain. Despite the limitations of these models, it is important to discuss them briefly, as some aspects could be useful in creating a robust, accurate, and reliable GHG model for water reuse or desalination facilities.

Table 5.12. Examples of Other GHG Emission Models in Water and Wastewater Sector

Model (Reference)	Provider	Description of Model	Utility	Applicable to Water Reuse or Desalination
CHEApet (Crawford et al., 2011)	Water Environment Research Foundation	Web-based model, quantifies energy/GHG emissions of wastewater treatment plants and specific unit processes during operational phase	Wastewater	Applicable only to water reuse facilities using pre-established activated sludge systems, limited advanced treatment options (e.g., tertiary filtration and UV)
Environment Agency Model (Reffold et al., 2008)	Environment Agency (UK)	MS Excel model, uses LCA to assess GHG emissions of various water supply options and demand management options	Water supply options (water reuse, desalination)	Applicable only to supply options in uk; model uses uk-specific data
Bridle and BSM2G Model (Corominas et al., 2012)	Corominas et al. (Université Laval, Québec, Canada)	Simple, comprehensive, and process-based GHG estimation model using steady-state and dynamic simulations	Wastewater	Applicable to wastewater treatment facilities
System Dynamics (Shrestha et al., 2011, 2012)	Shrestha et al. (University of Nevada, Las Vegas)	System dynamics model, consists of stock, flows, and connectors; used to model Las Vegas Valley's water management system	Varies	Applicable only to las vegas valley water management system, uses region's water network
GPS-X Model (Goel et al., 2012)	Hydromantis Environmental Software Solutions	GHG emission model incorporated into a dynamic process simulator, uses mechanistic models to design/optimize wastewater treatment processes	Wastewater	Not applicable; focuses on wastewater processes

Table 5.12. Examples of Other GHG Emissions Models in Water and Wastewater Sector *(continued)*

Model (Reference)	Provider	Description of Model	Utility	Applicable to Water Reuse or Desalination
Carbon Accounting Workbook, 5th Version (UKWIR, 2008)	UK Water Industry Research	Commercially available MS Excel spreadsheet, quantifies operational GHG emissions in water sector (latest version provides guidelines for accounting for embodied carbon and non-CO_2 emissions)	Water	Applicable to water treatment facilities in UK
mCO2 (MWH Global, 2012)	MWH	Commercially available GHG emission model, quantifies direct/ indirect emissions from water and wastewater sector	Wastewater	Applicable to wastewater treatment facilities

Notes: CHEApet= Carbon Heat Energy Analysis plant evaluation tool; CO_2=carbon dioxide; GHG=greenhouse gas; LCA=life cycle assessment; UK=United Kingdom; UKWIR=United Kingdom Water Industry Research; UV=ultraviolet

5.5.1 CHEApet

According to Crawford et al. (2011), CHEApet was released by the Water Environment Research Foundation in 2010 and is available online for free (http://cheapet.werf.org). This web-based model quantifies the energy consumption and GHG emissions of WWTP and specific unit processes during the operational phase. CHEApet is a preliminary evaluation model that allows WWTPs to evaluate how operational modifications, process changes, or combined heat and power alternatives can reduce energy use and GHG emissions. The model was designed to compare potential alternatives at preliminary design stages. Consequently, results cannot be used for reporting requirements, and a detailed analysis of specific site conditions are recommended for an actual project.

CHEApet currently includes preliminary/primary treatment, solids processing, secondary treatment (eight activated sludge biological processes), and some advanced treatment processes (tertiary clarification, UV disinfection). In addition, emissions associated with facility electricity requirements (e.g., lighting and heating) are included. Outputs are shown on the interface but are also downloadable as Excel files to facilitate the comparison of alternatives. Emissions considered include the following:

- Scope 1: direct GHG emissions from unit processes, fuel use, and methane from fugitive sources
- Scope 2: indirect emissions from electricity for treatment processes, lighting, HVAC, and miscellaneous uses

- Scope 3: other indirect GHG emissions from outside of the utility yet related to the operation of the wastewater treatment facility (e.g., methanol, other chemical entered as methanol, sludge hauling).

This model excludes emissions from other secondary treatment processes not included in the default biological processes, phosphorus removal processes, and the production of other chemicals (e.g., polymers, flocculants, and disinfection chemicals).

5.5.1.1 System Boundary

The system boundary includes the operational phase of a WWTP. This includes primary, secondary, and some tertiary treatment processes in addition to solids handling, building energy requirements, chemical use, and transportation (e.g., sludge hauling, chemical transport). Current biological process configurations include (1) activated sludge; (2) anoxic–aerobic Modified Ludzack Ettinger; (3) pre-anoxic aerobic–post-anoxic aerobic (four-stage Bardenpho^{TM}); (4) anaerobic–aerobic; (5) anaerobic–anoxic–aerobic (A^2/O^{TM}); (6) anaerobic–anoxic–aerobic (University of Cape Town); (7) anaerobic–pre-anoxic aerobic–post-anoxic aerobic (Modified Bardenpho^{TM}); and (8) anoxic anaerobic–anoxic aerobic (Johannesburg; http://cheapet.werf.org).

5.5.1.2 Data Sources

Data sources for calculation procedures come from various published literature sources. This model also draws from Local Government Operational Protocol (LGOP) methods, IPCC guidelines, and the National Greenhouse and Energy Reporting System (NGERS) Submissions for the University of Queensland (Australia).

5.5.1.3 Model Inputs

Input parameters include the biological treatment process type, flows, concentrations, liquid processing data, solid processing data, and miscellaneous inputs. Users may select from eight pre-established secondary biological treatment processes. Influent water quality data typically collected at WWTP (e.g., BOD, total suspended solids, ammonia-nitrogen) and influent flow data are also required. The liquid processing section allows users to enter data related to the unit processes used (e.g., influent pumping, screening and grit removal, primary treatment, secondary treatment, tertiary filtration, UV disinfection, effluent pumping, odor control).

Inputs for tertiary treatment by filtration include pump data for filter pumps and blowers, such as number of pumps, flow capacity, total discharge head (TDH), pump efficiency, number of blowers, and power per blower. Inputs for UV disinfection include data on number of channels, banks, modules, and lamps installed, UV transmittance, and lamp power. Solid processing information can also be entered, including data associated with waste-activated sludge thickening, sludge stabilization, and sludge dewatering. Miscellaneous inputs are used to calculate thermal energy and include boiler efficiency and cogeneration inputs.

5.5.1.4 Method for Calculation

Users may select from three different calculation approaches: the LGOP approach (based on California entities and IPCC), the Australia approach (based on IPCC and NGERS), and the informal approach (based on IPCC and mass balance equations). The approaches vary in methodology used to estimate direct process emissions and are recommended for different purposes.

The LGOP approach is based on IPCC methods and is recommended for users interested in establishing an initial baseline of GHG emissions. This approach calculates CH_4 from digester gas and N_2O from population data and emission factors or, alternatively, BOD data and nitrogen uptake for cell growth. The Australia and informal approaches are recommended for utilities implementing process changes that want to assess pre- and post-effects on GHG emissions. The Australia approach relies on a COD mass balance, the fraction of anaerobic treatment, and CH_4 emission factors to estimate CH_4 emissions. N_2O is calculated from a mass balance of nitrogen or population and protein intake data.

The informal approach is considered more precise because direct emissions are calculated using mass balance equations as opposed to emission factors, with the exception of N_2O emissions, which are calculated with emission factors. The informal approach estimates combustion and non-combustion CO_2 emissions, anaerobic digestion and fugitive CH_4 emissions, and N_2O off-gas from nitrification and denitrification processes. The informal approach also includes biogenic CO_2 emissions from the combustion of biomass.

Calculations for indirect emissions differ as well. For indirect emissions, the LGOP method uses emission factors from the California Air Resources Board to estimate transportation emissions. Power emissions are calculated using factors from the California Climate Action Registry (CCAR) or 2004 eGRID factors obtained from the EPA Inventory of U.S. GHG Emissions and Sinks. The informal approach uses transportation emission factors from CCAR and power emission factors from 2004 eGRID. Factors and assumptions for the production of methanol are included in Crawford (2011). The Australia approach describes only process emissions.

5.5.1.5 Model Outputs

Outputs display both indirect emissions per source (e.g., biosolid hauling, power consumption, chemical transportation and use) and direct emissions per source (e.g., process CO_2 emissions, process N_2O emissions, process CH_4 emissions, biosolid incineration from CH_4 and N_2O) in tons of CO_2eq per year. Additional outputs include carbon footprint, electricity consumption, mass and calorific balance, and thermal consumption and potential thermal capture. It is important to note that carbon footprint results are estimates and not designed to adhere to formal reporting requirements.

5.5.1.6 Model Limitations

This model is limited to select activated sludge biological unit processes, tertiary clarification, and UV disinfection. It excludes unit processes such as phosphorus removal or chlorine disinfection and does not cover many tertiary and advanced treatment processes used in reuse projects (e.g., tertiary membrane filtration, RO, ozonation). It is also limited to operational-phase emissions, thereby excluding GHGs associated with construction and decommissioning activities. Finally, various assumptions are made for CHEApet's calculations, and results do not provide accurate, site-specific estimations of GHG emissions.

5.5.1.7 Applicability to Water Reuse and Desalination

This model is only applicable to water reuse facilities that use one of eight pre-established biological processes, tertiary filtration, and UV disinfection. Future versions of CHEApet will include upgrades such as the inclusion of biological and chemical phosphorus removal, step-feed biological nutrient removal (BNR), chlorine disinfection, and additional recovery technologies, according to the CHEApet tutorial (http://cheapet.werf.org). Given the limited amount of advanced treatment options available, this model is more applicable to wastewater treatment facilities. It is not applicable to desalination.

5.5.2 Environment Agency Model

The Environment Agency Model (Reffold et al., 2008) was developed by a UK governmental agency, the Environment Agency. This model uses LCA to assess the GHG emissions of various water supply options (including desalination, water reuse, and reservoirs) and demand management options (e.g., water meters and rainwater harvesting). This MS Excel model estimates the carbon footprint in CO_2 equivalents over a 60-year lifetime. Construction, manufacture, installation, and O&M phases are included. The cost of carbon is also assessed using Department for Environment Food and Rural Affairs (Defra) in UK guidance, which accounts for the shadow price of carbon (SPC). The SPC methodology values GHG emissions by accounting for the estimated cost of damage associated with each ton of CO_2 equivalents emitted. A sensitivity analysis of the price of carbon is included in this model.

The model was used to evaluate various desalination and water reuse options for water supply in the Reffold et al. (2008) study. Desalination options evaluated included RO, electrodialysis, nanofiltration, and offshore treatment of brackish water and seawater, whereas indirect water reuse options included conventional options, RO, reed bed, and nanofiltration. This study assessed 12 desalination schemes and 6 effluent reuse options.

5.5.2.1 System Boundary

The system boundary includes GHG emissions from construction and operation phases. During construction, emissions from transportation, materials, and manufacturing are considered. During operation, emissions from transportation, energy use, fuel use, and maintenance activities are included. Emissions from water supply and treatment, distribution, leakage, customer use, collection, and wastewater treatment are evaluated for demand and supply options.

5.5.2.2 Data Sources

Data sources include information from various publications. Energy use data are obtained from the Environment Agency and other manufacturers and organizations. Carbon emission data (e.g., conversion factors, embodied carbon of materials) come from the UK government, academic research, and other published data from manufacturers. See Reffold et al. (2008) for further detail on data sources.

5.5.2.3 Model Inputs

Inputs include water resource planning data, common water supply option data, and site-specific data. Default or user-defined data can be used for water resource plans. Common data from water supply options include frequency of operation and discount rates. Specific data include dimensions of tanks, pump capacity, and pipeline length and diameter.

5.5.2.4 Method for Calculation

This model uses an LCA methodology over the construction and operation phases. Life cycle GHG emissions are calculated by using the mass of materials and conversion factors for embedded carbon/mass data. A Defra guideline electricity conversion factor of 0.43 kg CO_2eq per kWh is used to estimate GHG emissions.

5.5.2.5 Model Outputs

Outputs are given as total tons of CO_2 equivalents and CO_2eq per volume of water supplied for each water supply scheme. The total CO_2eq per volume of water supplied is based on the water produced over a 60-year timeframe. Other outputs correspond to the cost of carbon. Plots of total carbon cost versus yield and total CO_2eq versus water supplied over the 60-year timeframe are displayed.

5.5.2.6 Model Limitations

Operational electricity is the only energy source considered; other fuel types (e.g., renewable sources, oil, and gas) are not considered. An electricity conversion factor of 0.43 kg CO_2eq per kWh is used to estimate GHG emissions from electricity consumption. This conversion factor is specific to a Defra guidance document and thus not transferable to the United States. In addition, this model does not directly assess the construction of new WWTPs. The GHG emissions from construction are assumed to be the same as water treatment plants because focus is on water supply and demand, not wastewater treatment. Finally, this model excludes the decommission-phase emissions.

5.5.2.7 Applicability to Water Reuse or Desalination

According to Reffold et al. (2008), the MS Excel model is available for free for academics and practitioners. For further information regarding this report, interested parties can contact the Environment Agency at enquiries@environment-agency.gov.uk. Data for water supply options come from actual data from water supply schemes, but users can modify specific data to make it applicable to a new facility. This model can be used to estimate GHG emissions, although it uses data specific to the United Kingdom. Because it uses UK data, this model is not geographically applicable.

An aspect of the model that could be applicable to water reuse and desalination is the inclusion of the GHG emissions produced on the demand side. This study highlights the importance of demand-side action (e.g., metering) to reduce GHG emissions associated with electricity usage because 89% of the emissions in the supply, use, and disposal system were found to be associated with water used in homes (Reffold et al., 2008).

5.5.3 Bridle and BSM2G Models

Corominas et al. (2012) discuss two models used to evaluate GHG emissions at a virtual wastewater treatment facility: a simple comprehensive model (Bridle Model) and a process-based model (BSM2G Model) applied to the Benchmark Simulation Platform No. 2 (BMS2). The Bridle Model uses empirical factors to estimate direct N_2O emissions from secondary treatment and a simplified approach to calculate direct CO_2 and CH_4 emissions from an anaerobic digester. This simplified approach is based on assumptions related to the production of biogas and the CH_4 and CO_2 content. In contrast, the BSM2G Model uses a process-based approach to calculate direct N_2O and CH_4 emissions from secondary treatment (e.g., activated sludge) and sludge processing (e.g., anaerobic digestion) based on the mechanistic generation of these emissions. This mechanistic approach describes direct GHG emissions dynamically.

In addition, steady-state and dynamic simulations were applied to the process-based BSM2G Model to assess their impact on GHG emissions. Steady-state simulations represent a flow-based average, whereas dynamic simulations account for changes in the system over time. This model uses software to run both simulations. This study illustrates the benefits of dynamic, process-based GHG models that account for changes in the system (e.g., hydraulic load, influent water quality, temperature, operational modifications) in contrast to simple comprehensive and steady-state GHG estimation techniques that do not capture dynamic shifts in emissions.

The virtual WWTP under investigation consists of primary and secondary clarification, activated sludge, a thickener, anaerobic digestion, a storage chamber, and dewatering. For both models, emission sources come from secondary treatment (direct N_2O and biogenic CO_2), sludge treatment (direct CH_4 and biogenic CO_2), net power usage, chemical usage, and sludge disposal and reuse. These models focus on operational-stage GHG emissions only and express emissions as CO_2eq.

5.5.3.1 System Boundary

The system boundary includes operational stage GHG emissions from a virtual WWTP. This includes secondary treatment, sludge treatment, power usage, chemical production, and sludge handling.

5.5.3.2 Data Sources

Data sources include a wide range of previous literature in which Bridle et al. (2008) and Monteith et al. (2005) provide details on the Bridle Model. The BSM2G Model is based on a modified activated sludge model described by Hiatt and Grady (2008), Nopens et al. (2009), and other sources (Corominas et al., 2012).

5.5.3.3 Model Inputs

Model inputs include influent water quality concentrations (e.g., organic load), influent flow, sludge retention time, hydraulic retention time, dissolved oxygen, mixed liquor suspended solids, volatile suspended solids, influent and effluent anaerobic digester flow, energy consumed, and chemicals used.

5.5.3.4 *Method for Calculation*

The Bridle and BSM2G models calculate GHG emissions from direct secondary treatment, sludge processing, net power, embedded chemicals, and sludge disposal and reuse. Calculations for net power, embedded chemicals, and sludge disposal and reuse are the same for both models; however, calculations for N_2O emissions from secondary treatment and N_2O and CH_4 emissions from sludge processing differ.

The Bridle Model calculates direct secondary treatment GHG emissions from biomass respiration, BOD oxidation, and N_2O emissions. CO_2 emissions generated from biomass respiration and BOD oxidation are calculated with factors of 1.947 kg CO_2/kg biomass respired endogenously and 1.1 kg CO_2/kg O_2 consumed for the degradation of BOD. A factor of 0.004 kg N_2O/kg N to aeration is used to calculate N_2O emissions.

A CO_2 credit from nitrification is also accounted for because nitrifying organisms consume CO_2. Factors of 0.308 kg CO_2/kg N nitrified are used to estimate credit nitrification (CO_2 consumption from nitrifiers).

The Bridle Model also calculates CO_2 and CH_4 emissions from sludge treatment, CO_2 produced from net power consumption, embodied CO_2 in chemicals, and CO_2 emissions from sludge disposal and reuse. For sludge treatment, assumptions are made on the production of biogas to estimate the CH_4 content of the biogas produced and the CO_2 produced when burning CH_4 gas. Net power is calculated by first estimating the energy consumption of equipment (e.g., aeration, mixers) using an operational cost index from previous literature. The energy content of CH_4 gas represents an energy credit, which is then subtracted from the energy consumed. After that, a conversion factor of 0.94 kgCO_2kWh is used to convert the net power to CO_2 emissions. An emission factor of 1.54 g CO_2/g methanol, expressed as COD obtained from previous literature, is used to calculate the CO_2 emissions from chemical production. Finally, transportation effects of hauling biosolids and carbon mineralization are calculated using Bridle et al. (2008) assumptions.

The BSM2G Model differs in its use of process-based models to estimate CH_4 and N_2O emissions for secondary treatment and sludge processing. For secondary treatment, a modified version of an activated sludge model (Hiatt and Grady, 2008) was developed. This model includes the reaction parameters specific to the denitrification process to calculate N_2O emissions generated in the anoxic stages of a modified Ludzack-Ettinger activated sludge unit. In addition, previous literature was used to incorporate the impact of temperature on microbial growth and the consumption of ammonia species as substrates by nitrifying organisms (e.g., ammonia oxidizing organisms and nitrite oxidizing organisms).

By modifying previous literature (Nopens et al., 2009), Corominas et al. (2012) link the activated sludge model and a sludge processing model for the BSM2G Model. This is accomplished by considering additional variables, such as COD, nitrogen, charge balance equations, and denitrification process species (e.g., NO, NO_2^-, N_2O, N_2), as well as autotrophic biomass. The COD consumed by nitrogen species during denitrification is subtracted from the total COD in the activated sludge system prior to going to the anaerobic digester. Upon transferring to the digester, only NO_3^- is considered because N_2 stripping is assumed to occur. Finally, steady-state (flow-based average) and dynamic simulations are run to calculate GHG emissions for the BSM2G Model for varying anaerobic digester volumes. For further details on calculations, refer to supplementary material provided by Corominas et al. (2012).

5.5.3.5 Model Outputs

Results are expressed in CO_2eq by converting CH_4 and N_2O emissions using their 100-year GWP (IPCC, 2006). This study shows that the N_2O emissions varied by a factor of three between simple comprehensive (Bridle Model) and process-based models (BSM2G). CH_4 emissions varied only by 4%. Also, average dynamic and steady-state simulations were similar, though variability in N_2O arose with changes in influent water quality (C/N ratio) and temperature. The steady-state and simple comprehensive models could not capture variations in N_2O dynamics.

5.5.3.6 Model Limitations

Both of these models have limitations. For example, calibration and validation have not yet been conducted using experimental data from a full-scale treatment facility. This would verify the accuracy of these GHG estimation methods. Another limitation is the exclusion of N_2O produced during aerobic treatment because only N_2O emissions from the anoxic phase of activated sludge are considered. In addition, whereas the BSM2G Model has dynamic simulation capability, neither the Bridle Model nor the BSM2G Model allows for capturing dynamic variations in water quality parameters (e.g., pH) or operating conditions in the estimation of CO_2 emissions from biomass respiration and the oxidation of BOD. These calculations could be modified to account for variations in pH and oxygen uptake rates. In addition, this model assumes that CH_4 that returns to the anoxic tank is completely removed, and N_2O emissions from land application of biosolids are not considered. Other limitations are related to the simple comprehensive versus dynamic process-based approaches. The simple comprehensive model cannot capture process changes and partially captures structural changes. In contrast, a limitation to the dynamic process model is its complexity and high computational power requirements.

5.5.3.7 Applicability to Water Reuse or Desalination

The BSM2G Model illustrates the benefits of dynamic process-based models that consider changes in operation. Models that account for changes in loads, temperature, and other operating parameters can lead to a more accurate GHG model, compared to GHG estimates based on empirical factors or steady-state averaged conditions. Dynamic process-based models can thus be used for optimization to mitigate GHG emissions. The simple comprehensive model (Bridle Model) does not capture variability in operating conditions and is more appropriate for process design.

These models are applicable to traditional WWTPs using activated sludge and anaerobic digester systems. No tertiary treatment modeling is provided, so these models are applicable only to water reuse facilities for modeling of GHG emissions from secondary treatment processes (e.g., activated sludge), sludge processing, net power, chemicals, and sludge disposal and reuse. Because these models focus on traditional wastewater treatment, they are not applicable to desalination facilities.

5.5.4 System Dynamics Model

System dynamics is a method that is used to determine the effects of a certain policy option based on the relationships between the system’s different components. This method has been used in the research by Shrestha et al. (2011, 2012) to compare the carbon footprints of two supply options: the transport of groundwater pumped from counties in northern Nevada to the

Las Vegas Valley and the transport of desalinated water from California. It was also used to study the future impacts in energy use and CO_2 emissions in 2035 if the amount of water reused were to increase. The studies were carried out at the University of Nevada, Las Vegas.

A system dynamics model consists of four types of components: stocks, flows, connectors, and converters. Stocks refer to something that can accumulate. Flows are components responsible for filling and depleting stocks. Connectors determine the relationship between components, and converters can contain data such as tables and equations that may require inputs to produce a certain output (Shrestha et al., 2011). This method has been applied to model the effects of the implementation of desalination (Shrestha et al., 2011) and water reuse (Shrestha et al., 2012).

This model comprises two sectors: water supply and energy and carbon footprint. The water supply and energy sector is based on the Las Vegas Valley's water management system, which draws water from Lake Mead and, through pumping, transports it to different reservoirs and laterals (connections to communities), and then distributes it to users. This sector also calculates the energy requirement through use of an equation for pumping power, which takes into account the head loss due to friction. Some inputs include population data, per capita water demand, lake level, water use rate (indoor and outdoor), amount of treated wastewater reused, urban runoff, energy mix, and emission rates. The carbon footprint sector uses the energy requirement of the system, information on the designated energy mix, and emission factors to estimate the CO_2 emissions. Because this model was developed for the Las Vegas Valley water management system using the region's water network, it is not geographically applicable to water reuse or desalination facilities in other regions.

5.5.5 GPS-X

GPS-X is a commercially available modeling and simulation software for wastewater treatment facilities. Hydromantis Environmental Software Solutions, Inc. (Hamilton, Canada) recently partnered with Mohawk College to develop a carbon footprint model that coincides with the software. The release of GPS-X 6.2 in December 2012 included a carbon footprint model. GPS-X is a dynamic process simulator used to design and optimize wastewater treatment processes through mechanistic modeling of the removal of carbon and nutrients. The carbon footprint modeling is an extension of the dynamic process simulator used by GPS-X (Goel et al., 2012).

According to Goel et al. (2012), this model includes emissions from biological unit processes (e.g., aerobic and anaerobic CO_2 emissions, anaerobic CH_4 emissions), offsets from process emissions (e.g., biogenic CO_2 sources, conversion of CH_4 when burning), emissions associated with the consumption of energy, offsets due to the recovery of energy, emissions from chemicals and material manufacturing, and transportation. Offsets that are due to fertilizers and carbon sequestration from land use will be incorporated in future versions of this model. This model can be used to evaluate how process changes affect GHG emissions. In addition, this software allows users to enter site-specific electricity mixes, so it could be applied to various geographical locations.

The GPS-X model focuses on wastewater treatment processes, so it is not applicable to water reuse or desalination facilities. Its applicability is limited to water reuse facilities that have wastewater treatment processes included in the GPS-X model.

5.5.6 Carbon Accounting Workbook

The UK Water Industry Research (UKWIR) is an organization that conducts research for water and wastewater operators in the United Kingdom. Through a partnership with Water UK and Carbon Trust, it developed the Carbon Accounting Workbook, which includes an MS Excel spreadsheet originally developed in 2004 to quantitatively estimate operational GHG emissions in the water sector. In 2007, additional research was conducted to improve the model by developing reports in three phases: (1) updating the existing MS Excel–based model, (2) providing guidelines to account for embodied carbon, and (3) accounting for non-CO_2 emissions (CH_4 and N_2O). Reports on Phases 1 and 2 are available to UKWIR members at no cost and to non-members at £500 and £1000. The fifth and latest version replaces previous iterations of the workbook and reports and is available for £250 (UKWIR, 2008).

Limited information on this model is available online, and it is unclear from the Website if the model is applicable to water reuse and desalination. It appears to be for water treatment facilities and thus is not pertinent to this project. The WateReuse Research Foundation is an international collaborator of this organization, so perhaps additional information about this model can be formally requested.

5.5.7 mCO2

The mCO2 model is a commercially available emission model that quantifies direct and indirect GHG emissions for the wastewater sector. The proprietary software uses a GHG Protocol methodology and was designed to help utilities reduce their carbon footprint to meet emission regulations. The price varies depending on the application, and this model can be used in the United States or United Kingdom.

The mCO2 model accounts for Scope 1 and Scope 2 emissions. Scope 1 emissions are calculated for biogenic carbon, N_2O, and CH_4 from WWTPs using EPA equations, which are consistent with IPCC calculations. In Scope 2, indirect emissions associated with electricity consumption are also taken into consideration. The model accounts for purchased electricity as well as the electricity mix. This model has no pre-configured unit processes, and user inputs are typically in kWh or btu for electricity usage. Fleet vehicles are also included, and user inputs for commercial vehicle use include fuel type and annual consumption of fuel.

The software generates a report that identifies direct and indirect emissions by asset and critical gap areas to meet emission criteria. Limited information is available about this model online (MWH Global, 2012). The model is not specifically designed for water reuse or desalination facilities. Users can, however, use this model if input data are available.

Chapter 6

Information Gap Identification

This chapter discusses the system boundaries, emissions considered, limitations, and applicability of the models reviewed to highlight pertinent information gaps in the current research on water reuse and desalination GHG estimation models. Currently, there are aspects from the LCA models, hybrid LCA models, specific models, and other related models that could be applied to water reuse and desalination. LCA models and other related models have limited applicability and are, therefore, discussed only briefly. The hybrid LCA models and specific models appear to provide the most applicable options and are the focus of this section.

Despite the transferability of certain aspects of these models, there are major information gaps preventing the development of a robust GHG estimation model specific to water reuse and desalination. An accurate, transferable model should be user friendly, geographically adaptable, require minimal inputs, and have the capability to estimate GHG emissions from advanced water reuse and desalination treatment technologies. This section provides further insight into gaps and limitations in the current body of literature, as well as applicable aspects from existing models that could be used to develop a GHG estimation model specific to water reuse and desalination.

6.1 LCA Models

The LCA models reviewed in Chapter 2 cover a wide range of literature sources that investigated GHG emissions from water reuse and desalination facilities. LCA software, such as Gabi and SimaPro, were used to investigate the life cycle emissions associated with various water treatment technologies over the construction, operation, and decommission phases. Some examples of transferable aspects of the LCA models include (1) the wide range of emission sources considered (e.g., material production, fuel consumption, sludge disposal, chemical production); (2) the inclusion of direct and indirect GHG emissions; and (3) the assessment of fertilizer abatement potential for agricultural reuse. Refer to Chapter 2 for further details on LCA models.

Although LCA software provides a useful model for assessing the GHG emissions of water reuse and desalination facilities, the methodologies, system boundaries, emission sources considered, data sources, and output emissions analyzed varied across studies. Because of variability in these factors, traditional LCA studies reviewed may not provide a model that is readily available for widespread adoption to water reuse and desalination facilities; they focus on specific case studies, hypothetical scenarios, or both. Using LCA can also be time- and data-intensive, and input parameters may not be readily available.

Some of the information gaps in the existing LCA models include their reliance on LCA databases that do not provide site-specific data, subjective selection of a system boundary, and the variation in parameters selected to estimate GHG emissions.

6.2 Hybrid LCA Models and Specific Models

This section summarizes some of the model attributes for hybrid LCA and specific models. Hybrid LCA models (WEST, WWEST, WESTWeb) and the specific models (Johnston Model and Tampa Bay Water Model) differ in terms of their system boundaries, emission sources considered, GHG emissions considered, limitations, and applicability. The following section discusses the commonalities and differences of these models and identifies knowledge gaps that need to be addressed to develop an accurate, robust, and transferable model.

6.2.1 System Boundary

Table 6.1 provides a comparison of the life stages considered by the hybrid LCA and specific models reviewed in this report. The system boundary for hybrid LCA models includes the construction, operation, and maintenance phases and upstream supply-chain processes (e.g., material extraction, material provision, and manufacturing). Both the Johnston Model and Tampa Bay Water Model focus solely on the operational life stage. This is the result of a fundamental difference in approach to GHG emission modeling, in which hybrid LCA models consider life cycle emissions and the specific models do not. None of the models include decommission within the system boundary.

Table 6.2 shows the water supply stages considered by both model types. Both hybrid LCA and specific models evaluated consider collection, treatment, and distribution water supply phases. None of the models separate GHG emissions during treatment by unit process (e.g., primary treatment, secondary treatment, tertiary treatment, disinfection). The separation of GHG emissions by unit process would allow utilities to identify high impact areas and focus mitigation efforts specifically on those. The separation of tertiary treatment processes, for example, could facilitate the comparison of GHG emissions from different tertiary treatment options (RO, nanofiltration).

Table 6.1. Life Cycle Stages Considered by Hybrid LCA and Specific Models

Life Cycle Phases Considered	Hybrid LCA Models			Specific Models	
	WEST	WWEST	WESTWeb	Johnston Model	Tampa Bay Water Model
Construction	X	X	X		
O&M	X	X	X	X	X
Decommission					

Notes: LCA=life cycle assessment; O&M=operation and maintenance; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

Table 6.2. Water Supply Phases Considered by Hybrid LCA and Specific Models

Water Supply Phases Considered	Hybrid LCA Models			Specific Models	
	WEST	WWEST	WESTWeb	Johnston Model	Tampa Bay Water Model
Collection	X	X	X	X	X
Treatment	X	X	X	X	X
Distribution	X	X	X	X	X

Notes: LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

6.2.2 Emission Sources Considered

The major differences in emission sources considered lie in material production and delivery. Neither of the specific models considers the upstream processes needed to produce and deliver materials; however, hybrid LCA models do consider these upstream emissions. Table 6.3 summarizes the emission sources considered by hybrid LCA models and specific models.

The only sources of CO_2 and GHG emissions considered by all of the hybrid LCA and specific models are those associated with electricity consumption; however, hybrid LCA models consider both construction and operation life stages, whereas the specific models consider only the operation life stage. In assessing electricity consumption, all of the models account for the electricity mix, which is important to accurately estimate GHG emissions in a specific location. The Johnston Model and all of the hybrid LCA models allow users to select a custom, state, or national electricity grid for the United States. A custom electricity mix specific to the energy consumption of a water reuse or desalination facility is the most accurate, followed by the state mix and the national mix.

The electricity mix is important because it can significantly impact the GHG emissions (e.g., mixes dominated by renewable sources have a lower GWP than fossil fuel–based electricity mixes). It is important to note that the hybrid LCA and Johnston Model have eGRID data embedded in the model, which facilitates the selection of an accurate electricity mix. The national database eGRID is compiled by the EPA and includes CO_2, CH_4, and N_2O emission rates, electricity mixes, and other environmental data for energy generating facilities throughout the United States (USEPA, 2012b). For these models, users can simply enter the location of the facility to obtain the correct statewide mix. The Tampa Bay Water Model was developed specifically for the Tampa Bay region with electricity mixes from three different utilities. This model could easily be modified, however, to apply it to another region using data provided by eGRID. Also, the hybrid LCA models allow users to consider life cycle emissions or emissions directly from the smokestack for the generation of energy.

As seen in Table 6.3, electricity consumption during the operation stage is the only source considered in the Tampa Bay Model. This differs from the Johnston Model and hybrid LCA models, which consider other emission sources (e.g., fuel consumption, chemical production). The Johnston Model and the hybrid LCA models (WEST, WWEST, and WESTWeb) consider emissions from fuel consumption, including on-site fuel consumption and fleet vehicles. The hybrid LCA models and the Johnston Model also consider sludge disposal and chemical production, although the emission factors used in the Johnston Model for sludge are from a potable water system.

The hybrid LCA models and the Johnston Model also consider direct process emissions; however, there are important distinctions for these emission sources. The WWEST model estimates emissions from various wastewater treatment processes based on water quality data and population served. The direct emissions accounted for in the Johnston Model come from published emission factors for potable water production. These Johnston Model emission factors are, therefore, not applicable to water reuse or desalination facilities, whereas the WWEST data provide a more accurate method to estimate direct CH_4 and N_2O emissions using IPCC estimation equations.

Process equipment and disinfection processes also vary between the hybrid LCA model and the Johnston Model. Some examples of relevant equipment and processes for the hybrid LCA approach include filter media, membranes, and UV disinfection. The Johnston Model allows users to estimate emissions associated with the energy consumption of processes such as MF/UF, UV, RO, and thermal desalination based on the average flow rate. A combination of hybrid LCA and specific model estimation methods could be used to develop a more robust, accurate, and transferable GHG emission model specific to water reuse and desalination.

Table 6.3. Sources of CO_2 and Other GHG Emissions Considered by Hybrid LCA and Specific Models

Emission Sources Considered	Hybrid LCA Models			Specific Models	
	WEST	WWEST	WESTWeb	Johnston Model	Tampa Bay Water Model
Material production	X	X	X		
Material delivery	X	X	X		
Electricity consumption	X	X	X	X	X
Electricity mix	X	X	X	X	X
Fuel use (on-site and fleet vehicles)	X	X	X	X	
Sludge disposal	X	X	X	X[1]	
Chemical production	X	X	X	X	
Direct process emissions		X[2]	X[2]	X[1]	
Process equipment			X[3]	X[4]	
Disinfection processes			X[3]	X[4]	

Notes: 1=direct emission factors for ozone generation, granular activated carbon, reservoirs, and sludge disposal from potable water production; 2=direct emission for various wastewater treatment processes (Section 5.3.2); 3=includes filter media (sand, gravel, anthracite, or other coal product), membranes, pumps, fans/blowers, motors and generators, turbines, metal tanks, UV lamps/lights, other industrial equipment, electrical, controls; 4=utilities can estimate energy consumption from mixers, flocculators, settlers, DAF, filtration, microfiltration/ultrafiltration, ultraviolet, ozone, hypochlorite, decarbonators, reverse osmosis, and thermal desalination by entering the average flow rate; LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

Table 6.4. GHG Output Emissions for Hybrid LCA and Specific Models

GHG Output Emissions	Hybrid LCA Models			Specific Models	
	WEST	WWEST	WESTWeb	Johnston Model	Tampa Bay Water Model
CO_2eq	X	X	X	X	X
CO_2				X	X
N_2O				X	X
CH_4				X	X

Notes: CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalent; GHG=greenhouse gas; LCA=life cycle assessment; N_2O=nitrous oxide; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

6.2.3 GHG Output Emissions

All models consider CO_2, N_2O, and CH_4 emissions in their calculations; however, the specific models (Tampa Bay Water Model and Johnston Model) present CO_2, N_2O, CH_4, and CO_2eq in the results. The hybrid LCA models present only CO_2eq as GHG emission results.

One key advantage to the Johnston Model is its presentation of results as Scope 1 (direct), Scope 2 (indirect), and Scope 3 (other indirect) emissions. This is consistent with published protocols for GHG classifications (e.g., LGOP and WRI/WBCSD GHG Protocol Corporate Standard). Existing and voluntary GHG reporting programs include Scope 1 and Scope 2 emissions, as will future required regulations or cap-and-trade programs (Huxley et al., 2009). The Tampa Bay Water Model considers only Scope 2 emissions from electricity consumption; however, this model does not present the results in this manner.

The hybrid LCA models include all three scopes, but they are not designed to separate direct emissions (Scope 1), indirect electricity consumption produced off-site (Scope 2), and other indirect emissions related to the treatment facility (e.g., chemical production and delivery) (Scope 3). This is a major disadvantage to the WEST, WWEST, and WESTWeb models because the results are not consistent with voluntary, mandatory, or projected future reporting requirements; however, more consistent methods to report carbon emissions are beginning to appear. For example, the Climate Registry (http://www.theclimateregistry.org) is a nonprofit collaboration that sets consistent and transparent standards to calculate, verify, and publicly report GHG emissions into a single registry. Table 6.4 compares the presentation of GHG emission results for hybrid LCA and specific models analyzed.

6.2.4 Data Sources

The hybrid LCA models (WEST, WWEST, and WESTWeb) and specific models (Johnston Model and Tampa Bay Water Model) all use eGRID electricity mix data and emission factors to estimate emissions associated with the consumption of electricity. The most up-to-date eGRID data available come from eGRID2007 Version 1.1, which provides GHG emission rate outputs for CO_2, N_2O, and CH_4 in lbs per GWh for different eGRID subregions throughout the United States (USEPA, 2012b).

In an effort to use more recent emission factors, the Tampa Bay Water Model also used 2010 emission factors from the CAM data program, which was recently replaced by the Air Markets Program Data Program (USEPA, 2012c). CAM does not provide N_2O emission factors, and eGRID is more commonly used to estimate GHG emissions in the United States. The hybrid LCA models and the Johnston Model also draw from the EPA for emission factors associated with fuel consumption. These are the only data sources that the hybrid LCA and specific models have in common.

The hybrid LCA models also draw from LCA databases, such as EIO-LCA and Gabi to estimate the life cycle emissions of materials, chemicals, equipment, energy, and transportation. Other hybrid LCA data sources include manufacturer Websites (e.g., Caterpillar) and the California Air Resources Board for additional equipment information. Finally, factors used to estimate emissions from wastewater treatment processes come from an IPCC method (IPCC, 2006); these are the same equations used in other wastewater treatment GHG estimation models (e.g., mCO2 and CHEApet).

These data sources differ from the Johnston Model, which relies primarily on factors from other literature sources to estimate emissions associated with treatment technologies and chemicals. Direct emissions from potable treatment processes, such as ozone generation, GAC regeneration, reservoirs, and sludge disposal, are calculated using emission factors from Huxley et al., (2009). Johnston (2011) also used energy use factors from Veerapaneni et al. (2011) for pressure filtration, MF/UF, UV, ozonation, RO, and other technologies and chemical production emission factors from Tripathi (2007).

These data sources vary because of vast differences in GHG estimation approaches. This makes sense because hybrid LCA models focus on life cycle emissions, and the specific models focus on operational GHG emissions. The only source common to all three models is eGRID data associated with electricity mix information and corresponding emission factors for electricity consumption. The Johnston Model and hybrid LCA model also used EPA emission factor data for fuel consumption. Aside from that, data sources for chemicals, sludge disposal, and equipment use vary between the Johnston Model and hybrid LCA models, whereas these sources are not considered in the Tampa Bay Water Model. This is caused by differences in the focus of the study, where the Johnston Model focused on potable water systems, and the hybrid LCA includes water and wastewater with applicability to water reuse and desalination. Differences in data sources also emerge from alternative system boundaries, water supply phases, and emissions considered.

6.2.5 Limitations

Table 6.5 shows a side-by-side comparison of the limitations associated with hybrid LCA models and the specific models. The nature of LCA requires a large amount of data to conduct a comprehensive analysis using WEST and WWEST.

WESTWeb is less data intensive but still requires a large amount of inputs for full utilization. Users are not required to enter all of the inputs and have flexibility on what data to enter; however, the arbitrary selection of data inputs could lead to difficulties in comparing systems comprehensively and consistently.

Some utilities may not have or collect input data used in the hybrid LCA models. For example, correspondence with San Elijo Water Reclamation Facility (3-MGD capacity) and Miami-Dade Water and Sewer Department Water Reclamation Facility (10-MGD capacity) indicated that both facilities do not currently collect all the data necessary for hybrid LCA models. (See Chapter 7 for further information on the ability to implement models for all utility partners.) Correspondence from the San Elijo facility indicated that only data related to electricity, flow, and water quality were available, whereas the Miami-Dade facility was able to provide only flow data. The lack of input data collected in practice could be a limitation to the successful implementation of the hybrid LCA models. Other considerations are that the inputs for the hybrid LCA models may not be consistent with the functionality of a utility in practice, and certain utilities may be more interested in operational emissions because these emissions are more significant for energy-intensive systems.

Table 6.5. Limitations for Hybrid LCA and Specific Models

Limitations	**Hybrid LCA Models**			**Specific Models**	
	WEST	**WWEST**	**WESTWeb**	**Johnston Model**	**Tampa Bay Water Model**
Requires understanding of hybrid LCA	X	X	X		
Data-intensive inputs	X	X	X		
Inputs not consistent with utility functionality	X	X	X		
Exclusion of life cycle emissions				X	X
Exclusion of decommission phase	X	X	X	X	X
Exclusion of administrative activities	X	X	X		X
No separation of direct and indirect emissions	X	X	X		X
Exclusion of model calibration	X	X	X	X	X
No separation of specific unit processes	X	X	X	X	X
No Web-based tool				X	X

Notes: LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web.

The Johnston Model and Tampa Bay Water Model, on the other hand, have fewer inputs than the hybrid LCA models. The specific models require fewer inputs because they focus on the operational phase only. This could be a disadvantage for utilities interested in life cycle emissions associated with construction or expansion projects. None of the models consider the decommission phase, and the hybrid LCA and Tampa Bay Water models exclude emissions from administrative buildings. Tracking the GHG emissions of administrative buildings may be useful for utilities interested in minimizing emissions through energy conservation efforts in other areas related to treatment.

The hybrid LCA models and the Tampa Bay Water Model do not separate direct and indirect emissions. In the case of the hybrid LCA models, the results can be separated by water supply phase (e.g., supply, treatment, and distribution), life cycle stage (construction, operation, maintenance, and decommission), and activity (material production, material delivery, equipment use, energy production, and sludge disposal); however, these categories do not distinguish between direct and indirect emissions. For the Tampa Bay Water Model, direct emissions from the treatment processes and Scope 3 emissions are not considered. The Johnston Model is the only model that separates direct and indirect emissions, including Scopes 1, 2, and 3 emissions.

Model calibration is not available for any of the hybrid LCA or specific models reviewed. A sensitivity analysis was conducted on the WEST model (Stokes and Horvath, 2006), but this feature is not embedded in the MS Excel or Web-based versions. Model calibration is important to assure that GHG estimation methods accurately quantify emissions.

Another limitation common to both model types is the omission of GHG quantification for specific unit processes. The hybrid LCA models and specific models can distinguish between collection, treatment, and distribution but are currently not set up to track GHG emissions from unit processes with a treatment train (e.g., secondary and tertiary treatment). Given input data availability, the hybrid LCA models could be used to assess specific unit processes.

A robust model would benefit from the tracking of GHG emissions of specific unit processes because this would allow utilities to identify high impact areas and make efforts to minimize GHG emissions in those areas. This would enable the comparison of GHG emissions from combinations of secondary treatment (e.g., activated sludge), tertiary treatment (e.g., RO), and disinfection (e.g., chlorination, UV, ozone) alternatives for water reuse or desalination facilities.

WESTWeb is a user-friendly model because of its Web interface and public accessibility. WEST and WWEST are incorporated into WESTWeb with less flexibility to customize certain data inputs. WEST, WWEST, Tampa Bay Water Model, and the Johnston Model are all MS Excel spreadsheets.

Table 6.6. Applicability of Hybrid LCA and Specific Models

Applicability and Availability	Hybrid LCA Models			Specific Models	
	WEST	**WWEST**	**WESTWeb**	**Johnston Model**	**Tampa Bay Water Model**
Previously applied to water reuse	X				
Previously applied to desalination	X				X
Designed for wastewater facilities		X			
Designed for water facilities				X	
Designed for regional water supply					X
Currently regionally transferable	X	X	X	X	
Custom, state, and national electricity mix	X	X	X	X	
Availability upon request	X	X	X	X	X

Notes: LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web

6.2.6 Applicability to Water Reuse and Desalination Comparison

Table 6.6 summarizes the applicability of hybrid LCA models and specific models to water reuse and desalination facilities. The table shows that the WEST model was previously used to estimate GHG emissions from water reuse and desalination facilities (Stokes and Horvath, 2006, 2009), whereas the Tampa Bay Water Model was used to estimate GHG emissions from a regional water supply system that included groundwater, surface water, and desalinated sources. WWEST was designed for wastewater treatment facilities and includes direct emissions from certain wastewater treatment processes. This model appears to be more beneficial to water reuse facilities concerned with estimating direct emissions from select wastewater treatment processes. The Johnston Model was designed for water treatment facilities but includes some disinfection and desalination processes that could be useful for water reuse or desalination facilities.

The Tampa Bay Water Model is the only model listed in Table 6.6 that is not regionally transferable because of its focus on eGRID data specific to the Tampa Bay region. This differs from the hybrid LCA models and the Johnston Model, which permit users to select a custom, state average, or national average electricity mix, allowing for greater geographical adaptability.

All of the hybrid LCA and specific models are either in the public domain or available upon request. The hybrid LCA Web-based model can be used online, and the MS Excel versions of WEST and WWEST can be requested by emailing the developers. The Johnston Model is from a master's thesis that could be available on request. The Tampa Bay Water Model is an internal document that could be available to utilities interested in obtaining further information on this method.

6.2.7 Additional Features

Table 6.7 includes some additional features that could be useful to the development of an accurate, robust, and transferable model. The Johnston Model has the distinct feature of using benchmarking techniques to establish energy estimation equations. These equations facilitate the estimation of GHG emissions for utilities that do not have these data readily available. Benchmarking techniques would be beneficial to the creation of a transferable model for water reuse and desalination applications. The Johnston Model also has a water–energy nexus tool that allows users to estimate the net water production based on water produced at the treatment facility and water consumed to produce electricity by the power provider.

A distinct feature of the Tampa Bay Water Model is the estimation of demand-side energy consumption. This could be useful for water reuse and desalination facilities interested in integrated water management. The Tampa Bay Water Model also has an energy data collection system that collects monthly electricity consumption data throughout its collection systems, treatment facilities, and distribution systems.

Finally, the WEST model has a water stress indicator tool specific to California that is helpful in identifying areas that are currently under water stress. An expansion of this tool nationally or internationally could provide useful data related to the locations in which water reuse and desalination are most needed.

Table 6.7. Additional Features Included in Hybrid LCA and Specific Models

Additional Features	Hybrid LCA Models			Specific Models	
	WEST	WWEST	WESTWeb	Johnston Model	Tampa Bay Water Model
Energy estimation equations[1]				X	
Quantification of demand-side emissions					X
Water–energy nexus tool				X	
Water stress indicator tool	X				

Notes: 1=Energy estimation equations are specific to potable water production; however, this could be applied to a water reuse or desalination model; LCA=life cycle assessment; WEST=water energy sustainability tool; WWEST=wastewater energy sustainability tool; WESTWeb=WEST for the Web.

6.3 Other Related Models

Other related models were determined to be less applicable (compared to hybrid LCA and specific models) but with certain applicable attributes. The only other related model that specifically assessed water reuse and desalination is the United Kingdom Environment Agency Model, which uses UK emission factors for electricity and thus does not account for regional variations in electricity mixes. The other models reviewed in this section focused on estimating emissions from conventional water or wastewater treatment facilities and were not transferable to unit processes specific to water reuse and desalination. Some of the attributes from these models that could be useful in the development of an accurate, robust, and transferable model are as follows:

- CHEApet is a Web-based model containing some capabilities for GHG estimation of tertiary filtration and UV disinfection. Future versions of CHEApet will include biological and chemical phosphorus removal, step-feed BNR, and chlorine disinfection GHG estimation abilities. The inclusion of tertiary treatment and disinfection GHG emission estimations would be useful to a water reuse or desalination model. The Web-based interface is also beneficial to transferability and user friendliness of a model.

- The UK's Environment Agency Model includes demand-side GHG estimation techniques. SPC is also assessed in this model, which values GHG emissions by accounting for the estimated cost of damage associated with each ton of CO_2 equivalents emitted. Both demand-side estimation techniques and the SPC methodology could be applicable to a water reuse or desalination utility interested in integrated water management and economic considerations.

- The BSM2G Model provides a dynamic, process-based model that captures variations in operating conditions, temperature, and influent loads over time. Modeling desalination unit processes or tertiary treatment processes for water reuse could be beneficial to a robust model.
- Future versions of GPS-X will include offsets because of fertilizers and carbon sequestration from land use. In addition, because this software coincides with a wastewater treatment process modeling program, it can be used to evaluate how process changes affect GHG emissions. The GPS-X GHG model was also tested against GHG data from a wastewater treatment facility. Using model calibration techniques, it was able to coincide with real data, thus validating the accuracy of the GHG estimation results. This is the only model that used calibration techniques, which would be useful to the development of a robust water reuse or desalination estimation tool.
- mCO2 is a user-friendly software that automatically produces a report identifying critical areas to meet emission criteria. User-friendly software is a crucial element to the successful development of a GHG model for water reuse and desalination facilities.

These transferable attributes from other related models are important to consider for the development of a future water reuse and desalination model.

The limitations and gaps of these models are not discussed in detail because they are beyond the focus of this study, being less applicable to water reuse and desalination compared to hybrid LCA and specific models. Generally, it can be stated that the other related models have limitations associated with geographical adaptability and the primary focus of the model (e.g., specific to water or wastewater).

6.4 Knowledge Gaps

A wide range of knowledge gaps was identified throughout the literature review. Some of the major gaps in knowledge that would prevent a robust, accurate, and precise GHG estimation model for water reuse and desalination facilities are listed herein. Further research in these areas is needed to develop a comprehensive model to estimate and minimize GHG emissions from water reuse and desalination facilities.

Input Data

- A model with limited data inputs consistent with the functionality of a water reuse or desalination facility
- Information on the input data availability in practice

Output Data

- Separation of direct emissions associated with processes (Scope 1), indirect emissions associated with electricity use (Scope 2), and other indirect emissions associated with material consumption and other related activities (Scope 3) specific to water reuse or desalination facilities
- Inclusion of specific emissions (CO_2, CH_4, N_2O) and CO_2 equivalents (reprinted with permission from Stokes and Horvath, 2011b)

System Boundary

- Consistent framework for water reuse and desalination system boundary selection
- GHG emissions produced during the decommission life stage

Methodology

- Method that provides enough detailed data to determine critical areas where GHG emissions can be minimized
- Consistent method for direct emission estimates
- Energy estimation equations for unit processes specific to water reuse and desalination facilities

Model Validation

- Model validation for Scope 1 and Scope 2 emissions
- Sensitivity analysis and model calibration integrated into a model

Emission Sources Considered

- Direct emissions from various unit processes specific to water reuse and desalination facilities
- Indirect emissions associated with membrane production, renewal, and disposal
- Indirect emissions associated with disposal of brine effluent

- GHG emissions associated with on-site renewable energy generation and integrated resource recovery
- Inclusion of biogenic emissions for water reuse and desalination processes when applicable

Transferability

- Regionally transferable model specific to water reuse or desalination
- Assessments of GHG emissions for different types of water reuse (agricultural, direct potable, indirect potable)
- Detailed data on energy consumption of different water reuse and desalination unit processes for facilities of varying capacities

Chapter 7

Availability and Implementation of Emission Models

7.1 Summary of Available Models

Table 7.1 shows a summary of off-the-shelf emission models analyzed in this report. This table includes the four model types, including LCA-based, specific, hybrid LCA, and other related models. The name of the emission model, tool type, availability, and Website or contact information are also provided in Table 7.1.

LCA-based models include SimaPro, Gabi, and SiSOSTAQUA software, all commercially available programs that can be used to estimate GHG emissions. These programs have previously been used to estimate life cycle GHG emissions from water reuse and desalination facilities. They provide a useful model for utilities interested in carbon footprinting over the life cycle of a system but can be data- and time-intensive.

Specific models include the Tampa Bay Water Model and the Johnston Model, both MS Excel–based GHG estimation models available upon request. The Tampa Bay Water Model has been applied previously to a desalination facility and could be used to estimate operational indirect emissions from electricity consumption of water reuse facilities. The Johnston Model is specific to water treatment facilities but provides a useful framework for estimating operational GHG emissions from water reuse and desalination facilities (e.g., energy estimation equations based on algorithms, inclusion of direct and indirect emissions, and presentation of results as Scopes 1, 2, and 3 emissions). Both of these models focus on operational-phase emissions only.

Hybrid LCA-based models include WEST, WWEST, and WESTWeb; WEST and WWEST are MS Excel–based models available on request, and WESTWeb is publicly available online. These models have been applied previously to water reuse and desalination facilities to estimate life cycle GHG emissions. Direct and indirect emissions from both the construction and operation phases are included. These models are the most comprehensive and are therefore more time- and data-intensive than those focused on operational-phase emissions.

Finally, other related models include CHEApet, UK Environment Agency Model, Bridle Model, BSM2G Model, System Dynamics, GPS-X, the Carbon Accounting Workbook (5th version), and mCO2. These models contain varying degrees of applicability to water reuse and desalination facilities but were deemed to be less applicable than others previously mentioned. CHEApet is a publicly available, web-based tool that focuses on GHG emissions from wastewater treatment facilities, whereas the Environment Agency and Carbon Accounting Workbook are MS Excel models specific to a UK context. The Environment Agency Model is available upon request, and the Carbon Accounting Workbook is commercially available. The Bridle Model and BSM2G Model are publicly available; however, software is required to run simulations. The remaining models (System Dynamics, GPS-X, and mCO2) are all commercially available software used to estimate GHG emissions. System Dynamics models have been applied previously to water reuse and desalination facilities, whereas GPS-X is specific to wastewater treatment, and mCO2 is designed for water and wastewater facilities.

Table 7.1. Summary of Model Availability

Model Type	Emission Models	Tool Type	Available	Website or Contact Information
LCA-based models	SimaPro	Software	Commercially	www.pre.nl
	Gabi	Software	Commercially	www.gabi-software.com
	SiSOSTAQUA	Software	Commercially	www.simpple.com
Hybrid LCA-based	WEST	MS Excel	On request	Dr. Jennifer Stokes at ucbwaterlca@gmail.com
	WWEST	MS Excel	On request	Dr. Jennifer Stokes at ucbwaterlca@gmail.com
	WESTWeb	Web based	Publicly	west.berkeley.edu
Specific models	Tampa Bay Water	MS Excel	On request	www.tampabaywater.org
	Johnston Model	MS Excel	On request	Dr. Tanju Karanfil at tkaranf@clemson.edu
Other related models	CHEApet	Web based	Publicly	cheapet.werf.org
	UK Environment Agency Model	MS Excel	On request	Environment Agency at enquiries@environment-agency.gov.uk
	Bridle and BSM2G Models	Software	Publicly	Dr. Lluis Corominas at lcorominas@icra.cat
	System Dynamics	Software	Commercially	www.iseesystems.com
	GPS-X	Software	Commercially	www.hydromantis.com/GPS-X.html
	Carbon Accounting Workbook, 5th version	MS Excel	Commercially	www.ukwir.org
	mCO2	software	Commercially	www.mwhglobal.com

7.2 Model Implementation Summary

On the basis of the review provided, the WEST, WWEST, WESTWeb, Johnston, and Tampa Bay Water models can be used to estimate GHG emissions associated with treatment processes at water reuse and desalination facilities. A major limiting factor to implementation of these models, however, is the availability of input parameters (e.g., amount of water pumped, electrical usage, energy production) collected in practice.

To evaluate the ability to implement these models, a list of input parameters was developed for the WEST, WESTWeb, and Tampa Bay Water models and provided to utility partners to determine which input parameters they collected. WWEST input parameters are included in WEST and WESTWeb parameters. Additional WWEST input parameters include water quality data that are collected by most water reuse facilities by regulation (e.g., influent BOD). Johnston Model input parameters are also included in the WEST input parameters; therefore, WWEST and Johnston Model input data parameters were not sent to the utility partners. The surveys were designed to determine if input parameters were assessed for collection, secondary treatment, tertiary treatment, and distribution phases. Input parameters included in the survey were those necessary for estimation of GHG emissions by the WEST, WESTWeb, and Tampa Bay Water models. (See Appendix C for a comprehensive list of all input parameters included in the survey.) Survey responses or correspondence were obtained from the following utility partners:

- Tampa Bay Water, Tampa, FL (desalination)
- Palm Beach County Water Utilities, Palm Beach District, FL (water reuse)
- San Elijo Joint Powers Authority Water Reclamation Facility, San Elijo, CA (water reuse)
- Miami-Dade Water and Sewer Department Water Reclamation Facility, Miami-Dade, FL (water reuse)
- City of Tampa Wastewater Department Howard F. Curren Advanced Wastewater Treatment Plant, Tampa, FL (water reuse)

Tampa Bay Water provided information related to its desalination facility. Correspondence with facility personnel indicated that sludge disposal information, some energy production information, and treatment chemical consumption information were available (Bracciano, 2012). These data do not provide enough information to fully use the WEST or WESTWeb models because material production information, equipment use, and material delivery information were not provided; however, the Tampa Bay Water Model is currently being used. WEST and WESTWeb could still be used, but not to their full capacity.

Based on the results provided in Table 7.2, Palm Beach County Water Utilities does not collect all of the input data necessary to use the WEST, WESTWeb, or the Tampa Bay Water models. Information missing from the Palm Beach facility includes data to separate secondary and tertiary treatment, the electricity mix, and electrical usage from pumping. All these parameters are necessary for at least one of the models. This facility would not be able to estimate GHG emissions from pumping or fuel use by equipment and vehicles based on the current data collected.

Table 7.2. Input Parameters Collected by Palm Beach County Water Utilities District[1]

Models and Input Parameters	Data Collected?	Comment
Tampa Bay Water Model		
Amount of water pumped annually (MGY)	Yes	Incl. Collection, treatment, distribution separately
Amount of water produced annually (MGY)	Yes	Incl. Collection, treatment, distribution separately
Electrical usage from pumping per yr (kWh)	No	
Annual gross load of power utility serving your facility (MWh used)	No	
Annual CO_2, CH_4, N_2O emissions from power utility serving facility	No	
Electricity mix of power utility serving your facility (% per energy source)	No	
WEST		
Material production information	Yes	Material type, cost/weight, year of purchase, service life, purchase frequency collected for collection, treatment, distribution
Material delivery information	Some	Cargo weight/deliveries per year collected for collection, treatment, distribution; excl. transportation mode, distance traveled
Equipment use information	No	
Energy production information[2]	Yes	Single meter records electricity for WWTP as a whole during O&M; fuel usage tracked by separate department
Sludge disposal information	No	
WESTWeb		
General information	Yes	
Material production information	Yes	
Infrastructure materials information	Yes	Purchase price, pipe length, amount of items (pipe, fittings, flow meters, valves) incl. in overall cost
Infrastructure piping information	Yes	Purchase price, total pipe length incl. in overall cost
Reinforced concrete materials information	No	
Process equipment information	Yes	Purchase price of filter media, membranes, motors/ generators, turbines, metal tanks, UV lamps, other industrial equipment kept for treatment; purchase price of pumps, blowers, motors/generators, other industrial equipment, electrical, and controls kept for collection, treatment, distribution
Energy production information[3]	No	
Energy use information[4]	Yes	Annual electricity consumption collected from single meter
Treatment chemical consumption info	Yes	Chemical consumption quantities collected for supply, treatment, distribution

Notes: 1=Palm Beach County does not collect separate data for secondary and tertiary treatment; 2=Includes electricity use, fuel use for equipment and vehicles, custom electricity mix; 3=Includes electricity mix data; 4=Includes electricity and fuel use; CH_4=methane; CO_2=carbon dioxide; N_2O=nitrous oxide; O&M=operation and maintenance; WEST=water energy sustainability tool; WESTWeb=WEST for the Web; WWTP=wastewater treatment plant.

Similarly, the San Elijo Joint Powers Authority Water Reclamation Facility (3-MGD capacity) and Miami-Dade Water and Sewer Department Water Reclamation Facility (10-MGD capacity) cannot use any of the current models because of a limitation in data availability. For example, from personal communication with personnel from the San Elijo facility, one electricity meter is installed for both secondary and tertiary treatment processes. Consequently, GHG emissions from electricity consumption for secondary and tertiary treatment cannot be separated. In addition, based on correspondence with San Elijo facility staff, it is unclear how often the electricity data are recorded or if the information is stored in a database. Data not routinely collected at this facility include amounts of water pumped, electricity mix, material production and delivery information, equipment use, and energy use. The only information provided by the Miami-Dade Water Reclamation Facility was flow data.

Aside from Tampa Bay Water, the Howard F. Curren Advanced Wastewater Treatment Plant was the only facility surveyed for this report that was capable of using at least one model—the Tampa Bay Water Model. Review of its survey (Table 7.3) shows that the facility is not capable of using the entire WEST or WESTWeb models because it lacks information regarding material production and delivery, process equipment, reinforced concrete materials, and equipment use.

This sample reveals that some limitations may come from data availability. These limitations prevent the successful implementation of a water reuse or desalination GHG estimation model. Further research is needed to determine if these input parameters are regularly collected at other facilities around the country and, if not, what would be the resource allocation required to obtain such data.

Table 7.3. Input Parameters Collected by Howard F. Curren Advanced Wastewater Treatment Plant

Models and Input Parameters	Data Collected?	Comment
Tampa Bay Model		
Amount of water pumped annually (MGY)	Yes	Incl. Collection, secondary, tertiary treatment/ distribution separately
Amount of water produced annually (MGY)	Yes	Incl. Collection, secondary, tertiary treatment/ distribution separately
Electrical usage from pumping per year (kWh)	Yes	Collected for wwtp but does not distinguish between unit process (e.g., collection, treatment, distribution)
Annual gross load of power utility serving your facility (MWh used)	Yes	Collected by power utility but not for specific unit processes
Annual CO_2, CH_4, N_2O emissions from power utility serving facility	Yes	Collected by power utility but not for specific unit processes
Electricity mix of power utility serving your facility (% per energy source)	Yes	Collected by power utility but not for specific unit processes
WEST		
Material production information	No	Aggregated records not kept for material production information
Material delivery information	Some	Available for chemical deliveries during operation; excludes delivery of construction materials
Equipment use information	Some	Equipment/miles logged by diesel station in aggregated monthly reports during o&m; not available for specific unit processes or during construction phase
Energy production information	Yes	Electricity recorded for WWTP as a whole during o&m; cannot be separated by unit process or specific machine
Sludge disposal information	Yes	Gas recovery efficiency, sludge disposal location varies
WESTWeb		
General information	Yes	Annual wastewater production in annual report
Material production information	No	Aggregated records not kept
Infrastructure materials information	No	Aggregated records not kept
Infrastructure piping information	No	Aggregated records not kept
Reinforced concrete materials information	No	Aggregated records not kept
Process equipment information	No	Turbines and uv lamps not used
Energy production information	Yes	Collected by power utility but not for specific unit processes
Energy use information	Yes	Collected for electricity, natural gas (formerly for sludge drying, no longer used), gasoline, and diesel; cannot be allocated to unit processes
Treatment chemical consumption information	Yes	

Notes: CH_4=methane; CO_2=carbon dioxide; N_2O=nitrous oxide; O&M=operation and maintenance; UV=ultraviolet; WEST=water energy sustainability tool; WESTWeb=WEST for the Web; WWTP=wastewater treatment plant

7.3 Analysis of Potential Application of Available Models

7.3.1 Comparison of Two Models

To assess the GHG emission models evaluated in this report, two models were compared in a case study: the Tampa Bay Water Model and the WEST Model. These two models were assessed because they represent the range of GHG estimation model capabilities. The Tampa Bay Water Model could be considered the simplest system because it requires the fewest inputs and only estimates GHG emissions associated with the generation of operational electricity (Scope 2 emissions). In contrast, the WEST Model could be considered the most comprehensive because it requires more inputs and estimates GHG emissions associated with the generation of electricity (Scope 2 emissions) in addition to other indirect emissions from materials, chemicals, and transportation (Scope 3 emissions). Both models yield an output expressed in mass of CO_2eq per volume of water produced; however, the Tampa Bay Water Model can also calculate specific GHG emissions (CO_2, CH_4, and N_2O). To compare the two models, input inventory data from a previous application of the WEST model (Stokes and Horvath, 2009) was applied to the Tampa Bay Water Model for (1) seawater desalination with membrane pretreatment, (2) brackish water desalination, and (3) a water reclamation facility. As defined in Stokes and Horvath (2009), these are potential sources to replace a facility in California with a capacity of 36 million m^3 per day. Results from the WEST Model were then compared to the Tampa Bay Water Model to highlight differences between them. Inputs for the Tampa Bay Water Model were obtained from Stokes and Horvath (2009) and the eGRID database and are shown in Tables 7.4 and 7.5. A summary of input data used in the WEST Model is shown in Table 7.6.

Table 7.4. Input Data Used in Tampa Bay Water Model

Model Input	Desalinated Seawater, Membrane Pretreatment	Desalinated Brackish Groundwater	Recycled Water
Water produced (m^3/yr)	36,000,000	36,000,000	36,000,000
Electricity use (kWh/yr)	183,600,000	103,680,000	77,040,000

Source: Stokes and Horvath, 2009

Table 7.5 Input Data Used in Tampa Bay Water Model, Collected from eGRID

Pollutant	Output Emission Rate (lbs/kWh)
Annual CH_4	0.000031
Annual N_2O	0.000005
Annual CO_2	0.540060
Annual CO_2eq[1]	0.542096

Source: http://cfpub.epa.gov/egridweb/view_st.cfm
Notes: 1=Used 100-year global warming potential factors from Tampa Bay Water Model to calculate CO_2eq of CH_4 and N_2O; CH_4=methane; CO_2=carbon dioxide; CO_2eq=carbon dioxide equivalent; eGRID=Emissions & Generation Resource Integrated Database; N_2O=nitrous oxide

Table 7.6. Summary of Input Data Used in WEST Model

Model Inputs	Desalinated Seawater, Membrane Pretreatment	Desalinated Brackish Groundwater	Recycled Water
Piping/aqueduct length	3.2 km	4.8 km	1 km
Electricity use (kWh/y/m^3) treatment	0.38	0.26	0.45
Description	membrane, filtration, RO, disinfection	filtration, RO, disinfection	filtration, disinfection
Chemical use (g/y/m^3)			
acid (hydrochloric or sulfuric)	81	65	
alum			53
aqueous ammonia	8.4	13	
calcium carbonate	26		
caustic soda		17	
chlorine			19
CO_2	26		
ferric chloride	0.4		
sodium hypochlorite	6.5	11	
other	7.5	3	4
Electricity use (kWh/y/m^3) distribution	4	2.4	0.19
Description and pipe length	1003 km	1000 km	nonpotable (35 km)
Electricity use (kWh/y/m^3)	0.72	0.22	1.5

Source: Stokes and Horvath, 2009
Notes: Data expressed per m^3 of water; CO_2=carbon dioxide; RO=reverse osmosis

The actual Tampa Bay Water Model requires specific energy and emission data from the local power provider. To make a fair comparison to the WEST application presented in Stokes and Horvath (2009), the state average electricity mix in California was used instead. Table 7.5 shows the eGRID emission factors used to estimate GHG emissions in this comparison study. The mass per year of the GHG emissions was calculated by multiplying emission factors by annual electricity use. The 100-year global warming potential figures for CH_4, and N_2O used in the Tampa Bay Water Model are 23 and 296, whereas the WEST Model uses 21 and 310. In this study, the values of 21 and 310 were used for fair comparison.

The Tampa Bay Water Model was used to calculate the carbon footprint of the three systems described with the input data listed in Tables 7.3 and 7.4. The output was compared to the carbon footprint calculated by Stokes and Horvath (2009), as shown in Table 7.7. The Tampa Bay Water Model estimates represent 52% of the total life cycle estimates from WEST for the desalinated seawater with membrane pretreatment and recycled water systems and 44% of the total life cycle GHG emissions from WEST for the desalinated brackish groundwater system.

Table 7.7. Output Comparison of Carbon Footprint Using Tampa Bay Water and WEST Models

Facility	Tampa Bay Water Model	WEST Model[1]	% Tampa Bay Water Model of WEST Model Estimate
	CO_2eq (kg)/m^3 Produced	CO_2eq (kg)/m^3 Produced	
Desalinated seawater, membrane pretreatment	1.26	2.40	52%
Desalinated brackish groundwater	0.71	1.63	44%
Recycled water	0.53	1.02	52%

Notes: 1=Base-case scenario in Stokes and Horvath (2009) based on California average life cycle emissions; CO_2eq=carbon dioxide equivalent; WEST=water energy sustainability tool

These values are lower than the percent contribution of energy consumption reported by Stokes and Horvath (2009), in which 92% of the life cycle GHG emissions for the recycled water system and 76% of the emissions for the desalinated brackish water system come from energy consumption. This is because the Tampa Bay Water Model includes only electricity consumption, whereas energy consumption in WEST Model also includes fuel use by equipment and vehicles during construction and operation phases. Another contributor to GHGs included in the WEST Model but excluded in the Tampa Bay Water Model is the production of chemicals, which ranged from 4 to 18% of the cumulative energy consumption (Stokes and Horvath, 2009).

As expected, the Tampa Bay Water Model yields a lower carbon footprint than the WEST Model for all facilities evaluated; however, the Tampa Bay Water Model does provide an accurate baseline estimate of the carbon footprint with minimal inputs required.

In summary, the Tampa Bay Water Model requires inputs of electricity consumption, eGRID data, and flow rate data, and it provides the GHG emissions data associated with on-site energy usage (Scope 2 emissions). In contrast, the WEST Model requires inputs related to material production, material delivery, energy production (fuel and electricity), equipment use, and sludge disposal and provides information on life cycle GHG emissions associated with these activities during the construction and operation phases (Scope 2 and Scope 3 emissions). In addition, WWEST provides a way to estimate Scope 1 emissions from select wastewater treatment processes, which is particularly useful for water reuse systems.

7.3.2 Potential Application of Available Models to Existing Facilities

This report identified those aspects from traditional LCA models, hybrid LCA models, specific models, and other related models that could be applied to estimate the carbon footprint and GHG emissions of existing water reuse and desalination facilities. A robust model should be user friendly, geographically transferable, require limited inputs, and have the capability to estimate GHG emissions from individual and integrated advanced water reuse and desalination treatment technologies. Although traditional LCA models and other related models have limited applicability, the hybrid LCA models and some of the specific models reviewed in this report provide some applicable options. Unfortunately, despite the transferability of certain aspects of these models, there are still major information gaps that will prevent the widespread adoption of these models.

One of the major gaps that prevent development and use of a robust and accurate carbon footprint and GHG estimation model for water reuse and desalination facilities is related to limitations in the type of input data that are currently collected by utilities. These data requirements include, at a minimum, the amount of water pumped, electricity usage, and practices associated with on- or off-site energy production. In addition, it appears it is not common practice to have electric meters associated with specific unit processes that would discern electricity use of different technologies during their operational life stage. This would be useful for identifying and mitigating high impact unit processes within a system. Utilities also currently do not collect data required to support sophisticated models, such as production and delivery of materials and process equipment use. In addition, not all models separate direct and indirect emissions or allow for breakdown of specific emissions (CO_2, CH_4, N_2O) and CO_2 equivalents. Models have limitations in terms of their regional transferability specific to water reuse or desalination, especially when accounting for regional differences in the electricity generation portfolio, the presence of on-site renewable energy, and assessing GHG emissions for different types of water reuse (e.g., agricultural, direct potable, indirect potable).

For existing facilities, the Tampa Bay Model with the statewide electricity generation portfolio information from eGRID can be used as a starting point to estimate the emissions of CO_2, CH_4, N_2O, and CO_2eq associated with purchased electricity (Scope 2 emissions) if the facilities currently collect data on the amount of water pumped and produced and electricity usage for the entire facility. Existing facilities need to start collecting data on electricity consumption associated with specific unit processes, chemical consumption, material consumption, process equipment usage, and on-site renewable energy production so the more sophisticated models can be applied to accurately estimate life cycle GHG emissions and carbon footprint, establishing baselines for water reuse or desalination comparisons.

7.3.3 Potential Application of Available Models to Future Proposed Facilities

For future proposed facilities, not only the capacity but also the potential electricity consumption should be considered in the design. If such information is available, the Tampa Bay Model with the statewide electricity generation portfolio information from eGRID can be used as a screening tool to estimate the Scope 2 emissions of the proposed facilities. Future facilities should establish a standard data collection template and collect required data (as described previously) for the more sophisticated models to accurately estimate GHG emissions and carbon footprint. The hybrid LCA models (WEST, WWEST, and WESTWeb) provide a more detailed estimation for future utilities to account for Scope 2 and Scope 3 emissions both prior to and during the construction phase.

7.4 Key Recommendations for Next Step

7.4.1 Recommendations for Data Collection

Among the models reviewed in this study, the Tampa Bay Water Model is the simplest one, requiring minimum data input, and the hybrid LCA models (WEST, WWEST, and WESTWeb) models are the most sophisticated, requiring extensive data input, as shown in Table 7.6.

Most existing facilities collect the data needed for the Tampa Bay Water Model, such as the amount of water pumped and produced and electricity usage for the entire facility. It is recommended that existing facilities extend their data collection efforts to include, at a minimum, information on electricity providers, amount of water pumped and produced, and facilitywide electricity usage. For both existing and future facilities, establishing a standard data collection template and collecting the following data are recommended: amount of water pumped and produced, name of electricity providers, electricity consumption associated with specific unit processes and entire facility, chemical consumption, material consumption, process equipment usage, and on-site renewable energy production.

7.4.2 Recommendations for Model Development

We observed the need for a user friendly and robust model that would allow utilities and design firms to plan, design, and manage water reuse and desalination facilities so they can account for rising energy costs and emissions of CO_2 and other GHGs over all life stages of an existing or proposed facility. This model should be applicable to different geographical regions that account for different energy mixes in the production of electricity as well as different end uses associated with water reuse (e.g., residential, agricultural, direct potable, indirect potable). This model should have an option that would require different levels of sophistication related to required input parameters; for example, a less input-intensive version could be used to provide a screening level analysis of carbon footprint and GHG emissions, and a second level could provide more detailed output and accordingly require a greater number of model inputs.

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Appendix A: Names and Affiliations of Individuals Contacted

Contact was made with the following to determine if they were aware of models and methodologies for estimating the carbon footprint and GHG emissions associated with water reuse and desalination facilities.

Name	**Affiliation (Country)**
Gregory M. Adams	Sanitation Districts of Los Angeles, United States
Nick Apostolidis	GHD, Australia
Gary Bickford	Nestis Consulting, Australia
David Butler	University of Exeter, United Kingdom
Francesc Castell	Rovira I Virgili University, Spain
Ni-bin Chang	University of Central Florida, United States
Jonathan N. Cooper	Calvin, Giordano & Associates, Inc., United States
John Crittenden	Georgia Institute of Technology, United States
Glen Daigger	CH2M Hill, United States
Francis A. DiGiano	University of North Carolina–Chapel Hill, United States
E. Friedrich	University of KwaZulu-Natal, Australia
Carl Hensman	King County WTD, United States
Arpad Horvath	University of California, Berkeley, United States
Tanju Karanfil	Clemson University, United States
Brian W. Karney	University of Toronto, Canada
Cindy Lee	Clemson University, United States
Manfred Lenzen	University of Sydney, Australia
Ke Li	University of Georgia, United States
Tom Love	Inland Empire Utilities Agency, United States
Sven Lundie	University of New South Wales, Australia
Tek Narayan Maraseni	University of Southern Queensland, Australia
Ivan Muñoz	University of Alcala, Spain
Linda Reekie	Water Research Foundation, United States
Lisa Rephlo	MWH Global, United States
Martin Rygaard	Institut for Vand og Miljøteknologi, Denmark
Michael W. Selna	Sanitation Districts of Los Angeles, United States
L. Serra	University of Zaragoza, Spain
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Thomas F. Speth	U.S. Environmental Protection Agency, United States
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Appendix B: Life Cycle Stages Included in Reviewed Studies

Table B.1. Life Cycle Stages Included in Literature on Desalination Facilities

Life Cycle Stages	No. of Papers	Reference(s)
O&M	4	Raluy et al., 2005a; Biswas, 2009; Beery et al., 2010; Shrestha et al., 2011
Construction, O&M, decommission	5	Raluy et al., 2004; Muñoz and Fernández-Alba, 2008; Jijakli et al., 2011
O&M and decommission	1	Peters and Rouse, 2005

Note: O&M=operation and maintenance

Table B.2. Life Cycle Stages Included in Literature on Water Reuse Facilities

Life Cycle Stages	No. of Papers	Reference(s)
O&M	3	Stillwell and Webber, 2010; Shrestha et al., 2012, Fine and Hadas, 2012
Construction and O&M	2	Tangsubkul et al., 2005; Zhang et al., 2010
Construction, O&M, decommission	2	Ortiz et al., 2007; Friedrich et al., 2009[1]

Notes: 1=Includes decommission of wastewater treatment plant but not water reclamation facility; O&M=operation and maintenance

Table B.3. Life Cycle Stages Included in Literature on Both Water Reuse and Desalination

Life Cycle Stages	No. of Papers	Reference(s)
O&M	3	Muñoz et al., 2009; Pasqualino et al., 2010; Meneses et al., 2010
Construction and O&M	5	Lundie et al., 2004; Stokes and Horvath, 2006, 2009; Lyons et al., 2009; de Haas et al., 2011
Construction, O&M, decommission	1	Muñoz et al., 2010[1]

Notes: 1=Includes construction, O&M, and decommission for desalination plants, construction and O&M for advanced WWTP and other treatment facilities; O&M=operation and maintenance

Appendix C: Survey Sent to Utility Partners

Survey Sent to Utility Partners	X=have or collect this input parameter						
Do you have or collect the following data at your water reuse or desalination facility? Please indicate if you have this information for the following water supply and life cycle phases	Water Supply Phase				Life-Cycle Phase		
Tampa Bay Model Input Parameters	Supply	Secondary Treatment	Tertiary Treatment	Distribution	Construction	O&M	Comments
Amount of water pumped annually (MGY)							
Amount of water produced annually (MGY)							
Electrical usage from pumping per year (kWh)							
Annual Gross load of power utility serving your facility (MWh used)							
Annual CO2 emissions from each power utility serving your facility (tons)							
Annual CH4 emissions from each power utility serving your facility (tons)							
Annual N2O emissions from each power utility serving your facility (tons)							
Electricity mix of power utility serving your facility (% per energy source)							

Do you have or collect the following data at your water reuse or desalination facility? Please indicate if you have this information for construction, operation, and/or maintenance phases.	Water Supply Phase				Life-Cycle Phase			
WEST Input Parameters	Supply	Secondary Treatment	Tertiary Treatment	Distribution	Construction	O&M	End-of-Life	Comments
Material Production								
Material Type (e.g. chemicals, piping, concrete, etc.)								
Material Cost or Weight (kg)								
Year of Purchase								
Service Life (years)								
Purchase Frequency								
Material Delivery								
Cargo weight (kg)								
Deliveries per year								
Type of Transportation and Distance (Primary mode)								
Type of Transportation and Distance (Secondary mode)								
Equipment Use								
Equipment Type (e.g. dump truck, excavator)								
Use Amount (hours)								
Frequency (hours, miles)								
Energy Production								
Electricity Use (kWh)								
Fuel Use by Equipment (gallons)								
Fuel Use by Vehicles (gallons)								
Custom Electricity Mix (% contribution from sources)								
Sludge Disposal								
Amount (tons/year)								
Facility Type								
Gas recovery Type and efficiency								
Transport Distance (miles)								

Do you have or collect the following data at your water reuse or desalination facility? Please indicate if you have this information for construction, operation, and/or maintenance phases.	Water Supply Phase				Life-Cycle Phase		
WESTWeb Input Parameters	Supply	Secondary Treatment	Tertiary Treatment	Distribution	Construction	O&M	Comments
General							
Annual wastewater production volume (gallons/year)							
Material Production							
Infrastructure Materials - Purchase price of the following items:							
Piping							
Fittings							
Flow meters							
Valves							
Infrastructure Piping - Material, length and diameter of piping							
Reinforced Concrete Materials - Total volume of concrete used							
Process Equipment - Purchase price of the following (if applicable):							
Filter media (sand, gravel, anthracite, or other coal product)							
Membranes							
Pumps							
Fans/Blowers							
Motors and Generators							
Turbines							
Metal Tanks							
UV Lamps/lights							
Other Industrial Equipment							
Electrical							
Controls							
Energy Production							
Electricity Mix - Percentage of fuel/energy from source:							
Coal							
Oil							
Natural Gas							
Nuclear							
Hydro							
Biomass							
Wind							
Solar							
Geothermal							
Energy Use – Annual consumption of:							
Electricity (MWh)							
Natural Gas (MMBtu)							
Gasoline (gallons)							
Diesel (gallons)							
Treatment Chemical Consumption - Quantities of chemicals used (lb/year) for:							
pH Adjustment (Hydrochloric Acid, Sulphuric Acid)							
Coagulants & Flocculants (Aluminum Sulfate, Aluminum Hydroxide, Caustic Soda, Ferric Chloride, Polymers)							
Disinfectants (Chlorine, Calcium Hypochlorite, Ozone, Aqueous Ammonia)							
Other (Fluorosilicic Acid, Other chemicals)							

Practical Solutions for Water Scarcity




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